HOTSPOTS FIRE PROJECT

Fire and the Vegetation of the Southern Rivers Region

Draft for Comment

Dr Penny Watson
Project Ecologist
December 2006

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This document has been prepared to help Hotspots and local NRM staff provide accurate information to landholders in the Southern Rivers region. The information it contains reflects our understanding at the time of writing. We are learning more about fire and the environment every day and anticipate that some recommendations may change as new information comes to hand.

Suggestions for improvement are welcome. Please contact Penny on (02) 9477-7361 or at pennyw@efa.com.au before 28 February 2007.

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1 Introduction

1.1 Scope of this review

This literature review forms part of a suite of materials that Hotspots aims to produce in each CMA region in which it works. While most Hotspots products are targeted to landholders, literature reviews are primarily directed towards a professional audience. Their major aim is to provide ecological background to underpin and inform the messages about fire that Hotspots and local NRM practitioners present. A secondary aim is to offer a platform for discussion and debate on the role of fire in regional vegetation types. In both cases we hope the outcome will be more informed fire management for biodiversity conservation.

The review considers literature relevant to a subset of vegetation classes in the Southern Rivers CMA region of New South Wales (NSW). It aims to help land and fire managers not only to understand the impacts of fire in the region, but also to place that understanding in a wider ecological context. While the companion document for the Northern Rivers (Watson 2006) was framed as a critical review of the several sets of fire frequency guidelines already in existence for that region, this document takes a more descriptive approach.

The Southern Rivers CMA region, which stretches from the coastal strip above Wollongong to the Victorian border and west onto the Southern Tablelands, covers an area of great climatic diversity. Climate on the Kiama-Illawarra escarpment is warm and moist, while the Tablelands are considerably cooler and drier. The Region also encompasses alpine areas which are cool, but also moist (Gellie 2005:238). Annual rainfall varies from 645 mm at Bombala to 2252 mm at Charlotte Pass, and from 1277 mm on the coast in the north of the region at Port Kembla, to 837 mm at Merimbula on the south coast. Average maximum temperatures in January for these four weather stations are 25, 17, 24 and 24°C, while equivalent figures for July are 11, 2, 17 and 16°C. “Frost days”, defined as days when the temperature drops below 0°C, average 74, 180, 0 and 1 per year, respectively (Bureau of Meteorology 2005). Vegetation naturally reflects these considerable gradients, and is in turn extremely diverse. Soil fertility also influences plant associations, adding to the rich array of vegetation in the Region (Gellie 2005:238).

Fire affects different plant and animal species differently, and fire regimes compatible with biodiversity conservation vary widely between ecosystems (Bond 1997, Watson 2001, Kenny *et al.* 2004). This document explores the role of fire in a sample of Southern Rivers vegetation classes. The selection process has been informed by:

- whether local research exists
- whether the vegetation class is one for which the Hotspots audience is likely to have responsibility
- whether the vegetation class is one with which the Hotspots audience is likely to be familiar
- whether exploration of issues in the vegetation class has the potential to further understanding of the role of fire in the Southern Rivers region and across the State.

Vegetation classes discussed in this review include:

- Heathlands: Sydney Coastal Heaths, South Coast Heaths (Chapter 2)
- Grassy woodlands and grasslands: Temperate Montane Grasslands, Coastal Valley Grassy Woodlands (Chapter 3)
- Dry sclerophyll forests: Southern Tableland Dry Sclerophyll Forests, South East Dry Sclerophyll Forests (Chapter 4)
- Wet sclerophyll forests: Montane Wet Sclerophyll Forests, Southern Lowland Wet Sclerophyll forests (Chapter 5).

Names and descriptions for these vegetation classes have been drawn initially from Keith (2004). However a more detailed, empirically-based vegetation typology covering much of the region is also available: Gellie (2005). While the two classification schemes have common ground, they also differ in various ways. I have attempted to link the two when introducing each vegetation class, however detailed consideration of similarities and differences is beyond the scope of this document. Judgement has also inevitably been exercised when deciding the relevance of research studies to particular vegetation classes.

As understanding of the role of fire in Australian ecosystems increases, the importance of fire cycles – that is of a series of fires rather than of any one single fire – becomes increasingly apparent. Issues related to fire frequency thus feature prominently in this review. However the influence of other aspects of the fire regime – season, intensity and extent – will also be discussed where research is available. While vegetation and plant species receive most attention, fauna studies are also discussed.

Before addressing the literature on specific vegetation classes, some ecological concepts and principles underlying current understanding of fire regimes are explored in the remainder of this chapter. Fire-related attributes that vary between species are canvassed, along with a range of concepts including disturbance, succession, interspecific competition, landscape productivity and patch dynamics. These ideas provide a framework which helps explain how fires have shaped the landscape in the past, and how fire management can best conserve the diversity of the bush in the future. They thus give context to specific research findings, and can assist understanding of differences between vegetation types.

1.2 Species responses to fire

Plant species differ in the way they respond to fire. Fire-related characteristics or attributes which vary between species include:

- Regeneration mode – the basic way in which a species recovers after fire
- How seeds are stored and made available in the post-fire environment
- When, relative to fire, new plants can establish
- Time taken to reach crucial life history stages.

Plant communities are made up of species with a variety of fire-related attributes. These differences mean plant species are differentially affected by different fire regimes; fire regimes therefore influence community composition. This topic is developed in Section 1.4.

1.2.1 Regeneration modes

In a seminal article in 1981, Gill classified plants as “non-sprouters” or “sprouters”, on the basis of whether mature plants subjected to 100% leaf scorch die or survive fire. Most adults of sprouting species, also called ‘**resprouters’** regrow from shoots after a fire. These shoots may come from root suckers or rhizomes, from woody swellings called lignotubers at the base of the plant, from epicormic buds under bark on stems, or from active pre-fire buds (Gill 1981). Some resprouters, ie those which regrow from root suckers or rhizomes (such as blady grass and bracken), can increase vegetatively after a fire. However other resprouters cannot increase vegetatively, and therefore need to establish new plants to maintain population numbers, as adults will eventually age and die.

On the other hand, adults of non-sprouting species, or ‘**obligate seeders**’, die when their leaves are all scorched in a fire, and rely on regeneration from seed.[1] Obligate seeder species generally produce more seed (Lamont *et al.* 1998), and greater numbers of seedlings (Wark *et al.* 1987, Benwell 1998) than resprouters, and seedling growth rates tend to be more rapid (Bell and Pate 1996, Benwell 1998, Bell 2001).

These categories are not invariant. Survival rates in the field for both resprouters and obligate seeders change with fire intensity (Morrison and Renwick 2000). Some species exhibit different regeneration strategies in different environments (Williams *et al.* 1994, Benwell 1998).

1.2.2 Seed storage and dispersal

Fire provides conditions conducive to seedling growth. Shrubs, grass clumps, litter and sometimes canopy cover are removed, allowing increased light penetration to ground level and reducing competition for water and nutrients (Williams and Gill 1995, Morgan 1998a). For plant species to take advantage of this opportunity, seeds need to be available. There are several ways in which this can be accomplished.

Some species hold their seeds in on-plant storage organs such as cones, and release them after a fire. These ‘serotinous’ taxa include species in the Proteaceae and Cupressaceae families, for example Banksia, Hakea and Callitris. Some eucalypts release seed in response to fire (Noble 1982, Gill 1997). The degree to which seed release also occurs in the absence of fire varies between species (Enright *et al.* 1998).

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[1] The term ‘fire sensitive’ is sometimes used as a synonym for obligate seeder, however I tend not to use this term as it can be taken to imply that all obligate seeder species are liable to damage when exposed to any fire. In fact, many obligate seeders need fire for reproduction, and are advantaged by moderately frequent fire (eg Watson 2005). By the same token, I tend not to use the term ‘fire tolerant’ for resprouters, as it can be taken to imply that any fire regime is okay for these species. In fact, frequent burning can cause substantial declines in populations of some resprouting species (eg Watson and Wardell-Johnson 2004).

A second group of species stores dormant seeds in the soil; dormancy requirements ensure germination occurs mostly after fire. Heat promotes germination in legumes (Shea *et al.* 1979, Auld and O'Connell 1991, Clarke *et al.* 2000), while smoke plays a role for many species (Dixon *et al.* 1995, Roche *et al.* 1998, Flematti *et al.* 2004). Some taxa respond best to a combination of these two fire-related cues (Morris 2000, Thomas *et al.* 2003).

A third strategy is to create seeds rapidly after a fire, through fire-cued flowering. *Xanthorrhoea* species are a well-known example of this phenomenon (Harrold 1979, McFarland 1990), however shrubs such as *Lomatia silaifolia* (Denham and Whelan 2000) and *Telopea speciossima* (Bradstock 1995) also flower almost exclusively in the years after a fire. Many grassland forbs[2] exhibit this characteristic (Lunt 1994).

Finally, some species rely to a greater or lesser extent on seed coming in from outside the burnt area. This strategy is not common in fire-prone environments: seed dispersal distances in Australian fire-prone vegetation seem to be limited to tens of metres or less in most species (Auld 1986a, Keith 1996, Hammill *et al.* 1998). However some wind and vertebrate-dispersed species do occur in these environments; examples include plants with fleshy fruits such as *Persoonia* species and some epacrids (eg *Styphelia viridis*, *Leucopogon* spp.). These species may have a different relationship to fire cycles than do taxa whose seeds are not widely dispersed (French and Westoby 1996, Ooi *et al.* 2006).

1.2.3 Recruitment relative to fire

Species also differ in when they establish new plants relative to fire. For many species in fire-prone environments, recruitment is confined to the immediate post-fire period (Auld 1987, Zammit and Westoby 1987, Cowling *et al.* 1990, Vaughton 1998, Keith *et al.* 2002a, Section 2.1.5), although this may vary between populations (Whelan *et al.* 1998) and with post-fire age (Enright and Goldblum 1999). Some species, however, recruit readily in an unburnt environment, and are therefore able to build up population numbers as time goes by after a fire.

1.2.4 Life history stages

The time taken to complete various life stages affects a species' ability to persist in a fire-prone environment. Time from germination to death of adult plants, time to reproductive maturity and, for resprouters, time to fire tolerance are important variables, as is duration of seed viability.

The time from seed germination to reproductively-mature adult is known as a species' '**primary juvenile period**'. Resprouting species also have a '**secondary juvenile period**': the time taken for vegetative regrowth to produce viable seed (Morrison *et al.* 1996). The length of these periods differs between species, and may even differ within a species, depending on location (Gill and Bradstock 1992, Knox and Clarke 2004). Once flowering has occurred, it may take additional years before viable seed is produced, and even longer to accumulate an adequate seedbank (Wark *et al.* 1987, Bradstock and O'Connell 1988).

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[2] A forb is a herbaceous plant which is not a grass, sedge or rush.

In resprouters, the primary juvenile period is often much longer than the secondary juvenile period, as well as being longer than the primary juvenile period in equivalent obligate seeders (Keith 1996, Benwell 1998). Resprouter seedlings are not immediately fire tolerant: it may take many years before lignotuber development or starch reserves are sufficient to allow the young plant to survive a fire (Bradstock and Myerscough 1988, Bell and Pate 1996).

The length of time seed remains viable is another important variable, but one about which not a great deal is known. It is clear, however, that species vary greatly (Keith 1996). The seedbanks of serotinous species are likely to be depleted more quickly than those of species with soil-stored seed, although much variation exists even here (Gill and Bradstock 1995, Morrison *et al.* 1996). Species also vary in whether seedbanks are exhausted by a single fire (either through germination or destruction). Species which store seed in the canopy, and species whose soil-stored seeds are relatively permeable (eg *Grevillea* species; Morris 2000), are unlikely to retain a residual store of pre-fire seeds through the next interfire interval. However some species, generally those with hard, soil-stored seeds such as peas and wattles, retain viable ungerminated seed through more than one fire; *Bossiaea laidlawiana*, from south-west Western Australia, is an example (Christensen and Kimber 1975). This gives these species a ‘hedge’ against a second fire within the juvenile period.

1.3 Disturbance, succession and a paradigm shift

In this section the focus shifts to broader matters. Disturbance and succession are basic concepts in ecology. Advances in understanding of these processes has informed several theories and models which are useful for understanding the way plant communities – groupings of species with different fire-related attributes – respond to fire.

Fire is a **disturbance**. A disturbance can be defined as “any relatively discrete event in time that removes organisms and opens up space which can be colonised by individuals of the same or different species” (Begon *et al.* 1990). The concept encompasses recurring discrete events such as storms, floods and fires, as well as on-going processes like grazing. Disturbance may stem from natural phenomena or human activities (Hobbs and Huenneke 1992), and is ubiquitous throughout the world’s ecosystems (Sousa 1984).

Succession follows disturbance. This concept has been of interest to ecologists since Clements outlined what is now called ‘classical succession’ in 1916. In classical succession “following a disturbance, several assemblages of species progressively occupy a site, each giving way to its successor until a community finally develops which is able to reproduce itself indefinitely” (Noble and Slatyer 1980:5). Implicit in this model is the idea that only the final, ‘climax’ community is in equilibrium with the prevailing environment.

A popular metaphor for this **equilibrium paradigm** is ‘the balance of nature’. Conservation practice aligned with this model focuses on objects rather than processes, concentrates on removing the natural world from human influence, and believes that desirable features will be maintained if nature is left to take its course (Pickett *et al.* 1992). Fire does not sit easily in the balance of nature approach, which influenced attitudes to burning, both in Australia and elsewhere, for many years. For example, forester C.E. Lane-Poole argued to the Royal Commission following the 1939 fires in Victoria for total fire exclusion on the grounds that this would enable natural succession to proceed resulting in a less flammable forest (Griffiths, 2002).

Over recent decades, however, a paradigm shift has been underway. Drivers include the realisation that multiple states are possible within the one community (Westoby *et al.* 1989), as are multiple successional pathways (Connell and Slatyer 1977). Most importantly from a conservation perspective, it has increasingly been recognised that periodic disturbance is often essential to maintain diversity, allowing species which might otherwise have been displaced to continue to occur in a community (Connell 1978).

This **non-equilibrium paradigm** can be encapsulated by the phrase 'the flux of nature'. **Scale** is important in this paradigm: equilibrium at a landscape scale may be the product of a distribution of **states** or **patches** in flux (Wu and Loucks 1995). Implications include a legitimate – or even vital – role for people in ecosystem management, and a focus on the conservation of processes rather than objects. This does *not,* of course, imply that all human-generated change is okay; it does mean human beings must take responsibility for maintaining the integrity of ecosystem processes (Pickett *et al.* 1992, Partridge 2005). Fire fits much more comfortably into the non-equilibrium paradigm, where it takes its place as a process integral to many of the world's ecosystems.

1.4 Theory into thresholds

The non-equilibrium paradigm forms the basis for a number of theories and models which have been used to inform an understanding of fire regimes in Australia. These include the **vital attributes model** of Noble and Slatyer (1980). This scheme employs the fire-related characteristics of plant species outlined in Section 1.2 to predict successional pathways. It can also be used to define disturbance frequency domains compatible with maintenance of particular suites of species. This model has recently been used to develop fire management guidelines for broad vegetation types in NSW (Kenny *et al.* 2004).

The basic idea is that, to keep all species in a community, fire intervals should vary within an upper and a lower threshold. Lower thresholds are set to allow all species vulnerable to frequent fire to reach reproductive maturity, while upper thresholds are determined by the longevity of species vulnerable to lack of burning. Species with similar fire-related characteristics are grouped into functional types (Noble and Slatyer 1980, Keith *et al.* 2002b). The vulnerability of each group, and of species within sensitive groups, can be assessed through consideration of their 'vital attributes'`.

Functional types most sensitive to **short interfire intervals** (high fire frequency) contain obligate seeder species whose seed reserves are exhausted by disturbance. Populations of these species are liable to local extinction if the interval between fires is shorter than their primary juvenile period (Noble and Slatyer 1980). The minimum interfire interval (lower threshold) to retain all species in a particular vegetation type therefore needs to accommodate the taxon in this category with the longest juvenile period (DEC 2002).

Species whose establishment is keyed to fire (Noble and Slatyer call these 'I species') are highly sensitive to **long interfire intervals** (infrequent fire): they are liable to local extinction if fire does not occur within the lifespan of established plants and/or seedbanks (Noble and Slatyer 1980). The maximum interval (upper threshold) therefore needs to accommodate the taxon in this category with the shortest lifespan, seedbank included (DEC 2002, Bradstock and Kenny 2003).

Data on plant life history attributes relevant to setting **lower thresholds** – regeneration modes and juvenile periods – are much more readily available than the information needed to set **upper thresholds** – longevity of adults and seeds. Kenny *et al.* (2004) note the lack of quantitative data on these latter attributes, and point out that as a result, upper thresholds in the NSW guidelines are "largely based on assumptions and generalisations" and are therefore surrounded by "considerable uncertainty" (Kenny *et al.* 2004:31). Work on these variables is an important task for the future. It can also be argued that upper thresholds need to consider not only the characteristics of individual plant species, but also competitive interactions between species. This issue is explored in the next section.

1.5 Competition and productivity

The effect of dominant heathland shrubs on other species has been recognised in Sydney's sandstone country (Keith and Bradstock 1994, Tozer and Bradstock 2002, Section 2.1.4). When life history characteristics alone are considered, a feasible fire frequency for the conservation of both these dominant obligate seeders and understorey species appears to be 15-30 years. However under this regime the dominant species form high-density thickets which reduce the survival and fecundity of species in the understorey, an effect which carries through to the next post-fire generation. Similar dynamics have been observed in other Australian heath communities (Specht and Specht 1989, Bond and Ladd 2001) and in South Africa's heathy fynbos (Bond 1980, Cowling and Gxaba 1990, Vlok and Yeaton 2000). An understanding of this dynamic has highlighted the need to include in heathland fire regimes some intervals only slightly above the juvenile period of the dominant species, thus reducing overstorey density for a period sufficient to allow understorey taxa to build up population numbers before again being overshadowed (Bradstock *et al.* 1995).

The competitive effect on understorey vegetation may be particularly profound where dominant shrubs resprout (Bond and Ladd 2001). Unlike obligate seeders, dominant resprouters will continue to exert competitive pressure immediately after a fire by drawing on soil resources, and once their cover is re-established, on light resources too. Their potential to outcompete smaller species in the post-fire environment may therefore be considerable. These dynamics have been documented in Western Sydney's Cumberland Plain Woodland, where dense thickets of the prickly shrub *Bursaria spinosa* are associated with a reduced abundance of other shrub species, particularly obligate seeders (Watson 2005, Section 3.2.2).

Bursaria has the unusual advantage of being able to recruit between fires, whereas most sclerophyllous (hard-leaved) shrub species recruit almost exclusively after a fire (Purdie and Slatyer 1976, Cowling *et al.* 1990, Keith *et al.* 2002a). The vital attributes model explicitly identifies species able to recruit between fires – Noble and Slatyer call them 'T species' – and their propensity to dominate in the absence of disturbance is also explicitly noted (Noble and Slatyer 1980). However to date little emphasis has been placed on the role of T species when determining fire frequency guidelines.

The importance of competition between plant species, and thus the importance of disturbance to disrupt competitive exclusion, is likely to vary with **landscape productivity**. A second non-equilibrium paradigm offshoot, the **dynamic equilibrium model** (Huston 1979, 2003, 2004), considers the interaction of productivity and disturbance in mediating species diversity. In harsh environments where productivity is low, interspecific competition is unlikely to be great. Here, a-biotic factors such as low rainfall, heavy frosts and infertile soils limit the number of plant species able to grow, and also limit their growth rates. The need for disturbance to reduce competitive superiority is therefore minimal. In fact, a high disturbance frequency is predicted to reduce diversity in these ecosystems, as organisms will be unable to grow fast enough to recover between disturbances. In highly productive, resource-rich environments, however, competition is likely to be much more intense, as many species can grow in these areas, and they grow quickly. Here, diversity is predicted to decline where disturbance frequency is low, as some species will outcompete others, excluding them from the community.

Landscape productivity, as defined by plant biomass as an example, is likely to increase with rainfall, temperature, season of rainfall – where rainfall and warm temperatures coincide, there is a greater potential for plant growth – and soil fertility (clay soils are often more fertile than sandy soils, however they also tend to support more herbaceous, and fewer shrub, species; Specht 1970, Prober 1996, Clarke 2003). Relatively frequent fire may thus be more appropriate in wet, warm, productive fire-prone systems than in those whose productivity is limited by poor soils, low rainfall or a short growing season.

This discussion brings us back to the concept of succession. South African fire ecologists Bond *et al.* (2003, 2005) divide global vegetation types into three categories:

- **Climate-limited systems**. These communities are not prone to either major structural change, nor to succeeding to another vegetation type in the absence of fire, although fire frequency may influence species composition to some extent. In South Africa these communities occur in arid environments, and also in areas nearer the coast where rainfall is moderate but occurs in winter.

- **Climate-limited but fire modified systems**. These vegetation types do not succeed to another vegetation type in the absence of fire, but their structure may alter from grassy to shrubby. The Cumberland Plain Woodland described above fits into this category.

- **Fire-limited**. These vegetation types will succeed to a different community in the absence of fire. In South Africa, these communities occur in higher rainfall areas, and include both savannas and heath.

Climate-limited but fire-modified systems can occur in at least two ‘states’, for example grassy woodland and Bursaria-dominated shrub thicket woodland on the Cumberland Plain (Watson 2005). Fire-limited vegetation types could also be said to be able to exist in different states, although the differences between them are so great that they are rarely thought about in this way. For example, in north Queensland, *Eucalyptus grandis* grassy wet sclerophyll forest is succeeding to rainforest, probably due to a reduction in fire frequency and/or intensity (Unwin 1989, Harrington and Sanderson 1994). However rainforest and grassy wet forest are not generally considered as different states of a single vegetation type, but rather as two different types of vegetation.

1.6 Patch dynamics

The examples in the last paragraph illustrate how dynamic vegetation can be in relation to fire. In some productive landscapes, variation in interfire intervals within broad thresholds, that is variation in *time*, may not be sufficient to maintain all ecosystem elements. Variation in *space* may also be needed to ensure all possible states, and the plants and animals they support, are able to persist in the landscape. Fire can mediate a landscape of different patches, whose location may change over time.

For example recent studies in north-eastern NSW indicate that some forests in high rainfall areas on moderately fertile soils can exist in more than one ‘state’. Relatively frequent fire – at intervals between 2 and 5 years – is associated with open landscapes in which a diverse flora of tussock grasses, forbs and some shrubs thrives (Stewart 1999, Tasker 2002). Nearby areas which have remained unburnt for periods over 15 or 20 years support higher densities of some shrub and non-eucalypt tree species, particularly those able to recruit between fires (Birk and Bridges 1989, York 1999, Henderson and Keith 2002). Each regime provides habitat for an equally diverse, but substantially different, array of invertebrates and small mammals (York 1999, York 2000, Andrew *et al.* 2000, Bickel and Tasker 2004, Tasker and Dickman 2004).

The concept of ‘states’ provides options for the creation and maintenance of habitat across space as well as time. It can reduce conflict between those who see the value in particular states (such as grassy or shrubby vegetation in sub-tropical wet sclerophyll forests), by pointing out the value of each and the need for both. Of course, it also raises questions as to the proportion of each state that may be desirable in the landscape, the scale of mosaics, and various other factors. These questions represent fertile ground for research and discussion in future.

1.7 Conclusion

The concepts and models described in this chapter provide a framework for viewing the findings of the numerous fire ecology studies which have been conducted over the past 50, and particularly the last 20, years. Subsequent chapters in this report summarise some of these findings, particularly those relating to eight vegetation classes found in the Southern Rivers region. These findings and the discussion that accompanies them both reinforce, and draw upon, the concepts and models presented here.

2 Fire in heathlands

"Heathlands cover only a small portion of the Australian continent, yet they are associated with some of its most inspirational coastal and mountain scenery" (Keith 2004:172). Distinguished by their lack of trees (other than short multistemmed mallees or occasional emergents), heathlands generally grow in sandy soils which are very low in nutrients (Keith 2004). While the number of people who live in or next to heathlands is limited, many people visit and love these wild and colourful places.

Heath is highly fire-prone, and fire plays a vital role in maintaining the diversity of heathland ecosystems. Sydney's sandstone-based Coastal Heaths were the setting for much of Australia's initial fire ecology research, and it is with this vegetation class that we begin (Section 2.1). The focus here is on vegetation; the aim to use the rich research literature to illustrate some core concepts and findings of fire ecology. Section 2.2 focuses on South Coast Heaths. Fire-related research in this environment has been more concerned with animals than plants. Again the aim is to illuminate basic concepts and findings.

2.1 Sydney Coastal Heaths

2.1.1 Introduction

Sydney Coastal Heaths occur in the north-east corner of the Southern Rivers CMA region, north from Jervis Bay. This vegetation class is dominated by a diverse complement of shrubs including many species from the Proteaceae, Fabaceae, Myrtaceae, Epacridaceae and Rutaceae families. Sedges and herbs are also important components of the plant community (Keith 2004). Although the infertility of the shallow sandstone soils which underlie this vegetation class limits productivity, growth is promoted by relatively abundant rainfall along the coast, of the order of 1100-1400 mm per annum (Bureau of Meteorology 2005).

Gellie (2005) identifies two vegetation groups which appear to coincide with Keith's Sydney Coastal Heaths:

- Group 140 – Northern Coastal Tall Heath. Common shrubs in this vegetation group include *Banksia ericifolia* ssp. *ericifolia*, *Allocasuarina distyla*, *Hakea teretifolia*, *Epacris microphylla*, *Darwinia leptantha* and *Leptospermum squarrosum.* It has an open cover of tall shrubs, a diverse intermediate layer of smaller shrubs, as well as sedges and herbs. This vegetation group covers an estimated 40,100 ha and according to Gellie (2005) has not been cleared in the area covered by his study. Eighty-five percent occurs in conservation reserves.
- Group 141 – Northern South Coast and Escarpment Swamp Heath. A tall shrub layer of *Leptospermum attenuatum*, *L. juniperinum, Banksia ericifolia* and *Melaleuca squarrosa* covers an intermediate shrub layer in which *Sprengelia incarnata* and *Epraris obtusifolia* are common. The ground layer is mostly made up of sedges. This vegetation group covers an estimated 8,300 ha, has not been cleared and is 83% reserved.

As noted above, Sydney Coastal Heaths have hosted many fire ecology studies, and their basic dynamics are well understood. Fire frequency thresholds for this vegetation type were proposed by Bradstock *et al.* in 1995:

> "A decline in populations of plant species can be expected when:
>
> - there are more than two consecutive fires less than 6-8 years apart (fire-sensitive shrubs decline);
> - intervals between fires exceed 30 years (herbs and shrubs with short-lived individuals and seedbanks decline);
> - three or more consecutive fires occur at intervals of 15-30 years (sub-dominant herbs and shrubs decline);
> - more than two consecutive fires occur which consume less than 8-10 tonnes ha^{-1} of surface fuel (species with heat-stimulated seedbanks in the soil decline)" (Bradstock *et al.* 1995:328).

Recent analyses using plant species characteristics (Sections 1.2, 1.4) has reinforced these thresholds (Bradstock and Kenny 2003, Kenny *et al.* 2004). As the fire regime thresholds in this vegetation class are clear and undisputed, a comprehensive review of the literature will not be undertaken here. Rather some of the key concepts and findings reflected in the Bradstock *et al.* (1995) guidelines will be explored, as they illustrate ideas introduced in the previous chapter.

2.1.2 Sensitivity to frequent fire

> "A decline in populations of plant species can be expected when there are more than two consecutive fires less than 6-8 years apart (fire-sensitive shrubs decline)" (Bradstock *et al.* 1995:328).

A number of field studies in Sydney Coastal Heaths have identified several shrub species which are eliminated or reduced in abundance on frequently burnt sites (Siddiqi *et al.* 1976, Nieuwenhuis 1987, Cary and Morrison 1995, Morrison *et al.* 1995, 1996, Bradstock *et al.* 1997). These species include the dominant obligate seeders *Banksia ericifolia*, *Allocasuarina distyla* and *Hakea teretifolia.*

What makes these species vulnerable to frequent burning, where others are less sensitive? First, they have relatively long juvenile periods – they can take six to eight years to flower (Benson 1985). As these plants are obligate seeders, a second fire before seedlings germinating post-fire have matured sufficiently to set seed will leave no seeds to establish a new generation. Canopy storage increases this vulnerability; while soil-stored seed may survive through more than one fire ungerminated, and thus ready to burst forth after a second burn, serotinous species do not have this capacity. In addition seeds of these species, although winged, do not generally travel far from the parent plant (Hammill *et al.* 1998), limiting the potential to re-establish from unburnt patches after a second fire.

Demographic studies show some Sydney sandstone resprouters are also likely to decline under repeated short interfire intervals, as fire tolerance can take many years to develop (Bradstock and Myerscough 1988, Bradstock 1990).

Bradstock and Kenny (2003) used information on juvenile periods of species in Brisbane Water National Park just north of Sydney[3] to derive a domain of 'acceptable' fire intervals. Noble and Slatyer's vital attributes model was used to classify species into functional types. The maximum estimates for juvenile period from demographic and anecdotal sources were 6.0 and 6.5 years respectively, giving a minimum threshold of 7 years.

2.1.3 Sensitivity to infrequent fire

> "A decline in populations of plant species can be expected when … intervals between fires exceed 30 years (herbs and shrubs with short-lived individuals and seedbanks decline)" (Bradstock *et al.* 1995:328).

At the other end of the spectrum, field research around Sydney has identified shrub species which are disadvantaged if fire is too *infrequent* (Fox and Fox 1986, Nieuwenhuis 1987, Morrison *et al.* 1996). For example, Morrison *et al.* (1996) found lower abundances of the shrubs *Acacia suaveolens* and *Zieria laevigata* in sites with long interfire intervals than in sites where there had been less than seven years between burns. Fox and Fox (1986) speculate that fire may be necessary to prevent senescence in a number of resprouters which they found reduced in abundance after a 12 year interfire interval. Nieuwenhuis (1987) identified several resprouting herbaceous species as well as a number of obligate seeder shrubs including *Grevillea buxifolia* and *Conospermum ericifolium* whose abundance was significantly lower in infrequently burnt sites than in paired sites which had burnt more frequently.

The majority of species in Sydney Coastal Heaths recruit after fire (Keith *et al.* 2002a and references therein), making them I species in Noble and Slatyer's terms. These species depend on fire occurring either while adults are still alive or, if the species stores seed in the soil, before that seed loses viability. Fire-cued obligate seeder I species may be at particularly risk under low fire recurrence, as these plants will form even-aged stands after a fire (Auld 1987), and may die some years later as a group.

Bradstock and Kenny (2003) used anecdotal sources and calculations based on juvenile periods to predict longevity of plant species in Brisbane Water National Park. When estimates of seedbank longevity were considered, the serotinous obligate seeders *Banksia ericifolia* and *Petrophile pulchella* were considered more at risk of decline under long interfire intervals than short-lived species with soil-stored seed such as *Acacia suaveolens*. A predicted lifespan of 28-30 years for *Banksia ericifolia* defined an upper threshold of 30 years for the acceptable domain of fire intervals.

2.1.4 Variability within thresholds

> "A decline in populations of plant species can be expected when … three or more consecutive fires occur at intervals of 15-30 years (sub-dominant herbs and shrubs decline)" (Bradstock *et al.* 1995:328).

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[3] The vegetation in this Park includes both heath and open forest with a heath-like shrubby understorey. Several studies cited in this section cover this range of vegetation.

As time goes by after a fire in Sydney Coastal Heath, a small number of large shrub species, particularly the slow-growing serotinous obligate seeders *Banksia ericifolia, Allocasuarina distyla* and *Hakea teretifolia* gradually come to dominate many patches (Keith 1995, Figure 1a). Small shrubs and herbs decrease in abundance as resources are increasingly captured by the dominants (Morrison *et al.* 1995, Figure 1b). When fire occurs after 15 to 30 years, the large amount of seed stored in the canopy of these species produces abundant post-fire seedlings, which rapidly re-establish dominance, emerging above the understorey by five or six years post-fire (Tozer and Bradstock 2002). Two studies have confirmed that many understorey species are negatively affected by these thickets.

Figure 1. a (LHS). Dense thicket of *Banksia ericifolia* and *Allocasuarina distyla* many years after fire. b (RHS). Few other species growing under thicket canopy.

Keith and Bradstock (1994) studied understorey plants in the second year after a fire in places where overstorey characteristics had varied before the fire. Pre-fire overstorey density had a very significant negative association with the species richness of understorey shrubs. In addition almost all resprouting species were significantly more abundant where the overstorey had been absent prior to the fire, while obligate seeders varied in their responses to pre-fire overstorey characteristics. The authors conclude that “a non-equilibrium state which promotes coexistence of all species” would best be achieved through “varying the frequency and spatial extent of fires according to observed population levels. For example, a fire interval of less than 8 years may be required, at least over part of an area, if overstorey is dense and adversely affecting understorey over a wide area” (Keith and Bradstock 1994:353).

A similar post-fire study by Tozer and Bradstock (2002) which like Keith and Bradstock (1994) took place in Royal National Park south of Sydney, also found many species were less abundant in patches where overstorey had previously been dense. This study added a dimension to the previous work by assessing competitive effects separately in wet and dry heath: the effect was most pronounced in dry heath.

Additionally, this study found that some species were *more* abundant in overstorey patches, apparently because of suppression of the grasstree *Xanthorrhoea resinifera* which tended to dominated open patches. “We postulate that full diversity will be maintained when the density of overstorey shrubs fluctuates widely over a relatively short period of time,” the authors conclude. “This is most likely when fire frequency is highly variable” (Tozer and Bradstock 2002:213).

Finally, direct evidence for the importance of variability in interfire intervals comes from a multi-site study by Morrison *et al.* (1995) in Ku-ring-gai Chase National Park in Sydney’s north. Here increased variability in interfire intervals was associated with an increase in the species richness of both obligate seeders and resprouters.

2.1.5 Fire intensity

> “A decline in populations of plant species can be expected when … more than two consecutive fires occur which consume less than 8-10 tonnes ha^{-1} of surface fuel (species with heat-stimulated seedbanks in the soil decline)” (Bradstock *et al.* 1995:328).

The final point in the Bradstock *et al.* (1995) recommendations addresses fire intensity. The concern here is that a proportion of fires be sufficiently intense to provide good conditions for the germination and growth of seedlings of fire-cued species.

What does the research from Sydney Coastal Heath tell us about the role of fire in seedling establishment?

We have already noted that the majority of species in this vegetation type do all, or most, of their recruitment in the months following a fire. Studies which confirm this include Auld and Tozer (1995) for *Acacia suaveolens*, *Grevillea buxifolia* and *Grevillea speciosa* and Vaughton (1998) for *Grevillea barklyana*, a rare obligate seeder which occurs mostly near Jervis Bay. This is not simply a matter of fire cues promoting germination, however. Studies have found that seedlings of serotinous species survive better in burnt areas. In fact, all seedlings of *Banksia ericifolia* and *Banksia oblongifolia* which germinated from seed planted out by Zammit and Westoby (1988) into sites burnt 3, 7 and 17 years previously died within six months of germination, whereas seeds placed into recently burnt sites had much higher rates of both germination and survival. Bradstock (1991) found seedlings of four Proteaceous species placed into unburnt sites completely failed to survive due to predation, while in burnt areas the majority of seedlings survived, possibly due to reduced densities of small mammals (see Section 2.2.4).

Fire-related germination cues which operate across a wide variety of species found in Sydney Coastal Heath include heat and smoke. In the mid 1980s Auld (1986b) showed that heat broke seed dormancy in the obligate seeder legume *Acacia suaveolens*. A larger study of 35 species from the Fabaceae and Mimosaceae families confirmed the ubiquitous nature of heat cues in these families (Auld and O’Connell 1991), although species varied somewhat in the temperatures which were associated with a maximal response. These data, in conjunction with information on soil temperatures relative to fire intensities, led to the conclusion that repeated low intensity fire should be avoided. More recent studies have demonstrated the dormancy-breaking properties of smoke. Kenny (1999) and Morris (2000) found smoke increased germination of several *Grevillea* species found on sandstone, with some species also responding positively to heat. Thomas *et al.* (2003) documented a range of responses to heat and smoke cues amongst a group of Sydney sandstone species, including interactions between cues. Relatively moderate heat shock produced maximum germination in several species, leading these authors to conclude that "low-intensity fire or patches within fire" may be important for recruitment of some plants.

At a community level, a study by Morrison (2002) found that floristic composition in Sydney's sandstone country varied with fire intensity. Species favoured by relatively high intensity fire included peas and monocots. Species from the Proteaceae and Rutaceae families were most abundant where fire had been of low to medium intensity, while low intensity burns favoured species in the Epacridaceae family. Recent work by Ooi *et al.* (2006) shows that a number of obligate seeder *Leucopogon* species appear to rely at least in part on persistence of adult plants in unburnt patches and places where fire intensity is low.

The message here appears to be that some species benefit from intense fires, while others will be more abundant where fire intensity is low. Variability again appears to be the key to allowing species with different attributes to co-exist.

2.1.6 Conclusion

Research in Sydney Coastal Heaths reveals a classic environment-limited but fire modified vegetation type (Bond *et al.* 2003, 2005, Section 1.5). While low soil fertility limits plant growth, relatively high rainfall allows competition dynamics to develop. Fire frequency profoundly affects community composition and vegetation structure: Morrison *et al.* (1995) attributed almost 60% of the floristic variation in their samples to this variable. Variation in fire frequency, and to a lesser extent in intensity and perhaps also in season (Clark 1988), facilitates the co-existence of the wide diversity of large and small plants which grow in this environment. Of course variability in factors such as topography and drainage also plays an important role in providing opportunities for plant co-existence in Sydney Coastal Heaths (Bradstock *et al.* 1997).

The findings outlined above illustrate many of the aspects of non-equilibrium systems discussed in the previous chapter. Sydney Coastal Heath *is* dynamic – it changes with time and disturbance history; both too frequent and too infrequent burning can cause species to be lost from the community; long intervals between fires allow competitive exclusion of smaller by larger species; diversity across the landscape relies on a shifting mosaic of patches in flux. Application of the vital attributes model to data on individual plant species produces fire frequency thresholds which have face validity given findings of field studies at community level. Variability in interfire intervals within these thresholds is clearly essential in allowing species to coexist. Keith (1995:205) puts it like this:

> "Heathland mosaics are interactive and dynamic systems. The properties that characterize mosaics: spatial heterogeneity; temporal dynamics; and interactive processes, are inherent in all natural systems and are the ultimate means of sustaining their biodiversity. Mosaics should therefore be considered the norm of natural systems, rather than the exception."

2.2 South Coast Heaths

2.2.1 Introduction

In the far south-east of the Southern Rivers region on exposed, flat coastal plateaux grow South Coast Heaths. Soils are low-nutrient Tertiary alluvial sands and gravels. Rainfall is lower than in the Sydney region, averaging 750 mm a year at Green Cape lighthouse (Bureau of Meteorology 2005). Vegetation on these heathland 'moors' rarely exceeds 1 m, and contains a diverse complement of shrubs and sedges. Degree of exposure to sea winds and drainage patterns create a vegetation mosaic variously dominated by shrubs such as *Westringia fruticosa*, *Correa alba* var. *alba*, *Epacris impressa*, *Allocasuarina paludosa*, *Banksia paludosa* and *Xanthorrhoea resinifera.* A variant of this vegetation type is found on nearby coastal ranges, where *Leptospermum trivervium, Epacris microphylla* and *Lepidosperma gladiatum* are prominent (Keith 2004).

Research into South Coast Heaths in NSW has concentrated on Nadgee Nature Reserve. Here, the need to study the effects of fire came into sharp focus in 1972 when an intense and extensive wildfire swept through the surrounding forests and into the heath (Fox 1978). A number of studies of birds and animals commenced at that point, studies which "have had a seminal influence on our understanding of the fire ecology of these groups" (Keith 2004).

This vegetation type is also found in Victoria, where it extends from the NSW border along the Gippsland coast as far as Wilson's Promontory. Findings from a number of Victorian studies are therefore mentioned in this section.

The discussion below covers vegetation dynamics in South Coast Heaths (Section 2.2.2), fauna recovery after wildfire (Section 2.2.3), and time-since-fire effects on fauna (Section 2.2.4).

2.2.2 Vegetation dynamics

We are fortunate to have a careful longitudinal study of vegetation change with time-since-fire in South Coast Heath. Posamentier *et al.* (1981) had fortuitously sampled heathland vegetation at Nadgee just before the 1972 fire, and continued monitoring the same plots for the next six years, with sampling at 3 and 6 months, and at 1, 2, 4 and 6 years after the fire. By three years post-fire per plot species richness was almost twice as great as it had been before the burn. Many species which were either not recorded in the pre-fire survey, or which had been rare, appeared after the fire. The spread of species across the heath was also greatly enhanced by the fire: by the end of the study 45% of species had surpassed their pre-fire distribution, while a further 50% had equalled it. By six years post-fire vegetation was thickening. Thirty-seven percent of species had begun to decline by this time, while most monocots reached the peak of their post-fire distribution in only two years. Cover of the dominant shrubs recovered more slowly. Posamentier *et al.* (1981) noted that the intense burn at Nadgee had encouraged seedlings: "…the enhanced seed germination has resulted in a large number of what may be termed fugitive species which have taken the opportunity afforded by the fire to increase their populations and store of seed in the topsoil. Such an intermittent opportunity may be important to their long term persistence in the community" (Posamentier *et al.* 1981:172-3).

Russell and Parsons (1978) used a ‘place for time’ strategy to assess vegetation change over 21 years in heath at Wilson’s Promontory. Species richness along line transects showed no decline over the first post-fire decade, but dropped from over 60 species to 49 by 21 years post-fire. Recent inter-fire intervals in the ‘younger’ sites in this study varied between five and nine years.

Both these studies suggest that South Coast Heaths, like their Sydney counterpart, are dynamic ecosystems. That they change with time-since-fire is clear. That different fire frequencies will favour different plant species and species groups seems likely, although studies addressing this issue directly are, so far, lacking.

Are there any indications that competition dynamics may be an issue where South Coast Heath remains unburnt for extended periods? This question is relevant as a second wildfire at Nadgee in 1980 has been followed by an extensive fire-free period over much of the Reserve, giving a post-fire age of around 25 years.

The large serotinous obligate seeders which figure in Sydney Coastal Heaths are largely absent from the South Coast: *Banksia ericifolia* does not occur below Jervis Bay, and while the distribution of *Allocasuarina distyla* and *Hakea teretifolia* does include the far south coast, neither species is prominent (Keith and Bedward 1999).[4] However a description of coastal heathland vegetation in Nadgee (NPWS 2006) lists as common several species which are considered to have the potential to encroach to the detriment of other species. These include Coastal tea-tree *Leptospermum laevigatum* (Molnar *et al.* 1989, Offor 1990, Bennett 1994), *Acacia longifolia* ssp. *sophorae* (McMahon *et al.* 1996, Costello *et al.* 2000), and *Melaleuca armillaris* (R. Bradstock pers. comm. 2006). While it is not always clear that reduced fire frequency is the primary reason that these species gain dominance, this is generally considered part of the story. Past grazing by cattle may also be a factor.

2.2.3 Fauna recovery after fire

Turning now to animal studies, the primary message from the post-fire work at Nadgee is that fauna species can recover even after intense and extensive fire.

There is no doubt that the immediate effects of the 1972 wildfire on birds and animals were severe. Charred bodies of numerous birds, small mammals, macropods and possums were counted, although some live lizards, snakes, wombats, kangaroos and birds were observed soon after the fire (Newsome *et al.* 1975, Recher *et al.* 1975, Fox 1978). Post-fire predation and lack of food further reduced population numbers. Keith *et al.* (2002a) speculate that the effects of heathland wildfires on fauna may be more severe at Nadgee than in heath on sandstone, due to a relative lack of post-fire refuges and nest sites which in sandstone country are provided by rock outcrops.

However “after a few years there was no evidence that any species of plant or animal present before the fire had been lost from the Reserve”[5] (Recher 2005). In a survey of heathland birds in the spring of 1979, seven years after the 1972 fire, “not only were

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[4] In addition, Fox (1978) reports vigorous resprouting of both *Casuarina distyla* and *Hakea teretifolia* at Nadgee after the 1972 fire.

[5] Nadgee Nature Reserve contains forests as well as heathland. Some studies cover this range of vegetation. I have focussed on heath-related findings in this section; other findings are reported in Section 4.2.

Ground Parrots abundant at Nadgee, but so were a full complement of other heath birds, including the Eastern Bristlebird" (Recher 2005:24). Even after a second fire in 1980, avifauna recovery was rapid.

Where did recolonisation come from? Although the picture isn't completely clear, we know that some heathland species survived in unburnt patches along creeklines (Recher *et al.* 1975, Fox 1978). Others probably moved in from large unburnt heathland patches to the north and south of the Reserve once conditions again became suitable (Recher 1981, 2005).

Recher (2005:25) concludes: "It is clear that an intense wildfire need not be a disaster for the fauna or the vegetation. Even a second fire relatively soon after the first may not lead to the loss of species, so long as there is a source of colonists nearby."

2.2.4 Time-since-fire effects on fauna

Research at Nadgee also illustrates another common finding with respect to the effects of fire on fauna. As time-since-fire progresses, resources and habitat change, and so too does the suite of fauna species.

Newsome *et al.* (1975) recorded greatly reduced numbers of **small mammals** across the reserve in the two years after the 1972 fire. While small numbers of *Antechinus swainsonii*, *Antechinus stuartii*, *Rattus lutreolus* and *Rattus fuscipes* were trapped, they tended to be found in refugial habitats such as forested gullies which were not commonly used before the fire rather than in heath. In heathland patches recovery varied with type of heath. In general, few animals were found in the immediate post-fire years, although some *Antechinus swainsonii* were trapped in a "totally burnt out swamp" where *R. lutreolus* and *R. fuscipes* had previously been abundant. Upland heath housed various small mammal species before the fire, however post-fire recovery was slow (Newsome *et al.* 1975, Catling and Newsome 1981). Small mammal abundance recovered more rapidly in heathland close to forest, than in non-ecotonal areas (Catling and Newsome 1981). By six to eight years after the 1972 fire, all four native small mammals were found in a range of heathland habitats – including open dry heath, though only where scattered mallee thickets increased habitat complexity[6] (Catling 1986).

The second fire in 1980 again reduced the abundance of native small mammal species at Nadgee pretty much to zero, although again small numbers of *Antechinus swainsonii* were found in burnt out swamp (Catling 1986). Again post-fire recolonisation of heathland was slow relative to that in forest. Catling (1986:132) points out that at Nadgee "as habitats age after fire they increase in structural complexity." As this occurs small mammal abundance and diversity also increases – and this happens most rapidly in habitats which are by nature relatively complex. Catling (1986) found that the first native small mammal to colonise heath, *Rattus lutreolus,* was strongly associated with thickets of taller vegetation within the heath: these thickets added to habitat complexity. The association between time-since-fire, habitat complexity and fauna diversity is a recurring theme in the fire ecology literature, and will be discussed again in Section 4.2.

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[6] Catling (1986) scored habitat complexity by assessing cover of trees, shrubs, ground cover, rocks, logs and litter, and the presence of water.

Nadgee, however, is in one sense an a-typical environment for studying effects of fire on small mammals, as no *Pseudomys* species are found there. Unlike *Antechinus* and native rats, these native mice tend to be heathland specialists and are also often associated with the early years of post-fire recovery (Catling and Newsome 1981, Fox 1983, Friend 1993). Catling (1986) suggests that the early success at Nadgee of *Rattus lutreolus*, which is a relatively late coloniser in other heathland environments (Fox 1982, 1983, Friend 1993), may be due to lack of competition from *Pseudomys* species.

Nadgee does, however, host a variety of heathland **birds**. Recher (2005:26) monitored bird populations in an area of heath burned after the 1980 fire, and compared results with those from an adjacent area that had burnt in 1972 but not in 1980. In the first year after fire large numbers of quail, bronzewing pigeons and finches 'invaded' the moor, "responding both to the seed released after the fire from the burnt heath plants and then to the seed produced by the abundant growth of native grasses." Forest birds were also prominent in heath during the first 12-18 months, taking advantage of the abundant food provided by insects. Within two years, however, the original heathland avifauna had re-established itself and the numbers of seed-eaters and forest birds diminished. As the height and density of the heath increased, birds such as the Australian Pipit declined and disappeared. In 1995, 15 years post-fire, Baker (1997) recorded a small number of Bristlebirds in heath to the south-east of the Reserve. By 2000, 20 years after the 1980 fire, numbers of Tawny-crowned Honeyeaters, which forage on the ground, had fallen, while the abundance of Ground Parrots and Striated Fieldwrens also appeared to have diminished. However other species, including several pairs of Bristlebirds, had established territories in the thick 20-year regeneration (Recher 2005).

Findings of time-since-fire studies on the Ground Parrot, an 'icon' heathland bird, have varied: while some studies have found this species in greater abundance in the early to middle years of post-fire regeneration than in later post-fire years (eg Meredith *et al.* 1984, McFarland 1988), others have found that parrot densities have held steady or continued to increase a decade or even two after fire (Baker and Whelan 1994, J. Baker pers. comm. 2004). Possible explanations include that studies were conducted in different kinds of heaths – heaths with a high proportion of sedges may provide better food resources for the birds (Meredith *et al.* 1984, Keith *et al.* 2002a). Growth rates may vary between heath types, varying the time to development of dense closed stands that Ground Parrots may not find ideal. Predator density may alter utilisation patterns. The length and pattern of interfire intervals 'behind' the different times-since-fire surveyed may affect vegetation composition and thus habitat suitability. Catling and Newsome (1981) present data from Nadgee that show recovery of Ground Parrots to pre-fire levels by four to five years post-fire. The heath they surveyed in the year prior to the fire had been burnt ten years previously, so this finding tells us that parrots were more abundant at 5 and 10 post-fire years than in the early post-fire years, but doesn't provide a comparison to heath of greater than 10 year post-fire age. In Croajingolong National Park just across the Victorian border from Nadgee, Meredith *et al.* (1984) failed to find Ground Parrots in 28-29 year old heaths and sedgelands, although they were abundant between 4 and 7 years post-fire in closed heaths with a diverse shrub complement, and between 5 and 17 years in heath dominated by grass-trees and sedges.

On the other hand, research into the even more at-risk Eastern Bristlebird paints a picture of a cover-dependent species whose survival depends on the existence of dense, long-unburnt heathland habitat (Baker 2000). Bristlebirds are poor flyers given to breeding failure and low fecundity, characteristics which means their chances of surviving a fire are low, as is their ability to recolonise over more than short distances

(Baker 1997). A single, extensive fire has the potential to wipe out a fair percentage of remaining individuals, as this species is only found at a few locations (Baker 1997). Fire exclusion has been recommended as a key conservation strategy for the survival of the species (Baker 2000). However the need to ensure unburnt patches containing Bristlebirds remain when a wildfire does come through implies that limiting fire extent may be as important as low fire frequency to the survival of these birds. Bristlebird populations in 1997 at Barren Grounds west of Kiama were in relatively good shape after a series of patchy fires over the preceding 17 years: fires occurred in five years during this time, 10 and 50% of the reserve was burnt in each of these years (Baker 1997). Recolonisation from unburnt into adjacent burnt areas is also the probable explanation for the findings of Pyke *et al.* (1995) at Jervis Bay. These researchers found no differences in numbers of birds in habitat burnt less than seven years previously, to numbers in areas burnt over 11 years ago. Their burnt transects were all, however, within about 300m of long-unburnt areas.

2.2.5 Conclusion

Kenny *et al.* (2004) recommend a fire frequency of between 7 and 30 years for heathlands throughout the state. While studies have not directly addressed the effects of fire frequency in South Coast Heaths, this range seems compatible with the findings of the time-since-fire studies summarised above. Studies after the 1980 fire show recovery of a wide range of fauna species following the preceding eight year interfire interval, suggesting that occasional relatively short intervals should not be harmful. Indeed the presence of "fugitive" plant species which thrive in the early post-fire years, and the existence of plant species with the potential to dominate to the detriment of other species suggests some short intervals may be important for maintaining diversity. At the other end of the fire frequency spectrum, long intervals in some parts of the landscape are clearly important for some bird species, particularly the endangered Eastern Bristlebird.

Variability in time-since-fire is clearly important in creating the habitat diversity needed to support the full range of plants, animals and birds in South Coast Heath. A range of age-classes may also help limit the risk of extensive wildfire, which could be detrimental to Bristlebirds. As noted above, the strength of the time-since-fire dynamics for both plants and animals suggests variability in fire frequency is also likely to be important. Varying fire frequency in South Coast Heath may be the best way to ensure plants and birds whose abundance is greatest in the early post-fire years have opportunities for growth and recruitment, while also safeguarding areas of dense vegetation for cover-dependent species such as the Eastern Bristlebird.

As Recher (1981:39) points out, "heaths will burn whether we want them to or not." Whether there may sometimes be a role for planned fire can be debated. The most likely time is when large areas are approaching the upper reaches of the recommended fire interval domain. The primary aim would be to provide habitat for early-successional plants and animals. Research into the effects of interfire intervals of various lengths, and particularly those of long interfire intervals on early successional species, would help clarify this issue.

3 Fire in grasslands and grassy woodlands

Grassy woodlands and grasslands were once widespread in Eastern Australia. Growing on fertile soils in areas of moderate rainfall, grassy vegetation quickly attracted European settlers seeking pastures for sheep and cattle. Many areas were cleared for crops and towns. Pastures grazed by domestic animals were often 'improved' through addition of exotic species. Today, high quality native grasslands and grassy woodlands exist mostly as isolated remnants, and even these have often been modified (Sivertsen 1993, Prober and Thiele 1995, Sivertsen and Clarke 2000, Yates and Hobbs 2000). Because of their association with agriculture, remaining native grasslands and grassy woodlands are often found on private property.

Research into the role of fire in Australia's temperate grassy vegetation has focussed on the *Themeda*-dominated grasslands of Victoria's lowland plains, with substantial gaps elsewhere (Hobbs 2002, Lunt and Morgan 2002). This Victorian work informs the discussion of Temperate Montane Grasslands in Section 3.1. More recently investigations into the effects of fire in Coastal Valley Grassy Woodlands, which are the subject of Section 3.2, has begun.

3.1 Temperate Montane Grasslands

3.1.1 Introduction

Temperate Montane Grasslands once covered an extensive area on the Monaro Tableland from Cooma south to Bombala, with outlying patches elsewhere, for example around Bungendore and Braidwood (Cambage 1909, Benson 1994, Keith 2004). These grasslands, which are related to those found on the basalt plains west of Melbourne (Keith 2004), have been extensively used for grazing. Modification through the use of fertilizer and sowing of exotic herbs and grasses is widespread (Benson 1994, Marshall and Miles 2005). Remaining native species-dominated remnants are therefore a valuable conservation resource (Eddy *et al.* 1998).

Temperate Montane Grasslands occur between 600 and 1500 m above sea level, and receive between 500 and 750 mm of rainfall each year. Composition varies with altitude, topography and soil type. Clay soils are generally dominated by the tussock grasses *Themeda australis* (Kangaroo Grass) and *Poa sieberiana* (Snowgrass), drainage lines are often filled with dense *Poa labillardieri* (Tussock), while the sandy clay loams formed from granite tend to be dominated by *Austrostipa* (Speargrass), *Austrodanthonia* (Wallaby Grass) or *Bothriochloa* (Red Grass) (Keith 2004). This latter community, or parts of it, may be a degraded form of the former, having lost its *Themeda* to grazing over time (Benson 1994, Marshall and Miles 2005, Wong *et al.* 2006). Amongst the tussock grasses which dominate Temperate Montane Grasslands grow forbs, including daisies, lilies, peas and orchids (Keith 2004). Unfortunately, exotic grasses and forbs are also common (Benson 1994, Dorrough *et al.* 2004).

Gellie (2005) identifies a vegetation class called Southern Tablelands Temperate Grasslands. Vegetation groups within this class include:

- Group 152 – Tableland Herb/Grassland. This ecosystem is dominated by "grasses such as *Themeda australis* and *Poa sieberiana* var. *sieberiana.*" A moderately diverse cover of herbs may include *Chrysocephalum apiculatum*, *Asperula conferta*, *Hypericum gramineum* and *Convulvulus erubescens*. Estimated pre-European extent was 38,400 ha, 86% of which has been cleared. Thirty-one percent of the remaining 5,200 ha is found outside conservation reserves.

- Group 153 – Tablelands and Slopes Herb/Grassland/Woodland. This vegetation type is similar to Group 152, although a low overstorey of *Eucalyptus pauciflora* and/or *E. rubida* is sometimes present. *Themeda australis* is the primary grass in the tussock matrix; *Dichelachne micrantha*, *Microlaena stipoides* and *Elymus scaber* are also quite common. Herbs include those listed for Group 152 above, as well as *Microtis unifolia*, *Leptorhynchos squamatus* and *Tricoryne elatior*. Pre-1750 extent is estimated at 206,000 ha, of which 3,900 ha, or 2%, is believed to remain. None of this, according to Gellie (2005), is in dedicated reserves.
- Group 157 – ACT/Monaro Dry Grassland. Grass dominants in this vegetation group include *Bothriochloa macra*, *Poa sieberiana*, *Themeda australis*, *Danthonia caespitosa* and *Austrostipa bigeniculata*. Common herbs include *Chrysocephalum apiculatum*, *Convulvulus erubescens*, *Acaena ovina*, *Goodenia pinnatifida*, *Wahlenbergia communis* and *Desmodium varians*. This vegetation group covered an estimated 159,900 ha prior to European settlement, of which only 2,800 ha (2%) remains today, 85% outside conservation reserves.
- Group 158 – Monaro Dry Grassland. This vegetation type is dominated by *Austrostipa* and *Austrodanthonia* species, along with *Enneapogon nigricans*. Herbs include *Wahlenbergia communis*, *Vittadinia muelleri*, *Rumex brownii*, *Convulvulus erubescens* and *Desmodium varians*. The estimated 485 ha which remain represent less than 1% of the pre-1750 distribution of this vegetation group. It is not found in conservation reserves.

While little research has addressed the effects of fire in Temperate Montane Grasslands, fire-related vegetation dynamics in Victoria's lowland grasslands have been extensively studied – although fauna issues have as yet received little attention. Many species in this ecosystem are the same as, or similar to, those found in Temperate Montane Grasslands. Victoria's grasslands are dominated by *Themeda australis*[7], which is also an important component of Temperate Montane Grasslands (Eddy *et al.* 1998, Keith 2004, Dorrough *et al.* 2004, Gellie 2005). Between the tussocks formed by this species grow forbs and subdominant grasses (Tremont and McIntyre 1994, Kirkpatrick *et al.* 1995).

Our discussion of fire in Temperate Montane Grasslands begins with an account of fire-mediated interactions between matrix grasses and smaller species (Section 3.1.2). Research into the relationship between fire and the matrix species *Themeda australis* follows (Section 3.1.3). Because herbaceous exotics are a major issue in remnants grasslands, the effects of fire and *Themeda* on weeds is discussed in this section. The relationship between fire and shrubs in grassy vegetation – grasslands can contain some shrub species (Eddy *et al.* 1998) – will be addressed in the section on Coastal Valley Grassy Woodlands later in this document (Section 3.2.2).

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[7] I have used the designation *Themeda australis* throughout this document, as this name is used throughout NSW (Harden 1993). Others, including the Victorian researchers whose work is cited here, prefer *Themeda triandra*, a name which recognises the close affiliation between the African and Australian forms.

3.1.2 Grass/forb dynamics

The need to consider the role fire in conservation of temperate grasslands first became salient through a study by Stuwe and Parsons in 1977. A comparison of three management regimes found that the patchy annual burning undertaken on railway reserves was associated with a higher richness of native plants than was grazing or fire exclusion. All sites were dominated by *Themeda australis* (this was a selection criterion), leading Stuwe and Parsons (1977:473) to hypothesize that the differences in species richness might be because regular firing of the rail sites prevented *Themeda* from "attaining maximum size and vigour," depositing a deep litter layer and thus outcompeting smaller, less competitive herbaceous species.

More recent work has confirmed aspects of Stuwe and Parson's theory. *Themeda australis* does indeed grow rapidly after fire (Morgan 1996, Lunt 1997b), so that by three years post-fire, gaps between *Themeda* tussocks in Victoria's lowland grasslands have mostly disappeared (Morgan 1998a). A study by Lunt and Morgan (1999a) confirmed that species richness is significantly reduced in patches where *Themeda* is dense. Studies have found that forb seedlings need gaps to survive and grow (Hitchmough *et al.* 1996, Morgan 1997, 1998a), and that short interfire intervals are important for maintaining populations of adult interstitial species (Coates *et al.* 2005).

However attempts to encourage native species through burning have been less successful. Lunt and Morgan (1999a) found that although intermittent burning in a previously-grazed grassland reserve was associated with a slight increase in species richness, most colonisers were 'weedy' species – native and exotic – with wind-blown seeds. Morgan (1998b) found that fire frequency was not reflected in differences in species richness or vegetation composition in five grasslands with different burning histories over the last 10 years. Results were more promising in a grassland reserve managed with six burns over 17 years (Lunt and Morgan 1999b). Here, comparison with an unburnt control plot found that "The frequently burnt zone was dominated by native species (72% cover) with relatively little cover of exotics (7%), whereas the rarely burnt zone was dominated by exotic species (49% cover) with just 40% cover of native species" (Lunt and Morgan 1999b:85).

The differences found in this last study appear to relate more to the effects of fire on *Themeda* (which are discussed below) than to encouragement of seedling establishment in native forbs. Seedling establishment appears to be a relatively rare event in productive temperate grasslands. Morgan (1998d), who counted seedlings emerging eight months after a fire in a regularly-burnt grassland remnant, found that only six native species had seedlings. A second study of recruitment patterns in four remnants (Morgan 2001) found few native seedlings over the four-year study period. However what native seedling recruitment there was almost all occurred in sites which had been burnt, with virtually none in the absence of fire.

Other studies point to an important characteristic of the great majority of native species in these grasslands: they tend to rely on persistence of existing individuals, rather than on recruitment of new individuals. Although there are some annual and biannual species, most are perennials, and *all* perennial species resprout after fire (Lunt 1990, Morgan 1996). Unlike many heathland species, grassland perennial forbs tend *not* to have a large permanent store of seed in the soil (Morgan 1995, 1998b). Many species germinate easily and rapidly, and are not inhibited by darkness (Willis and Groves

1991, Lunt 1995, Morgan 1998c), characteristics which imply that seedbanks will be rapidly depleted by germination. On the other hand, grassland species almost all flower within the first year after a fire (Lunt 1990, Morgan 1996, 1999), and flowering effort for many forb species is concentrated in the first post-fire year, dropping considerably in year two (Lunt 1994). These species are therefore using the third strategy discussed in Section 1.2.2 for ensuring seed is available after a fire: creating seeds rapidly. Post-fire rainfall is also almost certainly an important determinant of recruitment success for grassland species (Morgan 1998c, 2001).

3.1.3 *Themeda,* fire and weeds

Research has shown that in many situations, fire plays a vital role in maintaining the vigour of the grassland matrix species *Themeda australis*. *Themeda* dominated large areas of temperate Australia prior to European settlement (Prober and Thiele 1993, Nadolny *et al.* 2003, Prober and Thiele 2004), and in some places, continues to do so today (Section 3.1.1). Research has also found that a healthy *Themeda* sward can, in turn, limit or reduce weed invasion. Studies from grasslands and grassy woodlands are included in this section.

Maintaining *Themeda* vigour

Both time-since-fire and fire frequency studies have documented a positive relationship between fire and *Themeda*.

A decline in *Themeda* abundance and vigour with increasing time-since-fire has been noted by several researchers. Morgan and Lunt (1999) studied *Themeda* at various post-fire ages in a grassland remnant near Melbourne. Numbers of tussocks, numbers of tillers per tussock, and numbers of inflorescences all declined with time since fire. Significant declines were first observed at five years post-fire. By 11 years without disturbance, almost all vegetative matter in tussocks was dead, and tussock numbers per unit area were half those in recently-burnt areas. Long-unburnt tussocks were significantly slower to recover when a fire did finally occur, and had fewer tillers. Also in Victoria, Robertson (1985) found a decrease in the abundance of *Themeda* in unburnt woodland areas, while *Microlaena stipoides* increased. S. Clarke (2003), working in grassy woodland near Sydney, found cover-abundance of *Themeda* was higher in recently-burnt than in unburnt sites. Similar responses have also been reported from South Africa, where some forms of *Themeda triandra* (synonym *T. australis*) “become moribund in the absence of fire” (Bond 1997:434).

Fire frequency studies have linked abundant *Themeda australis* to regular burning. In a grassland reserve near Melbourne, *Themeda* density was three times as great in areas burnt six times in 17 years, that in a control area which had had 17 years between fires (Lunt and Morgan 1999b). In grassy woodland near Sydney, *Themeda* dominated the ground layer in sites burnt at least once a decade, but not in sites with over 20 years between the last two fires (Watson 2005). Long-term experiments in South Africa, where *Themeda* is a common savanna grass, have also recorded considerably more *Themeda* in frequently than in infrequently or long-unburnt areas (Uys *et al.* 2004, Fynn *et al.* 2005).

Why does fire maintain *Themeda* vigour? Periodic defoliation appears to prevent the self-shading which suppresses tiller production (Bond 1997, Morgan and Lunt 1999). Fire removes the thick mulch of dead material which prevents seedling regeneration (Morgan and Lunt 1999). Smoke may play a role in cuing seed germination in *Themeda* (Baxter *et al.* 1994, Wood 2001), although not all studies have found this to be the case (Clarke *et al.* 2000). Summer-growing C_4 grasses such as *Themeda* use water more efficiently and have lower nutrient requirements than all-season and winter-growing C_3 grasses like *Microlaena stipoides* and *Austrodanthonia* species[8] (Ojima *et al.* 1994, Nadolny *et al.* 2003), and these characteristics may give C_4 species a competitive advantage in a frequently-burnt environment (Fynn *et al.* 2003).

The vulnerability of *Themeda* to lack of burning may vary with environmental or genetic factors. Bond (1997) notes that some forms of *Themeda triandra* in South Africa appear to be immune to self-shading, as tillers are borne on aerial shoots. Vigorous flowering of *Themeda australis* was observed in late 2005 along roadsides burnt in the February 2003 fires south of Canberra, presumably in areas where fire had not been frequent.

An additional factor in the *Themeda* story concerns its response to grazing. Researchers have noted a decline in *Themeda* with moderate to heavy grazing (Moore 1953, Prober and Thiele 1995, Chilcott *et al.* 1997, Fensham 1998, McIntyre *et al.* 2003); locally in basalt grasslands on the Monaro Dorrough *et al.* (2004) found a substantial decline in *Themeda* with increased grazing frequency. Where grazing has reduced but not eliminating *Themeda* from the grass sward, fire may play a useful role in its retention and regeneration (Prober and Thiele 2005).

Are other matrix grasses likely to respond in a similar fashion to fire? The ability of C_4 species to use nitrogen and water efficiently suggests these species are more likely than C_3 grasses to have a positive response to fire. Some C_3 grasses, particularly *Poa sieberiana*, may also be encouraged, or at least not discouraged, by regular burning. Tasker (2002) found that *Poa sieberiana* was considerably more abundant in frequently burnt sites in the New England Tablelands than in sites which had not had a fire for many years. In Snow Gum woodland in Namadgi National Park Kelly (2004) was unable to detect a trend in the abundance of *Poa sieberiana* in plots subject to between two and 11 fires in a 41-year period.

Reducing weeds

Themeda australis is of particular interest because it may be one of a small number of native grass species able to compete successfully against exotics (Cole and Lunt 2005). Morgan (1998d), and Lunt and Morgan (1999b) reported a significant negative correlation between *Themeda* abundance and species richness of exotic herbs in

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[8] Photosynthesis in grass species involves one of two metabolic pathways. In C_3 species photosynthesis takes place in cells just below openings in the leaf surface called stomates, and produces as a first step a product containing three carbon atoms. In C_4 species carbon is fixed in cells arranged in bundles (Kranz anatomy), and produces as a first step a product containing four carbon atoms. The C_4 pathway uses CO_2 more efficiently and is favoured where light levels are high. Because C_4 plants have a lesser need for open stomates, they lose less water than C_3 species and so may do better in arid environments. C_3 grasses often grow better in winter and are sometimes called 'cool season' grasses, whereas C_4 grasses tend to grow in summer and are sometimes called 'warm season' grasses.

Victorian grasslands, while Watson (2005) reported similar findings in grassy woodland near Sydney.

Recent work in White Box woodlands suggests that *Themeda* may play a key role in ecosystem function, regulating nitrogen to the advantage of native perennials over exotic annuals (Prober *et al.* 2002b, Prober *et al.* 2005). Prober *et al.* (2002b) found degraded sites in Western Slopes rangelands had higher soil nutrient levels, in general, than undegraded grassy woodlands. Exotic plant cover was highest in more fertile soils, particularly where nitrate was high, while *Themeda* and *Poa sieberiana*-dominated reference sites had the lowest levels of nitrate. These authors suggest that one way to restore a healthy native understorey in degraded pastures is to break the nutrient feedback loop set up by annual exotics (which die, releasing nutrients) by regenerating perennial tussock grasses such as *Themeda*. A recent test of this proposition (Prober *et al.* 2005) found reductions in levels of both nitrate and exotics in experimental plots seeded with *Themeda* (Prober and Thiele 2005). These reductions were most impressive when *Themeda* seeding was combined with burning (two fires one year apart). Prober *et al.* (2005:1084) conclude:

> "Our results indicate that *Themeda* may be a keystone species, able to drive and maintain the soil understorey system in a low-nitrate condition that, if appropriately managed, remains resistant to weed invasion."

Although many exotics may be deterred either directly or indirectly through regular burning, other weed species may be well equipped to take advantage of the 'stable invasion window' provided by frequent fire (Morgan 1998d, Setterfield *et al.* 2005). Concerns that this might apply to perennial grass species with high invasive potential such as *Eragrostis curvula* (African Lovegrass), *Nassella neesiana* (Chilean Needle Grass) and Serrated Tussock (*Nassella trichotoma*), have been expressed (Stuwe 1994, Nadolny *et al.* 2003, Marshall and Miles 2005).

Where perennial exotic grasses occur amongst or alongside native tussock grasses, fire may have both benefits and risks. Enhancing the vigour of native tussock grasses may help keep exotic perennials in check through competition. On the other hand, some exotic perennials flower rapidly and profusely after fire (pers. obs.), and may have more extensive and permanent seedbanks than native species (Odgers 1999).

Research on the capacity of fire and/or *Themeda* to influence the rate of spread of invasive exotic grasses is limited, but what there is gives cause for cautious optimism. One study, by Lunt and Morgan (2000) found that dense stands of *Themeda australis* significantly slowed, but did not eliminate, invasion by Chilean Needle Grass (CNG) in a Victorian grassland. Hocking (nd), who followed up various initiatives to address infestations of CNG in Victorian grasslands, found that late Spring burning reduce the area occupied by CNG tussocks, and curtailed seed production and recruitment. Oversewing of areas where CNG had been sprayed out with *Themeda* met with varying degrees of success in terms of tussock establishment, however where establishment did occur *Themeda* was able to hold its own against, though not eliminate, CNG over a five-year period.

Integrated weed management, combining fire with other strategies, may provide answers in these situations. The post-fire environment presents opportunities to target weed species while in an active growth phase, before they flower, and while they can easily be disentangled from natives. Hocking (nd) recommends a combination of periodic spot spraying, late Spring burning, and *Themeda* establishment for managing

Chilean Needle Grass. In grassland remnants in north-east Victoria, managers follow ecological burning in early summer with herbicide applications to post-fire regrowth of perennial exotic grasses, particularly *Paspalum dilatatum* and *Phalaris aquatica* (Johnson 1999). The extent to which various combinations of fire and other strategies can assist in deterring aggressively invasive weeds in temperate grassy vegetation would be an excellent subject for adaptive management (Bradstock *et al.* 1995, Lunt and Morgan 1999a, Gill *et al.* 2002).

3.1.4 Conclusion

The findings outlined above have led to the conclusion that:

- Temperate grassland species are well able to cope with frequent fire.
- Fire or some other disturbance which removes biomass of the dominant grass species is essential if forbs and less competitive grasses are to persist (Lunt and Morgan 2002). Whether fire and grazing are interchangeable is discussed in Section 3.2.6.
- Frequent fire is important for plant recruitment. Fire ensures that two of the three conditions for seedling establishment – gaps in the grass canopy and seed availability – are fulfilled. The third requirement, adequate moisture, may not be met after every fire, but it is argued that with relatively frequent fire, seeds, gaps and rainfall will coincide often enough to maintain forb populations (Morgan 1998a).
- Relatively frequent fire is needed to ensure the *Themeda australis* matrix which characterises many native temperate grasslands remains healthy (Section 3.1.3).

In NSW, the statewide thresholds for interfire intervals in grasslands are currently 2 and 10 years. The upper threshold is based more upon recognition of the need for periodic biomass removal discussed above than on vital attributes data, which was insufficiently detailed (Kenny *et al.* 2004). Intervals as low as 1-3 years have been recommended for Victoria's productive basalt grasslands (Morgan 1998a, Coates *et al.* 2006). Growth rates in the Temperate Montane Grasslands under consideration here are likely to be slower than those in the Victorian lowlands. This means that gaps between *Themeda* clumps may close more slowly, leaving more opportunities for forb growth and recruitment, and also that *Themeda* collapse may be delayed. Local ecologist John Briggs, who has experimented with various interfire intervals in grasslands and grassy woodlands around Canberra, suggests 5-8 year intervals may be appropriate in *Themeda*-dominated vegetation on the Southern Tablelands. However detailed work on the endangered daisy *Rutidosis leptorrhynchoides* (Button Wrinklewort), which is found in some sites in the Goulburn area (Eddy *et al.* 1998) has led researcher John Morgan (1997:574-5) to conclude that remnant populations in *Themeda* grasslands "will need to be burnt at a maximum of 3-4-year intervals to ensure that large canopy gaps are regularly created to favour seedling recruitment and to minimize deep shading that will disadvantage the standing population."

On poorer soils where *Austrodanthonia* and *Austrostipa* species dominate, the need for biomass removal is likely to be less than in *Themeda*-dominated grasslands on more productive clay soils, suggesting a lower fire frequency may be suitable (Stuwe 1994, Johnson 1999, Kenny *et al.* 2004). These hypothesized differences in grass-herb dynamics on different substrates, and subsequent differences in recommended disturbance regimes, reflect the ideas presented in the discussion of competition and productivity in Section 1.5. However as the role of fire in Temperate Montane Grasslands on different substrates is yet to be studied, these differences remain speculative.

3.2 Coastal Valley Grassy Woodlands

3.2.1 Introduction

This vegetation type covers "a suite of highly diverse plant assemblages isolated in different dry coastal valleys that occupy rainshadows among the surrounding hills" (Keith 2004:86). These valleys are low-lying, receive 700-1000 mm of rainfall annually, and have moderately fertile soils. In southern NSW Coastal Valley Grassy Woodlands occur on the plains surrounding Lake Illawarra and in the valleys of the Araluen, Moruya and Bega rivers. These woodlands have a diverse ground layer of grasses and herbs, and "scattered or clumped shrubs, which are mostly sclerophyllous" (Keith 2004:86). *Eucalyptus tereticornis* (Forest Red Gum) is found throughout the range of this vegetation class; other overstorey species include *E. moluccana* (Grey Box), *E. crebra* (Narrow-leaved Ironbark), *E. eugenioides* (Narrow-leafed Stringybark) and *Angophora floribunda* (Rough-barked Apple). These woodlands were settled rapidly and are now fragmented by land-clearing (Keith 2004).

Gellie (2005), who uses the JANIS criterion of "an overstorey canopy cover greater than 20%" (JANIS 1997:22) to define 'forest ecosystems', identifies a vegetation class which he names South Coast Forest Red Gum Dry Grass Forests. Vegetation groups within this class include:

- Group 54 - Far South Coast Red Gum Grass/Herb Dry Forest/Woodland. This vegetation group occurs "from the Bega Valley all the way up the coast onto the Cumberland Plain" (Gellie 2005: Appendix 5, p58). *E. tereticornis*, *Angophora floribunda* and *E. globoidea* overlay a grassy understorey dominated by *Themeda australis* and *Microlaena stipoides*. Forbs include *Dichondra repens, Glycine clandestina*, *Desmodium varians* and the fern *Cheilanthes sieberi*. *Acacia mearnsii*, *Bursaria spinosa* and *Dodonaea viscosa* are common shrub species. 3,900 ha (17%) of this vegetation group remains uncleared, of which 56% is found outside conservation reserves.
- Group 52 – Bega Valley Shrub/Grass Forest. Gellie (2005) notes but does not map the extent of this vegetation group, as its occurrence falls outside his study area. The tree species which dominate Group 54 are also the dominants in this group, and while *Acacia mearnsii*, *Bursaria spinosa* and *Dodonaea viscosa* are still important shrub species, a number of other shrubs, including *Kunzea ericoides*, *Trema aspera* and *Leptospermum* spp. are also common. *Themeda australis* and *Danthonia longifolia* dominate the grassy ground layer.

South Coast ecologist Jackie Miles (2000, 2005) citing work by Keith and Bedward (1999), discusses two dry grassy forest vegetation types in the Bega Valley Shire which appear to fall into the category under discussion. Both occur in agricultural areas on private land, both have been heavily cleared, both have recently been listed as endangered ecological communities, and neither is well-reserved. These communities are:

- Bega Dry Grass Forest. Found on undulating country around the edges of the Bega Valley, tree dominants in this vegetation type are *Eucalyptus tereticornis* and *Angophora floribunda*. *Themeda australis* and *Microlaena stipoides* dominate a ground layer with a diverse herbaceous complement. *Acacia mearnsii*, *Bursaria lasiophylla* and *Ozothamnus diosmifolius* are common shrub species.
- Candelo Dry Grass Forest occurs in the dry, central part of the Bega Valley, as well as in the upper Towamba Valley. Many species found in Bega Dry Grass Forest are also found here. Yellow Box, *Eucalyptus melliodora* is an additional tree species found particularly in this vegetation type. Daisies such as *Calotis lappuleacea*, *Chrysocephalum apiculatum* and *Brachycome ciliata* are found in the grassy understorey, which is usually dominated by *Themeda australis*. *Bursaria lasiophylla* forms a dense shrub layer in some areas, while exotic shrubs including African Boxthorn (*Lycium ferocissimum*) and Hawthorn (*Crataegus monogyna*) frequently invade.

Fire ecology research in Coastal Valley Grassy Woodlands has been confined to the last 15 years, to the effects of fire on plants, and mostly to the variant of this vegetation class found on Western Sydney's Cumberland Plain, which lies just to the north of the Southern Rivers CMA region. This vegetation type, Cumberland Plain Woodland (CPW), grows in areas of moderate rainfall and low relief topography on clay soils. Dominated by *Eucalyptus tereticornis* and the Grey Box *E. moluccana*, the CPW understorey contains a diverse range of herbs, including most of those listed above, within a grassy matrix which is often dominated by *Themeda*. The shrub layer includes wattles and peas but is dominated by *Bursaria spinosa* (Bursaria) which can form dense thickets (Benson 1992, James *et al.* 1999, Benson and Howell 2002). CPW thus resembles the more southerly Coastal Valley Grassy Woodland variants in both structure and species composition.

The discussion below first addresses what is known about the fire-related dynamics of three structural components of Coastal Valley Grassy Woodlands: shrubs (Section 3.2.2), trees (Section 3.2.3) and the ground layer (Section 3.2.4). The relationship between fire and exotic weeds is then discussed (Section 3.2.5). Section 3.2.6 asks whether grazing and fire are interchangeable disturbances in grassy vegetation, from the point of view of biodiversity conservation.

3.2.2 Shrubs

The importance of fire in the life-cycle of many Coastal Valley Grassy Woodland shrubs is apparent from an experimental study of fire and grazing in CPW. Hill and French (2004) found both species richness and abundance of shrubs was significantly greater in plots burnt 18 months earlier, whether by planned or unplanned fire, than in unburnt plots.

The influence of fire cycles on shrubs in CPW was first studied by Thomas (1994) in a single site. This study found few significant differences between frequently and infrequently burnt areas, although trends for two shrub species, *Bursaria spinosa* and the native legume *Pultenaea microphylla* were towards greater density where burning had been less frequent.

More recently, a survey in nine CPW remnants with differing fire histories found a high abundance of *Bursaria spinosa* in sites where fire frequency was low (these sites had been unburnt for at least 20 years prior to a recent fire), to the point where this species dominated much of the landscape. Sites in two other fire frequency categories (high, most intervals between 1 and 3 years; and moderate, most intervals between 4 and 10 years) had some Bursaria thickets, but much of the landscape was open and grassy (Watson 2005). This finding accords with those from productive grassy ecosystems around the world, where increases in the density of woody plants in the absence of fire have been observed (Lunt 1998a,b, Roques *et al.* 2001, Allen *et al.* 2002, Bond *et al.* 2005). It also suggests that fire may be a factor in the differences in the density of *Bursaria lasiophylla* observed in grassy woodlands around Bega (Miles 2000, 2005).

In the multi-site study mentioned above, patterns for native shrubs other than Bursaria were somewhat different: these shrubs were more abundant in sites burnt once or twice a decade than in either low, or high, fire frequency sites. Obligate seeder shrubs were particularly influenced by fire cycles: the abundance of these species was lowest in sites whose last interfire interval had been over 20 years, and highest where fire frequency was moderate (Watson 2005).

As discussed in Section 2.1.2, the relatively low abundance of obligate seeders in very frequently burnt sites is easily explained: if a second fire occurs before these species have grown sufficiently to set seed, then only ungerminated seed from before the first fire will be available to keep them in the community. Although obligate seeder shrubs on the Cumberland Plain mature rapidly, flowering by three or four years post-fire (Watson 2005), they are still vulnerable to very short intervals.

The low abundance of obligate seeder shrubs in low fire frequency sites probably owes something to competition from Bursaria, which resprouts vigorously after a fire and thus is in a good position to capture resources in the post-fire environment (Section 1.5). Obligate seeder shrubs on the Cumberland Plain may also decline in long unburnt areas because they are not particularly long-lived. Once a generation of fire-cued obligate seeders dies, the population exists only as soil-stored seed. Seeds will germinate after the next fire, but may decay if the interval between fires becomes too long. A moderate fire frequency allows these shrubs to mature and build up a soil-stored seedbank of fresh, viable seed. In these circumstances fire represents an opportunity to increase population numbers through prolific germination.

The story of CPW shrubs illustrates how differences between species in *when* they establish in relation to fire affects their response to fire cycles. Bursaria has an ‘edge’ in long unburnt sites because it can establish between fires: in Noble and Slatyer’s terms it is a T species (Section 1.5). Most other CPW shrubs, on the other hand, are I species: their seeds have dormancy mechanisms which ensure they mostly germinate after fire; many of them are legumes. Long-lived T species are the exception in fire-prone environments (Section 2.1.3), but can do very well in the absence of fire, as gradual recruitment, even at low levels, can lead over time to a considerable build-up of numbers. The results for I species under long interfire intervals may, however, be less felicitous. A series of long intervals, or fire exclusion, may tip the ecosystem away from I species and towards T species. It is interesting to note that a number of the shrubs in Gellie’s description of Bega Valley Shrub/Grass Forest (Gellie 2005) belong to a species or genus which has demonstrated the potential to encroach in grassy vegetation and/or displace other native species (*Bursaria spinosa* – Watson 2005; *Dodonaea viscosa* - Hodgkinson 1979, Noble 1997; *Kunzea ericoides* - Kirschbaum and Williams 1991, Singer and Burgman 1999; *Leptospermum* – Molnar *et al.* 1989, Offor 1990, Bennett 1994; *Melaleuca* – Crowley and Garnett 1998). Both absence of fire and creation of gaps through domestic grazing may be implicated in these situations.

3.2.3 Trees

The dominant tree species of Coastal Valley Grassy Woodlands are epicormic resprouters. Little is known about the role of fire in their recruitment, although Hill and French (2004) found greater numbers of eucalypt seedlings after a summer wildfire in CPW than in unburnt areas.

In the multi-site study reported above, fire frequency did not significantly affect either adult tree density, adult tree basal area, or the density of suppressed seedlings or saplings (Watson 2005). Trends suggested that frequent fire may have been associated with an increased density of juveniles, but also with a decrease in the number of saplings ‘getting away’ into the canopy. Summer wildfire may also have limited sapling numbers. In CPW, tree density in today’s remnants, which averaged 279 trees over 10 cm in diameter per hectare in Watson’s study, is almost certainly greater than tree density prior to European settlement (Benson 1992, Watson 2005). Tree populations are probably in a regrowth phase after logging, and density may be decreasing as post-logging regrowth thins. Support for this contention comes from the finding that there was a high degree of variability between remnants in tree density, but a much smaller range in average basal area (Watson 2005), so that remnants tended to have either a large number of small trees, or a smaller number of large trees. If tree density is decreasing as woodlands recover from logging, there is no need at this point to encourage recruitment of additional trees. In fact, death of some trees after fire may be part of the post-logging recovery process, as it may free up resources that allow other trees to grow larger.

Cameron (2006) reports interesting observations of Port Jackson Fig (*Ficus rubinginosa*) growing in Bega Dry Grass Forest. Three large spreading specimens recovered after an intense wildfire near Bermagui, partly by avoiding total leaf scorch “through the blanket effect of the non-combustible foliage” (Cameron 2006:18) and partly through vigorous resprouting. Cameron (2006:19) suggests that managers using fire as part of their management strategy do not need to worry about “apparently fire-sensitive figs.”

3.2.4 Ground layer

What little is known about the effects of fire on ground layer grasses and forbs strongly suggests there is common ground between Coastal Valley Grassy Woodlands and temperate grasslands. This is not too surprising, as many herbaceous species are found in both environments.

Woodland herbs, like their grassland counterparts, often do not form persistent soil seedbanks. In CPW Hill and French (2003) found over 100 species represented in the above-ground vegetation did not emerge from soil samples. Odgers (1999), in grassy woodland in south-east Queensland, found most native grasses had transient seedbanks. A second characteristic which grassy woodland herbs share with those in grasslands is a restricted response to fire-related germination cues. Several researchers have explored this question in CPW (Wood 2001, Clarke and French 2005, Hill and French 2003); all found that although some herbaceous species responded to heat and/or smoke, many did not. Similar findings have been reported for the grassy woodlands of the New England Tablelands (Clarke *et al.* 2000, Grant and Macgregor 2001). Dormancy in many grassy woodland herbs is either non-existent or short-lived (Clarke *et al.* 2000). Finally, ground layer species in CPW flower more profusely soon after fire than in later post-fire years, as they do in Victoria's temperate grasslands (Lunt 1994). Watson (2005) found that native forbs produced almost seven times more flowers and fruits in CPW burnt four to six months prior to sampling, than in nearby areas which had not had a fire for at least five years.

Studies showing a similar response to fire by *Themeda australis* (Kangaroo Grass) in grassy woodlands and grasslands have already been cited (Section 3.1.3). Studies in Coastal Valley Grassy Woodland, which again focus on CPW, are in line with trends elsewhere. S. Clarke (2003) found cover-abundance of *Themeda* was higher in recently-burn than in unburnt sites. Watson (2005) found *Themeda* dominated high and moderate fire frequency sites, but not sites where fire frequency was low.

The most profound effects of fire on herb species composition in Coastal Valley Grassy Woodlands may, however, be the product not of fire-related characteristics of herbaceous species *per se*, but of the influence of the larger lifeforms – trees and shrubs – on ground layer species. Watson (2005) assessed the effects of fire frequency and microhabitat in six CPW sites. She found no evidence of a direct effect of fire frequency on ground layer species richness or floristics. However open patches, patches around trees and patches under Bursaria varied significantly in species composition. Ten of 14 species with a statistically significant connection with a particular microhabitat favoured open patches, including several native lilies. Given the finding that Bursaria abundance increased massively when fire frequency was low (Section 3.2.2), Watson (2005) concluded that fire frequency would likely affect ground layer composition in CPW indirectly, with ground layer species with a preference for open areas declining as Bursaria density increased. Miles (2005), drawing on local observation, describes this dynamic in dry grassy forest remnants around Bega.

It is sometimes suggested that fire may be less important for maintaining the herb complement in grassy woodlands than it is in grasslands, because tree canopies reduce the competitive advantage of dominant grasses, allowing forb species richness to remain high (Kenny *et al.* 2004). Support for this contention comes from research in the grassy Box woodlands of the Western Slopes, where Prober *et al.* (2002a) found increased herb species richness under trees. However Watson (2005) found no significant difference between CPW microhabitats in native herb species richness; this variable was in fact somewhat lower under trees. 'Under tree' microhabitats did differ from open patches, however, in having a significantly greater amount of leaf litter.

Findings from Watson's microhabitat study suggest that the forbs growing under trees in Coastal Valley Grassy Woodlands may not be the same ones at risk of being outcompeted by tussock grasses in open patches. Other researchers have also found that herb composition differs with canopy cover (Chilcott *et al.* 1997, Gibbs *et al.* 1999, Costello *et al.* 2000, Facelli and Temby 2002). It may be that fire plays a somewhat different role in mediating ground layer diversity in open woodland patches than it does under trees. In open areas, grass/forb interactions may be similar to those in grasslands; under trees, the mulching effects of litter may be more of an issue for herb growth and reproduction. Somewhat different suites of species may be involved in each environment.

3.2.5 Exotics

As Miles (2000) notes, woody weeds such as African Boxthorne can be a major problem in Coastal Valley Grassy Woodland remnants.

In CPW, African Olive (*Olea europaea* subsp. *cuspidata*) is a major invasive exotic species. Recently, von Richter *et al.* (2005) showed that fire has helped control Olive in one CPW remnant, by killing young plants before they became large enough to survive a burn – which they found took around six to eight years. These findings were reinforced by Watson (2005), who encountered considerably more woody exotics in low fire frequency sites than in areas which had burnt at least once a decade. Very frequently burnt sites had virtually no woody exotics. Olive, like Bursaria, can recruit between fires, an attribute which probably characterises other exotic woody weeds. Often seeds of these species are brought in to remnants native vegetation patches by birds, and establish below trees.

The link between fire, *Themeda* and herbaceous weeds has already been discussed (Section 3.1.3). In Coastal Valley Grassy Woodland on the Cumberland Plain, Watson (2005) found significantly fewer herbaceous weed species in very frequently burnt areas than where fire frequency had been low. There was a significant negative association, at a small scale, between the abundance of *Themeda australis* and the species richness and abundance of exotic herbs: more *Themeda*, less weeds. Again these results echo those from grasslands and grassy woodlands elsewhere (Section 3.1.3).

Regular burning may therefore not only help maintain the balance between I and T shrubs, and between shrubs and herbs in Coastal Valley Grassy Woodlands, it may be a potent tool in the fight against weeds.

3.2.6 Are grazing and fire interchangeable?

It is sometimes suggested that where grassy vegetation is grazed, fire is not needed to conserve native plant diversity. This suggestion springs from the observation that both fire and grazing can remove built up grass biomass that limits space for the forbs and small-statured grasses that grow between tussocks of dominant grass species (Kirkpatrick and Gilfedder 1999, Johnson and Matchett 2001, Lunt and Morgan 2002).

In this section I argue that despite the above similarity, periodic fire and stock grazing are not equivalent disturbances in other regards, and are therefore likely to lead to different biodiversity outcomes over time. Differences include:

- Fire removes vegetation in a non-selective manner, while grazing animals select more palatable, and accessible, plant species.
- Fire cues or catalyses processes in the life cycle of some plant species in a way that grazing is unlikely to replicate.

- Fire is a periodic disturbance which recurs at a scale of years to decades, while grazing is often continuous. Even 'crash' grazing is likely to involve defoliation at least once or twice a year.
- Fire and grazing are probably associated with fundamentally different nutrient levels and nutrient cycling processes, which will in turn affect plant species complements.

Selectivity in vegetation removal

Many studies attest to the fact that plant species are differentially affected by grazing, and by different levels of grazing (eg Landsberg *et al.* 1990, McIntyre *et al.* 1995, 2003, Clarke 2003, Dorrough *et al.* 2004). Species which are preferentially grazed are likely to decline in abundance, while unpalatable species increase. Some species which were once abundant in regularly burnt grassy woodlands have almost disappeared with grazing – the Murnong Daisy (*Microseris scapigera*) in Victoria is one example (Gott 1983), the Button Wrinklewort (*Rutidosis leptorrhynchoides*) is another (Morgan 1995). Once-dominant tussock grasses such as *Themeda australis* and *Sorghum leiocladum* are very sensitive to stock grazing but are encouraged by fire (Section 3.1.3). In Tasmania, shoot numbers of the endangered forb *Stackhousia gunnii* generally increased after fires, but tended to decrease with grazing (Gilfedder and Kirkpatrick 1998).

Interactions between plant lifecycles and disturbance

In fire-prone ecosystems, aspects of the life histories of many plant species are cued to, or catalysed by, fire. Examples include increased post-fire flowering, a trait found in shrubs such as *Lomatia silaifolia* (Denham and Whelan 2000) and in many herbaceous species (Lunt 1994, Watson 2005); post-fire seed release, an attribute found in some eucalypts (Gill 1997) and shrubs (Bradstock and O'Connell 1988, Enright and Lamont 1989, Lamont and Connell 1996); and heat- and smoke-cued germination, a characteristic of many shrubs (Auld and O'Connell 1991, Roche *et al.* 1998, Thomas *et al.* 2003) and also of some grasses and herbs (Read *et al.* 2000, Hill and French 2003). While many grasses and herbs are not dependent on fire-related cues (even though some may respond to them when present), and thus produce seeds which should germinate readily in gaps produced by grazing animals, others may not do so.

Fires also assist seedling establishment through their effects on competition, and on nutrient and water availability. While grazing may also reduce competition from dominant grasses, it may not provide the same establishment opportunities as fire.

Williams *et al.* (2005) used cutting – which simulates grazing – to explore the differential effects of burning and defoliation without fire on seedling emergence in a savanna woodland near Townsville. Both cutting and burning produced much higher levels of seedling emergence when rain arrived than occurred in undisturbed savanna, where virtually no seedlings were found. The number of seedlings emerging was significantly greater after burning than after cutting, a difference which was also found for some individual species. Seedling survival over the next couple of years was significantly higher in burnt than cut plots, in fact virtually none of the seedlings which germinated in the cut plots survived (Williams *et al.* in preparation). From this and previous work, Williams *et al.* (2005:493) concluded that in these woodlands "multiple fire-related cues promote germination… including exposure to heat shock, smoke, enhanced nitrate levels" as well as removal of competition from the herbaceous layer.

Frequency of disturbance

Much of the literature on fire and biodiversity is concerned with the effects of fire frequency. Where fires are too frequent, many species, particularly shrubs, will be reduced in abundance and may even become locally extinct due to their inability to reach life history milestones or to survive multiple episodes of defoliation (Section 2.1.2). Where these shrub species are palatable, grazing at short intervals is likely to have similar effects. Even crash grazing would constitute a very high frequency disturbance regime relative to the lifecycle of many native shrubs. A regime of fires tailored to plant species vital attributes allows time for shrubs to recover between disturbance episodes and reach maturity.

Some herbaceous species may also be unable to complete their lifecycles when grazed. Dorrough and Ash (2004) found that sheep selectively removed flowers and seedheads of the daisy *Leptorhynchos elongatus* in grasslands on the Monaro. Seedling recruitment was lower, and mortality of adult plants was higher, in grazed areas. While continuous grazing may be more problematic for native plant species than seasonal or crash grazing, even occasional grazing may not provide enough time for some herbaceous species to grow up and reproduce. For example the orchid *Diuris punctata* flowers in the second, but not the first year after defoliation (Lunt 1994).

Nutrient cycling

Burning and grazing may be associated with differences in nutrient cycling. While frequent burning is associated with low levels of available nitrogen and deep-rooted C_4 tussock grasses which use nitrogen efficiently (Ojima *et al.* 1994, Fynn *et al.* 2003), heavy grazing can increase nitrogen availability (Bromfield and Simpson 1974) and often disadvantages native C_4 tussock grasses (Section 3.1.3).

Johnson and Matchett (2001) investigated the effects of fire and grazing in prairie grasslands in North America. Grazing decreased growth of grass roots, while frequent burning encouraged it. Nitrogen concentration in roots was higher in grazed areas than in ungrazed and burnt exclosures, and the C:N ratio was lower. These researchers concluded that the two disturbance processes were associated with fundamental differences in nitrogen cycling, and that this was likely to be reflected in the species complements supported under each disturbance regime. Previous work in the same ecosystem concluded that frequent fire encouraged C_4 grasses which were efficient users of N, while lack of fire allowed N to build up, tipping the balance towards C_3 species. Prober *et al.* (2002b) report similar dynamics in grassy Box woodlands on the Western Slopes. Woodlands which had not been degraded by heavy grazing were dominated by native tussock grasses, particularly *Themeda australis* and *Poa sieberiana* (under trees). These sites had much lower nitrate levels than more degraded sites, which were dominated either by the C_3 grass taxa *Austrodanthonia* and *Austrostipa* or, in less naturally fertile areas, by *Aristida* and *Bothriochloa*, C_4 taxa which do not form large tussocks. The most degraded sites had the highest levels of nitrate and were dominated by annual exotic weeds.

Other studies have also found that herbaceous exotics tend to increase with grazing pressure (McIntyre *et al.* 2003), but may decline with fire, particularly if fire helps maintain a healthy sward of *Themeda* (Lunt and Morgan 1999b, Prober *et al.* 2004).

That these differences between grazing and burning can lead to different biodiversity outcomes is well illustrated in a study from Gippsland in Victoria. Lunt (1997a) compared frequently burnt but ungrazed grassy remnants with high-quality grassy forest remnants which had rarely been burnt but which were intermittently grazed. Although originating from the same species pool many years previously, areas subject to the two different management regimes differed considerably in species composition. While native species richness was higher in the unburnt quadrats, burnt quadrats had double the number of native geophytes, a category which includes native lilies and orchids. Numerous species were significantly more abundant under one regime or the other. Notably, *Themeda australis* was found in all frequently burnt sites, but was not recorded from the unburnt and grazed sites. On the other hand the C_3 grasses *Danthonia geniculata, D. racemosa, Microlaena stipoides*, *Poa sieberiana* and *Stipa rudis* were all significantly more abundant in the grazed but unburnt remnants.

In terms of fauna habitat, the biomass removal properties of grazing probably assist in providing suitable habitat for some fauna species which also favour environments generated by periodic burning (Redpath 2005, Wong *et al.* 2006). However the differential effects of fire and grazing on some habitat features, such as C_4 tussock grasses, mean that the two disturbances are unlikely to be interchangeable for all native fauna.

3.2.7 Conclusion

Fire plays an important role in conserving plant diversity in Coastal Valley Grassy Woodlands – a role which grazing is unlikely to replicate. Fire provides opportunities for heat and smoke-cued shrubs and grasses to recruit, it limits the abundance of gap-recruiting native shrubs which may encroach to the point where woodlands are no longer grassy, it ensures the open patches favoured by many herbaceous species remain available, it helps keep Kangaroo Grass swards healthy, it can play a role in limiting woody and herbaceous weeds.

How often should fire occur in this vegetation type? Coastal Valley Grassy Woodlands are at the ‘high end’ of the woodland spectrum in terms of productivity; it is therefore likely that plants will grow relatively rapidly and that competition dynamics will not take long to cut in. Watson (2005) predicts that variable interfire intervals between 4 and 12 years will maintain *Themeda* woodland with Bursaria thickets, open areas, and obligate seeder shrubs on the Cumberland Plain. She suggests that a simple monitoring program focused on flowering of obligate seeder shrubs, *Themeda* health and Bursaria expansion, could help managers tailor thresholds to particular site and climatic conditions. These recommendations are congruent with those of South Coast ecologist Jackie Miles (2005), who suggests burning Bega or Candelo Dry Grass Forest remnants every few years, using an adaptive management approach, to create gaps for sun-loving herbs and keep dense shrub regrowth within bounds. Cameron (2006) also suggests a regime of cool, frequent fires to limit dense shrub growth in Bega Dry Grass Forest.

Watson’s lower threshold of 4 years was based on analysis of the vital attributes of CPW shrubs, using the Noble and Slatyer (1980) framework also used by Kenny *et al.*

(2004), and is one year below the grassy woodlands statewide threshold of 5 years. This difference is small, and unlikely to be of great significance. A minimum interval of five years may be more appropriate than four years in south coast grassy woodlands, as temperatures may be cooler than on the Cumberland Plain.

Watson's upper threshold of twelve years reflects her findings on the vulnerability of *Themeda* to infrequent burning, the much reduced abundance of obligate seeder shrubs where fire frequency has been low, and the readiness with which Bursaria and exotics appear to expand in the absence of fire. This threshold is much lower than the statewide upper threshold for grassy woodlands, which is 40 years (Kenny *et al.* 2004). As already discussed in Section 1.4, Kenny *et al.* (2004) acknowledge the uncertainty surrounding their upper thresholds. These authors bypassed grassy woodland species in their 'most sensitive' category with estimated lifespans of 10, 20, 30 and 35 years, because these estimates were considered dubious. The need to consider interactions *between* species, as well as characteristics of individual species, has also been noted (Section 1.5). Certainly fire-mediated interactions are important in CPW, and are likely to be so throughout the Coastal Valley Grassy Woodland spectrum. Taking account once again of the cooler southern climate, an upper thresholds of 15 years is suggested for Coastal Valley Grassy Woodlands in the Southern Rivers region.

Although a moderate fire frequency should maintain a balance between different vegetation components in Coastal Valley Grassy Woodlands which presently contain both open grassy areas and a range of shrubs, where this is not the case, fire return intervals below 4 or 5 years, or above 15 years, might sometimes be appropriate. Intervals below five years may help to restore open grassy patches in areas where exotic or native shrubs have become so thick that ground layer species are struggling to survive. This proposition has not been tested so a cautious adaptive management approach is suggested. Short intervals may also be appropriate in areas where the aim is to maintain a very open grassy environment, perhaps to provide habitat for particular animal or plant species. On the other hand, it may be desirable to maintain limited patches of thick shrubs in some woodland areas, again because they provide fauna habitat not available elsewhere. In these patches some intervals above 15 years may help keep thicket thick.

4 Fire in dry sclerophyll forests

Dry sclerophyll forests are icons in the Australian landscape. Growing on infertile and often rocky soils, these forests support many sclerophyllous (hard-leaved) shrubs which flower spectacularly under a canopy of hard-leaved trees, almost all of which are eucalypts. Dry sclerophyll forests vary considerably both in composition of canopy species and in structure and make up of the understorey. Where grasses are a conspicuous element of some classes in this formation, in others shrubs rule and ground cover is sparse and more likely to consist of hard-leaved sedges than of grasses *per se*. The shrub/grass dry sclerophyll forests represent a transition between grassy woodlands and shrubby dry sclerophyll forests; soils and composition of the tree layer also form a continuum (Keith 2004).

“Bushfires play a vital role in the dynamic ecology of the dry sclerophyll forests … provid[ing] a critically important cue for regeneration by periodically stimulating the release of seeds, liberating resources such as nutrients and light, and by creating the open space essential for slow-growing sclerophyllous seedlings to survive” (Keith 2004:120). Research into the role of fire in the dry sclerophyll forests of the Southern Rivers region – outside of those in the Sydney region which have already been mentioned in Section 2.1 – has not been extensive. Some work has been carried out, however, in the two vegetation classes selected for discussion in this section: Southern Tableland Dry Sclerophyll Forests (Section 4.1) and South-East Dry Sclerophyll Forests (Section 4.2). Keith (2004) places both these classes in the shrubby subformation of dry sclerophyll forests.

4.1 Southern Tableland Dry Sclerophyll Forests

4.1.1 Introduction

Southern Tableland Dry Sclerophyll Forests occur on the Southern and Central Tablelands between 600 and 1100 m above sea level, in rugged, rocky terrain on shallow, infertile soils. This is not productive country: trees reach only 15-20 m in height, sclerophyllous shrubs form an open, species-poor understorey, the open ground cover of tussock grasses contains *Lomandra* species and some forbs but not the wide range of herbs found in adjoining grassy woodlands on more fertile soils. Trees are mostly stringybarks, peppermints and scribbly gums. *Eucalptus macrorhyncha* (Red Stringybark) and *E. rossii* (Scribbly Gum) occur throughout the range of this vegetation class, while other eucalypt species are distributed according to soil type, rainfall and altitude. *E. dives*, for example, co-occurs in elevated, wetter areas. Shrubs are mostly peas, wattles and epacrids; common species include *Daviesia mimosoides*, *Daviesia latifolia*, *Acacia falciformis*, *Hibbertia obtusifolia* and *Melichrus urceolatus*. *Joycea pallida* (Silvertop Wallaby Grass) is commonly found in the ground layer.

Although this vegetation class is found as far north as Mudgee, its distribution in the Southern Rivers CMA region is mostly confined to the south-west, where extensive areas are found in the foothills of the Kosciusko main range and in the gorges around Byadbo. To the north it occurs on hills near Bathurst, Goulburn and Yass, and also on Black Mountain near Canberra. Though poor soils discouraged clearing these forests were often selectively logged, and grazed (Moore 1953, Keith 2004). Current distribution includes substantial areas in reserves as well as patches on private property (Keith 2004).

About ten of the vegetation groups identified by Gellie (2005) could be categorised as Southern Tableland Dry Sclerophyll Forest. Most, however, are distributed outside the Southern Rivers CMA region. Those most likely to be found within its boundaries include:

- Group 115 – South East Tablelands Dry Shrub/Tussock Grass Forest. This open forest up to 18 m in height is dominated by *Eucalptus rossii*, together with *E. mannifera* and occasional *E. macrorhyncha* or *E. dives*. Common shrub species include *Hibbertia obtusifolia*, *Melichrus urceolatus*, *Pultenaea procumbens* and *Brachyloma daphnoides*. Ground cover is dominated by thick *Joycea pallida* and *Poa sieberiana* and is not species rich. This vegetation type once covered 74,200 ha of which 81% remains uncleared; of this, 91% occurs outside conservation reserves.
- Group 119 – Western Tablelands Dry Shrub/Grass Forest is dominated by *E. macrorhyncha* with *E. polyanthemos* or *E. nortonii* as subdominants. The few shrub species include *Acacia dealbata*, *Hibbertia obtusifolia* and *Brachyloma daphnoides*. A sparse ground cover includes a range of grasses, particularly *Danthonia pilosa* and *Elymus scaber*, and a variety of forbs. Eighty-one percent of the estimated pre-European extent of this veg type has been cleared, and less than one percent of what remains is reserved.
- Group 121 – Western Slopes Grass/Herb Dry Forest. *E. macrorhyncha* and *E. nortonii* dominate this vegetation type, which ranges from 15 to 22 m in height. The sparse shrub layer includes *Hibbertia obtusifolia*, *Brachyloma daphnoides* and *Melichrus urceolatus*. *Joycea pallida* and *Poa sieberiana*, which are the most common tussock grasses, are interspersed with a variety of smaller grasses, graminoids and forbs. Sixty-two percent of the 90,800 ha once covered by this vegetation group remains uncleared; 70% of what remains is outside conservation reserves.

4.1.2 Fire on Black Mountain

Thirty years ago, Rosemary Purdie conducted some of Australia's first fire ecology studies in the shrubby forests on the slopes of Canberra's Black Mountain (Purdie and Slatyer 1976, Purdie 1977a,b). This careful work demonstrated the varied responses of plant species to fire, and is still highly relevant today.

Although not in the Southern Rivers region, the vegetation on Black Mountain definitely falls into the Southern Tableland Dry Sclerophyll Forest category. *Eucalptus macrorhyncha*, *E. rossii* and *E. mannifera* are the dominant tree species. Common shrubs include *Daviesia mimosoides*, *Acacia genistifolia*, *Dillwynia retorta* and *Brachyloma daphnoides*. Herbaceous species include grasses *Joycea pallida* (then *Danthonia pallida*) and *Poa* species, with occasional forbs, particularly orchids, appearing in winter and spring. Elevation is about 650 m above sea level, annual rainfall averages 626 mm (Purdie and Slatyer 1976).

Experimental fires were carried out in three sites, which had previously been burnt 9, 11 and 33+ years ago respectively. Purdie and Slatyer (1976:225) note that prior to the experimental fire in the long unburnt site, “many of the larger shrubs were senescent or in varying stages of decay,” whereas shrub density at the other two sites was considerable. At each site 60 small quadrats were set up, 30 in an area which was then burnt (in summer), and 30 in an area which was left unburnt. Post-fire regeneration was monitored over several years (Purdie and Slatyer 1976). Species were classified as ‘obligate seed regenerators’ (obligate seeders), ‘obligate root resprouters’ (resprouters which didn’t produce seedlings), or ‘facultative root resprouters’ (resprouters which did produce seedlings).

Here are some of the findings:

- Almost all shrubs resprouted; the only shrubs which were clearly obligate seeders were *Acacia genistifolia*, *Dillwynia retorta* and *Pimelea linifolia*. *Pimelea linifolia* germinated in the long-unburnt site after fire, though it was not recorded there prior to the burn. All three species flowered within three years of fire (Purdie and Slatyer 1976).
- Most resprouting shrubs also produced seedlings. Some, such as *Daviesia mimosoides*, *Acacia buxifolia* and *Indigofera australis* were able to increase population numbers by root suckering – which was stimulated by fire. Flowering on resprouts occurred within two years of fire. Seedlings were slower to reach maturity, with many not flowering by the end of the three year monitoring period (Purdie and Slatyer 1976).
- Shrub and herb seedlings arose from seeds stored in the soil (Purdie 1977b).
- All tree species resprouted and also produced seedlings (Purdie and Slatyer 1976). Tree seedlings developed in both burnt and unburnt plots plots (Purdie 1977b).
- All the resprouters which didn’t produce seedlings were geophytic monocots, a category which included many orchids. All orchids flowered in the first year after fire. Several herbaceous species in this category – three Lomandras and *Dianella revoluta* – were able to increase through vegetative regrowth (Purdie and Slatyer 1976).
- Almost all species showed much higher seed germination in burnt compared with unburnt areas (Purdie 1977a, Purdie and Slatyer 1976), and survival rates were also generally higher in burnt vegetation (Purdie 1977b).
- A few species, including the dominant grass *Joycea pallida* and the exotic herb *Hypochoeris radicata*, produced more seedlings in unburnt than in burnt areas (Purdie and Slatyer 1976).
- Obligate seeders had more seedlings and higher seedling survival rates than resprouters (Purdie and Slatyer 1976).
- Many resprouters which weren’t able to sucker flowered rapidly after fire on regrowth, and this produced some seedlings in Year 2 (Purdie 1977b, Purdie and Slatyer 1976).
- Resprouter recovery stabilised at a lower level in the previously long-unburnt site (60% of pre-fire numbers – which we have already noted were relatively low), than in the two more recently burnt sites (85-90% of pre-fire numbers) (Purdie 1977a).

- In the unburnt plots, many species declined with time. Obligate seeder population numbers declined 16-22% in these plots in the first year of monitoring, and an additional 5-10% in the second (Purdie 1977a).
- "With the exception of rare species, all the tree, shrub and herb species represented in the quadrats prior to burning … were present in the regenerating communities in the first year after burning" (Purdie and Slatyer 1976:233).

Purdie and Slatyer (1976) conclude that changes in floristics as time-since-fire go by are merely changes in dominance due to differences in species growth rates and longevity, and, sometimes, reflect a move from visibility above-ground to presence only in the under-ground seedbank. 'Relay floristics', where additional species join the community as the period without disturbance lengthens, does not appear to characterise Southern Tableland Dry Sclerophyll Forest. This conclusion is consistent with what one might expect from a vegetation type that is, in Bond *et al.* (2003, 2005) terms, primarily climate limited, with some degree of modification by fire (Section 1.5).

Purdie's findings above tell us something about the effects of fire on plants in general, and also about species responses in Southern Tableland Dry Sclerophyll Forests in particular. Higher germination and seedling survival rates in burnt than in unburnt patches is a common finding in fire-prone vegetation, as is the tendency for obligate seeders to outdo resprouters in numbers of post-fire seedlings (Section 1.2). The small obligate seeder shrub complement and the rapid post-fire flowering, while by no means unique to this ecosystem, distinguishes it from some others such as the Sydney Coastal Heaths discussed in Section 2.1.

4.1.3 Post-fire grazing

Another classic study, by CSIRO researchers Leigh and Holgate, was published in 1979. This study assessed the effects of post-fire grazing by mammals on seedlings and resprouts in three dry sclerophyll forest and woodland environments on the Southern Tablelands. One site, at Mundoonen Nature Reserve near Yass, was classic Southern Tablelands Dry Sclerophyll Forest. Dominant trees were *Eucalyptus rossii*, *E. macrorhyncha* and *E. bridgesiana*. Shrubs included *Daviesia virgata*, *Dillwynia retorta*, *Gompholobium huegelii* and *Melichrus urceolatus*. *Poa sieberiana* dominated the ground layer. The main grazing animals at this site were grey kangaroos, with a few swamp wallabies and rabbits (Leigh and Holgate 1979).

Some of the plots at each site were open to grazing, while the rest were fenced to prevent access. Plots were not large. Half of those in each treatment were burnt while others remained unburnt. In the Mundoonen site the effects of grazing were monitored for three years before a single low intensity fire in early autumn was added to the experimental design.

Grazing alone reduced shrub density at Mundoonen by 19% over four years, while on ungrazed plots shrub density increased by 8% due to recruitment by the native pea *Indigofera australis*. Grazing effects were not uniform across shrub species. Fire increased grazing-related mortality in some species. Across the three sites, grazing after fire produced a range of effects on different species, but generally increased the mortality of both shrub seedlings and resprouts, creating a more open, grassy environment. Leigh and Holgate (1979) conclude that the interaction between fire and grazing may be of considerable importance and deserves further study.

To what extent post-fire grazing effects were exacerbated by the small size of the fires in this study is not clear. A number of other studies have shown that grazers, both native and domestic, often concentrate on recently burnt areas (Robertson 1985, Andrew 1986, Southwell and Jarman 1987). Where burnt areas are extensive, animals may spread out, reducing grazing pressure at any one point (Archibald and Bond 2004). Conversely, where burnt areas are small, grazing may be particularly heavy. It may therefore be a good idea to burnt relatively large patches at a time (Trollope and Trollope 2004).

The need to consider the effects of post-fire grazing by native animals may be growing in the Southern Tablelands as numbers of macropods and wombats build up (pers. comm. various land managers in the SR region, 2005).

4.1.4 Conclusion

Although fire ecology research in Southern Tablelands Dry Sclerophyll Forests has not been extensive and is now some decades old, it provides indications as to the probable effects of different fire frequencies. Kenny *et al.* (2004) recommend intervals between 7 and 30 years for dry sclerophyll shrub forests. From the little we know and can surmise, this recommendation is likely to be appropriate for Southern Tableland Dry Sclerophyll Forests.

Thinking first about lower thresholds, the risks of very short interfire intervals described for other vegetation types (Sections 1.4, 2.1.2) of course also exist in Southern Tableland Dry Sclerophyll Forests. However the species characteristics observed by Purdie and Slatyer (1976) suggest that moderately frequent fire is unlikely to be problematic. All known obligate seeder species on Black Mountain flowered within three years of fire, and all resprouting species flowered, on resprouts, within two years. A seven year minimum is well above these juvenile periods. Seven year intervals, and even occasional intervals down to 4 or 5 years are thus unlikely to cause species to be lost from this community.

Upper thresholds need to ensure that I species have opportunities to recruit before adult plants and seedbanks decline, and that competitive interactions that occur in the absence of fire are managed (Sections 1.4 and 1.5).

From the little we know, competitive exclusion does not appear to be a major issue in Southern Tablelands Dry Sclerophyll Forests. Net seedling recruitment in unburnt plots on Black Mountain, and particularly in the plot which had not had a fire for over 30 years, was not high (Purdie 1977b), and species which recruited preferentially in unburnt areas were often herbaceous (Purdie and Slatyer 1976). It appears that the shrub complement in this vegetation type does not contain any, or many, of the T species shrubs that increase when vegetation in warmer, wetter areas is left unburnt for a long time (eg Section 3.2.2). Nor, it seems, are many shrubs sufficiently large and long-lived to dominate to the detriment of smaller species after many years without fire, as occurs in coastal heaths (Section 2). Rather, shrubs in Southern Tableland Dry Sclerophyll Forest are generally fire-cued and decrease in abundance as time-since-fire progresses. This vegetation type is thus likely to look more ‘open’ when long unburnt than when a burn has occurred within the last couple of decades. The grass *Joycea pallida* may recruit between fires (Purdie 1977b), and that may assist the development of a grassy understorey in the absence of fire. Whether competitive interactions between dominant grasses and smaller herbaceous species such as orchids exist in long unburnt areas Southern Tableland Dry Sclerophyll Forest is not known. It is also possible that some exotics or native species from other environments may be able to establish and multiply between fires, and so could present a threat to native plant diversity. *Kunzea ericoides*, whose ability to encroach has been documented in the ACT and Victoria, may be a candidate (Kirschbaum and Williams 1991, Singer and Burgman 1999). This possibility should be monitored.

The greater risk, at this end of the fire frequency spectrum, is that I species native to the Southern Tablelands Dry Sclerophyll Forest may decline under very long intervals. The recommended upper threshold of 30 years is several years below the maximum interfire interval involved in Purdie's study (33 years). Shrubs in this long unburnt site were senescent prior to the experimental fire, and total plant density, at 8.0 plants/m^2, was considerably lower than in the more recently burnt sites, which averaged 30.4 and 19.2 plants/m^2 respectively (Purdie and Slatyer 1976). The difference in density was particularly marked for small shrubs and herbs. Although lack of replication of the times-since-fire in this study means these observations should be treated with caution, they are congruent with other findings and observations. Recent inspection of one of Purdie's sites, now unburnt for 34 years, found shrubs were sparse, though cover of *Joycea pallida* was considerable (pers. obs. 2005; pers. comm. Margaret Kitchin, Environment ACT, 2005). Monitoring in unburnt plots over the years of Purdie's study showed declines in plant numbers (Purdie 1977a), and post-fire resprouting was less prolific on the previously long unburnt site than in sites with shorter interfire intervals (Purdie 1977a). If numbers of adult plants of many species decline with time-since-fire, after several decades regeneration will increasingly depend on the longevity of seedbanks – an unknown quantity. Seedling regeneration in Purdie's site with the long interfire interval was of the same order as that in the more recently burnt sites (Purdie 1997b), suggesting seedbanks at 30-35 years post-fire are still adequate. However this may not be the case four or five decades post-fire. Grazing before fire may exacerbate the problem by reducing seed input by adult plants, while post-fire grazing may eliminate seedlings and stress resprouting plants, particularly if burns are small (Section 4.1.3). In very long unburnt sites with few shrubs it may be necessary to supplement seedbanks with seed from nearby more frequently burnt areas if the aim is to restore the pre-existing species complement.

An additional issue for fire frequency concerns how long the seedlings of the many resprouting species take to become fire-tolerant. We do know that they grow slowly relative to their obligate seeder counterparts, and Purdie (1977a) suggests that, from her observations, even after a 9 to 11 year interfire interval young plants may be vulnerable. Intervals above 15 years within a variable regime of intervals between seven and 30 years should allow opportunities for these seedlings to fully develop.

4.2 South East Dry Sclerophyll Forests

4.2.1 Introduction

This vegetation class dominates a large area of south-eastern NSW on shallow, infertile soils from sea level to 1300 m. *Eucalyptus agglomerata* (Blue-leaved Stringybark), *E. globoidea* (White Stringybark) and *E. sieberi* (Silvertop Ash) occur across the altitudinal range. *Corymbia gummifera* (Red Bloodwood) and *E. consideriana* (Yertchuk) are common near the coast, while *E. dives* (Broad-leaved Peppermint) and *E. smithii* (Ironbark Peppermint) are found at higher altitudes. *Allocasuarina littoralis* often forms a subcanopy. The understorey consists of sclerophyll shrubs including *Acacia obtusifolia*, *Acacia terminalis*, *Monotoca scoparia* and *Persoonia linearis*, with a sparse cover of sedges and grasses, particularly *Joycea pallida* (Silvertop Wallaby Grass). Relative to similar forests around Sydney, species richness in South East Dry Sclerophyll Forests is not high (Keith 2004).

Gellie (2005) lists many vegetation groups in several vegetation classes which may fall into Keith's South East Dry Sclerophyll Forest class. One of the most obviously relevant is Gellie's vegetation group 1. This group occurs between 300 and 800 m on exposed slopes and ridges. It is dominated by *E. sieberi* and *E. agglomerata* and sometimes has a small tree layer of *Allocasuarina littoralis*. Common shrubs include *Persoonia linearis* and *Acacia obtusifolia*. Within the area covered by Gellie (2005) this vegetation group spans an estimated 48,700 ha, is uncleared, and two-thirds of it is reserved. Gellie (2005) notes that a similar vegetation type, map unit 47, occurs in the Eden CRA Region.

Fire-related research in South East Dry Sclerophyll Forests has focussed on vertebrate fauna, on species responses to wildfire and on changes with time-since-fire. Effects of fire cycles on plants are being studied in forests near Eden, where replicated trials of low-intensity patchy fuel reduction burning have been underway since 1984. Results of this work are currently being collated, but are not yet publicly available (Trent Penman, Forests NSW, pers. comm. 2006). Fauna studies come from Nadgee Nature Reserve, which contains large expanses of forest in addition to the heathland discussed in Section 2.2 (eg Newsome *et al.* 1975, Recher *et al.* 1975, Catling *et al.* 2001). They also come from *E. sieberi*-dominated State Forest around Bega where in the early 1980s Dan Lunney and his colleagues surveyed a wide range of fauna species.[9] Though the primary aim of this work was to understand the effects of logging, a wildfire within the sampling period provided the opportunity for observations of the effects of fire (eg Lunney and O'Connell 1988, Lunney and Leary 1989). Work on birds has been carried out just over the border in Victoria by Loyn (1997).

4.2.2 Vegetation change with time-since-fire

Although available studies have not focussed on vegetation, there are indications that shrub dynamics in South East Dry Sclerophyll Forests may have common ground with those in the Southern Tablelands Dry Sclerophyll Forests discussed in Section 4.1. Fire, particularly intense fire, stimulates shrub germination, so that the open environment

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[9] Parts of Mumbulla State Forest where Lunney's research was centred have now been incorporated into Biamanga National Park (Penn *et al.* 2003).

created by incineration of the understorey is replaced over the next few years with thick regeneration of sclerophyll shrubs. With increasing time-since-fire it appears that many shrub species mature and then start to die off, opening the landscape up once again. This process takes at least two decades.

Catling *et al.* (2001) assessed various components of habitat complexity, including shrub cover, at Nadgee Nature Reserve over 18 years after the 1980 fire. The initial measurement one month after the fire reflected an open environment with less than 30% shrub cover in most survey sites. By six years post-fire shrubs had thickened, a finding confirmed by Recher *et al.* (1975) who report that by three years after wildfire in Nadgee forests, wattle regrowth had already reached 3-4 m and was so thick it was difficult to walk through. Catling *et al.* (2001) didn't measure the vegetation between 1980 and 1986, but then did so every two years to 1998. Shrub cover scores increased steadily between six and 12 years post-fire, then remained steady over the subsequent six years. On the other hand peak values for ground cover were reached by six years post-fire then declined in many sites, a result attributed to the decrease in light reaching lower vegetation strata as shrubs and trees thickened up.

Although there were few signs of a decrease in shrub cover by 18 years post-fire in the data gathered by Catling *et al.* (2001), Loyn (1997) noted that prior to fire in his forest sites shrubs were few. Dense shrub regrowth developed, however, after the fire. In more recent years scientists from the Department of Environment and Conservation have noted the understorey 'thinning out' at Nadgee, where time-since-fire is now mostly either 26 or 34 years (Ross Bradstock, pers. comm. 2006). Finally, unpublished data from the Eden Forestry trials shows a gradual decline in plant species richness over the course of the study across *all* treatments, ie those burnt at low intensity at 2 and 4 year intervals, and those left unburnt since the start of the study in the mid 1980s. The entire study area was burnt in an extensive, intense fire in 1983, and it seems likely that the decline in species richness reflects a time-since-wildfire effect (Trent Penman pers. comm. 2006).

Some shrub species in South East Dry Sclerophyll Forest may be able to regenerate between fires, and may have the potential to expand in some areas to the detriment of smaller plants. One obvious possibility is *Allocasuarina litoralis*, as this species is known to form dense thickets in the absence of fire in some environments (Withers and Ashton 1977, Smith and Smith 1990, Lunt 1998a,b). Observation suggests it tends to occur in groves which may be less flammable than the surrounding forest, enabling tall adults to avoid 100% leaf scorch in a fire, and thus survive. A matrix of Casuarina thickets and more open patches of sclerophyll shrubs may be desirable from a conservation point of view.

4.2.3 Small mammals

The abundance and distribution of small mammals in relation to fire has been studied at Nadgee Nature Reserve (Recher *et al.* 1975, Catling and Newsome 1981) and in forest north-east of Bega (Lunney *et al.* 1987, Lunney and Leary 1989, Penn *et al.* 2003). Small mammals generally appear to be most abundant some years after fire, with post-fire population recovery occurring more rapidly where the understorey is either not greatly affected, or where it can recover quickly. At least one exception to this rule may exist, however.

At Nadgee, forest plots along the river supported two Antechinus and two native rat species prior to the extensive 1972 wildfire. In the first year after the fire the Swamp Rat *Rattus lutreolus* and the Dusky Antechinus (*A. swansonii*) disappeared completely from this area, while Bush Rats (*Rattus fuscipes*) and Brown Antechinus (*A. stuartii*) were found only in small numbers. All four species, however, occurred in nearby unburnt areas and produced young in the first year after the fire (Recher *et al.* 1975). The rats recovered faster than the Antechinus: according to Recher *et al.* (1975) by two years post-fire Bush Rats were found everywhere in forest where they had been found prior to the fire, while Swamp Rats were also common, even where they had not been found pre-fire. Catling and Newsome (1981) also note that the rodents recovered quickly, and add that this occurred most rapidly in moist habitats. Their data show that native rat numbers generally peaked after five years, and then declined somewhat. *Antechinus stuartii* recovered to post-fire levels by the fifth post-fire year, with *A. swainsonii* taking a year or two longer. Recher *et al.* (1975) point out that this difference reflects the ecology of the two animals. *A. stuartii* is noctural, partly arboreal, eats insects on the ground but also on shrubs and trees, and thus is less at risk of predation and less dependent on cover than *A swansonii*, which is active by day, forages on the ground, and likes logs, dense shrubs, and lots of litter. The differential impact of severe fire on the two Antechinus species was also reflected in their condition on the Nadgee River plots in the first post-fire year. *A. swansonii* weighed up to 50% less than in the years before fire, whereas *A. stuartii*, which could find insects on eucalypt shoots, didn't lose weight at all.

Recher *et al.* (1975) also note that in the very early post-fire period, Pygmy Possums came down out of trees and foraged on the ground, where they got trapped by the researchers. "As the vegetation has regrown, [this species] has been caught less frequently, an indication that it is spending an increasing proportion of its time in the trees" (Recher *et al.* 1975:161).

In Mumbulla State Forest north-east of Bega studies have included pre- and post-fire surveys after a wildfire in 1980 (Lunney *et al.* 1987), and after the next fire in the area, which was a low-intensity fuel reduction burn in 1999 (Penn *et al.* 2003). As at Nadgee the Dusky Antechinus *Antechinus swansonii* was more affected by fire than other small mammal species: *A. swansonii* was not caught at all over the three years following the wildfire (Lunney *et al.* 1987), and was scarce after the planned burn (Penn *et al.* 2003). The second Antechinus species in this environment, *A. agilis* (called *A. stuartii* at the time of the earlier study), declined after each fire, particularly where the burn was more intense, but did not disappear. After the planned fire in 2000 this species maintained population numbers on less severely burnt south-east slopes but not on north-west slopes where burn was more intense (Penn *et al.* 2003). Bush Rats (*Rattus fuscipes*) at Mumbulla recovered slowly after wildfire, reaching 31% of pre-fire numbers by 2.5 years post-fire; Lunney *et al.* (1987) attribute this slow recovery to the sustained drought which followed the fire. The later planned burn, however, had little effect on *R. fuscipes*: numbers were slightly greater 10 months after the fuel reduction burn, particularly on the less-affected south-east slopes, while by 20 months post-fire the population was similar to what it had been prior to the fire (Penn *et al.* 2003).

Unlike the rodent and Antechinus species discussed so far, all of which are associated with cover to a greater or lesser extent, the rare and vulnerable White-footed Dunnart in Mumbulla Forest strongly favours open habitat such as logged, burnt ridges, and prefers sites with sparse ground and canopy cover (Lunney and Ashby 1987, Lunney *et al.* 1987, 1989). It survived the wildfire in 1980, reproduced successfully in the first post-fire year, but disappeared by the third post-fire winter as vegetation rapidly thickened up (Lunney and Ashby 1987, Lunney *et al.* 1987, 1989; Lunney and Leary 1989). These small animals vary in how far they travel, with some 'explorer males' covering distances of over a kilometre (Lunney *et al.* 1989). Researchers Lunney and Leary (1989) speculate that suitable habitat for this species may occur naturally only as disjunct and temporary patches, making the ability to travel an important attribute for survival.

The different relationship to habitat complexity of most small mammals vis-à-vis Dunnarts in the south-east was also noted by Catling and Burt (1995), who surveyed a range of forest types from Ulladulla to the Victorian border. Habitat complexity was positively correlated with the number of Brown Antechinus, Dusky Antechinus and Bush Rats captured; the Common Dunnart, however, was only trapped in habitats of low to moderate complexity.

4.2.4 Large and medium-sized mammals

Much what we know about large and medium-sized mammals and fire in the forests of the south-east comes from multi-site studies over broad areas which have looked for associations between indicators of animal abundance and habitat-related variables (Claridge and Barry 2000, Catling and Burt 1995, Catling *et al.* 2001). As for small mammals, much of the focus has been on cover. The models developed by Catling *et al.* (2001) are of particular interest as they are based on observations over 20 years in 99 heath and forest sites at Nadgee Nature Reserve. There are also post-fire observations from Nadgee (Newsome *et al.* 1975, Recher *et al.* 1975, Catling and Newsome 1981) and a post-fire study in a range of habitats in Mumbulla State Forest (Lunney and O'Connell 1988).

Available evidence generally indicates that the larger mammals of the south-east forests – kangaroos, wallabies and wombats – tend to be associated with fairly open, grassy environments that are more likely to be found in the early post-fire years and perhaps also where fire has been more frequent, than in later post-fire or rarely burnt environments. Medium-sized potoroos and bandicoots have a greater association with understorey cover. Predation almost certainly plays an important part in determining cover dependence.

Kangaroos, wallabies, wombats

There are hints that kangaroos, wallabies and wombats often survive even intense fire. Lunney and O'Connell (1988) counted dung pellets of the Red-necked Wallaby, Swamp Wallaby and Common Wombat at four and 17 months after an extensive wildfire in dry sclerophyll forest in Mumbulla State Forest north-east of Bega. No dead animals of these species were found after the fire, and all three taxa were present at both post-fire sampling periods, leading the authors to suggest that they had generally survived the blaze.

How does the abundance of kangaroos, wallabies and wombats vary with time-since-fire? According to the models developed by Catling *et al.* (2001) from the Nadgee data, kangaroos and wallabies are more abundant in early and middle than in later post-fire years, when declines are substantial. The models show these large animals responding more to time-since-fire *per se* than to habitat complexity, over a 25 year post-fire period. Catling and Burt (1995) found an association between large native herbivores and low habitat complexity scores, and comment that these species like forest with an open grassy understorey with few shrubs.

Shrub cover may still play a role for these species, however, particularly in the first year or so after fire when the openness of the environment means shelter from predators may be more than usually difficult to find. Newsome *et al.* (1975) report that kangaroo and wallaby numbers declined in the early post-fire years at Nadgee, due to increased predation by dingos. That food was not the problem is attested by Recher *et al.* (1975), who stated that kangaroos and wallabies at Nadgee were well-fed and that by the summer after the fire all females had young. A few years later, Catling and Newsome (1981) reported that macropod numbers remained steady for about three years after the fire and then rose sharply to well above pre-fire levels. Lunney and O'Connell (1988) found a significant positive association between shrub cover and wallaby dung at four months post-fire (Red-Necked Wallaby only), and at 17 months (Swamp and Red-Necked Wallabies). Shrubs can provide food as well as shelter, particularly for the Swamp Wallaby which is considered a browser rather than a grazer.

Wombats were relatively abundant after fire at Nadgee (Newsome *et al.* 1975, Catling *et al.* 2001). Modelling by Catling and Burt (1995) found a strong relationship between wombat abundance and low habitat complexity scores, however Catling *et al.* (2001) found just the opposite, while Lunney and O'Connell (1988) found no relationship between the abundance of wombat scats and cover measures. The model developed by Catling *et al.* (2001) showed time-since-fire was a significant variable for wombats, however it was difficult to interpret as abundance was high immediately after fire, decreased over the next decade then increased again. Lunney and O'Connell (1988), who examined fox and dingo scats, found few traces of wombats being taken in the first two post-fire years at Mumballa. However Catling *et al.* (2001) consider the decline in wombat numbers at Nadgee was probably due to dingo predation, and cite unpublished data showing that although wombat formed a very small part of dingo diet in the first five years after fire, the proportion rose substantially in years 7 to 9, after which dingo numbers declined.

Bandicoots and potoroos

The relationship between time-since-fire and abundance of bandicoots and potoroos was assessed by Claridge and Barry (2000) in a multi-site and multi-factor study over a wide area of eastern Victoria and south-east NSW. Post-fire observations from Nadgee have been published by Newsome *et al.* (1975) and Catling and Newsome (1981).

Claridge and Barry (2000), who studied factors associated with diggings of bandicoots and potoroos, found more diggings of both animals in sites unburnt for over 20 years than in sites burnt 0-10, or 10-20 years previously. Bandicoot diggings were also associated with a high density of ground cover, a measure which apparently included shrubs up to 2 m, leading the researchers to suggest that recently burnt habitats may not provide sufficient protection from foxes and dogs. Time-since-fire may also influence food resources, which for bandicoots and potoroos consist primarily of the underground fruiting bodies of fungi with an association with particular plant species. The relationship between fire, these fungi, and their host plants, however, is not at all clear.

Claridge and Barry (2000:683) conclude that burning in bandicoot and potoroo habitats "should be done mainly with long intervals."

There is also evidence, however, that the environment in the years immediately after a fire may suit Bandicoots quite well. According to Newsome *et al.* (1975) bandicoot activity increased post-fire at Nadgee, while Catling and Newsome (1981) report that bandicoot abundance increased in the early post-fire years, quickly reaching greater than pre-fire levels.

In the study by Claridge and Barry (2000), although time-since-fire was a significant variable for potoroos, cover was not. Potoroos appear to need a mosaic of dense vegetation, for shelter, and open areas for foraging.

Predators

A number of times in the discussion above the importance of dense shrub cover in allowing native animals to minimize the ravages of predators has been noted. How do predators themselves respond to time-since-fire and habitat complexity?

Findings for dingos vary. Catling and Burt (1995) found a positive correlation between dingo abundance and habitat complexity, whereas Catling *et al.* (2001), using data from Nadgee and including time-since-fire in their suite of variables, found no relationship with habitat complexity but a strong decline in dingo numbers in later post-fire years. Cats, in this study, increased with both time-since-fire and habitat complexity, while foxes were everywhere, irrespective of these variables. There are no doubt complex relationships between the abundance of predators and prey, which may alter as time-since-fire and/or cover increases.

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4.2.5 Arboreal mammals

Lunney (1987) surveyed possums and gliders at Mumbulla State Forest before a wildfire in 1981 and again six and 18 months later. With the exception of the Feathertail Glider which was rare before the fire, all species present pre-fire were recorded six months after the burn: these species were Greater Glider, Sugar Glider, Yellow-bellied Glider and Ring-tailed Possum. The Bushtail Possum, which was not found in the pre-fire survey, was located after fire. Unburnt and lightly burnt gullies provided critical refuges for arboreal mammals in the first post-fire year. For example before the fire, Yellow-Bellied Gliders were found on both slopes and gullies, whereas after fire they were confined to unburnt gullies.

Little is known about effects of fire on koalas. In North-eastern NSW Cork *et al.* (1997, cited in Cork *et al.* 2000) found that koalas were more likely to occur some years after fire than in the early post-fire years. Jurskis and Potter (1997) consider severe fires are liable to have at least short-term detrimental effects on koalas, and quote reports of loss of koalas from bushland around settlements after fires near Eden in 1952. However they also cite instances of recovery of populations after wildfire in forested land. These authors report that the home ranges of seven out of eight koalas they followed through radio-tracking had been affected by intense fires in the past, as evidenced by high frequencies of sapling regrowth trees in their home ranges, however information on time-since-fire is lacking.

4.2.6 Lizards

Lunney *et al.* (1989) studied three skink species in Mumbulla. A survey had just been completed when wildfire burnt through the study area in 1980. A second survey within weeks of the fire showed all three species were still present, in fact one, the Delicate Skink *Lampropholis delicata* was found in greater numbers after the fire than before it – a result put down to the difficulty of finding it in thick vegetation prior to the fire. Grass Skinks (*Lampropholis guichenoti*) were less abundant after than before the fire. Examination of the stomach contents of this species showed little food was available in the post-fire weeks, and the authors surmise that individuals died from starvation together with heat stress and vulnerability to predation resulting from lack of cover. Both *Lampropholis* species need relatively open habitat, however, and were less abundant where post-logging regrowth was thick. A third species, the Water Skink (*Eulamprus heatwolei*), showed no significant changes in numbers as a result of the fire.

Almost 20 years later the same area was once again surveyed before and after a low intensity planned burn in 1999; post-fire surveys were conducted at 10 and 20 months post-fire (Penn *et al.* 2003). The Grass Skink again showed a decline in the initial post-fire survey, however by 20 months it had recovered to greater than pre-fire levels. Delicate Skinks again showed little effect of fire in the initial survey, but by 20 months were less abundant in burnt than unburnt areas. Water Skinks stayed true to their previous form and were not significantly affected by the 1999 fire.

4.2.7 Birds

Smith (1989) studied bird populations in a 13 ha site in Mimosa Rocks National Park between Bega and the coast, before and after a moderate intensity wildfire. This site, which had a history of frequent burning, had a rich bird fauna, with 96 species recorded over three years. The forest bird community was “surprisingly resilient” to the effects of the fire and of the drought which was associated with it. Smith (1989) puts this down to the role played by gullies: the site encompassed a ridge and two gullies one of which contained some rainforest plants. The number of bird species recorded in the year before the fire, and in the first post-fire year as well as in the second, hardly altered, and changes in species composition were confined to occasional visitors to the site. Effects within the site, however, were noteworthy. Insectivorous species, particularly those which used dense shrubs for feeding, shelter and nesting, generally contracted into the unburnt and lightly burnt vegetation in the gullies; examples included the Brown Thornbill and the White-browed Scrubwren. On the other hand species with plant foods in their diet expanded their distributions and numbers as particular trees and shrubs flowered and fruited – although whether this was in response to the fire is unclear. Another feature of the year after the fire was a flurry of out-of-season breeding activity, again probably reflecting increased food resources. Recher *et al.* (1975) and Fox (1978) note similar behaviour at Nadgee, where some birds, particularly insect-eaters such as fantails, warblers and wrens, nested two and three times during the spring and summer.

Loyn (1997) studied birds just before, and for three years after, a very large wildfire in East Gippsland not far from the NSW border. Although this study wasn’t limited to dry sclerophyll forests it is included here because it provides insight into the way different groups of birds respond to fire.

Across all sites bird abundance declined, immediately after the fire, to an average of 60% of pre-fire levels. Numbers recovered over the next three years, doing so more rapidly in wetter environments, and faster in forest than heath. Gullies provided refuge in the early post-fire period, even when they had been burnt.

Canopy insectivores followed the pattern for all birds but changes were not significant. Understorey insectivores as a group also showed few changes, however some individual insect-eating species, particularly those associated with heathlands, showed major declines. Many insectivores that fed from tall shrubs continued to get food from scorched shrub branches and foliage, as did canopy insectivores. By one year post-fire regenerating shrubs and epicormic growth were providing plentiful insect food.

Birds which feed from open ground increased significantly in abundance, as a group, after the fire. Some species not recorded pre-fire were found for two or three years post-fire then disappeared again: Flame Robins, Scarlet Robins, and Buff-rumped Thornbills declined as shrub regrowth reduced the availability of open ground. Superb Fair-wrens continued to increase over the three year post-fire period – they “thrived after fire in all habitats … where they were able to exploit the low shrub regrowth that developed when the canopy was opened” (Loyn 1997:228).

Large hole nesters, mostly seed-eaters, declined after fire to 30% of pre-fire levels, stayed scarce for the first two years post-fire, then increased to pre-fire levels in the third year. Resident carnivores showed few changes as a group, though Laughing Kookaburras were most numerous one and two years post-fire when there was plenty of open ground. There was a “small influx” of birds of prey during and after the fire.

Honeyeaters virtually disappeared after fire, due to loss of their normal food sources: concentrations of eucalypt and shrub blossom. Abundance of species in this group increased with time-since-fire, reaching 60% of pre-fire levels by three years post-fire. Some taxa such as Yellow-faced and Lewin’s Honeyeaters had fully recovered by this time, while others, including Crescent and New Holland Honeyeaters were still scarce. These results differ from those of Smith (1989), who found these latter two species in considerably greater abundance in Winter and Spring surveys in the year after the fire, relative to pre-fire surveys in those seasons, probably due to flowering in the unburnt canopy of trees and gully shrubs.

It is interesting that many of the species which declined most after fire were mobile species of around honeyeater size. Loyn (1997) considers these birds well equipped to avoid immediate death in flames and smoke. Their problem was that their food sources, particularly nectar, were virtually eliminated. “This suggests an evolutionary history of adaptation to fire-prone environments. Species exploiting ephemeral or patchy food supplies need to be mobile and have intrinsic capacity to escape fire by flight: fire may be one of many factors that make those food supplies ephemeral or patchy. Conversely, weak-flying species can only survive where food supplies are resilient to major disturbances such as bushfire, or in habitats where such disturbance does not occur” (Loyn 1997:232). The discrepancy between the findings of Smith (1989) and Loyn (1997) accords with this interpretation, which also echoes the discussion of ‘explorer’ Dunnarts above (Lunney and Leary 1989; Section 4.2.3).

4.2.8 Conclusion

The rich array of studies of fire and animals in South Coast Dry Sclerophyll Forests tell a tale of resilience, and of a wide range of responses to time-since-fire – themes familiar from the discussion of South Coast Heath (Section 2.2).

There is no doubt that many species depend on the thick understorey which develops some years after a fire and persists for at least two decades. A need to shelter from predators appears to be a major driver of this habitat preference. One wonders to what extent the presence of feral predators – dogs, foxes and cats – has increased the need for cover over what pertained prior to European settlement. Certainly control of exotic predators needs to go hand in hand with fire management, to minimise post-fire vulnerability of native mammals large, medium and small.

There are also species which thrive in the years following a fire and decrease in abundance, or even disappear, as time-since-fire progresses. Some animals, including macropods, Dunnarts, some lizards and a number of birds, need open and/or grassy habitat which is generally associated with early post-fire regeneration or, as we have seen in previous sections, with environments which are burnt relatively frequently. Others thrive in a post-fire environment rich in new plant growth: “The decomposition of dead plant material releases nutrients slowly, but mature forests are less productive than young, vigorously growing plant communities… Wildfire massively releases these bound up nutrients and, as seen at Nadgee, there is a tremendous surge of life. Animal and plant populations explode and species diversity increases. In a way, fire is a renewal of the ecosystem” (Recher *et al.* 1975:163).

How are the different habitat requirements of the many fauna species in South Coast Dry Sclerophyll Forests to be accommodated? Here again we see the need for variability in time-since-fire, and probably also in fire frequency. Variable intervals between fires within the statewide thresholds for dry sclerophyll forests, seven and thirty years, should deliver a range of habitats including lots of dense vegetation, areas where new post-fire regeneration is available and areas where fires are more, and less, intense. Where native vegetation covers large areas it is likely that wildfire will fulfil this prescription. Where remnants have been cut off by clearing and urbanisation, or where fire suppression has been unusually effective, some ecological burning may need to occur. Recher *et al.* (1975:163) suggest that when planned fire is added to the wildfire regime, it should be done in a creative, rather than in a “monotonous and unimaginative” way. Hot fires have their place, as well as cool winter burns.

A further point which emerges from the South Coast forest studies is the importance of topography in providing refuge areas from which re-colonisation of the post-fire environment can occur. Not only do unburnt areas serve this function, places where fire is less severe also play this role. When thinking about the effects of fire and how best to manage it, it is instructive to consider landscape patterns: how does vegetation change with topography? How does topography affect how fire moves across the landscape? How does fire intensity vary over space and how does this enable plant and animals species to survive and thrive together in a fire-prone environment?

5 Fire in wet sclerophyll forests

"The giants of the Australian bush are the towering eucalypts of the wet sclerophyll forests" (Keith 2004:58). Found in areas of high rainfall on moderately fertile soils, these forests are characterised by a canopy of tall, straight-trucked eucalypts over a soft-leaved understorey of shrubs, herbs and ferns. Located floristically and often physically between rainforests and the woodlands and dry sclerophyll forests which occur where rainfall and/or soil fertility are lower, the wet sclerophyll forest understorey may be more, or less, shrubby. Keith places both the Southern Rivers forest classes discussed below in the grassy subformation of wet sclerophyll forests (Keith 2004).

"The wet sclerophyll forests not only tolerate but seem to promote one of the most fearsome and catastrophic ecological disturbances on earth – crown forest fires" (Keith 2004:59). Though devastating to humans, these fires play an essential role in forest renewal, allowing the light these trees need to regenerate to reach ground level, and holding back "the rising tide of mesophyllous plants" which would otherwise see succession to rainforest (Keith 2004:59).

Much of what we know about fire in wet sclerophyll forests comes from over 50 years of research in high altitude forests of Mountain Ash (*Eucalyptus regnans*), a species which occurs in Victoria and Tasmania but not in NSW. The ecology of this species has much in common with Alpine Ash (*E. delegatensis*), the dominant species of the Montane Wet Sclerophyll Forests discussed in Section 5.1. More recently studies in the rather different wet sclerophyll forests of Northern NSW have added to our understanding; this work will inform Section 5.2 on Southern Lowland Wet Sclerophyll Forests.

5.1 Montane Wet Sclerophyll Forests

5.1.1 Introduction

On steep slopes between 800 and 1400 m amongst winter mists, frosts and snow, grow Montane Wet Sclerophyll Forests. Ash eucalypts dominate these forests, which are found south from the Brindabella Range near Canberra, blanket the western and southern fall of the Kosciuszko plateau and extend into Victoria. The dominant tree species in this region is the Alpine Ash *E. delegatensis*, which occurs either in pure stands or with *E. dalrympleana* (Mountain Gum) or *E. pauciflora*. Patches of Montane Wet Sclerophyll Forest are also found on the eastern escarpment and in sheltered coastal mountain valleys; here the dominant is *Eucalyptus fraxinoides* (White Ash). Soft and hard-leaved shrubs including *Acacia dealbata*, *Bossiaea foliosa*, *Daviesia latifolia*, *Leucopogon lanceolatus*, *Coprosma hirtella*, *Olearia megalophylla*, *Polyscias sambucifolia*, *Tasmannia lanceolata* and *T. xerophila* form a variable shrub layer over a ground cover of ferns, herbs and grasses, including *Poa ensiformis* and *Poa helmsii*. Steep terrain has largely protected these forests from clearing and logging, however some have been grazed (Keith 2004).

The vegetation groups identified by Gellie (2005) which most closely match this description are:

- Group 87 – Western Escarpment Moist Shrub/Herb/Grass Forest. Co-dominated by *Eucalyptus delegatensis* and *E. dalrympleana* with an open shrub layer of

Polyscias sambucifolia, *Daviesia latifolia*, *Cosprosma hirtella* and *Tasmannia xerophila*, these forests, which are virtually uncleared, cover 76,000 ha. The grasses *Poa ensiformis*, *Elymus scaber* and *Poa helmsii* are interspersed with a wide range of forbs. Eighty percent of this vegetation group is in dedicated reserves.

- Group 86 – Western Sub-alpine Moist Shrub Forest. Dominated by *E. delegatensis* with *E. pauciflora* often forming a secondary tree layer, shrub cover in this vegetation group varies from sparse to dense. Shrub species include *Tasmannia lanceolata*, *Polyscias sambucifolia*, *Coprosma hirtella*, *Lomatia myricoides* and *Olearia megalophylla*. Ground species include *Poa ensiformis* and various herbs. This vegetation group covers an estimated 22,900 ha, has not been cleared, and is 89% reserved.
- Group 61 – Southern Escarpment Edge Dry Shrub Forest is dominated by *E. fraxinoides* with an open shrub layer containing *Leucopogon lanceolatus*, *Acacia obliquinervia* and *Platysace lanceolata*. A sparse, species-poor ground layer features *Dianella tasmanica* and *Stylidium graminifolium* with scattered tussocks of *Poa meionectes*. This vegetation group covers 10,500 ha, is uncleared, and is 78% reserved.

Gellie (2005) categorises all three vegetation groups as Ash Eucalypt Forests, which he describes as “typical fire sensitive montane tall forests, which can succumb to stand replacing fires at intervals as short as 40 to 60 years. Because of local site conditions, Ash Eucalypt Forests are fast growing and highly productive in their respective montane and sub-alpine environments” (Gellie 2005:240).

5.1.2 Obligate seeder eucalypts

Eucalypts are renowned for their ability to resprout, from lignotubers at ground level and from epicormic shoots on trunks and branches. A few eucalypt taxa, however, do not conform to this pattern; chief amongst them are the ash species *E. delegatensis*, *E. fraxinoides* and *E. regnans*. The response of these mavericks to fire is that of the classic obligate seeder with on-plant seed storage (Sections 1.2 and 2.1.2); adults receiving 100% leaf scorch are killed by fire and regeneration relies on seed released from fruits high in the canopy.

What do we know about the fire ecology of obligate seeder eucalypts? Some research into *E. delegatensis* has been carried out – mostly in Tasmania (eg Bowman and Kirkpatrick 1986a, Ellis 1985) but also in the ACT (O’Dowd and Gill 1984), while as far as I am aware *E. fraxinoides* is yet to be studied. In Victoria extensive work into the ecology of ash eucalypts has focussed on *E. regnans*.

Regeneration of *E. regnans* in the absence of fire has been researched in detail by Victorian ecologist David Ashton. In one early study – started in 1949 – both natural regeneration and seedlings arising from sown seed were followed in three sites, one a 50 m gap created by the fall of a number of trees, one a site with an intact canopy of mature *E. regnans* from which the understorey was removed, and the third a mature forest site with both tree canopy and understorey intact (Ashton and Chinner 1999). While seedling survival was better in the two more open sites than under full canopy, no

E. regnans seedling in any site survived beyond 13 years, and very few beyond six years. By two years after germination all seedlings in the intact forest were dead. Many factors contributed to the total failure of *E. regnans* to regenerate, including fungal attack, predation by insects, browsing by mammals, being smothered by litter, competition from dense understorey, desiccation due to the transpiration demands of established plants, and lack of light particularly as saplings developed pendulous foliage.

Studies of regeneration in Tasmanian *E. delegatensis* forests suggests that the controls on between-fire establishment are somewhat more relaxed there than in Victoria's Mountain Ash country: Bowman and Kirkpatrick (1986a) found *E. delegatensis* seedlings and saplings growing in unburnt areas. Adult plants were, however, very effective in suppressing young plants, largely through their influence on soil moisture (Bowman and Kirkpatrick 1986b,c). Bowman and Kirkpatrick (1986a) conclude that few saplings survive to join the adult population, and that "natural regeneration of *E. delegatensis* in dry forest appears to be largely dependent upon fire" (Bowman and Kirkpatrick 1986a:71).

On the other hand, the link between crown fire and regeneration of ash eucalypts has been apparent for many years (Ashton 1981 and references therein).

> "Crown fire eliminates most of the difficulties encountered by *E. regnans* regeneration by causing mass seedfall, creating a suitable seed bed and by temporarily changing the microclimate, soil moisture regime, soil nutrient status, microbial populations and ant foraging activity" (Ashton and Chinner 1999:265).

O'Dowd and Gill (1984) compared the behaviour of *E. delegatensis* on two sites in the Brindabella Ranges in the ACT, one burnt in an experimental fire which killed 93% of the Alpine Ash trees, one unburnt. In the unburnt site ants remove a large proportion of the small number of seeds which were released through the activities of Gang Gang Cockatoos in the canopy, and only a handful of seedlings emerged – none of which survived beyond the cotyledon stage. In the burnt site, however, many seedlings established.

Both *E. regnans* and *E. delegatensis* form even-aged stands which can be dated to severe bushfires, such as that in 1939 (Gilbert 1959, Ashton 1976, 1981; O'Dowd and Gill 1984, McCarthy and Lindenmayer 1998). Even when a wildfire does occur, regeneration of *E. regnans* is limited to areas where all, or almost all adult trees have been killed (Ashton and Martin 1996a,b). Similarly, Ellis (1985) found that post-fire *E. delegatensis* seedlings growing under a canopy of living mature trees in Tasmania did not develop, while in the ACT after the 2003 fires Vivian (2005) and Larkin (pers. comm. 2006) found that while *E. delegatensis* seedlings germinated abundantly after both low and high intensity fires, seedling height was considerably greater where burns had been severe.

If a fire kills some but not all trees in a patch, stands containing trees of more than one age class can result (Gilbert 1959, Ashton 1976, Bowman and Kirkpatrick 1984, Ellis 1993). For example a stand of *E. delegatensis* trees studied by O'Dowd and Gill (1984) in the Brindabella Ranges contained two age classes stemming from 1951-2 and 1939, both severe fire seasons in the ACT. Different age classes may also be dispersed through the landscape (Gilbert 1959) because even severe fires of considerable extent are patchy – see Ashton (1981) re 1939 fires, and Smith and Woodgate (1985) re 1983 fires. Doherty and Wright (2006), who assessed regeneration in the northern reaches of the Alps after the widespread fires of early 2003, found that while approximately 30% of *E. delegatensis* stands in the study area had been killed, the majority had survived. Bowman and Kirkpatrick (1986a) also report considerable post-fire survival of mature *E. delegatensis* in Tasmania.

After a 'stand-replacing' fire seed is released at unprecedented rates, temporarily overwhelming the seed-collecting capacities of ants (O'Dowd and Gill 1984). Rain provides the impetus for germination, and seedling density can be very high (Gilbert 1959, O'Dowd and Gill 1984, Ashton and Martin 1996a). Seedlings self-thin as they develop (Gilbert 1959, O'Dowd and Gill 1984, Ashton 1976, Ashton and Martin 1996a). Shoot growth is slow at first but increases rapidly (Ashton 1975), allowing *E. regnans* to reach 7 m in height in seven years (Ashton and Martin 1996a). *E. delegatensis* also grows rapidly: Doherty and Wright (2006) report that by 12 months after the 2003 fire some seedlings were already over 1.5 m high. Mountain Ash stands pass through a very dense thicket stage, and a stage when suppressed trees destined not to survive long-term are still living beneath the canopy, again creating a very dense forest. Canopy cover reaches a maximum at about 40 years post-fire, opens out somewhat as the stand matures, and continues to decline slowly but steadily to at least 220 years (Ashton 1976).

Like other obligate seeders with canopy-stored seed, ash eucalypts are vulnerable at each end of the fire frequency spectrum. A tree-killing fire within the juvenile period of these species will eliminate them, as there will be no seed available for regeneration. The juvenile period of *Eucalyptus regnans* is considered to be 20 years (McCarthy *et al.* 1999), while the NSW Fire Response Database (DEC 2002) gives a figure of 10 years for *E. delegatensis*. Gilbert (1959) reports that two fires 14 years apart in Tasmanian wet sclerophyll forest produced a dense fern thicket, from which a few sparse eucalypt seedlings struggled to emerge.

A very long period without fire also threatens these species. With respect to Mountain Ash, Ashton and Chinner (1999:278) consider that "If, by chance there is freedom from fire for another 2-3 centuries it is highly likely that the readily dispersible *Atherosperma moschatum* will invade from the gullies and form a type of rainforest with some of the more shade-bearing wet sclerophyll species such as *Olearia argophylla* and *Hedycarya angustifolia*." *E. regnans* trees are believed to live for at least 300 years (McCarthy *et al.* 1999). Given their inability to regenerate without a fire, trees left unburnt for several centuries are unlikely to replace themselves before they die. Studies have shown that the seed of these giants does not survive in the soil (Ashton 1979), so if a fire occurs after adults have gone from a site, there will be no capacity for them to regenerate.

Research in Tasmania suggests that in at least some circumstances, *E. delegatensis* trees in long-unburnt environments may be at risk of dying well before their potential lifespan, which DEC (2002) lists as 400 years. Ellis (1985) investigated vegetation patterns on a cool, moist, elevated plateau in the north of the state, using ring counts to determine tree ages and fire history. He concluded that while the entire area could potentially support rainforest, prior to European settlement fire had allowed *E. delegatensis* forest, and in some areas grassland, to survive. Long interfire intervals over the last 150 years were associated with the development of an increasingly mesic understorey of rainforest species, particularly *Nothofagus cunninghamii*. In many places where this transition was occurring, *E. delegatensis* trees were either dead or dying; growth curves developed from ring counts indicated that these trees had grown normally for 50-60 years after fire, then declined over 20 to 25 years before dying. Ring counts on fire scars found on dead eucalypts in rainforest stands suggested that prior to 1850, fires occurred at intervals of 15-50 years.[10] In NSW, Keith (2004) also notes the tendency for cool temperate rainforest to invade eucalypt forest, but points out that *Nothofagus* is not found in southern NSW, and that patches of cool temperate rainforest are few. The tendency for Montane Wet Sclerophyll Forest to succeed to rainforest is therefore likely to be much more limited than in Ellis's sites, and the need for fire to truncate this succession in order to keep *E. delegatensis* trees healthy is also less likely to be important. Ellis's work does, however, suggest that understorey composition and associated soil characteristics may influence survival of Alpine Ash, and that these factors should not be ignored when considering the impacts of long interfire intervals (Ellis and Pennington 1989, 1992).

5.1.3 Understorey dynamics

Three facts stand out from research into the effects of fire on the understorey of *E. regnans* and *E. delegatensis* forests. First, fires can, and do, occur in the understorey without impacting the tree canopy. Second, unlike the tree species, understorey plants in these forests often store seed in the soil. Third, these understoreys are dynamic: they change with time-since-fire and with fire frequency.

Many authors attest to the fact that fire can burn through the ash eucalypt forest understorey without unduly affecting adult trees. When this has occurred, understorey plants will be of a younger post-fire age than the trees above them (McCarthy *et al.* 1999). In a series of stands near Wallaby Creek, the site of much of Ashton's work, "the ages of *E. regnans* ranged from 15 to 220 years, whereas those of *P. aspera* [understorey shrub *Pomaderris aspera*] ranged form 15 to 75 years" (Ashton 1976:411). In Tasmania, Bowman and Kirkpatrick (1986a) studied fires in which adult *E. delegatensis* trees survived but mortality in the understorey was high, while Ellis (1985) identified understorey fires as a vital element in sustaining *E. delegatensis* forests on the Mt Maurice plateau. After the 2003 Kosciuszko fires Miles and Cameron (2006:23) found places where *E. delegatensis* had been killed, and others "where the understorey had been consumed with no canopy scorch." Ashton and Martin (1996b) noted considerable differences in the post-fire composition of the understorey in a site where only the understorey was burnt but the canopy remained alive, and a nearby site where intense fire had destroyed both canopy and understorey.

Many species in the understorey of ash eucalypt forests store seed in the soil (Ashton 1979, 1981). While the seeds of some species appear to react to fire cues (Ashton 1979), most if not all also respond to other forms of disturbance that allow light to reach ground level. Ashton and Chinner (1999), whose study did not involve fire, found prolific germination of both herbaceous and woody understorey species in two sites where canopy and/or understorey cover had been removed, while very little germination occurred under mature forest with trees and shrubs intact. After fire many species in the *E. regnans* understorey regenerate only from seed; this is particularly the case for

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[10] For further insight into the wet sclerophyll/rainforest dynamic from a vital attributes perspective see Noble and Slatyer (1980), who use observations in Tasmania to demonstrate the explanatory power of their model.

woody species (Ashton 1981). A number of shrubs and small trees, including *Olearia argophylla*, regrow from lignotubers.

Changes in understorey composition with increasing time-since-fire have been documented in *E. regnans* forest in Victoria, and in *E. delegensis* forest in Tasmania. In neither case is understorey composition quite the same as that described by Keith (2004) and Gellie (2005) for Montane Ash Wet Sclerophyll Forests in NSW, however there is much common ground.[11]

The early post-fire years are characterised by "a luxuriant growth of herbs" (Ashton 1981:353). The open conditions created by fire provide a window of opportunity for species which do not thrive in the later post-fire years. In *E. regnans* forest studied by Ashton and Martin (1996a) four new species appeared soon after fire, three of which persisted only 1-2 years. Understorey plants common in the first couple of post-fire years in Tasmanian ash forests include *Senecio* daisies and the Scrub Nettle *Urtica incisa* (Gilbert 1959). Miles and Cameron (2006) report that the rare perennial forb *Halorgis exalata* ssp. *exalata* regenerated massively in *E. delegatensis* forest after the 2003 Kosciuszko fires, flowering and fruiting profusely within the subsequent two years. Seedlings of this species were abundant where the eucalypts had been killed, and perhaps even more so where the canopy was intact but fire had burnt through the understorey. Ashton (1981) notes that plant species diversity tends to be greatest in the first post-fire decade.

Very quickly, however, shrubs which germinated after the fire become large enough to take over (Ashton and Martin 1996b). There is a progression in shrub species dominance as time-since-fire goes by. While to some extent this sequence reflects the different growth rates and life-spans of the species involved, it also owes something to differential ability to recruit between fires.

Several researchers have documented understorey composition 20 to 25 years post-fire. Middle-sized, relatively short-lived shrubs such as *Cassinia aculeata*, *Goodia lotifolia* and *Olearia lirata* are often present (Ashton and Martin 1996b). In *E. regnans* forests *Pomaderris aspera* has frequently gained dominance by this stage, and may be very thick (Ashton 2000). *Acacia dealbata*, *Olearia argophylla* and *Prostanthera lasianthos* are often important understorey components (Gilbert 1959, Ashton 2000). Once these shrubs mature their cover is very high; when added to the cover of mature crowns of *E. regnans* this produces a total forest cover of 75%, the equivalent of rainforest (Ashton 1976).

By 50 years post-fire the short-lived Asteraceous shrubs have disappeared and the density of *Pomaderris aspera* is declining (Ashton 2000). *Acacia dealbata* trees have emerged above the *Pomaderris* thicket and form a sparse subcanopy beneath the eucalypts (Ashton 2000). *Prostanthera lasianthos* dies out at around 60-70 years (Ashton 2000), *Pomaderris* at about 100, although it declines slowly from about 40 years post-fire (Ashton 1976, 2000). Gilbert (1959) estimates that few *Acacia dealbata* trees survive more than 70 years. While some shrubs are capable of between-fire recruitment, in most species this is limited and by 50 to 70 years post-fire the shrub

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[11] In fact, of the Wet Sclerophyll Forest vegetation classes described in Keith (2004), the one whose understorey appears to have most in common with the *E. regnans* understorey described by Victorian researchers is Southern Escarpment Wet Sclerophyll Forest. However only one of the five eucalypt species listed by Keith (2004) as indicative of this vegetation class acts as an obligate seeder in NSW (DEC 2002).

layer has often opened out leaving a dense understorey of ferns (Ashton 1976, 2000). Understorey species whose abundance increases in long-unburnt *E. regnans* forest include *Coprosma quadrifida* and the Rough Tree Fern *Cyathea australis* (Ashton 1976, Ashton and Martin 1996b).

Rainforest species may also increase in long-unburnt ash forests. This appears to occurs more rapidly in Tasmanian Alpine Ash forests than in ash forests in Victoria or NSW (Gilbert 1959, Ellis 1985). However even in Victoria where temperate rainforest patches are uncommon, rainforest species increase with time-since-fire, particularly where rainfall is high and/or seed is readily available from rainforest gullies (Ashton 1981, 2000).

Ashton (2000) notes that when a fire does come along after a long interval, many species will regenerate from soil-stored seed, but for some the period of soil storage may be too long. As is the case everywhere, the longevity of soil-stored seed is very poorly known. Some species appear to have the ability to survive as soil-stored seed for a very long time: Gilbert (1959) suggests that this must apply to *Acacia dealbata*, which appears after fire when even when stands of very old trees are disturbed.

Few studies have specifically addressed the effects of different fire frequencies on the ash wet sclerophyll forest understorey, however observations by Gilbert (1959) in Tasmania, and Ashton (2000) in Victoria, provide some clues. Gilbert (1959) describes an area which had been burnt by three or four understorey fires in the last 100 years. Here, the mesophyll understorey characteristic of Tasmanian forests was replaced by *Acacia dealbata, Olearia argophylla* and *Pomaderris aspera*. Ashton (2000) lists species in three understorey types arising from three different fire histories – one from a series of closely-spaced understorey fires, one from an understorey fire after a 28 year inter-fire interval, and one from a single intense stand-replacing fire 50 years previously. Shrub diversity was somewhat higher in the two more frequently burnt sites; grasses and forbs were most diverse in the very frequently burnt site, species richness of mosses and liverworts was highest in the site resulting from the stand-replacing fire. While at 25 years post-fire the obligate seeder *Pomaderris aspera* dominated the site with the 28 year interval, shrubs in the frequently burnt site were mostly lignotuberous, with *Olearia argophylla* and *Bedfordia arborescens* particularly well-represented. It seems likely that there are different understorey ‘states’ associated with fire frequency, as well as with time-since-fire, in these forests.

To what extent are the understorey patterns described above likely to pertain in the Montane Wet Sclerophyll Forests of NSW? These forests are more grassy than the Victoria’s *E. regnans* forests, and also contain more sclerophyll shrubs. They do, however, include a soft-leaved shrub component. It is likely that the Montane Wet Sclerophyll Forest understorey also regenerates abundantly after fire, and that there is some sort of progression in the dominance of shrub species. The ‘opening out’ seen in the later post-fire decades may also occur in this ecosystem. Different fire frequencies are likely to favour different understorey components, as they do in Tasmania and in Victoria. Moderately frequent fire would probably favour sclerophyll shrubs, while longer intervals may encourage the more mesic species. Carr and Turner (1959) report an increase in shrubs in *E. delegatensis* forests below the Bogong High Plains, a development they ascribe to regularly burning by graziers. It is hard to see how this would work if fires were very frequent, as obligate seeder shrubs would be eliminated by fires at intervals below their juvenile period, and resprouter seedlings would be unable to reach fire tolerance. However it is possible that moderately frequently burning would have this effect. From a conservation point of view, a mix of shrubby and grassy patches is probably desirable.

5.1.4 Forest fauna

Fauna research in ash eucalypt ecosystems again has focussed more on Mountain Ash habitats than on *E. delegatensis* forests. Studies either compare forests of different ages, or the same sites before and after fire.

Patterns of faunal succession after fire in *E. regnans* forests have much in common with those in South Coast Heaths (Section 2.2.1) and South East Dry Sclerophyll Forests (Section 4.2). Studies have identified a small number of early successional species; these animals generally need open habitat for one reason or another. The swift growth of trees and shrubs in Mountain Ash forests is reflected in a rapid increase in populations of species which prefer dense vegetation. Most of these species remain stable inhabitants of the forests as they age to over 200 years, although some decrease in abundance as the shrubby understorey declines and the forest opens out. Interestingly, these fauna studies highlight the habitat benefits of a mix of age classes: both old trees with hollows, and the dense shrub understorey characteristic of young stands (Section 5.1.3), play a vital role in providing the range of habitat features which maximises mammal and bird diversity.

Macfarlane (1988) studied mammals in Victorian Mountain Ash forests of different ages. He found that the presence and abundance of animals varied with forest age, but not dramatically so except in the very early years of regeneration, when many species were not present. Small mammals which thrive in dense habitat in other vegetation formations – Bush Rats, Brown and Dusky Antechinus – recolonised regenerating forest rapidly and increased in abundance as the forest matured (as in other ecosystems, the Dusky Antechinus took longest to return). Most arboreal mammals needed a well-developed canopy and large old trees – dead or alive – with hollows for nesting. Ringtail and Leadbeater’s possums also needed a well-developed middle stratum and so were not found in the oldest stands where shrubs had mostly died out. The greatest abundance and diversity of mammals was found in multi-aged forests which had both young and old Mountain Ash trees plus dense middle and low shrub strata with *Acacia*, *Pomaderris*, *Correa* and *Olearia*. This high structural diversity was often the result of a low intensity fire which had regenerated the understorey while leaving many old trees intact. Diverse habitat features were also found at the interface between stands of different ages.

The importance of the shrub layer for Mountain Ash forest birds is attested by the work of Loyn (1985), who found that just over half the birds in the *E. regnans* forest sites he studied fed from the foliage or blossom of shrubs and mid-storey trees, with another 19-25% feeding from the ground or low understorey. Birds found only or mostly in early regeneration included open patch foragers such as the Flame Robin, Superb Fairy-wren and Red-browed Firetail. The composition of the bird community showed few changes between 39 and 200 years post-fire, although a few successional changes relating to shrub abundance were noted. Hole-nesting birds were more common in forest older than 100 years, than in younger forest. Dead stags of old trees were important for hole-nesting birds while the next generation of trees was growing up.

A second study by Loyn (1998) compared patches of old-growth forest with 57-year-old regrowth from the 1939 fires. No significant differences in bird abundance, either overall or for different groupings of birds, were found. While a few individual species favoured one age class or the other, most found equally suitable habitat in each. Few birds that forage or take seed from open ground were found in this study, presumably because both 57 year regrowth and long-unburnt forest were too thick to meet their needs. While wet gullies were generally well-used by birds, some shrub foragers avoided the wettest gullies where cool temperate rainforest plants prevailed.

Brown and Nelson (1993) studied lizards in *E. regnans* forests in Victoria. Three species were relatively common. All responded to light, with two of the three species most commonly observed where light levels were high; these species needed patches of sun for basking. Both were most abundant on recently disturbed sites which were still relatively open, were not found in sites 11-63 years old which carried very dense vegetation, then reappeared in older sites (140 years plus) where trees and shrubs had thinned.

Because the Alpine Ash forest understorey is more grassy and has a greater sclerophyll component than that of Victoria's Mountain Ash forests, one might expect the effects of fire in Keith's Montane Wet Sclerophyll Forest class to reflect aspects of both the *E. regnans* fauna story, and elements of that found in dry sclerophyll forests and woodlands. Two studies from the ACT suggest this may indeed be the case. Both used data from long-term bird surveys to compare bird abundance in the years before and after fire.

Catling and Newsome (1981) report data on eighteen bird species from a long-term study in *E. dalrympleana* forest in mountains west of Canberra: while *E. dalrympleana* is not an obligate seeder eucalypt, it co-occurs with *E. delegatensis* in Montane Wet Sclerophyll Forest (Keith 2004, Gellie 2005). Data from the years before a high intensity fire was compared with post-fire data; there had been no fire in the area for many years before this burn. No species disappeared or appeared, however abundances changed for nine species, generally towards greater abundance after the fire. Ground foragers were particularly affected, with two species quadrupling their abundance in the first post-fire year. Overall, in the first five years after fire numbers increased by 48% on the forest floor, 24% in the shrub layer and 19% in the canopy. As time-since-fire increased, numbers of birds declined, particularly on the forest floor. Catling and Newsome (1981:295) conclude that "Fire appears to have produced favourable habitat for the birds, but habitat near the ground appears to require fire more often than the canopy habitat to be suitable."

Baker *et al.* (1997) used data from another long-term study in wet sclerophyll forest in the Brindabella Ranges to compare bird populations in the years before and after an understorey fire which followed a 46 year interfire interval. Wildfire significantly affected the population size of 13 of the 20 species studied: seven species including the Rose Robin, White-naped Honeyeater and Rufous Fantail were more abundant prior to the fire, while six including the Flame Robin, Superb Fairy-wren and White-browed Scrubwren were more abundant after it. An interesting finding was that the fire had very little effect on bird survival, implying that changes in population numbers were probably due to movement of birds between burnt and unburnt patches.

5.1.5 Conclusion

The fire interval range recommended by DEC for grassy subformation wet sclerophyll forest is 10 to 50 years, with the proviso that “Occasional intervals greater than 15 years may be desirable,” and that “Crown fires should be avoided in the lower end of the interval range” (Kenny *et al.* 2004:34).

From the information reviewed above, this range would certainly be very low for Victoria’s Mountain Ash forests, and of course as the guidelines proposed by Kenny *et al.* (2004) are only for NSW, this is not being suggested. McCarthy *et al.* (1999), who used a range of methods to determine the frequency of fires in Mountain Ash forests, estimated that the current interval between “tree killing” fires is 75 to 150 years, while if fires of any severity are considered (as we have seen not all fires kill trees), then the mean interval is 37-75 years. Even these intervals are probably somewhat higher than the ideal. For one thing, it is vital that some Mountain Ash trees survive well over 150 years, as it is only at this stage that the hollows essential to many fauna species begin to form in earnest (Lindenmayer *et al.* 1993 and references therein).

As already noted, the understorey of the Montane Wet Sclerophyll Forests in NSW is grassier than that of Victoria’s Mountain Ash forests, and contains more sclerophyll shrubs. Shorter intervals are therefore likely to be appropriate: sclerophyll shrubs such as wattles, peas and epacrids are more likely to be short-lived than the Mountain Ash shrub dominants which are known to live for over 50 years, while grasses and herbs are more at risk of being suppressed beneath dense shrubs and litter than are larger plants. The existence of forbs such as *Haloragis exalata* which thrive in the early post-fire years (Section 5.1.3), suggests that historically, intervals between fires must not have been too long – although Miles and Cameron (2006) suggest that this species may survive long fire-free periods through occasional recruitment in patches subject to other forms of disturbance. The increased use by quite large numbers of bird species, particularly ground foragers, of the post-fire environment in wet sclerophyll forests in the ACT (Catling and Newsome 1981, Baker *et al.* 1997*)* also suggests intervals were probably shorter here than in *E. regnans* forests, where this pattern of usage is not found (Section 5.1.4). And while the need for periodic fire to minimise rainforest takeover of the understorey, and ensure *E. delegatensis* trees do not succumb to dieback, is unlikely to apply in NSW to anything like the extent it does in Tasmania, again there are hints that intervals between fires should not extend too long. A frequency of 10-50 years might therefore cause few problems in the Montane Wet Sclerophyll Forest understorey.

For the obligate seeder trees of the overstorey, however, 10-50 years would be extremely short. While Alpine Ash grows rapidly and produces seed by 10 years post-fire (DEC 2002), several more years may be needed before canopy seedbanks are sufficient for replacement after a second fire. Kenny *et al.* (2004) address this issue through the caveat that crown fires should be avoided at the lower end of the interval range. How easy it would be to ensure fires were restricted to the understorey in ten-year-old stands of *Eucalyptus delegatensis*, is a topic worthy of study: presumably the separation between understorey and canopy is less complete in the early post-fire years and decades than it is later. Perhaps ‘stand-replacing’ fires should be followed by a fire-free period of somewhat more than 10 years.

The desirability of crown fires at the *upper* end of the interval range is also debatable, given that canopy fires are ‘tree killing’ in Montane Wet Sclerophyll Forests. Forty to

50 years would constitute a very short life-span for a eucalypt which could potentially live for 400 years. The issue of tree hollows which is so important in Mountain Ash forests may also be relevant here. It is likely that centuries-old Alpine Ash trees provide essential fauna habitat in Montane Wet Sclerophyll Forests: Lindenmayer *et al.* (1993) found the number of hollows in *E. delegatensis* trees increased as trees enlarged in diameter.

The answer to these apparent contradictions may lie in the degree to which fires in Montane Wet Sclerophyll Forests impact the canopy: perhaps in this vegetation class it is desirable that most fires are of relatively low intensity. This would allow obligate seeder canopy trees to survive for one, two or three centuries. Low intensity understorey fires at variable intervals within DEC thresholds, perhaps with an emphasis on the upper half of the range, would provide periodic opportunities for shrub and herb regeneration, and create a variety of time-since-fire-mediated understorey structures and resources (such as flowering shrubs) for an array of bird and fauna species. Some canopy impact by a subset of fires could have habitat benefits, as disturbance appears to play a role in hollow formation. Lindenmayer *et al.* (1993) found greater numbers of hollows in old trees within young stands, and in trees on steep slopes; the authors speculate that in both cases non-lethal damage by fire may have accelerated hollow formation.[12]

Low intensity understorey fires may also have a role to play in limiting the extent and/or impact of wildfire. As we have seen even extensive, high intensity fires kill only a proportion of *E. delegatensis* trees; it would be instructive to see whether there is any relationship between understorey age and the extent of obligate seeder tree death from wildfire in Montane Wet Sclerophyll Forests. Another matter which warrants investigation is the relationship between understorey fire frequency, wildfire intensity, speed and nature of understorey regeneration, and post-fire erosion. This issue may be particularly important in *E. delegatensis* forests which often grow on erosion-prone steep slopes (Good 1996). Although maintenance of a heavy litter load on the forest floor may be an excellent hedge against erosion (peak fuel loads in long unburnt Alpine Ash forests average 37 to 44 tonnes per ha), in a fire-prone environment there will inevitably be occasions when fire will remove litter and understorey cover. Erosion is most likely to be problematic when fires are intense (Adams and Simmons 1996, Prosser and Williams 1998, DeBano 2000, Dragovich and Morris 2002); in Montane Wet Sclerophyll Forests this will most likely occur during extensive ‘tree killing’ wildfires. Does a regime of low-intensity understorey fires mitigate damage in these circumstances, for example by encouraging thick grasses which bind the soil post-wildfire? Or do they simply add to the problem?

Of course, occasional high intensity fire, perhaps at a scale of centuries rather than decades, will be essential to ensure the obligate seeder eucalypts which dominate this vegetation class have the opportunity to regenerate before adults die.

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[12] See Lindenmayer *et al.* (1993:97-98) for a discussion of the role of fire in the creation, and destruction, of tree hollows.

5.2 Southern Lowland Wet Sclerophyll Forests

5.2.1 Introduction

Southern Lowland Wet Sclerophyll Forests inhabit a very different environment to the Alpine Ash forests discussed in Section 5.1. These are low altitude forests, growing along the coast and adjacent hills up to 300 m above sea level. Soils are moderately fertile loams derived from siltstones and mudstones. Most areas occupied by this forest type are dominated by *Corymbia maculata* (Spotted Gum) and/or the rough-barked eucalypts *Eucalyptus globoidea* (White Stringbark), *E. paniculata* (Grey Ironbark), *E. longifolia* (Woollybutt), *E. pilularis* (Blackbutt), *E. scias* (Large-fruited Mahogany) and *Corymbia gummifera* (Red Bloodwood). In wet, sheltered areas these forests may adjoin "tiny embedded patches" of Southern Warm Temperate Rainforest. Common trees in sheltered gullies are Sydney Blue Gum (*Eucalyptus saligna*), Bangalay (*Eucalptus botryoides*), and hybrids of these two species. *Syncarpia glomulifera* (Turpentine) may form a small tree layer (Keith, 2004:72-3).

Keith (2004:72) places these forests in the grassy subformation of wet sclerophyll forests, although they often also host "a tangle of vines and a variety of mesophyllous and sclerophyllous shrubs, including cycads – which are often in impressively high densities." Shrubs include *Leucopogon lanceolatus*, *Persoonia linearis*, *Hibbertia aspera*, *Synoum glandulosum*, *Acacia obtusifolia*, *Zieria arborescens* and *Elaeocarpus reticulatus*.

Southern Lowland Wet Sclerophyll Forests occur between Nowra and Bega and are particularly prominent between Narooma and Ulladulla, stretching up the Clyde River from Batemans Bay (Keith 2004).

The vegetation groups identified by Gellie (2005) which seem most closely aligned to Keith's description include:

- Group 21 – variously named South Coast, or Northern, Foothills Moist Shrub Forest. *Corymbia maculata*, *Eucalyptus pilularis* and *E. paniculata* dominate this group over the southern part of its range, with *E. saligna* and *Syncarpia glomulifera* more prominent in the north. A small tree layer is common; species here include *Synoum glandulosum* and *Elaeocarpus reticulatus*. The shrub layer contains *Macrozamia communis, Leucopogon lanceolatus, Persoonia linearis, Hibbertia aspera* and several other shrubs listed by Keith (2005) for Southern Lowland Wet Sclerophyll Forest. Vines, particularly *Cissus hypoglauca*, are common. There is a variable ground cover of sedges, vines and forbs. This vegetation group is found on rolling foothills in the Clyde-Kioloa area and further north in the Shoalhaven and in Kangaroo Valley. It once covered an estimated 87,700 ha, of which 67,100ha (77%) remains uncleared. Of this, 35% (23,400ha) is reserved.
- Group 9 – Coastal Lowland Cycad Dry Shrub Forest. *Corymbia maculata* is the dominant tree species, with *E. paniculata*, *E. globoidea* and *E. muelleriana* also present in some sites. *Macrozamia communis* dominates the shrub layer. Tall shrubs include *Allocasuaria littoralis* and *Persoonia linearis*; smaller species such as *Hibbertia aspera, Playsace lanceolata, Leucopogon lanceolatus* and *Oxylobium illicifolium* also occur. Grasses *Entolasia stricta*, *Imperata cylindrica* and *Microlaena stipoides* mix with sedges, *Lomandra* species and

forbs. This vegetation group occurs on foothill ridges and slopes between Termeil south of Ulladulla, and Tilba Tilba. Eight-five percent of the pre-existing area of 67,500 ha remains uncleared; 87% of this is outside conservation reserves.

- Group 10 – Southern Coastal Lowlands Shrub/Grass Dry Forest. *Eucalyptus globoidea* and *E. longifolia* are the most common trees in this vegetation group, although *Corymbia maculata* and *E. paniculata* are also not infrequently found. Shrubs include *Allocasuarina litoralis*, *Hibbertia aspera*, *Acacia irrorata*, *Pittosporum revolutum* and *Macrozamia communis*. The ground layer contains grasses *Entolasia stricta*, *Microlaena stipoides* and *Poa meionectes*, as well as *Lepidosperma laterale* and *Dianella caerulea*. This group is also found on the slopes and ridges of coastal foothills, south from the lower Clyde catchment. Once covering an estimated 18,900, it is 90% uncleared, and 13% reserved.
- Group 3 – Northern Hinterland Shrub Dry Forest. While this vegetation group features neither Spotted Gum nor cycads, other elements of its composition suggest placement in the Southern Lowland Wet Sclerophyll Forest vegetation group. Dominant tree species include *Eucalyptus scias*, *E. paniculata* and *E. agglomerata*. *Syncarpia glomulifera* is found throughout. Common shrubs are *Persoonia linearis*, *Acacia obtusifolia*, *Zieria arborescens* and *Elaeocarpus reticulatus*. *Entolasia stricta*, *Dianella caerulea* and *Lepidosperma laterale* dominate the ground layer. Found west of Ulladulla this vegetation group, which has not been cleared, occupies 25,300 ha, 74% of which is reserved.

5.2.2 Probable role of fire

So far as I am aware, no fire ecology research has yet been carried out in Southern Lowland Wet Sclerophyll Forest. However research in similar vegetation types provides fairly clear guidance as to the role fire is likely to play, which is:

- To limit the expansion of species able to increase population numbers in the absence of fire (T species – see Section 1.5), including rainforest taxa and semi-mesic, fauna-dispersed plants such as *Pittosporum revolutum*, *Elaeocarpus reticulatus* and *Cissus hypoglauca*.
- To maintain the understorey in a relatively open and grassy state, and
- To provide an environment conducive to the continued presence of hard-leaved species and those which recruit primarily after fire (I species).

Rainforest encroachment into wet sclerophyll forest has been documented from North Queensland to Tasmania. As Keith (2004) points out in his discussion of Southern Warm Temperate Rainforests, where fire has not occurred for some time, or where fires are very infrequent, rainforest species, with their ability to recruit into gaps, will slowly move into the eucalypt understorey. This will occur most rapidly in moist sites adjacent to embedded rainforest patches. Conversely, fire may sometimes penetrate rainforest edges, and a series of fires in short succession will allow eucalypt seedlings to establish. The boundary between wet sclerophyll forest and rainforest is therefore influenced by fire, and may move over time.

In North Queensland where wet sclerophyll forest occurs as a narrow belt between rainforest and savanna, rainforest encroachment is occurring so rapidly that a severe reduction or even a total loss is predicted within the next century (Unwin 1989, Harrington and Sanderson 1994). This loss is of concern, as a number of animal and bird species are at risk of local extinction if the wet sclerophyll habitat disappears (Chapman and Harrington 1997). In south-east Queensland an experimental study comparing frequent burning to fire exclusion has documented increased recruitment of the non-eucalypt Myrtaceous species *Lophostemon confertus* and *Syncarpia glomulifera* in long unburnt wet sclerophyll forest (Guinto *et al.* 1999), while an increase in the abundance of species such as *Lophostemon confertus* and *Alphitonia excelsa*, and a decrease in grasses, has been noted in unburnt forests and woodlands near Brisbane. Again, concern has been expressed that the loss of grassy habitat may be affecting fauna species, particularly Pretty Face Wallabies (Kington 1997). In Springbrook National Park near the NSW border ecological burns have been instigated to save specimens of the rare cycad *Lepidozamia peroffskyana* which appeared to be dying due to colonisation by crowsnest ferns and vines from adjacent rainforest.

In northern NSW, Smith and Guyer (1983) clearly demonstrated the replacement of *E. saligna* and *E. microcorys* tall eucalypt forest by rainforest east of Tenterfield. York (1999), working on plots set up near Port Macquarie as part of a replicated experiment comparing frequent burning to fire exclusion, found cover of shrubs over 1 m tall was greater on unburnt plots, while cover of ground herbs and shrubs under 1 m was greater on burnt plots. Plant species found only in unburnt plots included several broad-leafed shrubs and trees (Stewart 1999). After 30 years without fire a thick subcanopy of *Syncarpia glomulifera* had developed, particularly in wetter areas, and grasses had been shaded out (Doug Binns pers. comm. 2005). Plant species richness was greater in burnt plots; species unique to this habitat included grasses, forbs and sclerophyll shrubs (Stewart 1999). Each environment housed a wide range of invertebrate species, many of which were only found in one treatment or the other (York 1999, 2000; Andrew *et al.* 2000).

Tasker (2002), who studied plants, small mammals and invertebrates in wet sclerophyll forest on the Tablelands south and east of Armidale, also found big differences in vegetation structure between sites with different burning histories. Where shrubs and small trees dominated the understorey in sites which had not been burnt for over 15 years, grasses dominated frequently burnt areas. Plant species richness was higher in burnt sites, where herbaceous species were particularly well-represented, while unburnt areas supported more ferns, climbers, and small trees, particularly ones with rainforest affinities. As at Bulls Ground fauna composition varied considerably between the two environments, though species richness was similar. Bush rats (*Rattus fuscipes*) occurred in much greater abundance in unburnt areas, and Brown Antechinus (*Antechinus stuartii*) also tended to favour these sites. However three species were caught only on the frequently burnt sites, and another mostly there – and these were rarer species, including the New Holland Mouse (*Pseudomys novaehollandiae*) and the Hastings River Mouse (*Pseudomys oralis*) (Tasker and Dickman 2004).

In Tasmania, wet sclerophyll forest occurs in areas where climate and soils have the potential to support rainforest. In the absence of fire, an understorey of rainforest plants develops. Eventually, the eucalpyts die off leaving the rainforest species to take possession of the site (Section 5.1.2). Soils change during this process (Ellis and Pennington 1989), becoming less conducive to sclerophyll species (Ellis and Pennington 1992). Burning the rainforest understorey appears to restore eucalypt health (Ellis *et al.* 1980). The relationship between soils, time-since-fire, understorey composition and tree health is yet to be researched in NSW, and may or may not have common ground with what happens in Tasmania (Wardell-Johnson *et al.* 2005).

The potential for rainforest plants to move into wet sclerophyll forest in the absence of fire has thus been documented both to the north, and to the south, of the NSW south coast, making it likely that similar dynamics will apply in Southern Lowland Wet Sclerophyll Forests in the absence of fire. In forests elsewhere the move towards a more mesic understorey is accompanied by a decreased abundance of herbaceous and sclerophyll species. While some animals thrive in the long-unburnt environment, others are only found where fire occurs regularly. These findings suggest that considerable variability in fire regimes, including some quite short intervals, may be important to maintain the full range of plant and animal species in wet sclerophyll forests (Watson 2006).

According to Keith (2004:72), "Southern Lowland Wet Sclerophyll Forests do not appear to have been greatly affected by land clearing, livestock grazing or fire." Descriptions of vegetation groups within this class provided by Gellie (2005) suggest expansion of mesic species may be occurring, particularly in vegetation group Group 21 where *Synoum glandulosum*, *Elaeocarpus reticulatus* and *Cissus hypoglauca* are common (Section 5.2.1). In turn, Keith's comment suggests these changes may be linked to a relatively low fire frequency over recent decades.

5.2.3 Conclusion

Fire is likely to play an important role in maintaining a balance between mesic and sclerophyll species, and between herbs and shrubs, in the understorey of Southern Lowland Wet Sclerophyll Forests.

At present there is no compelling reason to question the DEC-recommended fire frequency thresholds of 10 and 50 years. Rainfall in the section of the coast where Southern Lowland Wet Sclerophyll Forests are found is moderate (1055 mm at Bodalla, 912 mm a Narooma, according to the Bureau of Meteorology 2005), suggesting any transition towards a more mesic species composition will occur on a moderate timescale. Variable interfire intervals between ten and 50 years may well allow mesophyll plants to persist in the landscape while still providing space and recruitment opportunities for herbs and sclerophyll I species.

Research into the fire-related dynamics of Southern Lowland Wet Sclerophyll Forests would assist in refining fire frequency thresholds. More information on the life history characteristics of plants species, and studies comparing areas with differing fire histories could be particularly useful. Questions which might usefully be explored include:

- How do sclerophyll species – particularly the iconic cycads – fare under various interfire intervals? Is there evidence that fire may need to occur within a certain time-frame in order to maintain the environment these species need?
- How rapidly do T species population numbers expand? Do these species shade out, or otherwise outcompete smaller plants? If so, what fire frequency range will allow T species to persist in the community while also maintaining abundance of smaller species?

- Do changes in soil conditions occur with time-since-fire? If so, are they also apparent at differing fire frequencies? What relationship, if any, exists between understorey floristics, soil properties and fire regime?
- What fauna species are associated with the environment created by various fire frequencies? Are the Northern NSW findings that different fauna species use different vegetation ‘states’ in wet sclerophyll forest duplicated in Southern Lowland Wet Sclerophyll Forest? If so, should conservation managers be aiming to maintain a mosaic of different vegetation states, mediated by variable fire frequencies, across the landscape?

6 Conclusion

While the context within which fire affects plants and animals in different vegetation formations and classes varies, there is often common ground in the processes involved. Thus while the findings of research in one vegetation class cannot be assumed to apply to others in detail, broad principles may well translate.

The research reported in this review is patchy in distribution, however taken together holds a wealth of information for those seeking to maximise conservation outcomes through appropriate management of fire. Themes which repeatedly emerge include:

- The wide range of fire-related characteristics exhibited by plant species in particular fire-prone environments, and the different ways populations interact with, and persist through, a series of fires.
- The need to ensure fire regimes allow time for plant species to mature – particularly obligate seeder shrubs and trees.
- The role of fire in providing opportunities for I species to recruit, and the need to ensure fire occurs before these plants and their seeds die out.
- The role of fire in mediating competitive interactions between plant species of different sizes and longevities, and between I and T species.
- The conservation benefits of variability in interfire intervals.
- The conservation benefits of variability in fire intensity.
- The ability of fauna to recover after even intense fire, given opportunities for refuge and recolonisation.
- The presence of different suites of fauna species at different times after fire, reflecting changes in resource availability as vegetation regenerates, matures and ages.
- The importance, for many fauna species, of the dense vegetation which characterises the middle and/or later post-fire years.
- The importance, for some fauna species, of the early stages of post-fire regeneration.
- The desirability, from a fauna conservation point of view, of a range of vegetation age-classes and perhaps also of a range of fire frequency-mediated vegetation states.
- The existence of interactions between plants and animals which are influenced by fire regime characteristics.
- The existence of interactions between topography, vegetation and fire behaviour which combine to provide habitat variability across the landscape.

Many of these themes are reflected in the ecological theory discussed in Chapter 1 as well as in the findings reported in Chapters 2 to 5.

Differences in responses to fire regimes between vegetation formations, and between classes within formations, relate to factors such as climate, soil and topography that affect plant growth, vegetation structure and fire behaviour. These factors interact in ways we are only just starting to understand. For example the role of fire in disrupting competitive exclusion is likely to be most salient in areas with fertile soils (eg grasslands on basalt, Section 3.1), and/or benign climate (eg coastal forest and heath, Sections 2.1 and 5.2). Where infertile soils, low rainfall and harsh temperatures limit plant growth, competitive exclusion is likely to be less of an issue (eg Southern Tableland Dry Sclerophyll Forests, Section 4.1). Plant growth will be faster in more productive environments, however life-forms may also be larger, and both factors will influence time to maturity and thus minimum interfire intervals. Fuel will build up more rapidly in more productive environments, but may be available for burning less often due to relatively high moisture levels. Those seeking to encourage conservation-friendly fire management need to acknowledge these differences, and recognise that fire regimes compatible with retention of plant and animal species will therefore vary widely.

In many places, fire will occur periodically as wildfire. With climate change increasing the incidence of days when extreme fire behaviour can be expected (Hennessy *et al.* 2005), the challenge in some places will be to keep ‘natural’ fire frequency within bounds. In other areas, however, vegetation remnants have been largely cut off from wildfire, and may well remain so. If fire in these areas is to occur at intervals within recommended thresholds, the challenge will be to introduce some fire.

The diverse landscapes of south-eastern NSW invite a nuanced understanding of fire ecology. While there is still much to be learned, the current state of knowledge is sufficient to provide considerable guidance to managers. It is hoped that the ideas and findings summarised here will contribute to productive discussion of the role of fire in the Southern Rivers region.

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