HOTSPOTS FIRE PROJECT

Fire and the Vegetation of the Central West Region

Draft 1, for Comment

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Project Ecologist

August 2007

Nature Conservation Council of NSW
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This document has been prepared to help Hotspots and local NRM staff provide accurate information to landholders in the Central West region of NSW. The information it contains reflects the author's 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 16 November 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 primary 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.

This review considers literature relevant to a subset of vegetation classes in the Central West 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. Companion documents covering the Northern Rivers and Southern Rivers regions are also available (Watson 2006a,b).

Fire affects different plant and animal species differently, and fire regimes compatible with biodiversity conservation vary widely between ecosystems (Bond 1997, Watson 2001, Bradstock *et al.* 2002, Kenny *et al.* 2004). This document explores the role of fire in a sample of the vegetation classes identified by Keith (2004) in his book Ocean Shores to Desert Dunes. All vegetation types covered in this review are found in the Central West, however not every vegetation class in the region has been included. The selection process has been informed by:

- whether relevant 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 Central West and across the State.

As the workshops being undertaken in the Central West during the pilot phase of the Hotspots project, in the latter half of 2007, are all in the eastern half of the CMA region, we have focussed on vegetation classes found on the tablelands and slopes. Vegetation classes discussed in this review, and the formations in which they fall (Keith 2004), are:

- Grasslands and grassy woodlands: Temperate Montane Grasslands, Western Slopes Grassy Woodlands, Southern Tableland Grassy Woodlands (Chapter 2)
- Dry sclerophyll forests: Southern Tableland Dry Sclerophyll Forests, Pilliga Outwash Dry Sclerophyll Forests, Western Slopes Dry Sclerophyll Forests (Chapter 3).

Although semi-arid shrublands and woodlands, and wet sclerophyll forests, also occur in the Central West, Hotspots' ability to address these vegetation formations was limited by time available: an omission which can hopefully be remedied in future years.

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. While vegetation and plant species receive most attention, fauna are also mentioned.

Before addressing the literature on specific vegetation classes, some ecological concepts and principles underlying current understanding of fire regimes are explored (Sections 1.2 to 1.6 in 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, DEC 2002).

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 (Ladd 1988, Enright *et al.* 1998).

A second group of species stores dormant seeds in the soil, with dormancy requirements which ensure germination occurs mostly after fire. Heat promotes germination in many legumes (Shea *et al.* 1979, Auld and O'Connell 1991, Clarke *et al.* 2000), while smoke plays a role for other 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 the Crinkle Bush *Lomatia silaifolia* (Denham and Whelan 2000) and the Waratah *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 very fire-prone environments such as coastal heaths, where seed dispersal distances seem to be limited to tens of metres or

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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 mean 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).

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

less in most species (Auld 1986, 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* and *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. Implications of this dichotomy are discussed in Sections 1.4 and 1.5.

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).

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 natural 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 a lower and an upper 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 resprouting shrub *Bursaria spinosa* are associated with a reduced abundance of other shrub species, particularly obligate seeders (Watson 2005, Section 2.2.4).

Bursaria has the 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. T species are almost certainly more prevalent in some environments than in others, for example in rainforests, and conversely in arid areas where recruitment may be geared more to periods of unusually high rainfall or flooding than to fire.

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 savanna and heath.

These three categories no doubt form a continuum. In NSW, limitation by climate rather than fire probably becomes more prevalent as average annual rainfall decreases. In some arid and semi-arid environments, droughts and/or floods may complement or even replace fire as the primary natural space-creating mechanism (Cunningham and Milthorpe 1976). Of course, climate and fire are everywhere intertwined, with major fires occurring during months and years when vegetation dries out with drought. *Higher* than average rainfall, however, is also intimately associated with fire in arid and semi-arid areas, as in many places fires will only spread when good seasons stimulate the growth of grasses and herbs which become cured, continuous fuel when rains retreat.

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, 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 The Central West and its landscapes

Bauer and Goldney (2000:108) describe the Central West as "a transition zone from the coast to the continental plains, modified by the complex elements of the Great Dividing Range." Three zones are commonly identified: the tablelands in the east, the plains in the west, and the slopes between them. The Central West thus covers an area of great diversity. Altitude ranges from 1000 m above sea level at Oberon in the far south-east, to 388 m at Dunedoo the slopes, to 170 m at Nyngan on the plains. Rainfall, which averages 846 mm a year at Oberon and 614 mm at Dunedoo, drops to 442 mm at Nyngan. Average maximum temperatures in January for these three weather stations are 24, 32 and 34℃ respectively, while equivalent figures for July are 9, 15 and 16℃. "Frost days", defined as days when the temperature drops below 0℃, average 55, 32 and 6 per year (Bureau of Meteorology 2006). Vegetation naturally reflects these considerable gradients, and is in turn diverse. Soil characteristics also influence plant associations, adding to the rich array of plant communities in the region (Metcalfe *et al.* 2003).

Woodlands are the predominant vegetation form in the Central West, and this, Goldney and Bowie (1990) believe, is one of the unifying themes of the region. Another is the effects of season: flushes of growth occur in autumn and spring, but not as a rule in winter when growth is restricted by cold, or in summer when drought is the limiting factor (Goldney and Bowie 1990).

Vegetation on the tablelands once consisted of a mosaic of grassy woodlands interspersed with dry shrubby forests on poorer soils, wet sclerophyll forests in the wettest most fertile areas, and grasslands around Bathurst and adjacent to some rivers and creeks. Grassy woodlands dominated the slopes, again with areas of shrubby forest on poorer soils (Semple 1997, Croft *et al.* 1997, Keith 2004). As the slopes give way to the plains the semi-arid zone, where rainfall averages between 250 and 500 mm a year, begins (Mitchell 1991). Rainfall in this part of country varies considerably from year to year, and droughts are common. Vegetation is, or was, a mosaic of grassy and shrubby woodlands and 'scrubs' of small trees (Croft *et al.* 1997).

Three major rivers flow through the regions covered by the Central West CMA, roughly from south-east to north-west: the Castlereagh which starts in the Warrumbungles, the Macquarie which begins on the slopes of the Great Dividing Range, and the Bogan which rises in the plains north of Parkes. Extensive flood plains are associated with these rivers, contributing yet more complexity to the mosaic of landscape features and vegetation communities (Metcalfe *et al.* 2003).

The Central West has a long history of human occupancy and use. Bauer and Goldney (2000) summarise evidence which suggests Aboriginal people may have occupied the region for 40 000 years or even longer. However Aboriginal control of the land did not persist for long after European settlement, which spread rapidly across the slopes and into the plains once pioneers made their way across the mountains to Bathurst in 1815. Grazing of sheep and cattle was accompanied by clearing of trees to encourage grasses (Goldney and Bowie 1990). By the time of Federation the combination of overgrazing and drought had had severe effects on native pastures. The sheep population peaked in 1890 at about 15 million, crashed to 3.5 million over the next eight years, eventually stabilising some years into the twentieth century at around 5 million (Bauer and Goldney 2000).

The twentieth century saw the expansion of cropping, particularly on the eastern edge of the plains, where the potential for growing wheat was first recognised towards the end of the nineteenth century. The first wheat was grown around Lake Cargelligo in 1894 (Bauer and Goldney 2000). Clearing for wheat farming was particularly extensive between 1920 and the 1970s. Meanwhile, pasture improvement had become common practice in the east.

European land use over the past 200 years has had major impacts on the native plants and animals of the Central West. By 1990, Goldney and Bowie (1990:431) argued that "A greater part of the regional environment has been irreversibly changed from its natural condition," with over 80% of the land area "substantially cleared," and remaining native vegetation in scattered, and often very small, remnants. Clearing continues (Fisher and Harris 1999, Benson *et al.* 2006): for example a study by Metcalfe *et al.* (2003:265) which mapped native woody vegetation on the plains of the Central West documented a reduction of nearly 10% in the total extent of vegetation between the 1980s, when work began, and 2000. Overall, 29% of the study area was mapped as native vegetation in 2000, down from 32% in the early 1980s.

While clearing of trees and other woody species may be the most obvious change to the landscapes of the Central West, other changes detrimental to many native plants and animals have also occurred. Even where trees persist, ground cover has changed. In some places, shrubs have probably declined; in other places they appear to have increased. Native animals have not only had to deal with loss of habitat, they have also had to contend with introduced predators, particularly cats and foxes. For many years rabbits had a major impact on vegetation, competing with both domestic and native herbivores. Rabbit populations peaked in the 1930s and 1940s, then declined from the 1950s due to myxomatosis. Native animals have also been subject to the direct effects of hunting (Bauer and Goldney 2000). It is not surprising, therefore, that many native fauna species have either disappeared, or have become much less common, in the region (Bauer and Goldney 2000, Sivertsen and Clarke 2000, Dickman 1994).

Initiatives to reverse some of these losses and protect remaining native vegetation are, however, underway (Goldney and Wakefield 1997). Perhaps to a greater extent in the Central West than elsewhere in NSW, it is vital that private landholders are part of these efforts. As Benson *et al.* (2006:431) explains, "The vast majority of the 213 plant communities in the NSW Western Plains are very poorly represented in protected areas." Only 0.35% of the plains section of the Central West is in conservation reserves, while 92% is freehold (Metcalfe *et al.* 2003). Conservation reserves on the slopes and tablelands are much more likely to contain shrubby vegetation types that were not valued for agriculture, than temperate grassy woodlands (Goldney and Bowie 1990).

Amongst the many factors that have changed over the last 200 years in the Central West, 'changed fire regimes' are sometimes mentioned (Fisher and Harris 1999, Bauer and Goldney 2000, Sivertsen and Clarke 2000). We have no way of really knowing what fire regimes pertained under Aboriginal management. However it is generally acknowledged that Aboriginal people used fire for a variety of purposes, and that it is likely fire played a part in maintaining the ecosystems which greeted European settlers as they moved into the Central West in the early 1800s (Fisher and Harris 1999, Prober and Thiele 2005). What part that might have been, and how and where fire may have a role in conserving biodiversity in the greatly changed landscapes of today, is something we are only just starting to discover. Clues can be found in the bush itself, and in the history of changes since European settlement. When these are added to the wider understanding represented in the theories and models of fire ecology that have been outlined in this chapter, we have enough to make broad predictions and suggestions for management. There is still, however, a great deal to be discovered by fire ecologists in the Central West.

Across NSW, the quantity of ecological research, and the degree of its relevance to fire, varies considerably between vegetation classes. Few studies have directly addressed the effects of fire frequency on plants and animals in the Central West. This review therefore draws on studies from outside the region as well as on local work.

Given the complexity of the factors impacting native plants and animals in the Central West, it is highly unlikely that changing one aspect of management – such as fire regimes – will provide a solution on its own. Fire regimes are one tool in the toolbox of the land manager seeking to enhance conservation outcomes. It is also very likely that in many situations, other disturbances – both natural ones such as drought and flood, and ones that have come with European settlement such as clearing, grazing by domestic herbivores and water regulation – will interact with fire regimes. Also, as Bauer and Goldney (2000:119) point out, when land is subject to a wide range of disturbances, particularly unfamiliar ones, ability to recover from additional disturbance may be impaired, as "The land system would be judged to have lost resilience." It is thus essential that when land managers think about fire regimes, they do so in the context of the nature, history and condition of the land.

Subsequent chapters in this report summarise research findings relating to six vegetation classes found in the Central West. We begin with the temperate grasslands and grassy woodlands of the tablelands and slopes.

2 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 initially focussed on the *Themeda*-dominated grasslands of Victoria's lowland plains, and substantial gaps still remain elsewhere (Hobbs 2002, Lunt and Morgan 2002). This Victorian work informs the discussion of Temperate Montane Grasslands in Section 2.1. More recently investigations into the effects of fire in grassy woodlands, including those of the western slopes and southern tablelands which are considered in Sections 2.2 and 2.3, have begun.

2.1 Temperate Montane Grasslands

2.1.1 Introduction

Temperate Montane Grasslands once covered an extensive area on the Monaro from Cooma south to Bombala, with outlying patches elsewhere, for example around Braidwood, Goulburn and Bathurst (Cambage 1909, Benson 1994, Semple 1997, Keith 2004). In the Central West, the upper Macquarie River valley may have had up to 20 000 hectares of grasslands when government surveyor George Evans followed Blaxland, Lawson and Wentworth over the Blue Mountains in 1813 (Semple 1997). Temperate Montane 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 has also been widespread (Benson 1994, Semple 1997). 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 (Lunt *et al.* 1998, Keith 2004). Unfortunately, exotic grasses and forbs are also common (Benson 1994, Dorrough *et al.* 2004).

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*[3], 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 herbaceous species (Section 2.1.2). Research into the relationship between fire and the matrix species *Themeda australis* follows (Section 2.1.3). Because herbaceous exotics are a major issue in remnant grasslands, the effects of fire and *Themeda* on weeds is discussed in Section 2.1.4.

2.1.2 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), that regular burns can increase gap size (Henderson and Hocking in prep.), and that short interfire intervals are important for maintaining populations of adult interstitial species (Coates *et al.* 2006).

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

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[3] I have used the designation *Themeda australis* in 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.

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, Lunt 1996, 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 forbs 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).

2.1.3 Maintaining matrix grass vigour

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. Research has also found that a healthy *Themeda* sward can, in turn, limit or reduce weed invasion; this is discussed in the next section. Studies from grasslands and grassy woodlands are included in this and the following section.

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* (Weeping Grass) 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*, *Poa* and *Austrodanthonia* species[4] (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. Many 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), Dorrough *et al.* 2004). Where grazing has reduced but not eliminating *Themeda*

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[4] 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.

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. However on the western slopes, fire frequency may mediate the balance between *Themeda* and *Poa* in grassy woodlands – see Section 2.2.2.

2.1.4 *Themeda* and 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 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*. 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 some situations. The post-fire environment presents opportunities to target weed species while in an active growth phase, before they flower, and while they do not overlap native species. 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).

2.1.5 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 vigorous populations of forbs and less competitive grasses are to persist (Lunt and Morgan 2002). Whether fire and grazing are interchangeable is discussed in Section 2.3.5.
- 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 2.1.3).

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. Ecologist John Briggs, who has experimented with various interfire intervals in grasslands around Canberra, suggests 5-8 year intervals may be appropriate in southern tablelands grasslands. However detailed work on the endangered daisy *Rutidosis leptorrhynchoides* (Button Wrinklewort), which is found in grasslands 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.

2.2 Western Slopes Grassy Woodlands

2.2.1 Introduction

Grassy White Box (*Eucalyptus albens*) woodlands were once "ubiquitous on fertile soils throughout the western slopes of the Great Dividing Range" from the Darling Downs in Queensland to central Victoria (Keith 2004:96). Fragments survive in a broad band across the Central West from Coonabarabran in the north, to west of Molong in the south.

Western Slopes Grassy Woodlands grow on fertile clay loams below 700 m above sea level, where rainfall averages 550 to 800 mm a year. As well as White Box, tree species can include Kurrajong (*Brachychiton populneus*), White Cypress Pine (*Callitris glaucophylla*), Yellow Box (*Eucalyptus melliodora*) and Blakely's Red Gum (*E. blakelyi*). Shrubs are not prominent in this vegetation class; those that do occur include Blackthorn (*Bursaria spinosa*), Sifton Bush (*Cassinia arcuata*) and the wattles *Acacia decora*, *A. dealbata* and *A. deanii*. The ground layer, on the other hand, is – or was – a species-rich mixture of native herbs and grasses, including Snow Grass (*Poa sieberiana*), Kangaroo Grass (*Themeda australis*), Wallaby Grasses (*Austrodanthonia* species) and Red-leg Grass (*Bothriochloa macra*). Herbs which grow amongst the grass clumps include lilies such as *Bulbine bulbosa* (Bulbine Lily), *Dichopogon* *fimbriatus* (Chocolate Lily) and *Dianella longifolia* (Blue Flax Lily), daisies such as *Chrysocephalum apiculatum* (Common Everlasting) and *Cymbonotus lawsonianus* (Bears-ear), and a wide range of other species (Keith 2004, Prober and Thiele 1993, 2005).

Unfortunately, "While White Box trees still remain abundantly scattered throughout the landscape, tree regeneration is limited and the native understorey community is very rare, either due to complete clearing for cropping, replacement by improving pasture, or altered floristic composition and weed invasion caused by livestock grazing" (Prober and Thiele 1993:30). Prober and Thiele (1995) found a decline in native species richness with increased grazing pressure, while the abundance of exotic species increased. Species which declined with any grazing included the original dominant grasses *Themeda australis* and *Poa sieberiana*, as well native herbs such as *Dianella*, *Glycine* and *Wahlenbergia* species. Ground layer species composition in landscapes subject to on-going heavy grazing follows a well-known path: from *Themeda/Poa* dominance, to dominance by *Stipa* and *Austrodanthonia* species, then to *Bothriochloa* dominance, and finally to annual exotics with patches of bare ground (Moore 1953). Pasture improvement and application of fertilizers almost certainly contribute to this progression (Goldney and Bowie 1990).

While it is generally acknowledged that fire was once part of these woodlands (Prober and Thiele 1993, Prober and Thiele 2005) there has been very little burning in grassy woodland remnants in the Central West, either through wildfire or by planned ignitions, for many decades (pers. comm. Geoff Selwood, RFS Orange, 2006).

Research into the ecology of grassy White Box woodlands, under the leadership of Suzanne Prober and Kevin Thiele, has been underway for approximately 15 years. Fire is a component of this work. Fire has also been studied in the closely-related grassy woodlands of Western Sydney's Cumberland Plain. This vegetation type, known as Cumberland Plain Woodland (CPW), grows on clay soils where rainfall averages 700-800 mm per annum. Dominant trees are box and gum eucalypts, specifically *Eucalyptus moluccana* (Grey Box) and *E. tereticornis* (Forest Red Gum). The understorey contains a diverse range of grasses and herbs, including all those listed in the description of Western Slopes Grassy Woodlands above, within a grassy matrix which is often dominated by *Themeda*. The shrub layer, which is dominated by *Bursaria spinosa* (Australian Blackthorn), includes various wattle and peas species. Sifton Bush (*Cassinia arcuata*), a familiar shrub on the western slopes, also occurs in some CPW remnants. Overall, Benson (1992) reports that 74% of the 439 plant species found in CPW also occur on the western slopes. CPW thus resembles Western Slopes Grassy Woodlands in both structure and species composition.

The discussion below draws on studies from a range of eucalypt woodland types to address what is known about the fire-related dynamics of three structural components of Western Slopes Grassy Woodlands: the ground layer (Section 2.2.2), trees (Section 2.2.3) and shrubs (Section 2.2.4). An exploration of the relationship between fire and exotic weeds follows (Section 2.2.5), as does a short section on fire and invertebrates (Section 2.2.6).

2.2.2 Ground layer

What little is known about the effects of fire on ground layer grasses and forbs strongly suggests there is common ground between Western Slopes Grassy Woodlands and the temperate grasslands discussed in the previous section. 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 2.1.3). Findings of studies of the effects of fire on *Themeda* in Western Slopes Grassy Woodlands are in line with trends elsewhere.

Recent experimental work on effects of fire frequency on the western slopes have found that while frequent burning favours *Themeda*, infrequent burning or lack of fire favours *Poa sieberiana* (Prober *et al.* in review). Small plots in two relatively undisturbed country cemeteries on the central western slopes were subject to burning at a range of frequencies: fires were either 2, 4 or 8 years apart. At the first site, which had a history of frequent burning and was initially dominated by *Themeda*, *Poa* cover increased as fire frequency decreased. At the second site, with a history of fire exclusion, swards were originally dominated by *Poa*. Here, *Themeda* increased in plots with the higher burning frequencies. Measures of sward health led the authors to conclude that biennial burning was not a good idea, and this conclusion was reinforced when the impacts on soils were assessed at one of the two sites. Prolonged exposure of the soil surface in the biennially burnt plots at this site was associated with increased soil crusting, decreased infiltration and decreased biological activity (Prober *et al.* in prep.). While the authors are cautious about making recommendations, they conclude that fires at 5-8 year intervals are unlikely to cause problems (Prober *et al.* in prep.). Periodic, though not too frequent fire, should ensure swards continue to support healthy populations of both *Themeda* and *Poa*. "These results indicate that different components of the dominant sward in temperate grassy eucalypt woodlands confer resilience in the face of different natural disturbance regimes. Functional complementarity between *Themeda* and *Poa* may thus engender resilience in a broader sense, i.e. resilience or adaptability to variable disturbance regimes" (Prober *et al.* in review).

Herb species composition in Western Slopes Grassy Woodlands may be influenced by the larger lifeforms – trees and shrubs – on ground layer species. This is certainly the case for trees. A survey of soils and plants under box eucalypts in high quality grassy woodland remnants on the central and southern western slopes (Prober *et al.* 2002a) found patches under trees had more fertile soil than patches out in the open away from trees. The composition of the understorey changed as well. Where *Themeda australis* dominated in gaps, *Poa sieberiana* was more abundant under trees. The species richness of native herbs was higher under trees; though few species were found only in this habitat, some preferred it. In a similar survey in CPW, Watson (2005) found significant differences in species composition between open patches, patches around trees and patches under *Bursaria* bushes. In contrast to the findings of Prober *et al.* (2002a), this study, which took place 9 -18 months after fire, showed no decline in species richness in open habitats; in fact ten of 14 species with a statistically significant connection with a particular microhabitat favoured open patches, including several native lilies. These differences between the woodlands of the western slopes and those of the Cumberland Plain may reflect differences in the vegetation dynamics in the two environments. They may also, however, reflect the different fire histories of the two sets of woodland sites. Where native herbs may have benefited, in the western slopes study, from the competition afforded by trees to *Themeda*, open patch herbs in CPW may have benefited from the openings in the grass sward provided by recent fire.

Researchers in other ecosystems have also found that herb composition differs with canopy cover (Scanlan and Burrows 1990, 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 eucalypt litter may be more of an issue for herb growth and reproduction. Somewhat different suites of species may be involved in each environment.

2.2.3 Trees

Trees in the grassy box woodlands of the western slopes have declined severely as a result of activities associated with rural land uses since European settlement (Prober and Brown 1994, Windsor 1999). Though scattered trees and groups of trees remain within the agricultural landscape, tree regeneration is often problematic (Prober and Thiele 1993).

The dominant tree species of Western Slopes Grassy Woodlands are epicormic resprouters (DEC 2002). Adults are therefore generally able to survive fire. For populations to survive and grow, however, new individuals must be recruited. While woodland eucalypts can recruit in the absence of fire, fire may enhance recruitment opportunities (Semple and Koen 2001, Li *et al.* 2003, Vesk and Dorrough 2006). For example on the Cumberland Plain Hill and French (2004) found greater numbers of eucalypt seedlings after a summer wildfire than in unburnt areas. The discussion below draws on studies from woodlands over a range of eucalypt woodland types.

Eucalypt seeds are stored in the canopy; the extent to which live seed can accumulate in the soil appears to be negligible (Ashton 1979, Vlahos and Bell 1986, Read *et al.* 2000, Hill and French 2003). Seed bed conditions play an important role in eucalypt establishment. For example in a study on the northern tablelands Whalley and Curtis

(1991, cited in Windsor 1999) found recruitment only occurred where the grass sward was thin and bare soil was available. Fire may enhance recruitment opportunities by promoting capsule dehiscence (Florence 1996), by reducing the competition that seedlings would otherwise experience from grasses or herbs (Noble 1980, Curtis 1990, Semple and Koen 2003), by killing some adult trees and thus creating canopy gaps (Wellington and Noble 1985), by enhancing seedbed conditions (Clarke and Davison 2001), or by triggering sufficient seed release to cause ‘predator satiation’ of ants (Ashton 1979, Andersen 1988). Eucalypt recruitment may be episodic, depending on the coincidence of seed availability, gap-creating disturbance, and rainfall (Wellington and Noble 1985, Curtis 1990, Clarke 2000).

While many eucalypt seedlings die within a year or two of establishment (Henry and Florence 1966, Wellington and Noble 1985, Clarke 2002), those that survive rapidly develop lignotubers which help them survive not only fire, but other disturbances such as drought and grazing (Curtis 1990, Semple and Koen 2001, Clarke 2002). Suppressed lignotuberous seedlings can persist in the understorey for many years (Noble 1984), even in the face of regular burning (Henry and Florence 1966). When conditions are right, individuals grow through the sapling stage and join the adult population (Florence 1996). Strategies which should enhance opportunities for eucalypt establishment include restricting grazing and reducing the density of exotic grasses and herbs above and below ground (Windsor 1999, Semple and Koen 2003).

The effect of fire on tree populations has been studied in savanna woodlands, particularly in Northern Australia and Africa. Some studies link fluctuations in tree density to fire frequency or intensity, although there is also evidence of considerable stability in eucalypt populations. For example in an experimental study in the Northern Territory, intense annual fires caused a reduction in tree stems, as did a wildfire after six years of fire exclusion. Mild annual burns, however, did not affect stem survival, and at whole tree level there was little difference between treatments (Williams *et al.* 1999). Frequent fire may limit tree recruitment by killing small diameter stems (Williams *et al.* 1999) and returning saplings to the basal-sprouting lignotuber pool. Very long-term fire exclusion may also cause a decline in woodland eucalypt populations (Withers and Ashton 1977, Lunt 1998b). In the CPW study reported above (Watson 2005), fire frequency did not significantly affect either adult tree density, adult tree basal area, or the density of suppressed seedlings or saplings. Trends suggested an association between frequent fire and an increased density of juveniles, but frequent burns also tended to decrease the number of saplings ‘getting away’ into the canopy. Summer wildfire may also have limited sapling numbers, by killing even quite large stems and reducing these plants to basal resprouts.

2.2.4 Shrubs

Although shrubs are not prominent in White Box woodlands, they do occur, particularly in less fertile sites (Prober 1996, Semple 1997).

The importance of fire in the life-cycle of shrubs in at least some temperate woodlands is apparent from an experimental study of fire and grazing in Cumberland Plain Woodland. 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. Similar findings have been obtained in grassy woodlands on the northern tablelands (Knox and Clarke 2006 – see Section 2.3.4).

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. 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). Patterns for native shrubs other than *Bursaria*, however, were 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).

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. 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, with soil-stored seed which eventually decays if fire-related germination cues are absent. A moderate fire frequency allows these shrubs to increase population numbers through prolific germination.

How relevant are these CPW findings to Western Slopes Grassy Woodlands? The literature suggests that native shrub abundance in these woodlands is generally low irrespective of fire regime: competition from abundant grasses may be the primary control (Semple 1997). *Bursaria* may thicken up in some places in the absence of fire. A recent survey by DEC (2006) on the eastern edge of the central west slopes found *Bursaria* thickets in one vegetation community allocated to the Western Slopes Grassy Woodland class (Capertee Rough-barked Apple – Regum – Yellow Box Grassy Woodland, DEC 2006:61). Where this is the case, fire frequency may regulate the balance between shrubby and open patches, as it does on the Cumberland Plain. Some sclerophyll species, particularly peas (eg *Indigofera australis*, *Pultenaea* spp.) and some wattles, may respond to fire in a similar manner to the CPW obligate seeders. If this is so, the absence of these species in long unburnt remnants may partially reflect lack of fire-related recruitment opportunities. However many of the native shrub species of the western slopes may use strategies for persistence which are relatively independent of fire. Shrubs whose seeds are distributed by higher animals (eg shrubs with fleshy fruits like *Lissanthe strigosa*, *Peroonia* species, and perhaps also some wattles), or wind (eg daisy bushes, dogwoods), are often able to establish in the absence of fire (Campbell and Clarke 2006, Ooi *et al.* 2006). The majority of shrubs found in Western Slopes Grassy Woodlands may fall into this category.

One native shrub species which has been very successful in building up its population in the Central West is *Cassinia arcuata*, or Sifton Bush (Campbell 1990, Semple 1990). Semple (1990) believes a major factor in the relatively recent prolific spread of this shrub is decreased competition from native grasses – a function of changed grazing regimes, and perhaps secondarily, in the case of woodlands once dominated by *Themeda*, of reduced fire frequency (see Section 2.1.3). Loss of overstorey cover may also play a role (Semple 1990, Sue Wakefield, pers. comm. 2007), although Sifton Bush can be found in considerable abundance even where the tree canopy is intact (Bower 2005). Sifton Bush produces copious seeds, which germinate readily with no sign of a need for fire-related germination cues (Campbell 1990). These characteristics give it a formidable strategy in any environment which provides conditions conducive to seedling establishment.

Might fire be a useful tool in the control of Sifton Bush? Semple and Koen (1993) explored this question through a series of experiments on the tablelands near Orange. They found that fire, particularly hot fire, was quite effective at killing existing Sifton Bush plants. Unfortunately, they also recorded extensive post-fire germination – like most native obligate seeder shrubs Sifton Bush has an effective strategy to stick around in a fire-prone environment. Semple and Koen (1993:329) regretfully concluded that "As a control technique … fire is of limited use unless followed up by some technique to control seedlings."

It is possible that in some situations, additional burns might constitute an effective method to control post-fire Sifton Bush seedlings.[5] However for this strategy to work, subsequent fires would need to take place before the new seedlings reached reproductive maturity (Semple 1990). This apparently takes about two years (Parsons 1973 cited in Semple and Koen 1993), a relatively short juvenile period. As noted in Section 2.2.2, repeated biennial burning may produce negative impacts on other ecosystem elements. However this trade-off might be worthwhile in circumstances where dominance of Sifton Bush is causing problems for biodiversity, particularly if only one or two short intervals were applied followed by a more moderate regime. A second or third fire would only be possible where sufficient grass fuel had developed. Strategies to encourage native tussock grasses such as *Themeda australis* might assist here (see Sections 2.1.3 and 2.1.4).

2.2.5 Exotics

Fire may have a part to play in limiting weeds, both woody shrubs such as African Boxthorne, Blackberry, Hawthorne and Sweet Briar, and herbaceous species.

In CPW, African Olive (*Olea europaea* subsp. *cuspidata*) is a major invasive woody exotic. 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 in this study had virtually no woody exotics. Olive can recruit between fires, an attribute which probably characterises some other exotic woody weeds as well. Often seeds of these species are brought in to remnant native vegetation patches by birds, and establish below trees.

The situation with fire and herbaceous weeds outlined for grasslands in Section 2.1.4 is also likely to apply in temperate grassy woodlands. An experimental study in degraded grassy woodland near Young (Prober *et al.* 2004) concluded that fire had the potential

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[5] An alternative strategy for Sifton Bush control in grassy woodland remnants, which doesn't involve fire, has been developed at Oolong Sanctuary near Dalton – see http://www.communitybuilders.nsw.gov.au/builder/volunteering/cvs.html.

to assist with the control of some weeds; the tree species in this woodland were *E. albens*, *E. melliodora* and *E. blakelyi*. Two spring burns a year apart dramatically reduced the abundance of exotic annual grasses, although perennial and dicot annual exotics weren't generally affected. In Cumberland Plain Woodland 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 2.1.4). Post-fire weed control strategies are also likely to be beneficial.

2.2.6 Invertebrates

Research into the effects of fire in Western Slopes Grassy Woodlands has not, in general, extended to fauna. An exception is a study by Greenslade (1997) into the effects on invertebrates of a single burn after a very long period without fire, in experimental White Box woodland plots near Cowra. Although invertebrate numbers were lower on burnt plots immediately post-fire, differences disappeared within six months. Community composition, however, was somewhat different on burnt and unburnt plots, with some species responding positively to fire and others being more abundant in unburnt areas. Most invertebrates in this study were Collembola (springtails). Although fire wasn't a major influence on the invertebrate community, there were large differences between samples taken in spring and autumn, probably due to differences in rainfall.

2.2.7 Conclusion

> "Burning was probably an important pre-European management tool in grassy White and Yellow Box woodlands … and some species are favoured by fire. Fire may also play a role in maintaining optimal tree densities, and thus it is still important to maintain a diversity of management regimes in grassy woodlands, including regular burning" (Prober *et al.* 2002a:695).

The studies outlined above strongly suggest that fire has an important place in Western Slopes Grassy Woodlands. Fire regulates the abundance of the two grasses that originally dominated this vegetation class, provides opportunities for heat and smoke-cued shrubs and grasses to recruit, may help young eucalypts to establish and grow, and can play a part in limiting and reducing the abundance of weeds.

In a recent article Prober and Thiele (2005) bring together what they have learnt from over a decade of research into grassy white box woodlands, and discuss implications for managers seeking to restore them.[6] These authors point out the importance of looking beyond species composition to an understanding of how ecosystems work. They recommend understanding the state before degradation, the reasons for ecosystem change, and the processes which can be used to restore ecosystem function. Fire is one

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[6] This article is recommended reading for anyone interested in restoring grassy box woodlands in the Central West.

of a suite of strategies which can be used to restore the low nutrient, *Themeda*- and *Poa*-dominated woodlands which are likely to favour native over exotic species.

How often should fire occur in this vegetation type? Kenny *et al.* (2004) recommend a fire frequency of between 5 and 40 years for grassy woodland ecosystems across NSW. Plants in Western Slopes Grassy Woodlands, with their moderate rainfall and moderately fertile soils, are unlikely to grow as fast as their counterparts on the Cumberland Plain where rainfall is higher, nor as slowly as plants in grassy woodlands at high altitude where temperatures are cooler. Intervals in the lower half of the statewide range may well produce the best biodiversity outcomes in this vegetation class. As we have seen, Prober *et al.* (in prep.) tentatively suggest fires at 5-8 year intervals may be compatible with maintaining a balance between the dominant grasses.

2.3 Southern Tableland Grassy Woodlands

2.3.1 Introduction

Keith identifies a second grassy woodland type in the Central West, the Yellow Box (*E. melliodora*) - Blakely's Red Gum (*E. blakelyi*)-dominated Southern Tableland Grassy Woodlands. These woodlands once extended over much of the central and southern tablelands, into Victoria. In the Central West they covered a wide area around Bathurst and Orange, stretching north almost to Mudgee (Keith 2004). Precipitation over the range of Southern Tableland Grassy Woodlands averages 550 - 900 mm a year, occasionally falling as snow. Soils are moderately fertile loams. Tree species which may co-occur with Yellow Box and Blakely's Red Gum in the Central West include *Eucalyptus bridgesiana* (Apple Box) and *E. rubida* (Candlebark) (Bower *et al.* 2002). Shrubs are not a major component of the flora; Keith (2004) considers this may be because of prolonged exposure to grazing. Those that do occur include Blackthorn (*Bursaria spinosa*), Silver Wattle (*Acacia dealbata*), Peach Heath (*Lissanthe strigosa*) and Native Cherry (*Exocarpos cupressiformis*). Most plant diversity is found in the ground layer, which is of similar composition to that in Western Slopes Grassy Woodlands. Herbaceous peas such as *Glycine clandestina*, *Desmodium varians* or *Swainsona* species may be present. Like the neighbouring White Box woodlands, this vegetation class has been extensively cleared and modified (Goldney and Bowie 1990, Banks 1997, Fisher and Harris 1999).

Given their similar composition and structure, Southern Tableland Grassy Woodlands are likely to have much in common ecologically with the Western Slopes Grassy Woodlands discussed in the last section. The somewhat higher altitudes at which the southern tableland woodlands are generally found are likely to slow plant growth to some extent, as winter temperatures will be very cool (Semple and Koen 2001).

Fragmentation in Yellow Box – Blakely's Red Gum woodlands has affected the fire regime to the point where very little fire now occurs in this vegetation type in the Central West. While lightning strikes continue, fires are easily extinguished. Reintroduction of fire on an experimental basis is taking place in some long unburnt remnants, for example in Boundary Road Reserve near Bathurst (pers. comm. Steve Woodall, NPWS, Bathurst, 2006).

In Sections 2.3.2 to 2.3.4 below, the fire-related dynamics of the ground layer, trees, and shrubs are briefly discussed. Research into the ecology of fire with a specific focus on Southern Tableland Grassy Woodlands has been very limited, however some studies in this and in the related New England Grassy Woodlands touch on the topic. Section 2.3.5 asks whether grazing and fire are interchangeable disturbances in grassy vegetation, from the point of view of biodiversity conservation. This latter discussion draws on information from, and is relevant to, a range of grassy vegetation classes.

2.3.2 Ground layer

The discussion of the effects of fire on ground layer vegetation in Western Slopes Grassy Woodlands (Section 2.3.2) is also relevant for the vegetation class under discussion here. In fact, the fire frequency plots studied by Prober *et al.* (in review) covered both White and Yellow Box woodlands.

In his discussion of Southern Tableland Grassy Woodlands Keith (2004) mentions the herbaceous native pea *Swainsona recta*. This rare plant has been studied by John Briggs, an ecologist who has worked on the southern tablelands around Canberra for many years. Populations of this species persist on railway easements, which were ungrazed and frequently burnt for many years. Observations during a yet-to-be-published experimental study of its response to various fire regimes has shown that *Swainsona recta* is well able to regenerate after fire, regrowing from deep, thick rootstocks, and flowering in the post-fire environment. This species, like many woodland forbs, relies mostly on persistence of existing individuals. While some seedlings have germinated in John's experimental plots, they are not common and are prone to insect predation, an effect which may be exacerbated by the small size of the plot burns. Over the 12 years of experimental burning, John has observed that plants burnt at 2-year intervals are looking somewhat stressed, and suggests that intervals in the 5-8 year range may be more sustainable (John Briggs, pers. comm. 2006).

2.3.3 Trees

The potential of fire to assist efforts to establish woodland eucalypts has already been explored in Section 2.2.3. In this section, four articles related to tree regeneration specifically in Yellow Box – Blakely's Red Gum woodlands are summarized. Three report on experimental studies, while Banks (1997) describes a survey of Yellow Box, and summarises what is known of this species.

According to Banks (1997), Yellow Box trees can live up to 400 years, although the mature individuals he surveyed around Canberra were between 110 and 180 years old. While Yellow Box is generally found in multi-aged stands, young trees are not particularly common. In healthy woodlands, there is a pool of suppressed lignotuberous seedlings; suppressed because of intense competition from established overstorey trees. These suppressed seedlings persist as low shrubs, resprouting from lignotubers when damaged, but not growing up into the canopy until a gap occurs, when a race for space will eventually result in one of them replacing the previous tree. According to Banks (1997:43) "Fire imposes little direct impact on these woodland trees, although it can have several important indirect effects." The thick bark at the base of Yellow Box trees protects them from damage by fire, which tends to be of relatively low intensity as fuels are mostly grasses. Yellow Box seedlings establish periodically, "but only when the necessary conditions are achieved and in the right sequence: seed, a sterilised mineral soil seed bed, moisture and freedom from competition" (Banks 1997:44). Fire can provide suitable seedbed conditions, and reduce competition.

Li *et al.* (2003) studied various aspects of the germination, survival and early growth of *E. blakelyi* seedlings in degraded Yellow Box - Blakely's Red Gum woodlands on the New England Tablelands. Temperatures below freezing inhibited germination, and cold killed seedlings. Seed-bed treatments affected seedling survival: grazing and fertilizer decreased survival, while cultivation and fire increased it. The positive effects of fire and cultivation were attributed to reduced competition from herbaceous species and creation of gaps in litter. The negative effects of fertilizer were attributed to increased grazing pressure, a factor that may also have delayed the positive response to fire – the effects of fire on seedling growth didn't become significant until 30 months after treatment. Messages from this careful study include the importance of limiting post-fire grazing so that young eucalypts have time to establish. It may also be a good idea to burn in spring rather than autumn, so that young eucalypts are not immediately subject to winter frosts (Banks 1997).

Although fire may aid regeneration, establishment can also occur in the absence of fire. Clarke (2002) tracked the survival of young eucalypts germinating from sown seed on the New England Tablelands over five years: species studied included *E. melliodora* and *E. blakelyi*. While most seedlings died, a few of each eucalypt species planted into experimental paddocks survived to form lignotuberous 'suppressed seedlings'. Clarke (2002) also remarks that widespread natural eucalypt recruitment occurred after significant rainfall in January 1996 on the New England Tablelands, presumably in the absence of fire.

Semple and Koen (2001) studied the effects of short-term crash grazing on small eucalypts in what was once Southern Tableland Grassy Woodland near Orange. Tree species in the vicinity included *E. melliodora*, *E. blakelyi* and *E. bridgesiana*. Lignotuberous resprouts and new seedlings arising after a fuel reduction burn in *Themeda* grassland were browsed more heavily when grazed by sheep in spring than in autumn. Seedling deaths were also greater after spring grazing. Growth measurements led to the conclusion that it would take at least five years for new seedlings to grow above the browsing height of sheep in this tablelands environment (Semple and Koen 2001). The authors suggest that in this ecosystem, occasional crash grazing in spring might be compatible with seedling survival. However they caution against assuming similar results would be obtained elsewhere.

While these studies do not provide definitive guidance for landholders wishing to use fire to encourage the establishment of young eucalypts in Southern Tableland Grassy Woodland remnants, they do suggest a starting point for experimentation. Spring burning in places where eucalypts are carrying a good supply of seed, followed by several years of minimal grazing pressure, looks like a plan. In some places it may be as important to limit access by native animals drawn to the post-fire green pick, as it is to exclude domestic stock. Good post-fire rainfall may be an essential element in post-fire eucalypt establishment. Although long-range weather forecasts may help plan for this contingency, it will remain at least partly a matter of luck. Thus establishment may be better after some fires (those followed by post-fire rainfall) than after others (those followed by drought).

2.3.4 Shrubs

Fire appears to play a role in the recruitment of a range of tablelands shrub species. Knox and Clarke (2006) studied the effects of fire on shrub recruitment in New England Grassy Woodlands, which share many species with their southern tablelands counterpart, including the dominant eucalypts *E. melliodora*, *E. blakelyi*, and *E. bridgesiana*. Fire enhanced emergence of seedlings in the majority of shrubs tested, particularly legumes including the wattle *Acacia dealbata* which is also found in Southern Tableland Grassy Woodlands. Post-fire seedling emergence was greater after fires in spring than after fires in autumn. It was also greater after hotter experimental burns. The authors conclude that “higher intensity spring fires that release seeds from canopies, break dormancy and reduce competition are likely to result in enhanced seed recruitment in these temperate woodlands” (Knox and Clarke 2006:738). In contrast, few seedlings of shrubs whose seeds were sown into unburnt woodland in an earlier study by Clarke (2002), survived.

Many years ago Leigh (1975) questioned why some shrub species seemed to be disappearing in certain woodlands, including communities dominated by *E. melliodora*, *E. blakelyi* and *E. mannifera*. He suggested that kangaroo grazing might be preventing regeneration of leguminous shrubs, a conclusion in line with research he later published (Leigh and Holgate 1979 – see Section 3.1.3). An alternative, or perhaps a complementary, explanation, is that recruitment in many woodland shrub species is uncommon in the absence of fire.

2.3.5 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 prep.). 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). Young eucalypts may need several disturbance-free years if they are to survive.

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.

2.3.6 Conclusion

From the small number of studies which touch on matters to do with fire and biodiversity in Southern Tableland and other Yellow Box woodlands, it appears that the ecology of this vegetation class is very similar to that of the Western Slopes Grassy Woodlands discussed in Section 2.2. Again, intervals in the lower half of the state-wide range – perhaps every 5 to 15 or 20 years – may produce the best biodiversity outcomes.

3 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 directly addressing the role of fire in the dry sclerophyll forests of the Central West has not been extensive. However studies relevant to the three vegetation classes selected for discussion in this section can be found – though much of this work is from outside the Central West region. These vegetation classes are Southern Tableland Dry Sclerophyll Forests (Section 3.1), Pilliga Outwash Dry Sclerophyll Forests (Section 3.2) and Western Slopes Dry Sclerophyll Forests (Section 3.3). Keith (2004) places the first and third of these classes in the shrubby subformation of dry sclerophyll forests, while the Pilliga Outwash forests are allocated to the shrub/grass subformation.

3.1 Southern Tableland Dry Sclerophyll Forests

3.1.1 Introduction

Southern Tableland Dry Sclerophyll Forests occur on the central and southern tablelands between 600 and 1100 m above sea level, on shallow, infertile soils often in rugged, rocky terrain. This is not productive country: despite reasonable rainfall trees generally reach only 15-20 m in height. Sclerophyllous shrubs form an open understorey which, while relatively species-poor when compared to that of dry sclerophyll forest classes such as those found around Sydney and the Blue Mountains, holds considerably more shrub species than the grassy woodland vegetation classes discussed in this review so far. 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* (Inland 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 *Acacia buxifolia*, *Brachyloma daphnoides, Daviesia mimosoides*, *Daviesia latifolia*, *Exocarpos strictus*, *Hibbertia obtusifolia* and *Melichrus urceolatus*. *Joycea pallida* (Silvertop Wallaby Grass) is commonly found in the ground layer.

In the Central West, Southern Tableland Dry Sclerophyll Forests reach the northern limits of their distribution. They occur, in remnants of various sizes, around Oberon, Orange, Bathurst, Rylstone and Mudgee, where they were cleared less enthusiastically than the surrounding grassy woodlands on more fertile soils (Fisher and Harris 1999, Keith 2004). From here this vegetation class stretches south, occupying extensive areas on the drier parts of the southern tablelands. Places where Southern Tableland Dry Sclerophyll Forests can be found include the hills north of Goulburn, Black Mountain in the ACT, the foothills of the Kosciuszko main range and the gorges around Byadbo. 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).

3.1.2 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 Central West 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 of resprouting shrub species 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 (Purdie 1977b).
- All the resprouters which didn't produce seedlings were geophytic monocots, a category which includes 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 goes by in this vegetation type 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 on Black Mountain. This conclusion is consistent with what one might expect from a vegetation type that is, in Bond *et al.* (2003, 2005) terms, climate limited but fire modified (Section 1.5).

Purdie's findings 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).

Post-fire observations in Southern Tableland Dry Sclerophyll Forests in the Central West suggest that the proportion of shrub species regenerating solely from seed (obligate seeders) may vary considerably across the range of this vegetation class. Where only three species in Purdie's Black Mountain sites regenerated solely from seed, the state forest south of Kandos appears to support quite a large variety of obligate seeder shrubs (pers. obs. 2007), including *Acacia buxifolia*, a species which resprouted from root suckers in the Black Mountain plots (Purdie and Slatyer 1976). Remnants around Orange may also contain a relatively large obligate seeder complement (Bower 2005).

Observation in long-unburnt sites in the Central West also suggests that a small number of shrub species may be able to recruit substantially between fires in this environment. Candidates include *Styphelia triflora* (Pink Five Corners), *Cassinia arcuata* (Sifton Bush) and perhaps some other *Cassinia* and *Leptospermum* species. If this is the case, these shrubs are likely to be more abundant in long unburnt areas than where fire has occurred recently, and may become a more prominent component of the shrub layer as short-lived fire-cued shrub species die out. A survey in South Mullion reserve after an extensive fire-free period (up to 100 years) found large populations of Sifton Bush and Five Corners, both obligate seeders (Bower 2005). The two other shrubs which were abundant in this long-unburnt environment, *Monotoca scoparia* and *Melichrus urceolatus*, both resprout in most environments where their regeneration mechanisms have been observed (DEC 2002). These species may survive through long-term persistence of existing individuals. Surprisingly, few researchers have explored whether periodic fire plays a role in allowing long-lived resprouting shrubs to continue to thrive, although the lower levels of post-fire recovery in Purdie's long unburnt plot (Purdie 1977a) suggests this may be the case. A range of other shrub species were also present in this long-unburnt reserve, suggesting that even fire-cued species have the ability to 'hold on' in the absence of fire, either through persistence of long-lived individuals or through some germination between fires. However as Purdie's work demonstrates, fire is likely to play an important role in maintaining robust populations of quite a few shrubs in Southern Tableland Dry Sclerophyll Forests. Although soil-stored seed can survive for many years, it will not remain viable indefinitely (Floyd 1976, Auld 1986, Auld *et al.* 2000, Moles *et al.* 2003).

3.1.3 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 tablelands. One site, at Mundoonen Nature Reserve near Yass, was classic Southern Tableland 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 some parts of the central tablelands as numbers of macropods and wombats build up (pers. comm. various land managers in the CW region, 2007).

3.1.4 Conclusion

Although fire ecology research in Southern Tableland 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 in the Central West.

Thinking first about lower thresholds, the risks to obligate seeders of very short interfire intervals (Sections 1.4) of course 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 four or five years are thus unlikely to cause species to be lost from this community due to inability to establish a seedbank.

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 Tableland 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 many T species shrubs that increase when vegetation in some other climates is left unburnt for a long time (Section 1.5) – though see caveat in last paragraph of Section 3.1.2. 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 patches of Southern Tableland Dry Sclerophyll Forest is not known.

The greater risk, at this end of the fire frequency spectrum, is that I species 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 1977b), suggesting seedbanks at 30-35 years post-fire are still adequate. However this may not be the case four or five decades post-fire, nor for all species. 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 3.1.3). 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, while still allowing species which produce most of their seed-crop in the early post-fire years the opportunity to build up population numbers through some short intervals now and then.

3.2 Pilliga Outwash Dry Sclerophyll Forests

3.2.1 Introduction

‘Outwash’ landforms are “sandy plains where alluvial sediments from the elevated rocky areas are deposited” (Keith 2004:140). This vegetation class, which is found to the west of the shrubby Western Slopes Dry Sclerophyll Forest class (Section 3.3) on relatively deep and fertile soils, falls within the shrub/grass subformation of dry sclerophyll forests. Pilliga Outwash Dry Sclerophyll Forests occur almost exclusively in the Central West and Namoi CMA regions, where they represent “a transition from the dry sclerophyll forests of the more humid regions in the east to the semi-arid woodlands of the dry interior of New South Wales” (Keith 2004:140). Trees are box eucalypts (White Box *Eucalyptus albens*, Grey Box *E. microcarpa*, Fuzzy Box *E. conica*, Pilliga Box *E. pilligaensis*, and Poplar Box *E. populnea* ssp. *bimbil*); ironbarks (Narrow-leafed Ironbark *E. crebra*, Silver Ironbark *E. melanophloia*, Blue-leaved Ironbark *E. nubila*, Mugga Ironbark *E. sideroxylon*), Tumbledown Red Gum (*E. dealbata*), White Cypress Pine (*Callitris glaucophylla*) and casuarinas (Bulloak *Allocasuarina luehmannii* and Belah *Casuarina cristata*). Shrubs include both sclerophyll and semi-arid species. Sclerophyll species include wattles (eg Dean’s Wattle *Acacia deanei*, *A. hakeoidea*, Streaked Wattle *A. lineata*, Golden-top Wattle *A. tindaleae*), peas (eg *Dillwynia sieberi* and *Indigofera australis*) and species from the Myrtaceae family (eg Broombush *Melaleuca uncinata, Homoranthus flavescens*). Semi-arid shrubs include Wilga (*Geijera parviflora*), Budda (*Eremophila mitchellii*) and Wild Orange (*Capparis mitchelii*). Grasses and forbs make up the ground layer (Keith 2004).

In this section we explore a phenomenon that has received a great deal of attention: the ‘thickening’ of some dry sclerophyll forest classes of the western slopes and plains by copious regeneration of White Cypress Pine. This story has been vividly told by Eric Rolls (1981) in his book *A Million Wild Acres*, and has been the subject of much debate (eg Norris *et al.* 1991, Bowman 2001, Griffiths 2002, Keith 2004). It has relevance both for the Pilliga Outwash forests which are the focus of this section, and for other vegetation classes which contain this species.

3.2.2 White Cypress Pine, *Callitris glaucophylla*

White Cypress Pine is a widespread species; it is found in all mainland states and is a component of many vegetation classes in the Central West (Bowman and Harris 1995, Metcalfe *et al.* 2003, Keith 2004, Thompson and Eldridge 2005b). It is well-known as a dominant tree species in the Pilliga Forest, particularly in the Western Pilliga. Unlike most trees in Australia *Callitris* species are not flowering plants (Angiosperms), they are conifers and Gymnosperms (plants which produce seed on the surface of cone scales, rather than in an ovary). *Callitris* is one of only two Australian conifer genera adapted to low rainfall environments (Bowman and Harris 1995). *C. glaucophylla* grows in areas with an average annual rainfall of 200 to 750 mm, on sandy, well-drained soils and in rocky areas where soils are skeletal (Thompson and Eldridge 2005a). Much of the land once covered by the vegetation classes containing White Cypress Pine has been cleared, with the result that it now often occurs as scattered trees, on rocky ridges or in small forested areas (Harris and Lamb 2004, Thompson and Eldridge 2005b).

White cypress pine recruits episodically, in years of good rainfall (Ross *et al.* in press). Rain in autumn and winter encourages germination, while follow-up rain the subsequent summer appears to be essential for seedlings to establish (Lacey 1972). When a recruitment event does occur, the density of young pines can be quite stunning: Lacey (1972) reports rates of up to 2.5 million seedlings per hectare. Unlike eucalypts, White Cypress Pine does not easily self-thin: thick stands of young spindly trees can persist for decades. Parker and Lunt (2000), for example, report densities of 92,500 plants per hectare in 40-year-old stands in fenced plots west of Echuca in Victoria, while in state forests south of Cobar Allan (1995) recorded many stands from the late 1800s which were still severely 'locked' 100 years later. Growth rates in these locked stands are very low (Horne 1990b, Ross *et al.* in press), but increase when stand densities are reduced (Henry 1960, Lacey 1973, Horne 1990a).

3.2.3 Forests of the past

What did the White Cypress Pine forests and woodlands of NSW look like 200 years ago?

A number of authors have used historical records to develop a picture of the landscape prior to European settlement. Many have focussed on vegetation structure: the density of trees and shrubs, and the balance between grassy and woody plants. Other researchers have used counts of old trees and tree stumps to reconstruct the forests and woodlands of the past. Here are some of their findings:

- Vegetation types over the Central West and adjoining regions (Namoi, Lachlan) formed a mosaic, as is commonly found in landscapes elsewhere. Much of the country was open woodland with a grassy understorey, however there were also areas where shrubs and/or small trees grew in abundance (Benson and Redpath 1997, Croft *et al.* 1997, Allen 1998, Keith 2004).
- Cypress pine was a prominent component of many vegetation types in central New South Wales at the time of European settlement. For example in a study south of Tottenham of trees used as markers by 19th century surveyors, Martin (2005) found 19% of trees used were cypress pines; only box eucalypts were used more often. As trunks had to be fairly large to allow room for survey markings, Martin (2005:170) concluded that "large pines were a prominent component of the vegetation during the 19th century on both the flatter country and the ridges."
- In many places, big, mature cypress pines co-occurred with big, mature eucalypts. The density of large trees appears to have been of the order of 20 to 40 per hectare (Benson and Redpath 1997, Martin 2005). Eucalypts appear to have been more abundant, at least in many areas, than cypress pine. Paull (2001), who used stump counts to estimate tree density in 1910 in 20 one hectare plots in the Pilliga West, found that while there was some variation in the relative numbers of the two canopy species, White Cypress Pine and Narrow-leaved Ironbark (*Eucalyptus crebra*), on average plots contained 25 large eucalypts to five large pines. Pine numbers never got above 9 per hectare, and the more ironbarks in a plot, the less pines. Lunt *et al.* (2006) also used stump

counts to estimate pre-European tree density across a wide area of central NSW south from Nyngan. The estimated average density of trees with a diameter of 20 cm or more at the time of settlement was 39 per hectare. Seventy-eight percent of these trees were eucalypts, 21% pines. Thus *Callitris glaucophylla* was an important, but sub-dominant, component of the pre-European canopy in these stands. Allen (1995), however, recorded approximately equal numbers of pre-1870 pines and eucalypts in two minimally disturbed state forests south of Cobar, suggesting that White Cypress Pine was a more important component of the canopy in the west than in the east of its distribution in NSW.

- Thickets of small cypress pines, and multi-aged pine stands, were a feature of the landscapes traversed by the first Europeans to explore the Central West. Several authors (eg Mitchell 1991:176, Martin 2005:170) cite extracts from Oxley's journals describing his 1817 exploration of the Lachlan River valley, which included phrases such as " …. after going through about eight miles of very thick cypress scrub ," and "it was a continued scrub, and where there was timber it chiefly consisted of small cypress…" Allen (1998:114-5) quotes Cunningham, who reported cypress pines "of various sizes and dimensions from seedlings, generally growing in clumps, to lofty trees of about 60 feet."
- Allen (1998) points out that pre-European cypress pine forests were dynamic, with *Callitris* cohorts going through a cycle from dense seedlings to scattered mature veterans. He suggests that the cycle would have played out more rapidly in some times and places than others, and that the forest in different parts of the landscape would have been in different stages of the cycle. Patch sizes no doubt also differed across the landscape. The patchy landscape encountered by the explorers, Allan points out, would have been a snapshot of this dynamic landscape at a particular point in time.

3.2.4 Changed forests

How have White Cypress Pine forests changed over the last 200 years?

Environmental historians agree that by the 1870s, extensive regrowth of cypress pine was occurring in what had formerly been relatively open woodland areas (Mitchell 1991, Norris *et al.* 1991, Allen 1998, Parker and Lunt 2000). Strong documentary support for this contention comes from reports requested by the New South Wales Surveyor General in 1880. Land Commissioners in the Lachlan and Murrumbidgee districts described the difficulties caused by pine scrubs which had sprung up over the past 15 years. The problem was less acute around Narromine and Dubbo, although "scrub patches were believed to be increasing in density" (Mitchell 1991:177). Pine regeneration in the Central West (Allan 1995) and the Pilliga appears to have occurred somewhat later than that further south. Rolls (1981) contends higher than average rainfall in several years between 1879 and 1887 was important for pine regrowth in the Pilliga, while Norris *et al.* (1991) argues that wet years between 1889 to 1894 are more likely to have formed the precursor.

There is also general agreement that very little cypress pine recruitment occurred in NSW during the first half of the 20th century (Lacey 1972, Rolls 1981, Norris *et al.* 1991, Allen 1998). In the 1950s, however, a series of wet years coincided with dramatic rates of pine seedling establishment (Mitchell 1991, Allen 1998).

These waves of regeneration were clearly distinguished by Allan (1995), who surveyed pine age and size classes in Cumbine and Nangerybone State Forests south of Cobar. These forests were unusual in having been subject to only limited logging and thinning. Pines haled either from before 1870 ('old greys', with an average density of about 29/ha), from 1870 to 1900 (mostly in dense, but patchy, locked stands), or from post 1950 (scattered individuals to dense clumps).

The other major change to forest and woodland structure over the past 200 years is, of course, the opening up and fragmentation of much of the landscape through ringbarking and clearing of trees. Conditions in scrub and improvement leases specified the removal of all mid-storey, and most non-pine canopy species, particularly eucalypts (Allen 1998). Forestry practice till quite recently involved removal of eucalypts in order to enhance the growth of cypress pine (Lacey 1973, Johnson and Jennings 1991, Allen 1998, Thompson and Eldridge 2005a). Paull (2001) estimated > 90% of the large, mature ironbarks had been removed from his plots in the Pilliga West Forest since 1910, leaving just 2 stems per ha over 50 cm diameter. Only three large *Callitris* remained in all Paull's study plots put together.

Where trees have regrown, forest structure and canopy composition is now very different to what it used to be. Lunt *et al.* (2006) compared current and pre-European tree densities in 39 minimally-disturbed stands across the *Eucalyptus*-cypress pine woodland belt of central NSW between Nyngan and Corowa. Living trees in current stands averaged almost 1500 per hectare, although there was a lot of variation between stands. Large trees (over 20 cm dbh) now number 198 per hectare on average, compared to 39 in pre-European times. Where at the time of settlement 78% of the basal area was contributed by eucalypts, 74% now comes from *Callitris*. Although numbers of trees have increased, basal area has stayed relatively stable. This is because eucalypt trees have larger stems and wider canopies than cypress pines. Allen (1998:v), who explored the history of three pine forests in the Lachlan, concludes that "through twentieth century management practices all three cypress pine forests are now converging to a similar stage and contain vastly fewer hardwoods than several centuries ago."

3.2.5 Causes of change

What caused the mostly open forests of the early 1800s to thicken up? A number of factors almost certainly combined to create the conditions that allowed White Cypress Pine seedlings to establish so profusedly.

Logging

The loss of many large eucalypts appears to have been almost ubiquitous across the forest and woodland landscapes west of the Divide, even in the least disturbed patches (Lunt *et al.* 2006). Large cypress pines have also been logged out. Thus reduction in competition from large, mature trees is one possible explanation for the massive establishment of pine seedlings.

In a paper prepared for the then Forestry Commission of NSW, Lacey (1972) summarises the relationship as follows: "In the absence of grazing there is an obvious relationship between stand density and the amount of regeneration which becomes established. Overstocked stands have very little regeneration while understocked stands are seldom without it." He attributes this relationship to both lower competition in understocked stands, and to the fact that seed set is higher when pine density is low. More recently, strong support for the ability of fully stocked stands to limit regeneration has been provided through modelling work by Ross *et al.* (in press). These researchers used data from thinning trials in state forests to develop and test their model, which simulates the dynamics of *Callitris glaucophylla* in Central Western NSW. Work to date has focussed on monospecific stands, over the period 1948 to 1998, in the absence of fire. Besides confirming the importance of high rainfall years for recruitment, both actual and modelled recruitment clearly decreased with increasing stand basal area. By 1998, recruits from the 1950s into understocked stands had only grown to between 2.5 and 5.5 m, and stand densities had stabilised at around 20 000 stems per hectare. This research team plans to add eucalypts and fire to their model in the future.

It is possible that mature eucalypts may be even more effective than mature pines at limiting pine recruitment. In Cumbine and Nangerybone forests patches with fewer big old cypress pines relative to large old eucalypts had considerably less regrowth from the late 1800s, although post-1950s regrowth was present in these patches (Allan 1995). In the Northern Territory Bowman *et al.* (1988) found reduced growth of *Callitris intratropica* saplings under a eucalypt canopy. In the Snowy River Valley, Clayton-Green and Ashton (1990) found that White Cypress Pine density was lower under White Box (*Eucalyptus albens*) canopies than between them.

There are indications, however, that dense pine regeneration can occur even where a mature tree canopy is intact. We have already noted that early explorers reported patches of dense pine scrub. Allen (1998), who documented the history of three forests in the Lachlan, found that all experienced broad-scale regeneration of cypress pine during the late 1870s and early 1880s, while only one had had any ringbarking at that stage (Allen 1998). Paull (2001) cites a forestry report from 1910, thought to date from before major logging took place, which talks of "an enormous forest" of immature White Cypress in the western Pilliga.

Thus while removal of canopy competition undoubtedly played a part in allowing the dense regrowth stands of the 1870s and 1880s to develop, it seems likely that other factors were also at work.

Changed grazing regimes

Grazing regimes have changed throughout the grassy forests and woodlands of the Central West. Where once kangaroos, wallabies and smaller herbivorous marsupials browsed, sheep, rabbits and cattle have progressively taken their place.

There is general agreement that rabbits were a major factor in preventing pine regrowth in first half of the 20th century. Their arrival in the late 1800s is close to the time when the first wave of pine regeneration ceased, and their rapid demise from myxomatosis in the early 1950s matches the commencement of the second wave of pine (Mitchell 1991, Allen 1995). Exclosure experiments have also been conclusive (Lacey 1972). For example in an experiment in south Queensland in the 1950s 27 young pines established in a plot where rabbits and domestic stock had been excluded, while no seedlings at all survived in plots grazed by rabbits and sheep and only one where sheep were excluded but rabbits could get in (Johnston 1969).

The effects of stock grazing are less clear-cut. It has been suggested that grazing may encourage pine regeneration though its influence on the composition and abundance of ground layer plants. A decrease in herbaceous cover might encourage the establishment of *Callitris* seedlings through reduction in competition. A change in composition from deep-rooted perennial grasses and herbs to shallow-rooted annuals might provide more sub-soil water which could assist cypress seedings to survive (Allan 1995). Allen (1998) points out the wave of regeneration in the late 19th century occurred during an era of widespread sheep grazing. Walker (1976) found more cypress pine regeneration on grazed than on ungrazed plots near Cobar.

There are strong indications, however, that in general, stock grazing is likely to work *against* the establishment of White Cypress Pine seedlings. These seedlings are palatable when young (Lacey 1972), and exclosure experiments have shown that sheep, at least, can readily destroy seedlings (Curtin 1987). For example Parker and Lunt (2000) report dense White Cypress Pine regeneration from the 1950s in exclosures west of Echuca in Victoria, while heavy grazing by sheep outside has prevented establishment. As to the impact of grasses, it appears that dense grass may encourage, rather than discourage, cypress seedlings. Lacey (1972) reports that abundant grass cover and abundant *Callitris glaucophylla* regeneration often occur together, both being greater in stands with a low abundance of pine. In the south Queensland exclosure experiment described above, ground cover facilitated the survival of pine seedlings by protecting them from the harsh effects of summer sun (Johnson 1969).

Also controversial is the role of native herbivores in reducing survival rates of cypress pine seedlings. Noble (2001) hypothesises, on the basis of many years of observation of shrub regeneration in semi-arid woodlands, that marsupial browsing was an important element in the regulation of scrub regeneration prior to European settlement.

Changed fire regimes

While we can never be sure what fire regimes pertained prior to European settlement, there is no doubt fire was a component of the landscapes where White Cypress Pine is found. Ignition sources were almost certain twofold: lightning strikes and burning by Aborigines (Allan 1998). Mitchell recorded Aboriginal fires in 1835 near the cypress pine forests studied by Allan (1995). In a review of the ecology of White Cypress Pine in Queensland, Harris and Lamb (2004:25) contend that “evidence, in the form of fire-scarred trunks, suggests that the majority of cypress forests incurred regular burning prior to being reserved as State Forests….” In the Northern Territory frequent low-intensity Aboriginal burns were, and in some places continue to be, a feature of the environment where *Callitris intratropica* grows (Bowman *et al.* 2001, Yibarbuk *et al.* 2001). It is likely that the frequency and intensity of fire varied across the landscape due to the effects of natural features (for example soil type would have affected grass growth which would in turn have affected fire). As well, Aboriginal management is unlikely to have been uniform.

Although again we can never know in detail, it appears that with European settlement, fire became increasingly less common in the White Cypress Pine landscapes of NSW.[7]

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[7] While this statement is likely to be accurate for the forests of the NSW slopes and plains, in the outlier White Cypress Pine woodlands of the Snowy River Valley, fire frequency appears to have increased with European settlement, with regular burning for green pick between 1840 and 1910. After this, fire frequency gradually declined (Pulsford *et al.* 1993).

We have already noted that stock grazing may have reduced fuels, and thus the intensity and frequency of fire. The need for grass to feed sheep and cattle probably also encouraged graziers to suppress fires wherever possible. Reduced fuel loads due to grazing, and fragmentation due to clearing, almost certainly made fire suppression increasingly viable. Thompson and Eldridge (2005b:564), in a review of the literature on White Cypress Pine, assert that "burning cycles ceased within the cypress pine zone with the expansion of agriculture and pastoralism by the early Europeans." Fire exclusion has been normal practice in cypress pine forests managed for forestry (Henry 1960, Lacey 1973, Johnson and Jennings 1991, Allan 1998).

From the early days of European settlement some observers have asserted that a reduction in fire frequency encourages thick pine regrowth. This view was expressed to the Surveyor General in 1880 (Mitchell 1991), and was put to the Western Lands Royal Commission in 1901 (Allan 1995). More recently, the potential for fire to help manage White Cypress Pine forests, and particularly its potential to help in reducing the density of young pines in locked stands, has been pointed out, as has the urgent need for research (Johnson and Jennings 1991, Date *et al.* 2002, Andrews 2003).

The potential of fire to regulate the abundance of White Cypress Pine is discussed in more detail in Section 3.2.7 below.

3.2.6 Impacts of change

What effects have changes in the structure of White Cypress Pine forests and woodlands had on biodiversity and other aspects of the environment? Effects on plants, on animals, and on soils are considered in turn.

Flora

It is often said that dense cypress regrowth suppresses grass cover (eg Lacey 1973, Harris *et al.* 2003, Cameron nd). Logically this makes sense, as grasses are in competition with the young trees. It also fits with the findings of many studies of effects of trees in general on grass production. Trees affect the growth of understorey plants by altering the availability of resources, particularly light, water and nutrients. While some studies have documented increased productivity under trees (eg Belsky *et al.* 1993), many others have found lower herbaceous biomass under trees and/or shrubs (Engle *et al.* 1987, Archer 1990, Scanlan and Burrows 1990), or that ground layer productivity increases when trees and/or shrubs are removed (eg Walker *et al.* 1986, Harrington and Johns 1990).

Studies seeking to document trends in ground cover specifically in relation to White Cypress Pine have produced equivocal results. That ground cover can be very low is attested by Shelly (1998a) who estimated cover of living ground plants in pine-box woodland with dense pine regrowth near West Wyalong at 5%, with litter covering an additional 45% of the ground surface. In Central Australia Bowman and Latz (1993) found less grass cover in quadrats containing White Cypress Pine than in quadrats without this species, however differences were not significant. Measurements by Greening Australia in a series of regrowth plots on the north-west slopes found big site-to-site differences, with grass and herb cover ranging from 4 to 55%. The average of 23% was much less than in adjacent pasture, where it was over 70% (Andrews 2003).

Work by Chris Nadolny reported in Andrews' paper found a reduction in the cover of ground layer vegetation as cover of White Cypress Pine increased, however the negative effect of grazing on ground cover vegetation was much greater than that of pine (Andrews 2003). Nadolny points out that there is probably an interaction here: grazed plants take longer to recover under dense Cypress Pine because of competition between the two life forms.

However a study designed to assess trends in understorey vegetation with varying density of cypress pine across a wide geographic area (Thompson and Eldridge 2005a) found no association between canopy cover and cover of plant species, whether mid-storey species were included or excluded. There was, however, an east-west trend across the rainfall gradient sampled (530 to 215 mm a year) with significantly more understorey cover in higher rainfall sites. The findings of this study were unexpected, and the authors consider that differences in grazing history, and also lack of rainfall during sampling, may have affected the results.

It may be that ground cover develops slowly after pines have been thinned – and equally may decline slowly as they thicken (Eldridge *et al.* 2003). Monitoring of thinned regeneration at the Western Plains Zoo near Dubbo showed much higher levels of ground cover in plots which had been thinned some years previously than in more recently thinned and unthinned stands (Cameron nd). Eucalypt regeneration appeared to have been encouraged by thinning, and the health of mature hollow-bearing eucalypt trees was better in thinned stands. Heavy grazing, mostly by native herbivores, probably limited recovery of ground layer plants in some places.

Even if cover of ground layer plants is reduced under cypress pine regrowth, this does not necessarily mean that diversity is also affected. Nadolny (in Andrews 2003) found no differences in the diversity of native species in plots with different levels of cypress pine cover. Similarly, in the study by Thompson and Eldridge (2005a) mentioned above, species richness did not decline with increasing Cypress Pine canopy cover, though again there was a trend for larger numbers of species in the east of the study area, and results may have been influenced by grazing and drought. Unfortunately, this paper did not distinguish between native and exotic species. Data were also collected on non-vascular plants – "complex assemblages of mosses, liverworts and lichens that form a biological crust on the soil surface" (Thompson *et al.* 2006:271). There were many non-vascular plant species in study sites – numbers varied from three to 33 with a mean species richness of 20 – but no clear relationship between canopy cover and cover or species richness of these plants emerged (Thompson *et al.* 2006).

On the other hand, Bowman and Latz (1993) found many species, particularly herbs, only in plots where *Callitris glaucophylla* was dead or absent. And after extensive vegetation survey work through the Pilliga forest, Date *et al.* (2002) found the understorey in heavily logged and long unburnt areas with cypress regeneration supported a much lower range of understorey species than lightly logged patches which had burned every 20 years or so.

Fauna

Little research directly addresses the effects of changes in forest and woodland structure on fauna. The most cited work comes from Cape York, where granivorous birds have declined as a result of encroachment by *Melaleuca* species into open habitats (Crowley and Garnett 1998, Crowley 2001). A study of "Woody Weeds and Biodiversity in

Western NSW" by Ayers *et al.* (2001) also found that the abundance of ground feeding birds decreased as shrub cover increased.[8] While fauna research in White Cypress Pine communities is not extensive, there are suggestions that some birds and other animal species may be similarly affected.

Changes in forest composition and structure have been invoked by Date *et al.* (2002) to explain their findings in the Pilliga. In the early 1990s these researchers surveyed 568 transects right across the forest, for birds. On 230 of these transects vegetation structure and plant species were also recorded, along with disturbance history. They found a much greater diversity of birds in patches along creeks where old trees were still to be found. These patches had a diverse shrub understorey, low cover in the subcanopy, and had experienced moderately frequent wildfires. The majority of forest, however, supported a smaller number of generalist bird species, which also occurred in the creeklines. These "non creek" patches had less shrub cover, more subcanopy cover, had been more heavily logged and had experienced very little fire over the last 40 years. Many of the species found only in the creeklines were either ground-foraging and/or ground nesting species, or species dependent on mature trees for food and nest sites. Date *et al.* (2002) believe that logging, grazing and fire exclusion have created a more homogeneous environment than once existed in the Pilliga, leading to a simplification of vegetation and bird communities. They recommend management strategies to bring back mature trees, and "intermediate fire regimes" which they suggest would involve variable intervals between 2 and 25 years. This study is discussed further in Section 3.3.2.

Bird surveys after thinning experiments at the Western Plains Zoo near Dubbo suggest similar processes may be operating in the Central West where Cypress Pine has thickened. Species richness and abundance of native birds was greater in thinned than in unthinned stands, with the largest number of bird species (59 species) recorded in stands which had been thinned for the longest period (8 years). "There appeared to be an edge effect, where birds were utilizing the edges of dense pine stands but making little use of central areas" (Cameron nd:6). Cameron (nd) points out that the majority of 20 bird species considered 'in decline' across the NSW wheat belt are ground or low shrub feeders unlikely to find suitable habitat in dense stands of regenerating pine.

Birds have also been surveyed in Back Creek State Forest near West Wyalong, in a patch of pure Belah and in adjacent White Cypress Pine-Box woodland with dense pine regrowth (Shelly 1998a). Both environments had not burnt for many years. Twenty-nine bird species were recorded in the Belah thicket, and 40 in the pine-box woodlands. Dense pine regrowth was used by some birds, with one unknown species nesting extensively (Shelly 1998a). The number of bird species found at Back Creek is higher than the number found in unthinned pine thickets at Dubbo Zoo (28 species), but lower than the 59 species in the thinned plots there (Cameron nd). Shelly (1998a) suggests that the lower fauna diversity in the belah thickets may reflect the homogeneity of this habitat, which contained virtually no shrubs or groundcover, and thus few resources for

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[8] The Ayers *et al.* (2001) study, which involved a survey of 35 sites with varying degrees of shrub cover up to about 40%, found no overall effects of shrubs on species richness or abundance, although some individual species and species groups responded positively or negatively to increasing shrub cover. This research focussed on *Eremophila*, *Dodonaea* and *Senna* species. Localities with shrub cover over 40% were not included because it was not possible to find open areas for comparison.

birds. Conversely, Cameron (nd) suggests that the higher bird counts in the thinned stands at the Zoo may reflect the more varied habitat there.

One species which may be disadvantaged where fire is excluded and pine regeneration is dense, is the Pilliga Mouse *Pseudomys pilligaensis*. Individuals of this rare species, which is closely related to the New Holland Mouse (*Pseudomys novaehollandiae*), "were found in greatest abundance in recently burnt moist gullies," and in areas with "extensive cover of low grasses and sedges, little shrub cover and large areas of ash-covered ground" (DEC 2005). The better-studied New Holland Mouse frequents areas in the early stages of recovery from fire where plant diversity is high and vegetation height is low (Tasker and Dickman 2004 and references therein).

The replacement of eucalypts by pines is likely to have affected some fauna through a reduction in nectar and hollows (Date *et al.* 2002, Lunt *et al.* 2006). For example Cameron (nd) reports that sugar gliders at the Zoo near Dubbo were associated with mature and regenerating eucalypts, but did not use unthinned pine stands. Koalas in the Pilliga are generally found in unburnt patches where the density of eucalypts exceeds 20 per hectare (Kavanagh and Barrott 2001).

So do dense stands of Cypress Pine regrowth enhance or reduce faunal diversity? Andrews (2003:15) suggests the answer depends on their location: "Where they occur in heavily cleared landscapes for instance, patches of regrowth cypress forest may be important as dense foliage refuge habitat for birds and other fauna." In other words, these patches are likely to be an advance, in biodiversity terms, on completely cleared land. Major *et al.* (2003) have found that even small patches of White Cypress Pine provide good habitat for arboreal insects. However "pure stands have relatively few fauna habitat resources compared to other native woodlands and forests" (Andrews 2003:15). As we have seen, thinning of closed dense stands may enhance habitat value for some plant and animal species. Measures which restore eucalypts and/or native perennial grasses whilst retaining some of the mid-story cover are likely to be even more effective.

Erosion

It is sometimes said that the soil under cypress pine thickets is at risk of erosion, due to loss of vascular plant cover. However as Thompson and Eldridge (2005b) point out, "Although groundstorey (vascular plant) cover may decline under dense stands of *C. glaucophylla*, other types of cover (biological soil crusts and litter) frequently compensate." Thompson *et al.* (2006) found that the soil surface in cypress pine woodlands overall was moderately stable, though scores for infiltration and nutrient cycling were low. There was a very significant positive relationship between the species richness of soil crust taxa and scores for stability and infiltration.

3.2.7 Cypress Pine and fire

Research into the impacts of fire on White Cypress Pine is surprisingly limited. No one, to my knowledge, has experimented with the effects of different fire intervals or intensities in *Callitris glaucophylla* forests or woodlands, and even the effects of a single fire have rarely been documented. Two sources, however, provide useful information on the fire-related attributes of the species. The first is the extensive research undertaken in NSW and Queensland with a view to informing forestry management of White Cypress Pine; the second is a series of studies on *Callitris intratropica*, a closely-related species in the Northern Territory. Here are some of the findings:

- White Cypress Pine is classified as an obligate seeder (NSW Fire Response Database – DEC 2002). Individual pines are certainly more likely to die in a fire than the eucalypts that grow with them (Lacey 1972).
- This does not mean, however, that all *C. glaucophylla* plants are killed in every fire. Even in hot, extensive wildfires, a proportion survives (Hawkins 1966). For example Mitchell (1991) cites the NSW Forestry Commission Annual Report for 1951/2, which indicates that many mature cypress pines survived the 1951 wildfires in the Pilliga Forest. Surviving veterans from a fire in 1926 could still be seen in Strahorn State Forest seventy years later (Allan 1998). In the Snowy River Valley many mature trees survived multiple fires in the early decades of European settlement (Pulsford *et al.* 1993).

Figure 1. White Cypress Pine four years after a wildfire near Eugowra.

- Several characteristics of the species encourage survival in the face of fire. First, some plants are tall enough to get above the flames (Figure 1). Second, even after complete leaf scorch, some individuals survive through production of epicormic shoots (Lacey 1973). Third, White Cypress Pine litter is probably less flammable than that of co-occurring eucalypts and grasses (Lacey 1973): fuel loads in *Callitris intratropica* stands are lower, moister, and contain much less grass than those under nearby eucalypts (Bowman and Wilson 1988).

Fourth, dense stands of *Callitris glaucophylla* may naturally discourage fire spread by reducing wind speed (Thompson and Eldridge 2005b).

- Small *Callitris* plants are more vulnerable to being killed by fire than large plants (Bowman *et al.* 1988, Bowman and Panton 1993, Allen 1998, unpub data D. Taylor QFRI 2007).
- A fire shortly after a regeneration event can eliminate an entire cohort of young seedlings. Allen (1998) reports that this is definitely known to have happened in Euglo South State Forest in 1957. Once saplings have managed to get a few meters above grass height, however, if a fire occurs some generally survive (Figure 1).
- White Cypress Pine, like several other *Callitris* taxa, is a T species, able to recruit between fires (Section 1.5). As we have already seen, recruitment occurs episodically, in periods of higher than average rainfall, irrespective of fire. It may be some years after fire before any recruitment occurs (Allen 1998).
- White Cypress Pine seeds develop in cones and are released in November and December (Hawkins 1966). Once released seed remains viable for only a short time. Seeds have "negligible dormancy" and there is no long term seed store either in the soil or on the plant (Lacey 1972, 1973).
- Seed crops vary considerably between years, although even in poor years some seed is produced (Hawkins 1966). It is unclear whether reduced seed supply in poor years limits regeneration; Bowman and Harris (1995) conclude that it does not. As already noted when conditions are right rates of seed germination can be extremely high.
- Stand density influences seed production considerably: the more open the stand, the more seed is produced (Lacey 1972).
- Fire inhibits seed production. Hawkins (1966), who compared seed fall in burnt and unburnt patches, found a considerable reduction in the quantity and viability of seed produced in moderately burnt areas for five years post-fire, after which levels returned to normal. Almost no seed was produced for at least seven years post-fire in a patch where intense fire had killed most cypress pine trees.
- Although seed of *Callitris* species does not generally appear to travel far from parent plants (Bowman *et al.* 1988, Bowman and Harris 1995 and references therein), seedlings of *C. glaucophylla* have been found up to 370 m from source trees (Lacey 1973).
- *Callitris glaucophylla* reaches sexual maturity at between 6 and 12 years of age (Bowman and Harris 1995). Lacey (1972) reports individuals in the field producing cones at 6 years old when less than 1.2 m in height.
- The life span of White Cypress Pine is at least 200 years (Lacey 1973, Bowman and Harris 1995).

Taken together, these characteristics paint a picture of a species whose abundance is likely to be readily regulated by fire. Elements of the life history of White Cypress Pine suggest extreme vulnerability to burning. These elements are its obligate seeder status, its lack of soil-stored seed which could potentially survive through more than one fire, the suppressive effects of fire on seed production and its relatively long juvenile period. On the other hand, elements of its life history indicate an ability to survive in a fire-prone environment, particularly where fire intensity is low. These elements include the capacity to create a local environment where fire may be precluded or at least reduced in intensity, the ability of larger individuals to survive fire while smaller competitors are eliminated, the capacity for seed dispersal over moderate distances, the ability to recruit massively outside the window of opportunity provided by fire, and its longevity.

These latter elements also point to a species likely to do very well in the absence of fire. Fire exclusion, or long periods between fires, should allow numbers of White Cypress Pine to build up. As at least some seed is always available, recruitment will occur whenever climatic conditions are suitable, particularly in canopy gaps. By 6 to 12 years after recruitment, seed crops from young pines will be adding to those produced by older trees. If mature trees should die of old age, young trees will continue to ensure a seed source for further recruitment.

In fact, as already noted, long fire-free periods have seen the development of dense, locked stands of young cypress. In pre-European forests and woodlands, fire – along with competition from large adult trees and perhaps also predation by native animals – probably limited sapling abundance, with fire selectively killing many small pines while leaving others to survive and dominate. The presence of eucalypts is likely to have both limited pine recruitment through direct competition for resources, and also encouraged fire through the production of flammable litter – which in turn would have discouraged the pine. Periodic fire may well have encouraged native tussock grasses (Section 2.1.3), which again would have provided flammable fuel for the fires that favoured eucalypts over pines.[9]

This fire-mediated regulatory process may have taken place primarily in clearings created by death of large, old trees – where grass growth and curing, as well as pine recruitment, were probably highest. Harris and Lamb (2004), in a review of the ecology of White Cypress Pine in Queensland, conclude that "The occurrence of regular fires in cypress forests apparently resulted in an open forest structure with eucalypt dominants and scattered small clumps of large cypress pine." Clumping may have been encouraged as small pines were killed off in open grassy areas and on the edge of dense pine stands, while individuals in the middle of stands survived (Lacey 1973). This small-scale mosaic of multi-aged stands probably existed within a larger-scale mosaic of grassy and shrubby vegetation, which would have been influenced by soil conditions as well as by fire (Allan 1998).

3.2.8 Managing for conservation

Managing Cypress Pine forests for conservation and/or restoration will require intelligence, experimentation, and a number of management tools. While fire is likely to be a useful component of the tool kit, other tools, particularly ones to restore and maintain canopy cover, will also be needed.

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[9] It has recently been hypothesized that this 'shackling' of cypress by eucalypts, grasses and fire was most effective in higher rainfall regions where grass fuels are continuous much of the time (pers. comm. Ross Bradstock, University of Wollongong, 2007). As rainfall decreases through the semi-arid and into the arid zone, grass continuity declines, reducing fire frequency and allowing cypress species, with their impressive ability to survive drought, to increase in abundance relative to eucalypts.

Thinning alone is unlikely to stop pine recruitment nor restore the forests and woodlands in which it is found to an open structure. Ross *et al.* (in press) found that individual *Callitris glaucophylla* saplings grew *less* when stands were thinned *more*, because gaps allowed young pine to recruit. In some thinning treatments, additional recruitment meant that stands approached or exceeded pre-thinning densities in as little as 5 to 10 years. Cameron (nd) reports secondary regrowth after thinning near Dubbo. The delicate art of thinning pine is discussed in a user-friendly fashion in the Greening Australia landholder guide "Regrowth White Cypress Pine and Natural Resource Management" (Andrews 2003). Given the difficulties of managing secondary regrowth when thinning alone is used, the addition of fire to the toolbox looks attractive.

However fire too is likely to have limitations, particularly where landholders hope to combine conservation aims with stock grazing or the development of Cypress Pine as a commercial resource. Sparse grassy fuels often make it difficult to get a cool fire into dense cypress regrowth. Hotter fires which will travel through cypress foliage, however, are likely to destroy or damage most trees, and may be difficult to control (Andrews 2003).

Given the complexity, outlined above, in the relationships between fire frequency, intensity and extent, canopy composition and density, and the effects of grazing, it is impossible to predict exactly what fire regimes would result in stable, healthy populations of White Cypress Pine and other native species. The challenge will be to find regimes which are sufficiently frequent and intense to prevent lock-up, but sufficiently infrequent and/or mild to keep *Callitris glaucophylla* in the landscape. Almost certainly the answer will differ with climate, soil type, nature and abundance of co-occurring species (grasses, shrubs, eucalypts), extent of degradation, and topography.

Some general principles for management can, however, be put forward as a basis for experimentation by those wishing to enhance biodiversity values in White Cypress Pine forests and woodlands:

- Management actions which encourage local eucalypt species back into the canopy should assist in bringing forests into balance.
- Management actions which encourage deep-rooted native perennial grasses should do the same.
- Where Cypress Pine is dense, conservative grazing of stock is recommended to allow grasses to recover from the double whammy of defoliation and competition for resources. Native and feral herbivores may also need to be controlled (Andrews 2003).
- Management actions which encourage both eucalypts and cypress pines to develop into large, old trees should be given a high priority. Where old trees exist, they should be preserved.
- Fire clearly does have a role in White Cypress Pine forests and woodlands. Periodic fires will thin regeneration and should help establish dominance of smaller numbers of large cypress pines.
- Fire may encourage eucalypt regeneration, and native grasses and shrubs (Hawkins 1966, Sections 2.2.3, 2.1.3 and 3.1.2). After a wildfire in 1957 in Euglo South State Forest "Eucalypts and wattles re-emerged quickly in dense stands … and grass grew particularly well" (Allen 1998:104).

- Low intensity fires will kill fewer pines than high intensity fires – and these fires will be easier to control. For this reason it may be best to begin with a cool fire – assuming, of course, that there is sufficient ground fuel to support a relatively low intensity fire. The next fire can always be of higher intensity if the survival rate of pines is higher than desired. Control of subsequent fires may be easier once pine thickets are broken up a bit through an initial cool burn.
- Where dense stands of small Cypress Pines have locked up, there may be scope for manual thinning followed by burning once retained trash has dried out. This approach has been recommended in native pine forests in the United States, where similar dynamics operate, as a way to get fire into forests which are otherwise difficult to burn due to lack of grasses (Allen *et al.* 2002, Metlen and Fiedler 2006). This strategy has also been tried at the Zoo near Dubbo (Cameron nd).[10]
- Statewide fire frequency guidelines recommend interfire intervals of between five and 50 years for shrub-grass subformation dry sclerophyll forests. As recruitment of *Callitris glaucophylla* is episodic and juvenile periods are long, intervals at the lower end of this range should be sufficient to prevent lock-up, and may also discourage recruitment through suppression of seed production. Some longer intervals would allow some young pines to reach reproductive age and grow tall enough to survive fires.
- Fires may be most effective, and most necessary, after good rains have encouraged both grass growth and pine regeneration. This may also be the time when they are most feasible in terms of both fuel loads and economics.

There is a great need for experimentation with a range of fire regimes in White Cypress Pine forests and woodlands. Landholders are encouraged to 'have a go', in partnership with people familiar with fire (eg RFS brigades), and where possible also with natural resource management professionals and researchers. Common sense, and careful 'reading' of the land, will be invaluable in this enterprise.

3.2.9 Conclusion

Because White Cypress Pine is a widespread species, the story detailed above has relevance for a variety of vegetation classes. In the Pilliga Outwash Dry Sclerophyll Forests which are the focus of this section, fire is likely to play a role in regulating populations of many plants other than *Callitris glaucophylla*. Other non-eucalypt trees and some shrubs in this vegetation class may have a similar ability to recruit between fires, either from seed or via suckers. Some of these species may have the potential to develop dense thickets which exclude other species – Belah (*Casuarina cristata*, Shelly 1998a), Bulloak (*Allocasuarina leuhmannii*, A. Deane pers. comm. 2005) and Budda (*Eremophila mitchellii*) may be examples. On the other hand sclerophyll shrub species may recruit primarily after fire – peas and wattles are likely candidates – and fire will be important for their survival. The balance between shrubs, grasses and forbs may be mediated by fire in this relatively fertile ecosystem, as it is elsewhere.

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[10] David Taylor of the Queensland Forest Research Institute is currently developing, on the basis of a series of trials, guidelines for prescribed burning in a range of fuel types found in production White Cypress Pine forests in that state (D. Taylor, pers. comm. 2007).

As noted above the recommended fire frequency for dry sclerophyll forests in the shrub-grass subformation is 5 to 50 years. The need to limit extensive recruitment of T species suggests a regime in the lower part of this range may be the precautionary option.

3.3 Western Slopes Dry Sclerophyll Forests

3.3.1 Introduction

As their name implies, this vegetation class occurs on the western side of the Great Dividing Range, on sandy, infertile soils – often derived from sandstone – where rainfall exceeds 500 mm. Patches occur right across NSW; in the Central West this forest class is particularly prominent around Dubbo (for example in Goonoo Community Conservation Area) and Wellington (eg Goobang National Park). The largest and best known tract occurs in the Pilliga north of Coonabarabran, to the east of the Pilliga Outwash Dry Sclerophyll Forests discussed in the previous section. Keith (2004) places Western Slopes Dry Sclerophyll Forests in the shrubby subformation of dry sclerophyll forests. Shrubs are mostly sclerophyll species including wattles (eg Spur-wing Wattle *Acacia triptera*, Streaked Wattle *Acacia lineata*), heaths (eg Urn Heath *Melichrus urceolatus*, Daphne Heath *Brachyloma daphnoides*), daisies (eg *Cassinia* species), and members of the Myrtaceae family (eg Broombush *Melaleuca uncinata*, Fringe Myrtle *Calytrix tetragona*). Trees are mostly eucalypts, particularly ironbarks, and cypress pines. Eucalypts include Narrow-leafed Ironbark (*Eucalyptus crebra*), Mugga Ironbark (*E. sideroxylon*) and Tumbledown Red Gum (*E. dealbata*). Both White Cypress Pine (*Callitris glaucophylla*) and Black Cypress Pine (*C. endlicheri*) occur in these woodlands. Herbs and grasses grow in the ground layer. These forests have been, and in some cases continue to be, a major source of timber (Metcalfe *et al.* 2003, Keith 2004).

Western Slopes Dry Sclerophyll Forests are prone to wildfire. In 2006 major fires occurred in both the Pilliga and Goonoo State Forests, while much of the eastern Pilliga also burnt in 1997 (Kavanagh and Barrott 2001). Fire frequency in the eastern and southern Pilliga, where the shrubby vegetation is concentrated, has been much higher than in the Western Pilliga where the grassy outwash forests are mostly found (Binns and Beckers 2001, Kavanagh and Barrott 2001, Date *et al.* 2002). Many understorey plants in Western Slopes Dry Sclerophyll Forests respond rapidly after fire, particularly when water is readily available (Reilly, nd).

3.3.2 Fauna studies

The discussion in the preceding section on relationships between fauna, White Cypress Pine, vegetation structure and fire is likely to have some applicability to this vegetation type as well. In particular, the extensive Date *et al.* (2002) study of birds, vegetation, logging and fire history mentioned in Section 3.2.6, covered the entire Pilliga Forest including the central, southern and eastern sections which Keith (2004) maps as Western Slopes Dry Sclerophyll Forest. This study located 170 bird species, compared to 220 species recorded in the Pilliga historically. The authors were particularly interested in woodland birds, many of which have declined. Many woodland birds nest and/or forage on the ground in low grassy or shrub/grass vegetation. It is likely that these birds have lost their core habitat due to clearing of land outside the Pilliga forest. This means that the albeit sub-optimal habitat of the forest is now very important if these species are to survive (Date *et al.* 2002).

Date *et al.* (2002) present a map showing the distribution of the two primary habitats they identified, species-rich 'creek' habitats, which were related to fire-prone sites on poor soils, and species-poor 'non-creek' patches, which covered much of the forest area. 'Creek' habitats, which supported a wide range of bird species including many generally considered to be in decline, had a diverse understorey of shrubs, grasses, and sedges, while the depauperate understorey of 'non-creek' habitats was the domain of a small number of generalist bird species. 'Creek' habitats had a history of less intense logging, and more frequent and recent fire, than 'non-creek' patches. Both habitat types were found throughout the Pilliga, although 'creek' habitat was considerably more prevalent in the south and the east. *Callitris glaucophylla* and *Allocasuarina leuhmannii* were significantly more abundant in 'non-creek' habitat, while the 'creek' assemblage included higher abundances of a range of shrubs including the sclerophyll species *Bossiaea rhombifolia* and *Brachyloma daphnoides*. Date *et al.* (2002:187) consider that there may be "a homogenizing effect on habitat conditions of logging, combined with changed fire regimes and stock grazing, and the dense regeneration of cypress pine, casuarina, and some shrubs… That is, the disturbance history of the Pilliga may have imposed a pattern on the biota which masks the effects of topography, soils and vegetation." As already noted, these authors recommend management strategies to return mature trees to the canopy, and suggest diverse "intermediate fire regimes (2-25 year intervals)," coupled with an adaptive management approach, to increase habitat heterogeneity and benefit avian diversity (Date *et al.* 2002:190). These authors note the importance of encouraging the relatively open, grassy habitat favoured by woodlands birds as part of the habitat mix.

Shelly (1998b) conducted a fauna survey in a patch of Goonoo State Forest dominated by Mugga Ironbark and Black Cypress Pine, in a fire-prone area which had not, however, burnt for at least 10 years. Animal species included 9 reptiles, 9 bats, 3 macropods and 51 birds, a fairly diverse avian fauna relative to that recorded by Date *et al.* (2002) in dense unburnt forest in the Pilliga. However few seed-eating birds were recorded, probably because by this time-since-fire there was almost no grass – the estimated cover of ground layer plants was only 5%. A small number of bird species such as silvereyes and warblers were confined to dense shrubby patches.

Both these studies accord with work from other Australian ecosystems which has found that different faunal assemblages favour different times-since-fire, and that differences in fire frequency also create variable habitat opportunities which are preferentially used by different fauna species. Some of this work has been summarised in previous Hotspots reviews - see particularly Sections 2.2.4 and 4.2 in the Southern Rivers review (Watson 2006b) and Sections 3.3 and 3.4 in the Northern Rivers review (Watson 2006a).

3.3.3 Black Cypress Pine, *Callitris endlicheri*

Both Black Cypress Pine (*Callitris endlicheri)* and White Cypress Pine (*C. glaucophylla*) are found in Western Slopes Dry Sclerophyll Forests. According to

Binns and Beckers (2001), the two are negatively associated, with Black Cypress more common in heathy areas, while White Cypress is more abundant in grassy vegetation.

Black Cypress Pine, like *C. glaucophylla*, is an obligate seeder which has the ability to survive fire when scorch is not too severe (DEC 2002). Whether Black Cypress recruits primarily after fire, or between fires as is the case with White Cypress, is unclear. According to Keith (2004:167), "Black cypress pines occur on the most barren soils in these forests and to the east extend to high elevations on the edge of the tablelands. This species regenerates from seedlings in an episodic manner when its woody cones release their seeds *en masse* after fire onto an open bed of ash. White cypress pines, in contrast, extend into the semi-arid interior of New South Wales and are less dependent on fire for regeneration." Ecologist Ross Bradstock suggests that there may be variation in the degree to which this species retains its fruits locked tight in cones, awaiting the trigger of fire: it may be a serotinous fire-dependent I species in some environments, but release seed and recruit between fires (ie act as a T species) in other situations.

3.3.4 Conclusion

From the little we know, the ecology of Western Slopes Dry Sclerophyll Forests shows every sign of sharing the ecological processes familiar from moderately productive dry sclerophyll forests elsewhere in NSW. Its sclerophyll shrub complement almost certainly contains a number of species which recruit primarily with fire; the balance between these species and those able to recruit between fires, like White Cypress Pine, is almost certainly fire-mediated; and variability in time-since-fire and fire frequency provide a range of habitat resources which enables a diverse fauna to persist.

Variable fire frequencies within the state-wide thresholds of 7 and 30 years are likely to be appropriate for this forest type. Where absence of fire has allowed extensive stands of species such as *Callitris glaucophylla* or *Allocasuarina leuhmannii* to develop, one or two short intervals in at least some parts of the landscape may help regenerate the grass and shrub layer, and provide more diverse fauna habitat – as we have seen, Date *et al.* (2002) suggest occasional intervals as low as two years, in this context. The finding that many bird species are more abundant in recently and/or moderately frequently burnt habitat suggests that intervals in the lower half of the state-wide dry sclerophyll forest interval range are likely to be an important element of fire regimes which promote plant and animal diversity.

4 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 valuable 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 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 some 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.

Many of these themes are reflected in the ecological theory discussed in Chapter 1 as well as in the findings reported in Chapters 2 and 3.

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 as one moves west and rainfall decreases, fuel continuity becomes an increasingly important factor controlling fire regimes. At the same time, plant species with an ability to recruit into gaps become more salient in the shrub and small tree flora – understandably, as bare ground is more readily available than in coastal and subcoastal areas where even after drought most of the ground is likely to be covered, in the absence of recent fire, by living or dead grasses (particularly in grasslands and grassy woodlands), by dense sclerophyll shrubs (particularly in heaths) or by thick persistent litter (particularly in eucalypt forests and woodlands). Opportunities for gap recruitment are also affected by rainfall: where there may be sufficient moisture to support the establishment of young tree and shrub seedlings almost every year in high rainfall coastal areas, these opportunities may only occur on a decadal timespan in semi-arid country. In coastal areas, fire plays a vital role in plant species' persistence by creating gaps; it is therefore not surprising that many species in these environments recruit almost exclusively after fire, often with the assistance of attributes such as serotiny, fire-cued dormant seeds, or post-fire flowering. In the semi-arid zone drought followed by rain brings opportunities for plant recruitment, but also for the fire that can then affect the abundance of germinating trees and shrubs. Grazing interacts with these processes, creating gaps in places where they might not otherwise have existed in high-growth environments, destroying seedlings directly, and reducing grass fuel loads and thus the intensity and frequency of fires – particularly, perhaps, in places where grass growth is episodic. 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 appropriate interfire intervals for biodiversity conservation. In highly productive environments such as those which support wet sclerophyll forests, fuel will build up rapidly, but may be available for burning only infrequently due to relatively high moisture levels. Those seeking to encourage conservation-friendly fire management need to acknowledge and study differences and interactions such as those suggested above, and recognise that because of differences such as these, fire regimes compatible with retention of plant and animal species will vary.

With climate change, the incidence of days of extreme fire weather can be expected to increase (Hennessy *et al.* 2005). The challenge in some places may be to keep 'natural' fire frequency above lower thresholds. This is most likely to apply where wildfires already occur periodically, and particularly where continuous fuels are found across large areas. In other places, however, vegetation remnants have been largely cut off from wildfire, and may well remain so even under a changing climate. If fire in these areas is to occur at intervals within recommended thresholds, the challenge will be to introduce some fire. In the west, if climate change brings drier conditions there may be reduced opportunities for wildfire or planned burning, due to lack of continuous grass fuels. Again, scientists are only just starting to consider how fire regimes in different ecosystems may be affected by climate change.

The diverse landscapes of the Central West invite a nuanced understanding of fire ecology. While there is still much to be learned, the current state of knowledge is sufficient to provide a reasonable degree of 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 Central West region.

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