HOTSPOTS FIRE PROJECT

Fire and the Vegetation of the Lachlan Region

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FIRE AND THE VEGETATION OF THE LACHLAN REGION

Scope of this review

This literature review forms part of a suite of materials that Hotspots aims to produce in each Catchment Management Authority (CMA) region in which it works. While most Hotspots products are targeted to landholders, literature reviews are 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 Lachlan 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 Central West, Hunter, Namoi, Northern Rivers and Southern Rivers regions are also available (Watson 2006 a, b 2007; Watson and Tierney 2008, 2009).

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 the vegetation formations of Keith (2004). All vegetation formations covered in this review also occur in the Namoi and Central West CMAs. However the literature is limited for some vegetation formations and this is reflected in this review. Fire is also of limited occurrence in most wetland types (though it can occur in Forested Wetlands), therefore wetlands are also not considered in this review. The broad vegetation formations of Keith (2004) can be further subdivided into classes (Table 1). Where literature permits, the fire ecology of classes that occur in the Lachlan CMA region are discussed (often there is no literature available at the class level or limited to only one study).

Table 1: NSW Vegetation Formations and Classes (Keith) occurring in the Lachlan Valley

VEGETATION FORMATION	KEITH VEGETATION CLASS
ARIDSHRUBLANDS ***Acacia subformation***	• Sand Plain Mulga Shrublands • North-west Plain Shrublands
Chenopod subformation	• Riverine Chenopod Shrublands • Aeolian Chenopod Shrublands
SEMI-ARID WOODLANDS ***Grassy subformation***	• Inland Floodplain Woodlands • Riverine Plain Woodlands • Brigalow Clay Plain Woodlands
Shrubby subformation	• Western Peneplain Woodlands • Dune Mallee Woodlands • Sand Plain Mallee Woodlands • Semi-arid Sand Plain Woodlands • Riverine Sandhill Woodlands • Inland Rocky Hill Woodlands
DRY SCLEROPHYLL FORESTS ***Shrub/grass subformation***	• Upper Riverina Dry Sclerophyll Forests • North-west Slopes Dry Sclerophyll Woodlands
Shrubby subformation	• Sydney Montane Dry Sclerophyll Forests • South East Dry Sclerophyll Forests • Southern Tableland Dry Sclerophyll Forests • Western Slopes Dry Sclerophyll Forests
HEATHLANDS	• Northern Montane Heaths
FORESTED WETLANDS	• Eastern Riverine Forests • Inland Riverine Forests
FRESHWATER WETLANDS	• Montane Bog and Fens • Montane Lakes • Inland Floodplain Shrublands • Inland Floodplain Swamps
SALINE WETLANDS	• Inland Saline Lakes
GRASSLANDS	• Temperate Montane Grasslands • Riverine Plain Grasslands • Western Slopes Grasslands
GRASSY WOODLANDS	• Western Slopes Grassy Woodlands • Subalpine Woodlands • Southern Tableland Grassy Woodlands • Floodplain Transition Woodlands
WET SCLEROPHYLL FORESTS ***Grassy subformation***	• Southern Tableland Wet Sclerophyll Forests

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The guidelines – a word of caution

A recommended fire frequency range is provided for each formation where it is possible to make recommendations. However, it is important to understand that these frequencies represent broad recommendations, based on the data that is available. Individual species (including threatened species) and identified endangered ecological communities may require specific fire regimes and this cannot be accurately assessed for these entities based on broad observations at the formation and class level. It should also be kept in mind that fire frequency is generally regarded as a powerful influence on ecological processes. However fire intensity, season, patchiness, fire history and a range of other factors interact to determine the impact of any given fire (see below). In some instances other factors (e.g. system productivity) or an interaction of factors determine system biodiversity outcomes. This may include recent anthropological changes (e.g. fragmentation effects) so that fire produces differing outcomes through time and space that go beyond predictions based on vegetation type. Biodiversity patterns are also much simplified by classification to vegetation formations and classes. Despite these complications, fire is an important driver of biodiversity in many systems in the Lachlan CMA and one that must be managed. There is thus a strong need for further research to fine tune the use of fire management in the region.

FIRE ECOLOGY SOME IMPORTANT CONCEPTS

Before addressing the literature on specific vegetation classes, some ecological concepts and principles underlying current understanding of fire regimes are explored. 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.

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.

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, i.e. 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.

Non-sprouting species, or ‘**obligate seeders**’, are plants that die when their leaves are all scorched in a fire, these species rely on regeneration from seed. 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 & 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 & Renwick 2000). Some species exhibit different regeneration strategies in different environments (Williams *et al.* 1994; Benwell 1998; DEC 2002).

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 & 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 & O’Connell 1991), 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 & Whelan 2000) and the Waratah *Telopea speciossima* (Bradstock 1995) also flower almost exclusively in the years after a fire. Many grassland forbs (herbaceous plants which are not a grass, sedge or rush) 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 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 (e.g. *Styphelia* and *Leucopogon* spp.). These species may have a different relationship to fire cycles than do taxa whose seeds are not widely dispersed (French & Westoby 1996; Ooi *et al.* 2006b).

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 & Westoby 1987; Cowling *et al.* 1990 Vaughton 1998; Keith *et al.* 2002a), although this may vary between populations (Whelan *et al.* 1998) and with post-fire age (Enright & 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.

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 & Bradstock 1992; Knox & 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 & 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 & Myerscough 1988; Bell & 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 & Bradstock 1992; 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 (e.g. *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 & Kimber 1975). This gives these species a 'hedge' against a second fire within the juvenile period.

Disturbance, succession and a paradigm shift

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 & 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 (Clements 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 & Slatyer 1980). 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 & 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 & 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.

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 & Slatyer (1980). It can also be used to define disturbance frequency domains compatible with maintenance of particular suites of species. This model has 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 & 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 & 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 & 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 & Kenny 2004).

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). 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.

Competition and productivity

The effect of dominant heathland shrubs on other species has been recognised in Sydney's sandstone country (Keith & Bradstock 1994; Tozer & Bradstock 2002). 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 & Specht 1989; Bond & Ladd 2001) and in South Africa's heathy fynbos (Bond 1980; Cowling & Gxaba 1990; Vlok & 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 & 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 *et.al* 2009).

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 & 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 & 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 nonequilibrium 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, abiotic 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 may be 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: Prober 1996; Clarke & Knox 2002). 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 savannah 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 & 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 & Sanderson 1994; Williams *et al* 2012). 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.

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 noneucalypt tree species, particularly those able to recruit between fires (Birk & Bridges 1989; Henderson & Keith 2002). Each regime provides habitat for an equally diverse, but substantially different, array of invertebrates and small mammals (Andrew *et al.* 2000; York 2000; Bickel & Tasker 2004; Tasker & 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.

Fire and climate change

Climate change is a challenging example of the non-equilibrium paradigm in operation. Altered fire regimes have been predicted over ensuing decades and are considered to be one of the major ecological management challenges for Australia (Morton *et al*. 2009). This presents a significant issue for fire ecologists who (like most ecologists) have operated under an assumption of an overall steady state (albeit with large variability) in fire regimes. General predictions of increasing fire frequency and intensity have been predicted to lead to major fire driven shifts in vegetation patterns. There are, however, a range of uncertainties around these predictions. Firstly, it is by no means certain what (if any) changes will occur. Bradstock *et al*. (2008) report that the two major drivers of fire are "fire weather" (hot, dry conditions; ignition rates) and fuel load. Whilst increases in fire weather may occur, drier conditions could lead to less fuel (Bradstock 2010; Penman & York 2012; Matthews *et al.* 2012), so that fire regime changes are by no means understood. Secondly, shifts in vegetation in response to fire will be limited by major habitat variables such as soil type (Westoby & Burgman 2006) and more generally by the overall suitability of adjacent areas (Keith *et al*. 2008). Additionally, selection and adaptation in the face of altered fire regimes may occur (Skelly *et al*. 2007).

THE LANDSCAPE, CULTURES AND CLIMATE OF THE LACHLAN VALLEY

The Lachlan Valley is composed of three distinct landscapes (*Figure 1*): tablelands (part of the Southern Eastern Highlands bioregion), slopes (part of the New South Wales (NSW) Southwestern Slopes bioregion) and plains (containing parts of the Cobar Peneplain, Riverina and Murray Darling Depression bioregions). Elevation in the catchment varies from over 1300m above sea level (asl) on the south-western slopes of Mt Canobolas near Orange to approximately 140m asl in the Great Cumbung Swamp at the confluence of the Lachlan and Murrumbidgee Rivers.

The Lachlan Valley is one of the most heavily agriculturally developed catchments in NSW. The majority of pre-European ecosystems across the catchment have been degraded or removed by over a century and a half of agricultural landuse (OEH VIS spatial data). Synergistic pressures leading to the extinction of numerous native mammal species include invasion and occupation by feral animals and weeds, serious degradation to land and water resources and the imposition of inappropriate fire regimes.

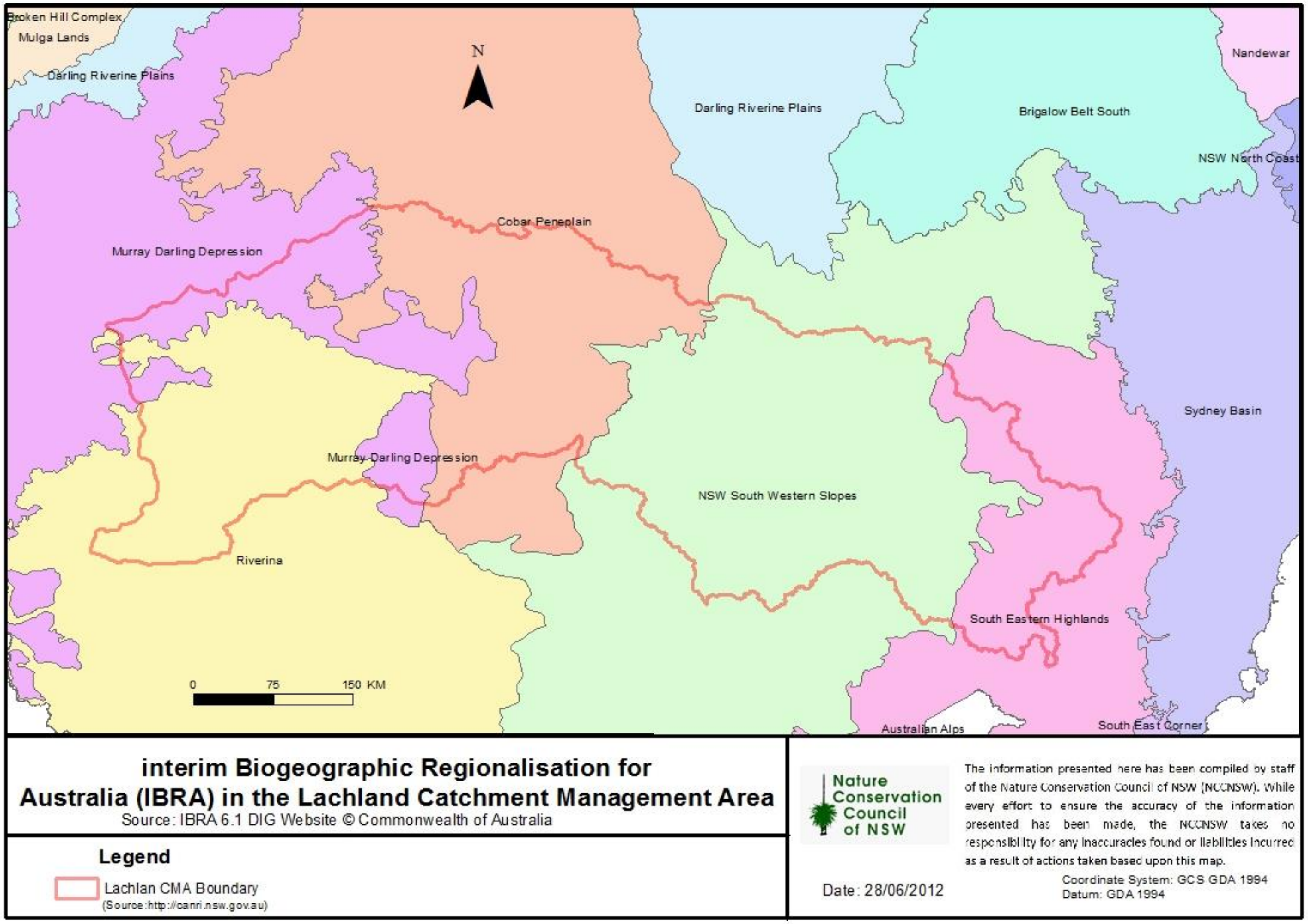
The Lachlan Valley contains 29 Local Government Areas (LGAs) with a collective population of approximately 100000 residents. Major population centres are Forbes, Parkes, Cowra, Young, Condoblin, Lake Cargelligo and Hillston. Because of the presence of large areas of highly productive land, a very small proportion of the catchment is Public Land (Conservation Reserves, State Forests and Crown Lands), but the majority of Public Land is vegetated.

There are several landscapes in the Lachlan Valley where relatively large areas of native vegetation remain (*Figure 2*). This includes the “discontinous corridors” of native vegetation linking the World Heritage listed Blue Mountains National Park and the Abercrombie River National Park. Further discontinuous forested corridors occur in a generally north-south alignment in the mid reaches of the Lachlan Valley between Goobang, Nangar, Conimbla, Dannanbilla and Weddin Mountains conservation reserves.

Many landscapes in the Lachlan Valley have been entirely cleared; as a result they have little biodiversity or conservation value. Notably the fertile and lower slope sections of the Southeastern Highlands and NSW Southwestern Slopes have no high quality remnant vegetation. The eastern and southern parts of the Cobar Peneplain Bioregion have been heavily cleared, as have the more fertile floodplains and lower slopes of the Riverina and Murray Darling Depression Bioregions. The western and southwestern parts of the catchment contain a relatively high proportion of native vegetation primarily chenopod shrublands, although this vegetation cover varies widely in respond to prevailing climatic conditions.

The Lachlan River has a highly variable flow that is quite unlike the greater and more reliable flows of the adjoining Murrumbidgee and Murray Rivers. This is due to the lack of spring snowmelt anywhere in the catchment. Productivity in the Lachlan Valley is driven by prevailing climatic conditions, with flows in the river ceasing in three years out of ten (CSIRO, 2008). The flows of the Darling River to the west have even greater variability than the Lachlan River, responding strongly to El Nino – La Nina cycles and deriving major flood pulses from Southern Queensland and the western New England Tableland. CSIRO (2008) predicts that rainfall patterns and hydrological regimes across the Lachlan catchment will be strongly influenced by global warming. Substantial declines in rainfall and increases in temperature are expected this century.

The Lachlan Valley is mostly contained within the Wiradjuri Nation, whose members have utilised and inhabited the landscape for tens of thousands of years (Flood, 1996). The Wiradjuri and other Aboriginal nationalities within the Lachlan Valley utilised fire for maintaining ecosystem processes and for economic purposes over these millennia. Much knowledge of traditional burning practices has been lost from the Lachlan Valley, however the Lachlan CMA is currently investing in research and re-establishment of traditional burning practices (Lewis 2010).



interim Biogeographic Regionalisation for Australia (IBRA) in the Lachland Catchment Management Area

Source: IBRA 6.1 DIG Website © Commonwealth of Australia

The information presented here has been compiled by staff of the Nature Conservation Council of NSW (NCCNSW). While every effort to ensure the accuracy of the information presented has been made, the NCCNSW takes no responsibility for any inaccuracies found or liabilities incurred as a result of actions taken based upon this map.

Figure 1: Bioregions of the Lachlan CMA

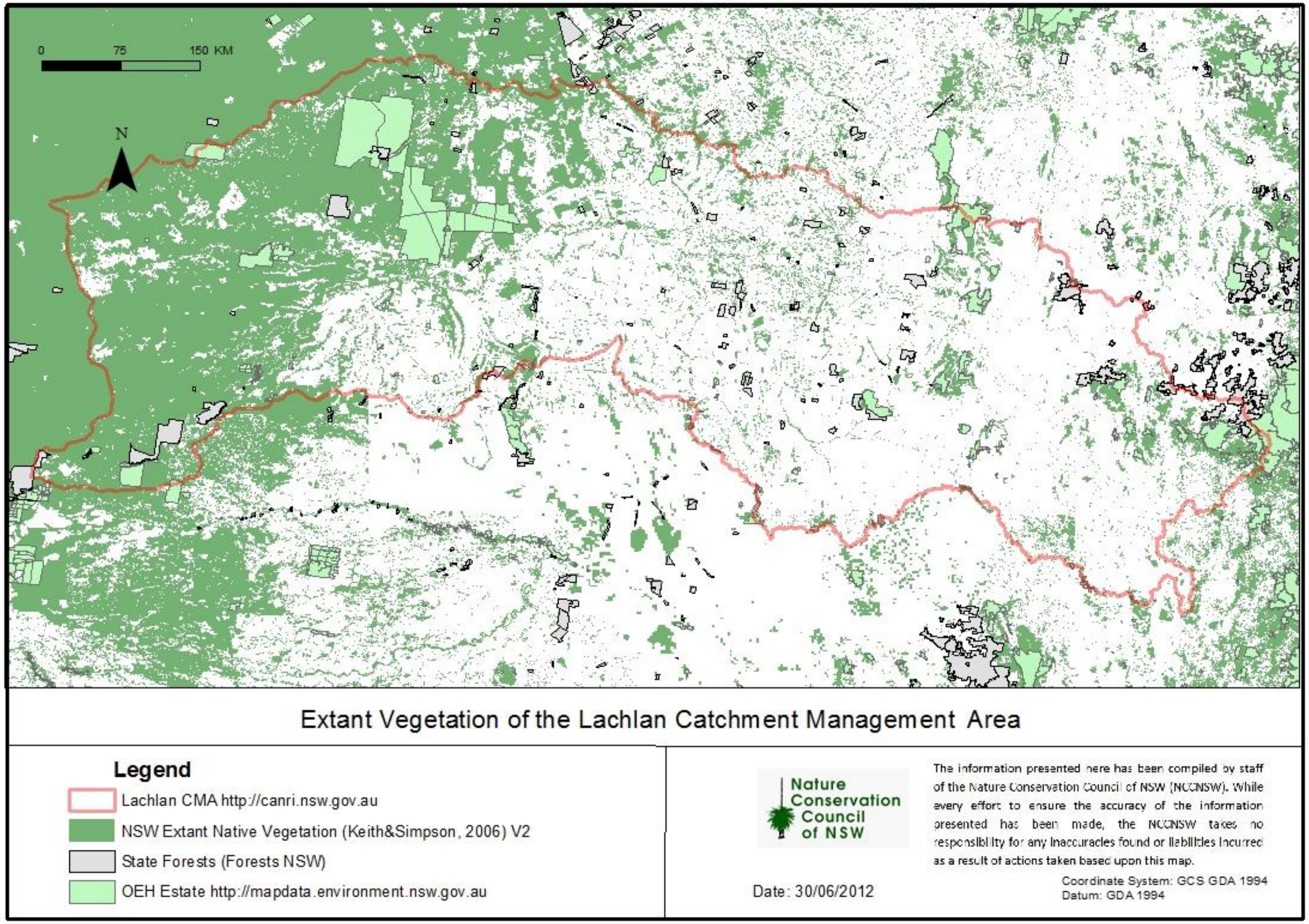


Figure 1: Extant native vegetation in the Lachlan CMA (OEH VIS data)

LANDSCAPE GRADIENTS AND DRIVERS OF BIODIVERSITY IN THE LACHLAN VALLEY

The substantial rainfall gradient across the Lachlan catchment from east (>1000mm p.a.) to west (<250mm p.a.), creates substantial variation in vegetation systems. These range from grassy wet sclerophyll forests on the high rainfall tablelands in the upper parts of the catchment, to mallee and sandplain semi-arid woodlands and inland floodplain shrublands dominated by lignum in the mid-reaches of the catchment, to arid shrublands dominated by mulga (*Acacia aneura*) and chenopods in the far west of the catchment.

A substantial east-west gradient of riparian forests occurs along the major drainage lines of the Lachlan River. Eastern Riverine Forests dominated by River Oak (*Casuarina cunninghamiana*) grade into Inland Riverine Forests dominated by River Red Gum (*Eucalyptus camaldulensis*) and Inland Floodplain Woodlands dominated by Black Box (*E. largiflorens*) in the far west.

The vegetation communities of the Lachlan catchment have evolved with fire as a prominent driver during the increasing aridity of the Tertiary and Quaternary periods, and many communities, populations and species are critically reliant upon fire to maintain essential life cycle components (White, 1999). Fire intensity and frequency vary greatly across the vegetation communities occurring within the Lachlan Valley, primarily as a function of variation in rainfall and site productivity. The wet sclerophyll forests in the east of the catchment have the greatest biomass of any vegetation community in the catchment, whilst the grasslands and chenopod shrublands on the far western plains have the least.

NATIVE VEGETATION PATTERNS IN THE LACHLAN VALLEY

A considerable proportion of the pre-European vegetation of the Lachlan catchment has been lost, mostly due to extensive clearance of, and ongoing degradation to, the vegetation occurring on more productive and fertile landscapes. The Lachlan Valley is a central component of the wheat belt that extends from Victoria through the western slopes and plains of New South Wales to southern Queensland. Formerly-extensive areas of grassy box gum woodland within these landscapes have been entirely removed. Fertile landscapes underlain by basalt and limestone in the Upper Lachlan Valley that previously supported Eucalypt forests have suffered a similar fate. Collectively, these landscapes are some of the most heavily fragmented and poorly reserved bioregions in Australia (Commonwealth of Australia, 2006).

Conversely, rocky and infertile landscapes supporting vegetation types such as Inland Rocky Hill Woodlands and Semi-arid Sand Plain Woodlands contain a much greater proportion of pre-European vegetation. The majority of remnant native vegetation of the Lachlan catchment is located within these landscapes.

The most extensive mosaics of mallee dune and sandplain woodland in NSW and some of the largest in Australia occur within the western parts of the Lachlan Valley. Conservation reserves including Nombinnie, Round Hill and Yathong protect the largest complex of these habitats in NSW. These conservation areas and adjoining private lands are a stronghold for mallee dependent species that have become extinct elsewhere in the state.

Substantial tracts of native vegetation are very limited in the Lachlan Valley. Other than the aforementioned mallee block, the only sizable fragments of native vegetation in the Lachlan Valley are in the far northeast of the catchment within the Blue Mountains National Park (a World Heritage property), Abercrombie River National Park and on the far western plains. All patches of native vegetation on the flatter and more fertile landscapes in the mid-reaches of the Lachlan Valley are very small (OEH VIS spatial data) or consist only of senescing trees in paddocks (Manning *et al.* 2006).

On the far western plains of the Lachlan Valley depositional patterns are complex and involve both alluvial and Aeolian processes. The floodplains and sandplains of the far west of the Lachlan Valley support a complex mosaic of chenopod shrublands, grassland and woodland communities.

The largest conservation reserves in the catchment are the combined Nombinnie, Yathong and Round Hill Nature Reserves, these reserves contain the largest remnants of mallee in New South Wales. Other significant reserves include Goobang, Weddin Mountains, Nangar, Conimbla and Woggoon.

FAUNA EXTINCTION IN THE LACHLAN VALLEY

Most Critical Weight Range (CWR) mammals are extinct in the Lachlan Valley. The extinction of populations of bandicoots, small wallabies and potoroids as well as native rodents and carnivores (Burbidge and Mackenzie, 1989) and replacement with sheep, cattle, foxes, rabbits and cats has led to a fundamental shift in ecosystem dynamics across the catchment. This shift in mammal community composition has resulted in a myriad of deleterious impacts on the diversity, structure and function of remaining native species. The consequences of such a large shift in the fauna community will likely resonate for many generations to come. Further extinctions of fauna species seem unavoidable and in some instances, such as the Red-lored Whistler (*Pachycephala rufogularis*), imminent (NSW Scientific Committee, 2008).

Extinction rates of woodland and forest bird species have historically been low, however in recent decades many species have experienced major declines, and this rate is accelerating (Reid, 1999). Few, if any, populations of Bush Stone Curlew, Barking Owl, Hooded Robin, Black Chinned Honeyeater and Speckled Warbler remain within the Lachlan Valley and substantial populations of Grey-crowned Babbler, Diamond Firetail and Brown Treecreeper are limited in number and declining (Reid, 1999).

The mallee dominated landscapes of the Lachlan Valley provide a nationally and state-significant refuge for several threatened fauna species. These species include the Western Blue-tongue Skink, Malleefowl, Chestnut Quail Thrush, Red Lored Whistler and Striated Grasswren (Benshemesh, 1992; Sass, 2006; NSW Scientific Committee 2009).

The Riverina population of the Glossy Black Cockatoo is listed as an Endangered Population under the TSC Act. The survival of this population is critically dependent upon the availability of fruiting She-Oak (*Allocasuarina verticillata*) trees for food. Many important stands of this species on the Cocoparra Range are in an advanced state of post-fire senescence and are no longer producing fruit (Milton Lewis *pers. comm*).

THREATENED SPECIES AND COMMUNITIES OF THE LACHLAN CMA

The Lachlan catchment contains a rich diversity of native flora and fauna species including many species listed as Vulnerable, Endangered or Critically Endangered, and two Endangered populations. Several species are endemic to the Lachlan catchment, including the critically endangered pea, *Bossiaea fragrans*. Most of these species occupy a limited range, are threatened by numerous processes and face a very high risk of extinction due to stochastic events.

Within the Lachlan Valley there are eleven Endangered Ecological Communities listed under the NSW Threatened Species Conservation Act (TSC Act) known or predicted to occur (OEH data, Table 2), as well as five Endangered Ecological Communities and one Critically Endangered Ecological Community (Table 3) listed under the Commonwealth Environment Protection and Biodiversity Conservation Act (EPBC Act).

Table 2: NSW Endangered Ecological Communities known and predicted to occur in the Lachlan Valley

Name	Level of Threat (TSC Act)	Occurrence
Acacia loderi shrublands – Neelia Shrublands	Endangered Ecological Community	Predicted
Acacia melvillei Shrubland (Yarran Shrubland) in the Riverina and Murray-Darling Depression bioregions	Endangered Ecological Community	Known
Fuzzy Box Woodland on alluvial Soils of the South Western Slopes; Darling Riverine Plains and Brigalow Belt South Bioregions	Endangered Ecological Community	Known
Inland Grey Box Woodland in the Riverina; NSW South Western Slopes; Cobar Peneplain; Nandewar and Brigalow Belt South Bioregions	Endangered Ecological Community	Known
Mt Canobolas Xanthoparmelia Lichen Community	Endangered Ecological Community	Known
Myall Woodland in the Darling Riverine Plains; Brigalow Belt South; Cobar Peneplain; Murray-Darling Depression; Riverina and NSW South Western Slopes bioregions	Endangered Ecological Community	Known
Sandhill Pine Woodland in the Riverina; Murray-Darling Depression and NSW South Western Slopes bioregions	Endangered Ecological Community	Predicted
Tableland Basalt Forest in the Sydney Basin and South Eastern Highlands Bioregions	Endangered Ecological Community	Known
Tablelands Snow Gum; Black Sallee; Candlebark and Ribbon Gum Grassy Woodland in the South Eastern Highlands; Sydney Basin; South East Corner and NSW South Western Slopes Bioregions	Endangered Ecological Community	Known
White Box Yellow Box Blakely's Red Gum Woodland	Endangered Ecological Community	Known

Table 3: Threatened Ecological Communities listed under the EPBC Act occurring in the Lachlan Valley

Alpine *Sphagnum* Bogs and Associated Fens	Endangered
Buloke Woodlands of the Riverina and Murray-Darling Depression Bioregions	Endangered
Grey Box (*Eucalyptus microcarpa*) Grassy Woodlands and Derived Native Grasslands of South-eastern Australia	Endangered
Natural Temperate Grassland of the Southern Tablelands of NSW and the Australian Capital Territory	Endangered

Weeping Myall Woodlands	Endangered
White Box-Yellow Box-Blakely's Red Gum Grassy Woodland and Derived Native Grassland	Critically Endangered

VEGETATION COMMUNITY AND FLORA SURVEYS OF THE LACHLAN VALLEY

Publications describing the vegetation of parts of the Lachlan Valley commence with the journals of the first European explorers of Australia, from the early 1800s until approximately the 1850s (Oxley, 1818) . Publications containing mapping and floristic descriptions of the vegetation of the Lachlan Valley commenced in the 1940s (Beadle 1948; Moore 1953). In recent times a number of statewide vegetation classifications have been completed (Keith 2004; Benson *et al* 2008; OEH 2011). These vegetation classifications describe distinct vegetation formations, subformations, classes, communities and associations that occur within the Lachlan Valley.

There is a substantial bias toward formal conservation reserves in the survey and analysis of native vegetation communities occurring in the Lachlan Valley, although this may reflect the general lack of native vegetation remaining on private freehold land. The majority of publications describing the vegetation communities and associations of the Lachlan Valley are restricted to the vegetation occurring within formal conservation reserves (ERM, 1996, Porteners 1993, 1997, 2000, 2001(a), 2001 (b), although Keith (2004), Benson (2006) and DEC (2006) all describe vegetation communities across the entire landscape. The lack of an extensive body of literature relevant to vegetation communities occurring on freehold tenure is primarily because little intact remnant vegetation remains across the majority of these lands in the Lachlan Valley and because (as in most of Australia) there has been little study of privately owned native vegetation.

VEGETATION COMMUNITIES OF THE LACHLAN VALLEY

The Lachlan River arises at an elevation of over 1200m in the Southern Highlands to the northeast of Gunning and Crookwell. The river then flows well over 1000km to its confluence with the Murrumbidgee River southwest of Booligal (*Figure 2*). The catchment is composed of three distinct landscapes: tablelands, slopes and plains. Sections of five bioregions are contained within the Lachlan Catchment Management Authority (CMA) Management Area. From east to west they are South Eastern Highlands, NSW South Western Slopes, Cobar Peneplain, Riverina and Murray Darling Depression.

The NSW Office of Environment and Heritage (OEH, 2011) determined that 126 Regional Vegetation Communities (RVCs) occur within the Lachlan CMA (*Appendix 1*). The 126 RVCs align with 35 State (Keith) vegetation classes and 13 formations or subformations (*Table 1*).

Of the 34 classes ten align with Final Determinations for listing as Endangered Ecological Communities (*Table 2*) under the NSW Threatened Species Conservation Act (1995), a further six are listed as Threatened Ecological Communities (*Table 3*) under the Commonwealth Environmental Protection and Biodiversity Conservation Act (1999). Of great interest is the occurrence within the Lachlan CMA of an endangered ecological community dominated by non-vascular species, the Mt Canobolas *Xanthoparmelia* lichen community.

VEGETATION CLASSES OF VERY LIMITED EXTENT OR QUESTIONABLE OCCURRENCE IN THE LACHLAN CMA

Several classes of vegetation identified as occurring within the Lachlan CMA by Eco Logical (2006) and OEH (2011) are of very limited extent. This includes Northern Montane Heaths (essentially limited to the western slopes of Mt Canobolas near Orange (Hunter 2000) and possibly on ridgetops of the western Blue Mountains National Park), and Montane lakes, bogs and fens which are restricted to several very small high elevation wetlands in the far east of the catchment (Phil Gilmour *pers. comm.*2011) None of these vegetation classes are mapped as occurring within the Lachlan CMA by DEC (2006 & 2008).

OEH (2011) also identifies the occurrence of of inland saline lakes, subalpine woodlands and brigalow clay plain woodland within the Lachlan CMA, but DEC (2006 & 2008) do not map any areas of these vegetation classes, although the existence of saline lakes within the Lachlan CMA is listed by several authorities (including Benson *et al* 2006; OEH 2011). Hunter (2000) describes and maps vegetation communities at Mt Canobolas that align with the description of Subalpine Woodlands in Keith (2004).

FIRE IN GRASSLANDS

Extensive natural grasslands were a prominent feature of the pre-European vegetation of the Lachlan Valley (Benson *et al*. 1996, DEC, 2006). These vegetation communities are considered to have contained isolated, or no, trees and shrubs (Benson 1996). Natural grasslands mostly occupied higher fertility and flatter landscapes that have been extensively converted to cropping or grazing uses within the Lachlan Valley (Keith 2004; Prober & Thiele, 2006).

The structural and floristic composition of the natural grasslands of the Lachlan Valley varies widely as a result of the interplay between climate, fire, landscape productivity and altitudinal influences. Most natural grasslands in the Lachlan Valley have a high proportion of exotic plants and few, if any, are free of weeds. Within the Lachlan Valley all the temperate montane grassland communities and the eastern most riverine floodplain grassland communities are critically fragmented and reduced to highly degraded and widely scattered remnants (Benson 1997). The sandplain grassland community that is the most westerly of the riverine floodplain grasslands and all the semi-arid floodplain woodlands of the Lachlan catchment have a higher proportion of remnant vegetation than the more easterly grassland classes (OEH VIS spatial data).

Grasslands within the Lachlan CMA have been described by NPWS (1998) Benson *et al.* (2006), Porteners (1998, 2000), Eco Logical (2006), DLWC (2000) and DEC (2006). DEC (2006) have mapped eight grassland communities within the Lachlan Valley. These comprise three Keith classes of natural grassland (Temperate Montane Grasslands, Riverine Plains Grasslands and Semi-arid Floodplain Grasslands) and one derived, Western Slopes Grassland (Keith 2004, OEH, 2011).

A substantial literature on the fire ecology of south-east Australian grasslands exists (e.g. Stuwe & Parsons in 1977; Morgan 1998a,b,c,d, 1999; Lunt 1995; Morgan 1997; Lunt 1997a; Lunt & Morgan 1999a,b; Verrier & Kirkpatrick 2005). This literature is largely based on studies of *Themeda australis* dominated grasslands from southern Victoria, Tasmania and the southern New South Wales ranges and slopes. However, grasslands do share a number of consistent ecological traits, even among continents (see O'Connor 1991) and the fire ecology of these southern *Themeda* grasslands is therefore an important body of literature.

A key feature of many grassland systems is that they are dynamic at relatively short time scales. In studied Victorian *Themeda australis* grasslands, *Themeda* grows rapidly post-fire (Morgan 1996; Lunt 1997d) and eliminates gaps among *Themeda* tussocks within three years. The diversity of the system is largely due to the diversity of forbs that grow in these tussock gaps. When burnt, many of the grassland forbs that exist in these tussock gaps also resprout and then flower at higher rates (Lunt 1990,1994; Morgan 1996). These species then produce seed which is generally non-dormant (Willis and Groves 1991; Lunt 1995, 1996; Morgan 1998b) and they colonise available gaps, provided post-fire conditions such as rainfall are favourable (Hitchmough *et al.* 1996; Morgan1997; 1998b, 2001). Hence, Lunt and Morgan (1999) document significant vegetation change in a *Themeda* grassland after a 11 year fire free period (which corresponds to a loss of forb species and a predicted probable decline in the non-dormant fraction of the seed bank).

This time since fire effect is also likely to interact with the spatial scale of a fire because seed dispersal capacity may be a limiting factor for some species (Lunt & Morgan 1999a). Hence completely unburnt sites (i.e. large sites completely unburnt or small isolated long unburnt sites) appear difficult to restore from the seed bank by burning alone (Lunt & Morgan 1999a; Morgan

1999). These sites are likely to have exhausted the seed bank of non-dormant seeds. However, some of the forb species in these grasslands employ a counter-intuitive life history strategy. Rather than being small short lived highly dispersive species, these species employ a persistence strategy, growing after disturbance ie. drought and frost from underground bulbs, tubers or rhizomes (Bond & Midgley 2001). Seedling establishment appears to be a relatively rare event in these *Themeda* grasslands but it is generally associated with fire (Morgan 1998d: Morgan 2001). Thus, life time fitness (total off-spring from an individual) is related to survivorship through time to maximize reproductive opportunity when a fire event occurs.

Frequent fire (intervals of a few years) thus appears to be an important management goal to maintain diversity in the studied *Themeda* grassland systems. However, the dominating role of *Themeda* in these systems means that factors that affect its persistence at a site are also important. Watson (2005) in grassy woodland near Sydney found that *Themeda* dominated the ground layer in sites burnt at least once a decade but not in sites which had more than 20 years between the last two fires. Watson (2007) also reviewed literature relevant to the health and persistence of *Themeda*. This review found that long-unburnt sites or sites with infrequent fire had lower densities of *Themeda* in both Australian grasslands and South African savanna compared to sites with frequent burns (based on Robertson 1985; Clarke 2003; Uys *et al.* 2004; Fynn *et al.* 2005; Prober and Thiele 2005). This difference was attributed to less self-shading suppression of growth in burnt sites (based on Bond 1997; Morgan & Lunt 1999**),** the fact that fire removes mulch which otherwise prevents seedling regeneration (based on Morgan & Lunt 1999) and also that fire may provide seed germination cues (based on Baxter *et al.* 1994; Wood 2001). Basic physiological differences among grass species (i.e. many lineages native to hot arid environments are “C4 grasses” and grow more effectively in dry conditions than other “C3 grasses”) may also contribute to different fire responses among grass species. Thus Watson (2007) suggested that because summer-growing C4 grasses such as *Themeda* use water more efficiently and have lower nutrient requirements than all-season and winter-growing C3 grasses like *Microlaena stipoides*, *Poa* and *Austrodanthonia* species (Ojima *et al.* 1994; Nadolny *et al.* 2003), these characteristics may give C4 species a competitive advantage in a frequently burnt environment (Fynn *et al.* 2003).

Watson (2007) also found compelling evidence that the competitiveness of *Themeda* may provide resilience to the grassland systems it dominates in relation to weed invasion. This evidence comes from negative associations of weediness with *Themeda* (based on Morgan 1998d; Lunt and Morgan 1999b; Cole & Lunt 2005; Watson 2005) and also from studies that have investigated nitrogen levels and regulation in woodlands with a *Themeda* dominated ground layer (Prober *et al.* 2002b; Prober *et al.* 2005). Regeneration of *Themeda* combined with high frequency fire may thus reduce the elevated nutrient status of weed infested sites and promote system resilience. However, there needs to be some caution attached to its widespread application as “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 & Miles 2005). There are examples of results (Lunt & Morgan2000) and applications (Johnson 1999) of integrated strategies that included fire in the management of weed species in grassy systems, and use of this approach within an adaptive management framework may be worth exploring in grassland remnants.

Research has also investigated the effect of grazing in these grasslands. Specifically some researchers have investigated grazing impacts compared to vegetation removal and found that there are differences among these treatments (e.g. Verrier & Kirkpatrick 2005). More generally grazing has been associated with a decline in *Themeda* (Prober & Thiele 1995; Chilcott *et al.* 1997; Fensham 1998; McIntyre *et al.* 2003; Dorrough *et al.* 2004).

An informative study by Prober *et al*. (2007) investigated the interaction of fire frequency and grazing (native grazers and mowing) in two contrasting systems, one with a *Themeda australis* dominated ground layer and one with a *Poa sieberiana* dominated ground layer. Key conclusions were that: 1. *Themeda* was resilient to 4 and 8 year fire frequencies but declined with biennial burning under drought conditions; 2. *Poa* reduced in dominance (was replaced by *Themeda*) with high fire frequency; 3. Low frequency fire (up to 14 years fire free) did not lead to sward collapse of *Themeda* (possibly because of increased *Poa* abundance), and 4. Grazing exclusion led to increased tussock abundance and inflorescence production. They concluded that a system with mixed dominants (*Themeda* and *Poa*) is likely to have increased resilience in the face of varying fire frequencies and grazing pressures.

1. Temperate Montane Grasslands, comprising 2 RVCs,

a) Wet tussock grasslands of cold air drainage areas of the tablelands
b) Kangaroo Grass – Redleg Grass – Speargrass dry grasslands of the South Eastern Highlands

In the Lachlan Valley Temperate Montane Grasslands occur between 600 and 1350m above sea level, and receive between 500 and 750mm rainfall each year. Composition varies with altitude, topography and soil type. Fertile soils and gentle relief are prominent features of the landscapes that Temperate Montane Grasslands occupy within the Lachlan Valley (Costin, 1954a; Benson, 1994, Keith & Bedward, 1999; Thomas *et al.*, 2000 Keith 2004).

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). 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 & 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). 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 occupy a highly limited area within the Lachlan Valley, essentially restricted to higher elevation landscapes in the far eastern and northeastern parts of the catchment (DEC, 2006; OEH, 2011). These grasslands are restricted to an area from Crookwell to Mt Canobolas along the higher ranges and plateaux across the eastern and northeastern edge of the Lachlan catchment.

“Natural Temperate Grasslands of the Southern Tablelands” is listed as an Endangered Ecological Community under the Commonwealth Environment Protection and Biodiversity Conservation Act (1999), although not listed under the Threatened Species Conservation Act (1995). Most Temperate Montane grasslands in the Lachlan Valley are highly degraded and none appear to be protected in formal conservation reserves. The resilience of these communities and their prospects for survival as a viable ecosystem are therefore limited.

A suite of threatened fossorial (ground burrowing) reptile species are mostly restricted to Temperate Montane Grasslands and occur in the Lachlan CMA including the nationally vulnerable Pink-tailed Legless Lizard (*Aprasia parapulchella*) and the Little Whip Snake (*Suta flagellum*). Populations of these species are small and suitable habitats are highly fragmented (Jenkins and Bartell 1980; Jones 1992, 1999 Osborne & McKergow 1993; Osborne & Jones 1995).

Two distinct Temperate Montane Grassland communities are known from the Lachlan Valley (OEH, 2011). One is a tussock grassland community restricted to swampy areas with impeded drainage on valley floors. This community is dominated by Tussock Grass (*Poa labillardierii*), Snow Grass (*P. sieberiana* var. *sieberiana*), Saw Sedge (*Carex appressa*) and Swamp Foxtail Grass (*Pennisetum alopecuriodes*). With increasing moisture in the lower parts of the landscape, tussock grassland communities intergrade into Montane Bogs and Fens.

The second of the Temperate Montane Grassland communities that occurs in the Lachlan Valley is dominated by Kangaroo Grass (*Themeda australis*), Redleg Grass (*Bothriochloa* spp.) and Speargrass (*Austrostipa* spp.). This community occurs on better drained soils than wet tussock grassland communities and occupies relatively fertile soils derived from various substrates.

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. 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*, 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 & McIntyre 1994; Kirkpatrick *et al.* 1995).

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) 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 & Hocking in prep.), and that short inter-fire 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 managed with six burns over 17 years (Lunt & 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 & Morgan 1999b).

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 & Groves 1991; Lunt 1995, 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 on *page 5* 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).

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 & Thiele 1993; Nadolny *et al.* 2003; Prober & 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. 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).

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, than in a control area which had had 17 years between fires (Lunt & 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 savannah 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 & Lunt 1999). Fire removes the thick mulch of dead material which prevents seedling regeneration (Morgan & Lunt 1999). Smoke may play a role in cueing 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 C4 grasses such as *Themeda* use water more efficiently and have lower nutrient requirements than all-season and winter-growing C3 grasses like *Microlaena stipoides*, *Poa* and *Austrodanthonia* species (Ojima *et al.* 1994; Nadolny *et al.* 2003), and these characteristics may give C4 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 (Prober & Thiele 1995; Chilcott *et al.* 1997; Fensham 1998; McIntyre *et al.* 2003; Dorrough *et al.* 2004). Where grazing has reduced but not eliminating *Themeda* from the grass sward, fire may play a useful role in its retention and regeneration (Prober & Thiele 2005).

Are other matrix grasses likely to respond in a similar fashion to fire? The ability of C4 species to use nitrogen and water efficiently suggests these species are more likely than C3 grasses to have a positive response to fire. Some C3 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. However, 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.

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 & 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 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) 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 & 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 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 aquatic* (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 & Morgan 1999a).

2. Riverine Plain Grasslands, 3 RVCs

a) *Plains Grass grassland on alluvial dark grey clays of central New South Wales (Benson 45)*
b) *Curly Windmill Grass – Speargrass – Wallaby Grass on alluvial clay and loam on the Hay Plain, Riverina Bioregion (Benson 46)*
c) *Corkscrew grass grassland/forbland on sandplains and plains in the semi-arid (warm) climate zone (Benson 165)*

The extensive riverine plains of the lower Murray, Murrumbidgee and Lachlan Rivers are composed of very fine grained grey-brown clay and clay loam soils. These floodplains carry short, open tussock grasslands with an abundance of herbs (Keith 2004). Riverine Plain Grasslands occupy floodplain and valley floor landscapes in the west of the Lachlan Valley. Riverine Plain Grassland communities occur on soils that are generally alluvial or aeolian in their formation and include some of the most fertile cracking clay soils of the catchment (Keith 2004).

Riverine Plain Grasslands have previously been described by Beadle (1948), Porteners (1993), Benson *et al.* (1997), Keith (2004) and Benson *et al.* (2006). Benson (2008) and OEH (2011) identify three distinct Riverine Plain Grassland communities as occurring in the Lachlan Valley. Riverine Plain Grasslands are primarily restricted to the western plains of the Lachlan Valley in the Riverina and Murray Darling Depression Bioregions. Each of the three RVCs known from the Lachlan Valley occupies a distinct landform and substrate.

Plains Grass (*Austrostipa aristiglumis*) grassland occurs on dark grey, self-mulching clays and clay loam soils in slightly low lying areas of the floodplains and alluvial plains of central NSW extending from the Riverina Bioregion to north of Warren in the north-central wheatbelt of NSW (Benson *et al* 1997). The majority of the extant and historic extent of Plains Grass grasslands is on the heavy cracking clay soils of the Liverpool Plains within the Namoi catchment. This grassland community often contains a high proportion of annual exotic species in spring. It is poorly represented in reserves and threatened due to cropping and grazing. Much smaller areas of this community are scattered through the Central West and the Lachlan CMAs.

The heavy basalt derived clay soils of the plains are regarded as having supported botanically distinct grassland within an area of generally lower rainfall (400-650 mm) compared to the Temperate Montane Grasslands to the east and south (Keith 2004). Today, these grasslands have generally been heavily modified by agriculture and remnants often occupy small roadside strips or fragmented patches in the landscape (Benson 1997). This grassland class is distinguished by the dominance of *Austrostipa aristiglumis* (Plains Grass) and common grass genera of the Temperate Montane Grasslands such as *Poa* are generally absent. The Liverpool Plains are regarded as naturally treeless and historical records compiled by Lang (2008) support this view. However, although fire may be important in restricting woody species dominance in Grassy Woodlands and Semi-arid Woodlands of the Lachlan CMA (see reviews of these ecosystems that follow), this may not be the case on the *Austrostipa aristiglumis* dominated plains. Lang (2008) reviewed the factors that drive grassland occurrence and concluded that fine textured soils, topography and climate commonly interact to restrict water availability to the subsoil which then restricts woody vegetation growth in many grassland systems. The heavy clay soils of the plains are thus considered to be a primary driver of the grassland occurrence.

Lang (2008) also reports on early records that mention high levels of native mammals with numerous diggings that would have provided natural soil disturbance to the plains grasslands and thus created colonisation opportunities for forbs. The dominance of *Austrostipa aristiglumis* is regarded by Lang to be an artefact of high grazing pressure and other changes post European settlement with the suggestion that these grasslands were possibly dominated by species such as *Themeda avenacea* (Native Oatgrass) and *Eulalia aurea* (Silky Browntop), with a range of forbs occupying the interstitial spaces There is limited historical evidence of Aboriginal burning on these plains, however, given the high productivity of the landscape and reported extensive use of fire by Aboriginal people elsewhere, this remains an open question.

There appear to be no studies into the fire ecology of the *Austrostipa aristiglumis* grasslands of the plains. These grasslands are ecologically distinct from other grasslands of the Lachlan CMA (i.e. they may not be "fire modified" where fire has removed woody vegetation, but may lack woody vegetation because of soil / water characteristics). Thus, it is problematic to infer appropriate fire regimes from other grasslands.

Benson (2006) describes Community 46 Curly Windmill Grass (*Enteropogon ramosus*) – speargrass (*Austrostipa* spp.) - wallaby grass (*Austrodanthonia* spp.) as occupying fertile alluvial clay and clay loam soils in the Riverina. This community is an open to closed tussock grassland generally about 0.3 m high dominated by Curly Windmill Grass, corkscrew grass (*Austrostipa nodosa* and/or *Austrostipa scabra*) and a number of wallaby grass species (*Austrodanthonia* spp.) with a range of forbs (Benson *et al.* 1997). This community probably would have been mixed with chenopod shrubs, particularly Bladder Saltbush, prior to European settlement. The saltbush has been eliminated by grazing and dieback leaving a grassland community. Annual weeds such as Wild Oats and Rye Grass can dominate in spring. In relation to fire in this community, Benson (2008) notes "Occasional wildfires sweep across plains perhaps every few decades or so". There do not appear to be any studies into the fire ecology of the community. The nationally vulnerable Plains Wanderer (*Pedionomus torquatus*) is highly reliant upon this grassland community (Roberts and Roberts 2001).

Corkscrew grass (*Austrostipa scabra*) communities (Benson 165) occupy sandy loam or clay loam soils in drainage depressions, swales or plains on aeolian dunefields, sandplains or higher alluvial plains in the semi-arid and arid climatic zones extending into South Australia and Victoria. The soil of these open herblands is heavier than that supporting mallee. Some areas are possibly natural but others have probably been derived through heavy grazing disturbance of woody plant communities such as bluebush shrubland or *Casuarina pauper* woodland (Benson *et al* 2006). Nothing appears to be published about fire in relation to this community, although Benson *et al. (*2006) note that this community is rarely burnt although every few decades or so fires burn adjoining mallee communities so some of these herblands may be burnt at the same time.

3. Western Slopes Grasslands, comprising a single RVC that is a derived community.

a) *Derived tussock grasslands of the central western plains and lower slopes of NSW (Benson 250)*

The only Western Slopes Grassland community known from the Lachlan Valley is a derived tussock grassland community. This community occupies fertile and higher productivity landscapes across the low hills and rises of the NSW Southwestern Slopes bioregion. This community has been created by the complete removal of box-gum woodland trees from the landscape.

4. Semi-arid Floodplain Grasslands

a) *Couch Grass grassland on river banks and floodplains of inland river systems (Benson 50)*
b) *Rat's Tail Couch sod grassland of inland floodplains (Benson 242)*

Although Keith (2004) describes Semi-arid Floodplain Grasslands as reaching their southern limit on the Darling River, Benson *et al* (2006) and OEH (2011) describe two Semi-arid Floodplain Grassland communities that are known from the Lachlan Valley. Both communities are restricted to alluvial landscapes adjoining the major drainage lines across the western plains. The biomass and floristic composition of these communities vary greatly over time and they are strongly driven by prevailing climatic conditions. Due to their landscape position, these grasslands are often highly impacted by grazing by domestic stock, rabbits and other herbivores. Few areas remain in good condition and it is unlikely that any weed free areas of this class of vegetation remain. Benson (1996, 1997) suggests that the extensive native grasslands of the Riverine Plain dominated by *Austrodanthonia - Austrostipa - Chloris* have most likely been derived since from a saltbush and *Acacia pendula* (Myall) shrubland as a result of agricultural clearance. Benson *et al* (2006) identify a Swamp Grassland of the Riverine Plain Community (Benson Community 47). Although a grass dominated community, this community is classified within the Inland Floodplain Swamp vegetation class of Keith (2004).

The primary drivers of plant biodiversity in Semi-arid Floodplain Clay Grassland communities in north-western NSW were investigated by Lewis *et al.* (2009). This work focused on Mitchell Grass (*Astrebla* spp.) dominated grassland communities occurring within a different climatic domain to those within the Lachlan CMA. Lewis *et al (*2009) found that of the disturbance-related variables, cultivation, stocking rate and flooding frequency were all influential. Total, native, forb, shrub and subshrub richness were positively correlated with increasing time since cultivation. Flood frequency was positively correlated with graminoid species richness and was negatively correlated with total and forb species richness. Site species richness was also influenced by environmental variables (e.g. soil type and rainfall). Despite the resilience of these grasslands, some forms of severe disturbance (e.g. several years of cultivation) can result in removal of some dominant perennial grasses (e.g. *Astrebla* spp.) and an increase in disturbance specialists. This study determined that environmental variables accounted for a greater proportion (61.3%) of the explained variance in species composition than disturbance-related variables (37.6%). Although the Lachlan region lacks the warm summer climate of the Namoi and Border Rivers-Gwydir regions necessary for the growth and reproduction of Mitchell Grasses (*Astrebla* spp.), the findings of Lewis *et al.* (2009) may be applicable to the Semi-arid Floodplain Grasslands of the Lachlan Valley because of similarities in landform and ecosystem function.

Benson Community 50 is a Couch Grass (*Cynodon dactylon*) low mat grassland often with *Juncus* spp., and sedge species such as *Eleocharis pallens*. Other grasses vary across a wide distribution but include Umbrella Grass (*Digitaria divaricatissima*), *Digitaria brownii* and *Sporobolus creber*. This community occurs on elevated rises and river banks on red to brown clay or loamy soils on the floodplains of inland river systems in a number of climate zones, associated species composition varies across its distribution but Couch Grass is a constant dominant species (Benson *et al.* 2006). This community grades into adjoining River Red Gum and Black Box dominated classes. Fire is not thought to play an influential role in the dynamics of this grassland due to its low biomass (Benson *et al.* 2006).

Benson Community 242 is a low, sod grassland dominated by Rat's Tail Couch (*Sporobolus mitchellii*) that usually covers >50% of the site (Benson *et al* 2006). Common ground cover species include grasses such as Windmill Grass (*Chloris truncata*) and Blown Grass (*Lachnagrostis filiformis*) (McGann & Earl 1999; Lewer *et al* 2003). This community occurs on silty clays on intermittent lake beds and depressions on alluvial plains and floodplains, because of this riparian position, weeds may be very common and include *Medicago polymorpha*, *Cirsium vulgare, Hordeum leporinum* and *Xanthium spinosum*. This grassland naturally occurs as patches in Black Box dominated woodland or may result from clearance of these box woodlands. This community requires periodic flooding, with many species occurring in semi-wetland sites. Fire is not likely to play a significant role in the dynamics of this grassland community because of the lack of biomass, dampness and sod-type of grassland (Benson *et al.* 2006). This also applies to adjoining Black Box woodlands.

FIRE IN GRASSY WOODLANDS

Grassy Woodlands are amongst the most productive of ecosystems in the Lachlan Valley. They mostly occupy fertile landscapes and flatter terrain. Because of the relatively high site productivity and deep soils in the landscapes occupied by grassy woodlands, they are ecosystems that are preferentially targeted for conversion to agricultural uses. Because of this most Grassy Woodlands in the Lachlan Valley are recognised as Endangered Ecological Communities under the TSC Act (*Table 2)*. Grassy Box-Gum Woodlands are recognised as Critically Endangered Ecosystems under the EPBC Act (*Table 3*).

Grassy woodlands are a critically fragmented ecosystem in Australia (Prober and Thiele, 2004; Commonwealth of Australia 2008). A great diversity of grassy woodland communities occupy the Lachlan Valley (Keith 2004; DEC, 2006, Benson *et al* 2006). The floristic composition of grassy woodland habitats in the Lachlan Valley is strongly driven by a climatic gradient varying from an average annual rainfall of 1200mm per annum on the central tablelands to less than 300mm in the far west of the catchment. This gradient encompasses the entire semi-arid as well as parts of the arid and temperate climatic zones (Bureau of Meteorology data).

The Lachlan Valley potentially supports the greatest diversity of Grassy Woodlands of any natural resource management region in Australia (*pers. obs* / OEH data 2011). This is primarily the result of the substantial variation in rainfall across the east-west gradient and the complex interactions of elevation, fire, topography, geology and soil formation processes. The climatic gradient present in the Lachlan Valley allows for the existence of a wide spectrum of grassy woodland classes from Southern Tablelands Grassy Woodlands in the east to Floodplain Transition Woodlands in the west.

Four classes of grassy woodlands are documented for the Lachlan Valley (OEH, 2011; Benson/RBG, 2008): Subalpine Woodlands, Southern Tableland Grassy Woodlands, Western Slopes Grassy Woodlands and Floodplain Transition Woodlands, collectively they span 26 Regional Vegetation Communities (Benson *et al* 2006 and OEH 2011). The composition and distribution of grassy woodlands in the Lachlan Valley has been described by Benson, 1996 2005; ERM, 1995; Porteners, 1996, 1998, 2000, Hunter 2000, DLWC, 2003, DEC, 2006 and Benson 2008 although most of this work has focused on the vegetation communities contained within formal conservation reserves and public lands. A considerable body of fire ecology research now exists for the Grassy Woodlands. This research has predominately focused on the Western Slopes Grassy Box Woodlands of New South Wales and on Cumberland Plain Woodland from Western Sydney. These woodlands are structurally similar and share a number of species from the lower strata but have distinct tree species. Thus the research in these Grassy Woodlands is of considerable value in relation to the Grassy Woodlands of the Lachlan CMA.

Keith (2004) presents both an extant and pre-clearing map of New South Wales vegetation. At this state-wide scale these once widespread woodlands (which ran as a contiguous vegetation band from Queensland to Victoria) have been subject to the highest level of clearing of any vegetation formation in New South Wales. These grassy woodlands are often highly fragmented and therefore have reduced ecological function for fauna. This is particularly well documented for woodland bird species (Reid 2000; Ford *et al*. 2001). Fragmentation is likely to also affect fire regimes and hence floristic patterns (Hobbs & Hopkins 1990). Research to date indicates that fire plays an important role in regulating these woodlands (Prober & Thiele 1993, 2005).

The highest elevation class of grassy woodlands, Subalpine Woodlands, is listed by Eco Logical (2006) as occurring in the Lachlan Valley and has been mapped by DEC (2006). In the course of this review no mapping was located for this class of grassy woodland. Because of the very limited existence of country over 1000m asl in the Lachlan Valley, it’s essentially limited to the Mount Canobolas area, Subalpine Woodlands are consequently of very limited occurrence. (Hunter 2000, Keith 2004)

1. Western Slopes Grassy Woodlands, comprising 13 RVCs

a) *Apple Box moist gully grass-forb open forest of the NSW South Western Slopes and South Eastern Highlands Bioregions (Benson 283)*
b) *Blakely's Red Gum - Long-leaved Box - Cootamundra Wattle shrubby woodland of the southern NSW South Western Slopes Bioregion (Benson 280)*
c) *Blakely's Red Gum - Rough-Barked Apple flats woodland of the NSW western slopes (Benson 281)*
d) *Blakely's Red Gum - White Cypress Pine woodland on lower slopes of hills in NSW South Western Slopes Bioregion (Benson 279)*
e) *Blakely's Red Gum - Yellow Box grassy woodland of the NSW South Western Slopes Bioregion (Benson 277)*
f) *Fuzzy Box - Inland Grey Box on alluvial brown loam soils of the NSW South Western Slopes Bioregion and southern BBS Bioregion (Benson 201)*
g) *Mugga Ironbark - Black Cypress Pine woodland on hillslopes and ridges of the Central Lachlan region of the South Western Slopes*

h) *White Box - Apple Box valley herbaceous woodland mainly of the NSW western slopes (Benson 275)*
i) *White Box - Black Cypress Pine - Tumbledown Gum - Mugga Ironbark shrubby woodland in hills of the NSW central western slopes (Benson 272)*
j) *White Box - Blakely's Red Gum - Yellow Box grassy woodland of the NSW South Western Slopes Bioregion (Benson 282)*
k) *White Box - White Cypress Pine - Inland Grey Box woodland on the western slopes of NSW (Benson 267)*
l) *White Box grassy woodland on well drained podsolic clay soils on hills in the NSW South Western Slopes Bioregion (Benson 266)*
m) *Yellow Box tall grassy woodland on alluvial flats mainly in the NSW South Western Slopes Bioregion (Benson 276)*

A diversity of Western Slopes Grassy Woodland communities occur in the Lachlan Valley, This region is a significant component of the South Western slopes of New South Wales, a bioregion characterised by its grassy woodlands. These landscapes are today the "Wheat Belt", a landuse that results from the conversion of millions of hectares of grassy box gum woodlands into agricultural landscapes (Prober and Thiele 2001).

Western Slopes Grassy Woodland communities generally occupy sites with deep and fertile soils on lower slopes and valley floors (Prober& Thiele 2001 Benson 2008). Exceptions to this are grassy woodland communities dominated by Mugga Ironbark, Long Leaved Box and Black Cypress Pine that occur on upper slopes and ridgetops. Slope and ridge top variants of grassy box woodland communities dominated by White Box, Blakelys Red Gum and Yellow Box have an understorey that generally contains more shrubs than communities occurring on lower slopes. Despite this, these communities are still classified as grassy box woodlands (Benson *et al.* 2006, OEH, 2011).

OEH (2011) lists 13 RVCs as occurring within the Western Slopes Grassy Woodland class in the Lachlan Valley. In the higher rainfall and more productive parts of the Lachlan Valley these communities are dominated by White Box (*E. albens*), Yellow Box (*E. melliodorai)* and Apple Box (*E. bridgesiana*), Blakelys Red Gum and Rough-barked Apple (*Angophora floribunda*). On floodplains with lower rainfall (500-600mm) further to the west, Fuzzy Box and Inland Grey Box dominate and in lower productivity sites and on slopes and ridges, Mugga (*E. sideroxylon*), Tumbledown Red Gum (, *E. dealbata)* and White Cypress (*Callitris glaucophylla*) dominate these woodland communities.

The Ground layer

In the last decade there has been extensive ecological research undertaken in the grassy box woodlands of NSW. Much of this research has concentrated on the ground layer and this work has been reviewed for the grassland formation above. This work has addressed the effects of fire on species diversity, system resilience, weed invasion and the interaction of fire with grazing. The work strongly implies that: 1. fire intervals of ~ 2 – 10 years will generally be optimal for maintaining diversity; 2. longer fire intervals will favour particular grass tussock species (e.g. *Poa sieberiana*) in the ground layer (rather than *Themeda australis*); 3. diversity in the grass species present may help buffer the ground layer against weed invasion; 4. increased nitrogen levels (e.g. associated with grazing) lead to a decline in the system (including declines in the grass species that underpin the system - Prober *et al.* 2002b; Prober *et al.* 2005); and 5. removal of grasscover (artificially) can act in a similar manner to fire, however, grazing is also associated with other changes to the system and is not an ecological replacement for fire.

Does the ground layer in grassy woodlands (which is floristically very similar to the grass / forb layer in grasslands of the slopes and Victorian plains) differ in its ecological response to fire from grasslands? A few studies have investigated the effects of tree and shrub cover on the ground layer of woodlands or other related systems. In African savanna, for example, tree clearing can lead to undesired shifts in the grass composition of the ground layer (Barnes 1979). Given that tree loss from the grassy woodlands has been ongoing until at least the recent past (Fisher & Harris 1999), this is an issue to consider in these woodland systems. Gibbs *et al*. (1999) studied tree and grass interactions on the New England Tablelands and found evidence that there was an association between *Poa sieberiana* and *Microlaena stipoides* and the dominant tree *Eucalyptus laevopinea*. The deep-rooted summer growing *Aristida ramosa*, in contrast, was more abundant away from tree canopies and root zones. This suggests that *Themeda* (also a summer growing grass) may be favoured away from tree canopies. This was confirmed in the study of Prober *et al*. (2002a) in which *Themeda* dominated in gaps and *Poa* dominated under tree canopy in box woodland. Watson (2005) also found significant differences in grass and forb species composition among open patches, patches around trees and patches under *Bursaria* bushes in Cumberland Plain Woodland. Similar interactions among ground layer species and shrubs have been found in other systems and nutrient patterns, water relations, herbivory levels and seed distributions may be associated with these patterns (e.g. Facelli & Temby 2002).

Because overstorey cover and fire can both influence grass dominance there are likely to be complex interactions of tree and shrub cover with fire and the ground layer. For example, tree cover may favour *Poa* which appears to be more resilient to longer inter-fire periods. Tree cover in grassy box woodlands was also associated with higher species richness. This could relate to altered nutrient status, water status or changes in the sward structure (i.e. recruitment opportunity) but it is also likely these factors interact. Watson (2005) did not find increased species richness under canopies in Cumberland Plain Woodland. It is likely that differences in the ground layer may result from differing interactions among vegetation layers and differing historical patterns of cover and fire.

Tree cover

Inferring pre-European vegetation patterns can be problematic because early records of vegetation patterns post-settlement may be biased (e.g. Benson & Redpath 1997). However, there is evidence that the woodlands of eastern Australia did exist in conjunction with large treeless grassland areas before European settlement (Goldney & Bowie 1990; Croft *et al*. 1997). Since settlement there has been a general reduction in tree cover in these woodlands (Prober & Brown 1994; Windsor 1999), but the pattern is not consistent across all areas (Fisher & Harris 1999). Much of this loss of trees from the landscape can be attributed to agricultural activities (Fisher & Harris 1999). Few studies directly investigate the effect of fire on the trees of box woodlands (but see Windsor 2000). However, given the dominating presence of *Eucalyptus melliodora* and *Eucalyptus albens* (Yellow Box and White Box) in these woodlands, understanding the fire ecology of these species is a critical component to understanding the effects of fire on the tree layer. This is particularly so given that the regeneration strategies of eucalypts in general are well understood (e.g. Nicolle 2006) and that there are both conceptual (e.g. Noble & Slatyer 1990) and population models (e.g. McCarthy *et al*. 1999; Tierney 2004) of species that provide a guide to the effects of fire on species with differing reproductive attributes. The key attributes that determine the response of eucalypts to fire are: 1) resprouting capacity post-fire; 2) seed production (time to reproductive maturity; seed set changes in response to fire); 3) seed longevity (as aerial or soil seed banks); 4) seed dormancy (and dormancy break cues), and; 5) seedling establishment, survival and maturation rates.

Eucalyptus melliodora is recorded as being a resprouting species (DEC 2002), and whilst the resprouting capacity of *E. albens* is not documented, it is likely to also be a resprouting species. Additionally, seed characteristics of the eucalypts are generally considered consistent within the group. Eucalypt seed generally lacks dormancy (Gross & Zimmer 1958; Langkamp 1987) and does not form persistent soil seed banks (Ashton 1979; Vlahos and Bell 1986; Myerscough 1998; Read *et al.* 2000; Hill & French 2003) although some aerial seed storage does occur (House 1997). Eucalypt seed also has morphological features associated with low dispersal capacity (small size; high density; no dispersal appendages – see van der Pijl 1982) and related species with these features do have limited dispersal abilities (Tierney 2003). Of these attributes, the capacity to resprout post-fire is perhaps the major driver of eucalypt fire response. Yates *et al*. (2007) modelled population persistence in the rare Myrtaceous species *Verticordia staminosa* and found that adult survival contributed the most to long-term population persistence. Similarly, *Eucalyptus caesia* populations appear to be most buffered from decline because this species resprouts (Yates *et al*. 2003). These species occur in vegetation with an open structure. The coastal woodland species *Angophora inopina* is another resprouting species from the Myrtaceae that has been modelled to be highly resilient to altered fire frequencies in the short to medium term (Tierney 2004). In contrast to the general trend amongst woodland eucalypts, a number of forest eucalypts are seeder species and allocate more resource to growth than to storage; this life-history strategy may be less successful in open woodland systems (see Bond & Midgley 2001 and references therein).

Over the long-term, loss of trees from woodlands may occur, even for resprouting species (Bowen & Pate 1993). Recruitment events are required for tree replacement. A number of key attributes can be postulated as likely to affect recruitment opportunity (e.g. gaps in the ground layer; competition with other species such as weeds; favourable weather conditions – rainfall for seedling establishment). Watson (2007) reviewed the triggers for eucalypt establishment in woodlands and noted the following: 1) fire has been associated with enhanced recruitment opportunities (Semple & Koen 2001; Li *et al.* 2003, Hill & French 2004); 2) recruitment may only occur where the ground layer has low cover (Curtis 1990; Semple & Koen 2003; 3) fire that kills some adult trees may enhance recruitment opportunity (Wellington & Noble 1985); 4) fire can enhance seedbed conditions for recruitment (Clarke & Davison 2001); 5) fire can trigger sufficient seed release to cause ‘predator satiation’ of ants (Andersen 1988; Florence 1996) and 6) recruitment may be episodic depending on the interaction of factors that affect it (Wellington & Noble 1985; Curtis 1990; Clarke 2000).

High mortality in the juvenile (seedling) stage is a feature of many tree species across a range of ecosystems, including eucalypts (Henry & Florence 1966; Wellington & Noble 1985; Clarke 2002). Eucalypt seedlings that do survive probably are fire tolerant within about seven years (e.g. Auld *et al*. 1993) and many species that grow in fire-prone systems will have developed a lignotuber within this timeframe (Semple & Koen 2001; Clarke 2002). These may, however persist as “advanced growth” seedlings for years (Noble 1984). Growth rates of tree seedlings can be highly variable in woodland systems (e.g. Fordyce *et al*. 2000; Tierney 2004). However, under optimal conditions many species probably grow quickly from a suppressed state (e.g. Fordyce *et al*. 2000; Florence 1996). High fire frequency is one factor that can suppress the progression of plants into adult stages (Williams *et al.* 1999).

In sum, woodland tree species that can resprout are likely to be resilient to variations in the fire regime in the short to medium (decades) term. Over longer timescales recruitment events need to occur to replace trees lost from the landscape. Ground layer condition (low sward density; low levels of weed competition) and variations in rainfall, grazing pressure, soil conditions (e.g. nutrient levels) and seed production levels will interact to determine seedling establishment and growth rates. Management actions which may enhance opportunities for eucalypt establishment include restricting grazing and reducing the density of exotic grasses and herbs above and below ground (Windsor 1999; Semple & Koen 2003). Very long-term fire exclusion may also cause a decline in woodland eucalypt populations (Withers & 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.

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 & Clarke 2006).

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; Allan *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. 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 woodlands of the Lachlan CMA?

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). 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 (e.g. *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 (e.g. shrubs with fleshy fruits like *Lissanthe strigosa*, *Persoonia* species, and perhaps also some wattles), or wind (e.g. daisy bushes, dogwoods), are often able to establish in the absence of fire (Campbell &Clarke

2006; Ooi *et al.* 2006). The majority of shrubs found in Western Slopes Grassy Woodland may fall into this category.

One native shrub species which has been very successful in building up its population in the Lachlan Valley is *Cassinia arcuata*, or Sifton Bush (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. 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. 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) 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. 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 & Koen 1993), a relatively short juvenile period. However, repeated biennial burning may produce negative impacts on other ecosystem elements. 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.

Weeds

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. 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 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 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. Post-fire weed control strategies are also likely to be beneficial.

Fauna

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.

An important consideration for these woodlands is the dramatic decline in woodland birds from this landscape (Reid 2000; Ford *et al*. 2001). Fragmentation is considered a key driver of this decline and loss of trees and shrubs from the landscape which contributes to further fragmentation should be avoided. Many of the bird species in decline require either structurally complex habitat or have specific foraging requirements which only persist in larger remnants with intact tree and shrub layers. Frequent fire may reduce these values. However Hannah *et al.* (2007), who report a significant positive association between recency of fire and the abundance of a number of woodland bird species in Central Queensland, recommend increased fire frequency in that grazed, fragmented eucalypt woodland landscape.

Conclusion

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 2005 article Prober and Thiele 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. 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 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. Prober *et al.* (2007) tentatively suggest fires at 5-8 year intervals may be compatible with maintaining a balance between the dominant grasses.

2. Subalpine Woodlands

OEH (2011) does not identify Subalpine Woodlands as occurring within the Lachlan Valley, although DEC (2006) mapped 800 discrete polygons of Subalpine Woodland between Mt Canobolas and the Blue Mountains National Park. All mapped areas are clustered on the Southern Tablelands along the far northeastern margin of the catchment. Although the mapping compiled by DEC (2006) did not identify dominant species in these habitats, Keith (2004) lists Subalpine Woodlands as being dominated by Snow Gum (*Eucalyyptus pauciflora*) with occasional Black Sallee (*E. stellulata*) and Mountain Gum (*E. dalrympleana*). Some areas mapped by DEC (2006) as Subalpine Woodland are likely to be equivalent to Southern Tablelands Grassy Woodlands, particularly Black Sallee – Tussock Grass open Woodland and Snow Gum – Candlebark grassland/woodland communities, and they are treated as such for the remainder of this review.

3. Southern Tableland Grassy Woodlands, comprising 4 RVCs

a) *Black Sallee - Tussock Grass open woodland of the South Eastern Highlands*
b) *Blakely's Red Gum moist sedgey woodland on flats and drainage lines of the South Eastern Highlands and South Western Slopes*
c) *Snow Gum - Candle Bark grassland/woodland of the South Eastern Highlands*
d) *Apple Box – Yellow Box dry grassy woodland of the South Eastern Highlands*

Southern Tablelands Grassy Woodlands occupy high elevation productive lower-slope landscapes in the eastern parts of the Lachlan Valley. Like Western Slopes Grassy Woodlands these communities have been critically fragmented, with most remnants consisting of isolated paddock trees (Manning *et al.* 2006). These woodlands once extended over much of the central and southern tablelands, into Victoria. In the Lachlan Valley they covered a wide area around Goulburn, Crookwell and Yass and north toward Bathurst and Orange (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 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).

The higher elevation variants of Southern Tablelands Grassy Woodlands are dominated by Snow Gum, Black Sallee and Candebark (*E. rubida*), as elevation increases these communities grade into Subalpine Woodlands at Mt Canobolas. The lower elevation variants of Southern Tablelands Grassy Woodlands are dominated by Yellow Box, Apple Box and Blakely's Red Gum. With decreasing elevation and rainfall these communities grade into Western Slopes Grassy Woodlands. Some areas mapped by DEC (2006) as Subalpine Woodland may be equivalent to Southern Tablelands Grassy Woodlands, particularly Black Sallee – Tussock Grass open Woodland and Snow Gum – Candlebark grassland/woodland.

Shrubs are not a major component of the flora; Keith (2004) considers this may be because of prolonged exposure to grazing. Shrub species 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 Southern Tablelands Grassy woodlands has affected the fire regime to the point where very little fire now occurs in this vegetation type in the Lachlan Valley. 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 in the Central West CMA (pers. comm. Steve Woodall, NPWS, Bathurst, 2006).

In sections 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. The grassland section of this review 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.

Ground layer

The discussion of the effects of fire on ground layer vegetation in Western Slopes Grassy Woodlands is also relevant for the vegetation class under discussion here. In fact, the fire frequency plots studied by Prober *et al.* (2011) 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 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 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).

Trees

The potential of fire to assist efforts to establish woodland eucalypts has been explored previously in this review. In this section, four articles related to tree regeneration specifically in Yellow Box – Blakely's Red Gum woodlands are summarized. Three reports 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 the 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 preferred option. 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 postfire 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).

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). An alternative, or perhaps a complementary explanation, is that recruitment in many woodland shrub species is uncommon in the absence of fire.

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 we 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. 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 (see *page 8*). 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 seed heads 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 C4 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 C4 tussock grasses (See Northern River Fire and Grazing review).

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 C4 grasses which were efficient users of N, while lack of fire allowed N to build up, tipping the balance towards C3 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 C3 grass taxa *Austrodanthonia* and *Austrostipa* or, in less naturally fertile areas, by *Aristida* and *Bothriochloa*, C4 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 C3 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 C4 tussock grasses, mean that the two disturbances are unlikely to be interchangeable for all native fauna.

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.

4. Floodplain Transition Woodlands, comprising 9 RVCs

a) *Inland Grey Box - Poplar Box - White Cypress Pine tall woodland on red loams mainly of the eastern Cobar Peneplain Bioregion (Benson 82)*
b) *Inland Grey Box - White Cypress Pine tall woodland on sandy loam soil on alluvial plains of NSW South-western Slopes and Riverina Bioregions (Benson 80)*
c) *Inland Grey Box tall grassy woodland on alluvial loam and clay soils in the NSW South Western Slopes and Riverina Bioregions (Benson 76)*
d) *Mixed box woodland on low sandy-loam rises on alluvial plains in central western NSW (Benson 248)*
e) *Mixed Eucalypt woodlands of floodplains in the southern-eastern Cobar Peneplain Bioregion (Benson 251)*
f) *Poplar Box - Belah woodland on clay-loam soils of the alluvial plains of north-central NSW (Benson 56)*
g) *Poplar Box grassy/shrubby woodland on alluvial clay-loam soils mainly in the temperate (hot summer) climate zone of central NSW (wheatbelt) (Benson 244)*
h) *Riverine Inland Grey Box grassy woodland of the semi-arid (warm) climate zone (Benson 237)*
i) *Riverine Yellow Box - River Red Gum tall grassy woodland of NSW South West Slopes and Riverina Bioregions (Benson 74)*

Floodplain Transition Woodlands occupy the fertile soils of the upper floodplains and peneplain margins where “the Western Slopes of the Great Dividing Range merge into the vast plains of the Murray-Darling River system. These woodlands are characterised by a largely continuous grassy ground cover and a sparse layer of mostly sclerophyllous shrubs” (Keith 2004). In the Lachlan Valley Floodplain Transition Woodlands occur on the far western slopes and the western plains and peneplains of the Lachlan Valley. These landscapes receive the lowest average annual rainfall (below 550mm) of any of the grassy woodland classes (Keith 2004), because of this they are floristically distinct from other classes of grassy woodland, with a prominent element of semi-arid plant species in the ground layer (Beetson 1980; Austin *et al* 2000; Metcalfe *et al*. 2003; Prober and Thiele 2004). To the west with diminishing rainfall Floodplain Transition Woodlands grade into several Semi-arid Woodland classes and to the east under a higher rainfall regime they grade into Western Slopes Grassy Woodlands (Prober and Thiele 2004).

The Lachlan CMA supports 9 Floodplain Transition Woodland communities (Benson *et al* 2006). This is potentially the highest diversity of Floodplain Transition Woodland communities for any CMA region in NSW (*per. Obs.*). These communities occupy several distinct landforms, including undulating peneplains atop sandstone, sandy alluvial rises, clay loam flood terraces and riverbanks. Species dominance varies accordingly, from Inland Grey Box (*E. microcarpa*) and Poplar Box (*E. populnea* ssp. *bimbil) to* White Cypress Pine (*Callitris glaucophylla*), Belah (*Casuarina cristata*), Yellow Box (*E.melliodora*) and River Red Gum (*E. camaldulensis*) (Austin *et al* 2000; Prober and Thiele 2004). Small variations in topography as well as subtle changes in fluvial and aeolian processes are the main drivers of variation in floristic composition and structure within the Floodplain Transition Woodland vegetation class (Austin *et al* 2000; Metcalfe *et al*. 2003).

Several Floodplain Transition Woodland communities including Fuzzy Box (*E. conica*) Woodland on alluvial soils and Inland Grey Box (*E. macrocarpa*) Woodland are gazetted as Endangered Ecological Communities. This gazettal recognises the very poor condition and reduced extent of these woodlands as well as their ongoing loss (NSW Scientific Committee, 2007). Remaining areas are highly fragmented and suffer the continued effects of grazing and invasion by agricultural weeds (Keith 2004). In many landscapes supporting floodplain transition woodlands, large “scalds” are frequent in those areas of the Lachlan Valley where overgrazing has been severe (Beeston 1980; Keith 2004; Prober and Thiele 2004).

Beeston *et al*. (1980) describe the considerable variation of vegetation communities across the range of Poplar Box. Siversten and Metcalfe (1995) and Priday (2006) describe several broadly distributed floodplain transition vegetation communities across the Lachlan Valley. These communities are defined by their landscape position and relative dominance of box (*E. populnea* ssp. *bimbil, E. conica, E. microcarpa* and *E. melliodora*), Belah (*C. cristata)* and White Cypress Pine (*C. glaucophylla*). The understorey composition changes along an east-west gradient from a dominance by temperate grassy woodland species to an increase in chenopods and other shrubs (Keith 2004).

Benson *et al* (2006) describe a diversity of Floodplain Transition Woodland communities known from the Lachlan Valley. These communities include White Cypress woodland on sandy-loams, Poplar Box – Belah woodland on clay-loam soils on alluvial plains, Western Grey Box tall grassy woodland on alluvial loam and clay soils, Western Grey Box - White Cypress Pine tall woodland on loam soil on alluvial plains, Western Grey Box - Poplar Box - White Cypress Pine tall woodland on red loams, Poplar Box grassy woodland on alluvial clay-loam soils, Mixed box eucalypt woodland on low sandy-loam rises on alluvial plains, Yellow Box - River Red Gum tall grassy riverine woodland, Riverine Western Grey Box grassy woodland and Mixed eucalypt woodlands of floodplains (a vegetation community that is potentially endemic to the Lachlan Valley).

In most instances the pre-European fire regimes of these communities is unknown and high levels of fragmentation have rendered them almost irrelevant, although Benson *et al* (2006) suggest for the Western Grey Box – White Cypress Pine tall woodland community that “the original fire regime may have been 10-40 years" and for most other Floodplain Transition Woodland communities that "fire is rare due to lack of ground biomass and floodplain position". Because of the significantly lower average annual rainfall experienced by the western variants of Floodplain Transition Woodland, there are limits to primary productivity. As a result the continuity of fuel is mostly insufficient for fires to carry (Beeston 1980; Noble 1989). Preceding wet seasonal conditions (La Nina) are generally required to ensure the continuity of fuel that will allow fire to carry within the western variants of this grassy woodland class.

FIRE IN SEMI-ARID WOODLANDS

The semi-arid zone covers over a third of NSW and includes most of the western plains and the drier half of the western slopes of NSW from the Darling River east to a line through Wagga Wagga, Dubbo and Narrabri (Keith 2004). Rainfall in the semi-arid zone is between 250 and 500 mm per annum, with much variability between years (Bureau of Meteorology data). Hence drought plays a major role in shaping vegetation and water availability is often limiting (Keith 2004). The woodlands of the semi-arid zone are dominated by sclerophyll trees (eucalypts, sheoaks, wattles, cypress pines), contain drought-resistant or 'xeromorphic' shrubs (widespread families include Myoporaceae and Chenopodiaceae), and are home to many ephemeral grasses and herbs.

Keith (2004) identifies two subformations within woodlands of the semi-arid zone: grassy and shrubby. Woodlands in the grassy subformation occur on the floodplains in areas subject to occasional inundation. These woodlands intergrade with grasslands and arid shrublands. To the east, grassy subformation semi-arid woodlands become grassy woodlands. The second subformation is the shrubby subformation. Trees in the semi-arid shrubby woodland subformation are generally shorter in stature than those in the grassy subformation, there is less grass cover but frequently xeromorphic shrubs are dominant. Soils vary from sandy outwash soils to red-brown loams on adjacent floodplains. The sandy soils naturally favour shrubs while the heavier floodplain soils are more likely to support a higher cover of grasses (although overall Keith considers this vegetation class as part of the shrubby sub-formation of semi-arid woodlands). As with other vegetation formations, disturbance regimes affect the density of trees, shrubs and grasses. Semi-arid woodlands are dominated by box eucalypts, particularly Bimbil or Poplar Box (*Eucalyptus populnea* ssp. *bimbil*) and Fuzzy Box (*E. conica*) on clay soils, mallee eucalypts, particularly Yorrell (*E.graciilis*), Red Mallee (*E.socialis*) and White Mallee (*E. dumosa)* and Mallee Pine (*Callitris verrucosa*) is also prominent on sandplain soils in the west, and in the east, Yellow Box (*E. melliodora*) and Grey Box (*E. microcarpa*). On stony ridges Dwyers Red Gum (*E. dwyeri*) White Cypress Pine (*Callitris glaucophylla*), and Kurrajong (*Brachychiton populneus* ssp. *populneus*) may also be prominent. Smaller tree species include Bulloak (*Allocasuarina leuhmannii*), Belah (*Casuarina pauper*), Leopardwood (*Flindersia maculosa*), Yarran (*Acacia homalophylla*), Wilga (*Geijera parviflora*), Sugarwood (*Myoporum platycarpum*), Wild Orange (*Capparis mitchellii*) and Whitewood (*Atalaya hemiglauca*). Large shrubs include Hopbush (*Dodonaea viscosa* syn *D. attenuata*), Budda (*Eremophila mitchellii*), Turkeybush (*Eremophila deserti*), Emubush (*Eremophila longifolia*), Needlewood (*Hakea leucoptera*). Smaller shrub species include Creeping Saltbush (*Atriplex semibaccata*), Silky Bluebush (*Maireana villosa*), and other chenopods. The understorey of semi-arid woodlands is dominated by a wide range of native grasses and herbs. In areas of sand dune and sandplain on the southern edge of the Cobar Peneplain Porcupine Grass (*Triodia scariosa* ssp. *scariosa*) is a prominent component of the understorey of the Dune Mallee Woodland class.

European land use has had lasting ecological impacts on these woodlands – including severe degradation from overgrazing. "Many of the current plant communities of the semi-arid region are as much a reflection of this ecological legacy, as of the historical landscapes viewed by Oxley and Sturt" (Keith 2004). Perennial plants are apparently a lynchpin in both the ecology of semi-arid ecosystems, and the sustainability of the pastoral industry (Friedel & James 1995). Ecologically, these plants have crucial roles in the interception and retention of scarce resources, such as moisture and nutrients, in the landscape (Keith 2004). However, the spread of shrub species in recent decades is reported to have significantly changed the structure of some of these woodlands (Cunningham *et al.* 1992).

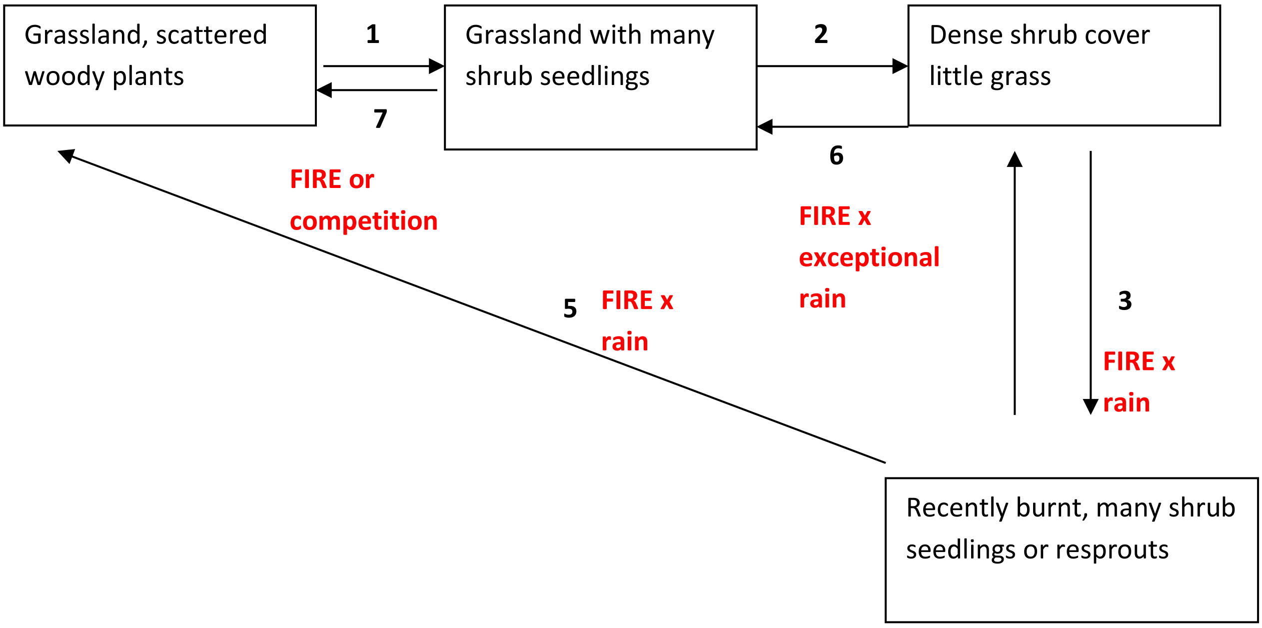
The Lachlan Valley contains the largest remnant areas of semi-arid woodland in NSW (Bradstock and Cohn 2002a;Keith 2004; DEC,2006; OEH, 2011). An exceptional diversity of semi-arid woodlands remains within the Lachlan valley, particularly in the extensive mallee sandplain and dune systems in the western and southwestern parts of the catchment. The block of native vegetation spanning Yathong, Nombinnie and Round Hill Nature Reserves and adjoining freehold lands is the largest complex of semi-arid woodlands in NSW. This area has nationally significant conservation values.

Semi-arid woodlands in the Lachlan Valley occupy a diversity of landforms, ranging from rocky hillslopes of the Cocoparra Range, to gravel plains of the Cobar Peneplain and aeolian sandplains of the western plains. The Lachlan CMA supports 34 regional vegetation communities (OEH, 2011) that combined represents nine classes of semi-arid woodlands as defined by Keith (2004): Inland Floodplain Woodland, North-west Floodplain Woodlands, Dune Mallee Woodlands, Sand Plain Mallee Woodlands, Semi-arid Sand Plain Woodlands, Western Peneplain Woodlands, Riverine Plain Woodlands, Riverine Sandhill Woodlands, Inland Rocky Hill Woodlands and Brigalow Clay Plain Woodlands.

Fire in semi-arid woodlands is a complex issue. Noble & Grice (2002), for example, reviewed the issue of fire in semi-arid lands and it is clear from this review that there are complex interactions of fire with grazing, browsing and rainfall in these systems. It is also true that there is likely to be variability in how fire functions within differing woodland types. This means that reducing fire effects to simplistic fire frequency guidelines is both difficult and potentially counterproductive. This review concentrates on the literature for fire in these systems and what it indicates. Firstly, however, broad ecosystem models that are used to understand these woodlands are explained.

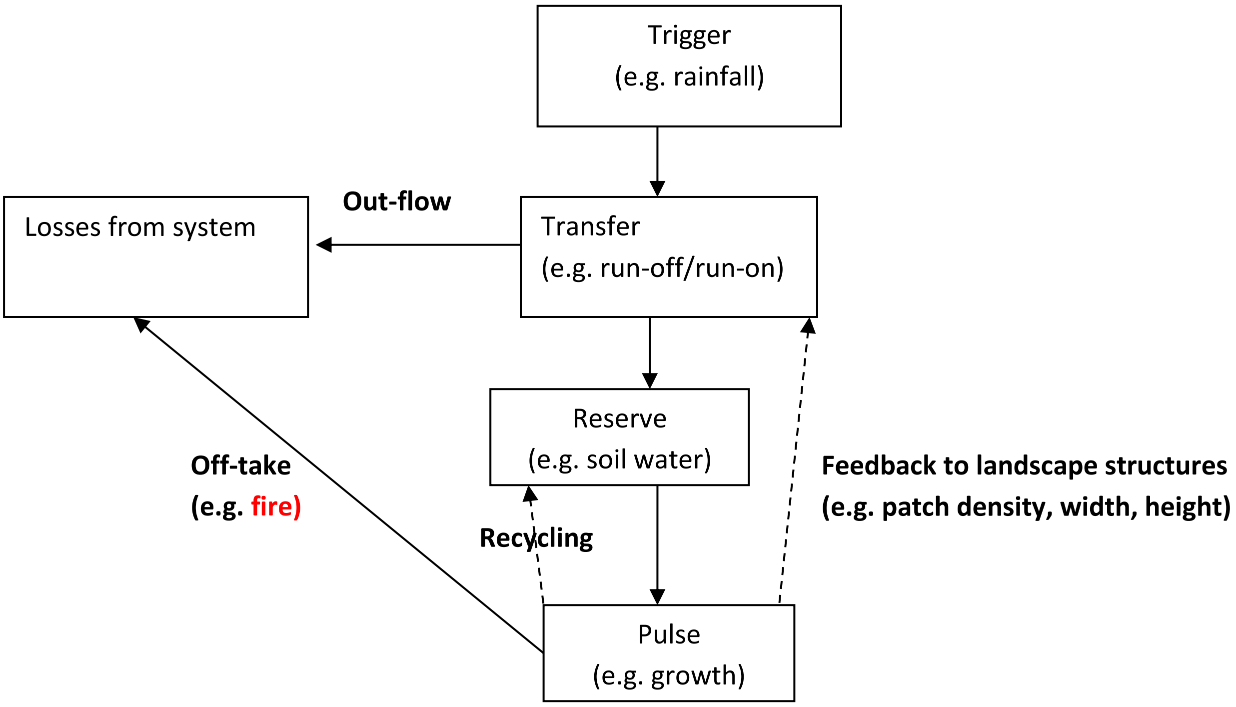
Then the contribution of the literature to understanding these woodlands in relation to fire can be elucidated.

Noble & Grice (2002) use the state-and-transition framework of Westoby *et al*. (1989) to model these woodlands. This defines four possible states and seven transitions (changes) that lead from one state to another (*Figure 3*):



Fire is required for four of the seven transitions (shown as numbers), but it must interact with rainfall for three of these transitions to occur and the other transition (transition 7) can occur alternatively via grass competition with shrub seedlings (or perhaps because of browsing). The requirement that fire must occur with other variables to cause ecosystem change is not unique to these semi-arid woodlands, however the variability in rainfall in the semi-arid zone means that there are likely to be high levels of stochasticity in these transitions. “Rainfall variability is typically between 25 and 30% of the mean, but can exceed 100%, and severe droughts have a frequency of recurrence of about 1: 8 to 1:10” (Mitchell 1991).

Whilst the majority of semi-arid woodland classes in the Lachlan CMA have a eucalypt dominated tree layer (Inland Floodplain Woodlands, Western Peneplain Woodlands, Sand Plain Mallee Woodlands, Semi-arid Sandplain Woodlands, Riverine Sandhill Woodlands, Inland Rocky Hill Woodlands), two classes are dominated by *Acacia* species (Riverine Plain Woodlands and Brigalow Clay Plain Woodlands). Hodgkinson (2002) used a different model: the trigger-transfer-reserve-pulse model (Ludwig *et al*. 1997) to explain the functioning of *Acacia* dominated woodlands (*Figure 4*):



This model emphasizes the role of rainfall as the master trigger, the event that exerts the major influence on the system. It is then the transfer or storage of water that exerts the predominant control of the system (and consequently alterations to water flow / storage can produce major system dysfunction). Fire, along with grazing, are secondary mechanisms (removal of plant biomass – off-take). A range of fire adaptations and life history strategies in the plant species of these *Acacia* systems are reported. Many of these studies, however, are in Mulga (*Acacia aneura*) shrublands.

This review covers semi-arid woodlands, including *Acacia* dominated woodlands (see below). Overall, Hodgkinson (2002) suggests that fire (or the use of prescribed fire) has not been proven by studies to be a tool to meet general biodiversity objectives in these *Acacia* systems. Studies generally support a consistent story of fire in semi-arid woodlands. Perhaps the most difficult aspect of this is the pre-European fire regimes. Although caution needs to be attached to accounts of early European settlers (Benson & Redpath 1997), settlers and explorers did report an open grassy woodland structure across much of the area (Rothery 1970) and frequent burning by Aboriginal people (Nicholson 1981). Fire frequency is considered to have been a factor in maintaining an open grassy structure before European settlement (Hodgkinson and Harrington 1985). This concept is supported by numerous studies that have demonstrated that many woody shrub species are susceptible as seedlings to fire (e.g. Hodgkinson & Griffin 1982) and that grass competition can restrict woody shrub establishment (e.g. Noble *et al*. 1986). However, drought coupled with overgrazing in the decades before 1900 dramatically altered the rangelands (e.g. Griffiths 2001;

Lunney 2001) and extrapolating from present day studies is therefore problematic. Hence imposing pre-European settlement fire regimes is untenable and fire needs to be managed to a set of currently defined objectives (Hodgkinson 2002; Hodgkinson *et al*. 1984).

Fire regime

Fire frequencies for semi-arid woodlands have been estimated by a few authors (at differing scales with differing approaches). Hodgkinson *et al.* (1984) estimate wildfire intervals of 1 in 50 years, and suggested prescribed fire intervals of 10 to 50 years. Walker (1981) estimated fire intervals of 1 in 25 years for Poplar Box woodlands. More recently Turner *et al*. (2008) provided an assessment of fire recorded across the arid and semi-arid zone as detected by satellite over a seven year period. Fire incidents were strongly skewed to the north-west of the study area with regression analyses showing rainfall as the best explanatory variable. That is, higher rainfall equated to higher fire frequency. About three quarters of the study area recorded no fire, predominantly in the south-east and south-west (i.e. this includes the semi-arid woodland zone of NSW).

Walker *et al*. (1981) provided a detailed review of fire behaviour in Poplar Box Woodlands. Fire frequency across the *full* area which these woodlands cover (from north of Longreach Qld to south of Hillston NSW) was estimated to vary greatly down to a minimum of every few years in northern regions. This was attributed to differences in fuel load of the grass species (estimated to vary from 0.5-6 tonnes/ha) and also differing rainfall and degree of shrub and tree cover. Native perennial grass species potentially accumulate fuel over years (depending on grazing etc.) and produce high fuel loads, compared to the annual species present. The invasive *Cenchrus cilliaris* (buffel grass) was noted as producing very high fuel loads leading to the potential for more frequent fire (potentially in most years). The minimum fuel to carry a fire depends on the grass distribution and moisture content, but was estimated to be 1 tonne/ha in the region that covers the Namoi CMA.

Wind speed, temperature, humidity and fuel moisture content also influence the rate of fire spread. Thus, for *Eragrostis – Aristida* dominated systems (carrying 150-300 g per m^2 of fuel) minimum wind speeds of 7 km per hr were required to carry fire, whereas for *Themeda* dominated systems (carrying of grass fuel) fire was reported to carry with wind speeds as low as 1 km per hr. Graphs provided by Walker *et al*. (1981) show the relationship among fire intensity, flame length and fire spread and also fire risk (according to fuel moisture content, wind and rate of spread) for these woodlands. Walker *et al*. (1981) also report a trend for fire to occur after high rainfall (which triggers plant growth) and hence tabulate large wildfire events (1 million hectares or more) in relation to years (i.e. fires in 1957 followed high rainfall in 1956; fires in 1974- 75 followed high rainfall in 1974). Fires in the Western Division were mostly started by lightning (77%) compared to estimates of 9-10% in other areas. The prevailing pattern in these systems is that fire follows periods of higher rainfall.

Fire ecology

The grass layer. A number of authors have investigated the fire ecology of plant species within these woodlands and the grass layer has drawn attention because of its importance in carrying fire. Walker *et al.* (1981) note the importance of the dichotomy among perennial species (e.g. *Themeda australis*; *Monochather paradoxa*) that persist (and therefore potentially can carry fire across long time periods) verses short-lived species that are only present for periods after rainfall. Other workers report similar findings. Harrington *et al*. (1984), for example, found that the main fuel in semi-arid woodland was the cool season speargrass (*Stipa* spp.) which has low persistence year-to year. These short-lived species may also not persist as a seed bank through drought periods (Anderson *et al*. 1996). Other short-lived species in semi-arid woodlands include *Aristida* spp. and *Enneapogon* spp. (Daly & Hodgkinson 1996) and these have been recorded as recovering poorly after fire, at least in some western vegetation types (Mulham 1985). In contrast, Shaw (1957) found that the grass *Heteropogon contortus* had relatively high survival, enhanced germination and was favored by the reduced competition created by annual burning and that this species increased in density in northern semi-arid (Poplar Box) woodlands. Tothill (1969) attributed the germination success of *Heteropogon contortus* to its seed's capacity to bury itself in soil and to germinate with increased soil temperatures which occur post-fire.

The grass x rainfall interaction. The importance of rainfall events in driving fire patterns is supported by numerous studies. Hence rainfall must be considered in understanding fire in the grass layer. Grass growth after rainfall has been reported (e.g. Hodgkinson & Harrington 1985) and studied (e.g. Orr *et al*. 2004) and is acknowledged as a generality across the semi-arid zone. Death of grass species appears also to be dominated by rainfall patterns (lack of rainfall) including rainfall differences measured among years (O'Connor 1994) but rainfall over the previous three month period has been found to be the best predictor of grass death (Hodgkinson & Muller 2005). Although factors such as landscape position influence the soil moisture status (and hence grass death rates), and there are inter-species differences (Hodgkinson & Muller 2005), overwhelmingly, low rainfall (below 75 mm) leads to grass death and reduced fuel for fires.

The grass x grazing interaction. Grazing can significantly reduce grass abundance and therefore affect fire in semi-arid woodlands (Harrington *et al*. 1979; Grice & Barchia 1992). Areas with a history of high grazing pressure can have a reduced capacity for grass establishment when a rainfall event occurs (Anderson *et al*. 1996). Thus a grazing x drought x grass interaction can occur where, for example, high levels of grazing during drought reduce grass seed set (Hodgkinson 1993) and perennial grass populations cannot be maintained (Silcock & Smith 1990).

The grass x shrub / tree interaction. The interaction among the grass layer and shrubs is well established at the landscape level (Daly & Hodgkinson 1996). There is a negative relationship at the landscape level among shrub cover and grass cover. Shrub recruitment can be decreased by grass cover (Harrington 1991), but there are seasonal and species differences so that for some species in some seasons grass cover has been found to facilitate shrub seedlings (Booth *et al*. 1996 b). Thus there are potentially complex interactions among grasses and shrubs (Booth *et al*. 1996 a, b).

Scanlan (2002) reviewed the literature on trees and their relationships with the grass layer in semi-arid systems. The main points of this review were that: 1) most tree species reduce grass production, but some introduced leguminous species increase grass production; 2) grass can reduce tree seedling survival but can have little impact on mature trees except via influencing fire events; 3) modeling indicated that differing outcomes can be produced from the tree / grass interaction and that tree clumping can significantly increase grass production. There is also evidence that trees are an important part of water dynamics and make a significant contribution to “water catching” rather than water shedding at the landscape level. These water dynamics are ecologically significant (Ludwig *et al*. 1997; Eldridge & Freudenberger 2005). Thus water flow / infiltration is considered a critical component to the functioning of the semi-arid woodlands (Ludwig *et al*. 1997) and this is therefore an important ecological role of trees (i.e. for rainfall driven grass growth that drives fire).

The shrub layer. In the absence of fire shrub cover increases (Hodgkinson & Harrington 1985). Since European settlement the evidence that there is suggests that decreased fire frequency has favored shrub species (Hodgkinson 1986), whilst increased grazing has decreased grass cover (Harrington 1991). These two changes strongly interact. These are, however, broad patterns and there are also a range of differing / competing responses (e.g. grazing may also reduce shrub establishment – Auld 1995; grazing effects on shrubs can vary according to the shrub species, its life-cycle stage and the grazer species – Harrington *et al*. 1979).

1. Inland Floodplain Woodlands, comprising 3 RVCs

a) Black Box - Lignum woodland of the inner floodplains in the semi-arid (warm) climate zone (Benson 13)
b) Black Box grassy open woodland of rarely flooded depressions in south western NSW (Benson 16)
c) Black Box open woodland with chenopod understorey mainly on the outer floodplains of the Riverina and Murray-Darling Depression Bioregions (Benson 15)

Inland Floodplain Woodlands occur away from the major inland rivers, often forming a halo around the Inland Riverine Forests that line the banks of the major channels of the Lachlan Valley. Inland Floodplain Woodlands are typical representatives of the grassy subformation of the semi-arid woodlands. The canopy height is up to 25m and the groundlayer is composed of a variety of grasses and herbs. River regulation, overgrazing and clearing have all heavily impacted upon this vegetation class, meaning that few, if any, areas are in good ecological condition (Keith 2004; Benson *et al* 2006). The understorey varies greatly as a result of differences in grazing history, latitude, frequency of flooding and rainfall (Keith 2004). The undestorey of many areas is dominated by exotic species, particularly in the more heavily impacted southern occurrences including the Lachlan Valley (Benson *et al.* 2006).

Inland Floodplain Woodlands in the Lachlan Valley occur on the semi-arid floodplains adjoining main riverine channels in the lower reaches of the Lachlan River. All are dominated by Black Box (*E. largiflorens*) with understorey composition varying from Lignum (*Muehlenbeckia florulenta*) on frequently flooded drainage depressions to chenopods on Lacustrine plains (Pressey *et al* 1984; Porteners 1993; Cohn 1995 Siversten and Metcalfe 1995; Dykes 2002). Three RVCs are recognised by OEH (2011) within this class. Subtle variations in topography, hydrology and substrate are the primary determinants of the floristic composition of these communities. This class occurs on clay, clay-loam or sandy loam, often gilgaied, soils on inner floodplains and on alluvial plains mostly in depressions that are frequently flooded, although variants with a chenopod understorey have been observed on sandy lunettes of some dry lakes in the semi-arid (warm) and arid climate zones of far south western NSW.

In many places the understorey may have once been dominated by Old Man Saltbush (*Atriplex nummularia*) but this has been reduced due to grazing leaving bluebush, Nitre Goosefoot and copperburrs to dominate today. Black Box requires intermittent flooding for seed germination and successional stages are strongly influenced by the intensity of grazing. Prior to European settlement and stock and rabbit grazing it is likely that there was a denser understorey of chenopod shrubs and native grasses (Benson 1998). Fire is rare in this community although Black Box can probably recover from intense fire better than River Red Gum (Benson 2008). However for those Black Box woodland communities where fire-sensitive chenopod shrubs are a dominant or significant component of the understorey, fire plays no major role in the dynamics of this community.

2. Riverine Plain Woodlands, comprising 2 RVCs

a) Yarran shrubland on peneplains and alluvial plains of central-northern NSW (Benson 77)
b) Weeping Myall open woodland of the Riverina and NSW South Western Slopes Bioregions (Benson 26)

Riverine Plain Woodlands are one of two Semi-arid Woodland classes that are dominated by Wattles (*Acacia* spp.) rather than Eucalypts. The structure and floristics of Riverine Plain Woodlands place them in the grassy subformation of Semi-arid Woodlands (Keith 2004). Subtantial areas of Riverine Plain Woodland remained intact until the 1890s when large numbers of Myall (*Acacia pendula*) were cut down for emergency stockfeed during drought and when the highly palatable saltbush understorey was decimated by prolonged overgrazing (Beadle 1948; Moore 1953). The loss of dominant canopy and understorey species converted Riverine Plain Woodlands to Riverine Plain Grassland and it is now nearly impossible to determine the original vegetation of these areas (Keith 2004).

Riverine Plain Woodlands in the Lachlan Valley are composed of two distinct regional vegetation communities, both dominated by the *Acacia* genus. The first community, dominated by Weeping Myall is a mid-high open woodland up to 8 m high. Other tree species include Belah (*Casuarina* *cristata*), while Black Box (*Eucalyptus largiflorens*) and River Red Gum (*Eucalyptus camaldulensis*) may occur in depressions. This community occurs on brown clays or loam soils on alluvial plains mainly in the Riverina and NSW South-western Slopes Bioregions of south-western NSW (Siversten and Metcalfe 1995). Prior to European settlement this community probably contained a dense understorey of saltbush. Much of its original extent has now altered to be a derived native grassland dominated by native grasses and forbs. Weeping Myall is an Endangered Ecological Community because of high rates of clearance and degradation of the community. Much of the present *Austrodanthonia* spp. grasslands of the Riverina and Murray Darling Depression may have been derived from a pre-European *Acacia pendula - Atriplex nummularia* woodland/shrubland (Austin *et al.* 2000). The chenopods, and presumably Weeping Myall, were eliminated from vast regions through a combination of clearing and over-grazing (Lewer *e. al.* 2003).

Occasional wildfire sweeps across the plains - an extensive fire burnt part of the Riverina in 1991. This resulted in mass germination of *Swainsona* and other legume species. Presumably, the seed of *Acacia pendula* is long-lived and may germinate after fire (Benson *et al.* 2006).

The second Riverine Plain Woodland community in the Lachlan is a tall shrubland or low woodland to about 10 m high dominated by Yarran (*A. homalophylla/A. mellvillei* complex) with a sparse small tree layer including Weeping Myall (*Acacia pendula*), Belah (*Casuarina cristata*) and scattered emergent trees including Western Grey Box (*Eucalyptus microcarpa*) and Poplar Box (*Eucalyptus populnea subsp. bimbil*) (Siversten and Metcalfe 1995; Dykes 2002). It occurs on red to brown earths (loams) and red podzolic soils and sometimes gravel on level plains and gentle rises mainly in central NSW (wheatbelt). Small occurrences remain in the mainly cleared northern wheatbelt extending to the Liverpool Plains with larger stands remaining in the Western Division on the eastern edge of the Cobar Peneplain. Seed of Yarran may be long lived and it suckers after fire or clearing (Benson 2008).

3. Brigalow Clay Plain Woodlands, comprising 2 RVCs

a) *Brigalow open woodland on red earth and clay plains mainly in the Mulga Lands Bioregion (Benson 29)*
b) *Belah woodland on alluvial plains in central-north NSW (Benson 55)*

Brigalow Clay Plain Woodlands mostly occur in the northwest of NSW as two main disjunct occurrences: east and west of the upper Darling River floodplain. There are also small scattered patches as far south as Roto near Nombinnie Nature Reserve in the west of the Lachlan Valley (Plantnet 2012). Brigalow Clay Plain Woodlands occupy rolling downs, outwash slopes and flats with deep fertile alluvial clay soils (Keith 2004). Brigalow Woodlands are mostly dominated by Brigalow

(*Acacia harpophylla*), although Belah (*Casuarina cristata)* and Bimble Box (*E. populnea* ssp. *bimbil*) are common associates in the east and Mulga (*A. aneura*) is a common associate in the west.

Due to extensive clearance that accelerated with the arrival of modern bulldozers and "scrub pulling" techniques from the 1950s onwards to convert the community to croplands, Brigalow Clay Plain Woodlands are severely reduced in extent with very few large remnant areas. This has resulted in the community being listed as an EEC under both NSW and Commonwealth legislation.Brigalow Clay Plain Woodlands include many species with rainforest affinities and have a very close relationship to Western Vine Thickets. Because of this, Brigalow Woodlands are one of the few fire intolerant vegetation classes west of the Great Dividing Range (Keith 2004). Brigalow Clay Plain Woodlands in the Lachlan Valley are the southern and southwestern-most occurrences of this iconic and evolutionarily informative vegetation community.

Two Brigalow Clay Plain vegetation communities, both listed as EECs, are known from within the Lachlan CMA, although neither was mapped by DEC (2006). Benson *et al. (*2006) identified the occurrence of a "Brigalow open woodland community on red earth and clay plains" (Benson Community 29) within the Cobar and Carrathool Local Government Areas, of the Lachlan Valley. Eighty five hectares was mapped in the Cobar Shire by Dykes (2002) and three central west sites were documented by Shelley (2008). Brigalow has short lived seed compared to most other *Acacia* species (Reichmann *et al.* 2006). Germination probably occurs after the first rains but grazing probably eliminates most seedlings. It is a species that is generally killed by fire.

The second Brigalow Clay Plain Woodland vegetation community in the Lachlan Valley is "Belah Woodland on alluvial plains" Benson *et al's* (2006) Community No 55. Belah Woodland has a height of up to 12 m high, and is dominated by Belah (*Casuarina cristata*). Other tree species include Black Box (*Eucalyptus largiflorens*) in depressions and on higher ground Western Grey Box (*Eucalyptus microcarpa*) and Poplar Box (*Eucalyptus populnea subsp. bimbil*). Weeping Myall (*Acacia pendula*) may be present as an associate but not as a dominant species (Lewer *et al* 2003). This community occurs on alluvial brown or grey clay soils that may be gilgaied on floodplains, on alluvial plains and on black loam soils derived from basalt, it mostly occurs along intermittent drainages and on flats (Benson *et al.* 2006). There do not appear to be any publications that detail the fire ecology of these woodlands.

On the basis of published evidence that Brigalow Clay Plain Woodlands are highly sensitive to fire, it is strongly recommended that fire is excluded from landscapes supporting this vegetation class.

4. Western Peneplain Woodlands, comprising 4 RVCs

a) *Poplar Box grassy woodland on flats mainly in the Cobar Peneplain and Murray-Darling Depression Bioregions (Benson 105)*
b) *Poplar Box - Gum-barked Coolibah - White Cypress Pine shrubby woodland mainly in the Cobar Peneplain Bioregion (Benson 103)*
c) *Smooth-barked Coolibah - Mulga open woodland on gravelly ridges of the Cobar Peneplain Bioregion (Benson 108)*
d) *White Cypress Pine - Poplar Box woodland on footslopes and peneplains mainly in the Cobar Peneplain Bioregion (Benson 72)*

Western Peneplain Woodlands are one of the most extensive and varied classes of semi-arid woodland. They are an open class of woodland with a canopy typically 10-15m in height, there is mostly an open shrub layer, placing it within the shrubby subformation of the semi-arid woodland vegetation formation. Poplar Box (*E. populnea* ssp. *bimbil*) and Gum Coolibah (*E. intertexta*) are dominant in drier and stonier soils, whilst White Cypress (*Callitris glaucophylla*) dominates on lighter soils (Beeston *et al.* 1980 Metcalfe *et al.* 2003). Western Peneplain Woodlands are widespread on the Cobar Peneplain, although more easterly areas have been heavily cleared. More westerly variants of Western Peneplain Woodland are confined to drainage lines where moisture availability is higher; they are replaced in the drier parts of this landscape by North-west Plain Shrublands (Keith 2004).

The understorey composition of Western Peneplain Woodlands varies greatly, the result of changes in soil texture, moisture availability and disturbance history (Keith 2004). These communities are highly modified by historic overgrazing, a process that is ongoing, particularly because of the extremely high densities of Feral Goats (*Capra hircus*) that occur on the Cobar Peneplain and surrounds (NSW Department of Primary Industries undated). Overgrazing and clearing has created large shifts in landscape structure and function including extensive erosion and scalding and the proliferation of native shrubs (including Hop Bush, *Dodonaea* spp.*,* Emu Bush , *Eremophila* spp. and Cassia, *Senna* spp.) that are unpalatable to domestic stock. These species are termed “woody weeds” by pastoralists due to their impacts upon the availability of grasses, a more recent name for this shift in understorey composition is “Invasive Native Scrub” (INS).

Four Western Peneplain Woodland communities occur in the Lachlan Valley. Three of these communities are dominated by Poplar Box in association with White Cypress Pine and Gum-barked Coolibah. The first of these, “Poplar Box grassy woodland” (Benson 105) occupies flats in the Cobar Peneplain and Murray Darling Depression Bioregions. The shrub layer is very sparse and includes Mulga (*Acacia aneura*), *Acacia colletioides*, Wilga (*Geijera parviflora*), Budda (*Eremophila mitchellii*), Tar Bush (*Eremophila glabra*) and *Bertya cunninghamii*. The ground layer may be mid-dense after rain and includes the small shrubs *Einadia nutans* and *Maireana microphylla*. Grass species include wire grasses such as *Aristida behriana*, Curly Windmill Grass (*Enteropogon acicularis*), bottlewasher grasses such as *Enneapogon avenaceus* and *Enneapogon polyphyllus*, *Austrostipa scabra* and occasionally spinifex *Triodia scariosa*. This community occurs mostly in the Cobar Peneplain Bioregion on red earth soils on lower slopes, run-on areas and flats on peneplain and plain landforms (Benson *et al.* 2006).

The second community "White Cypress Pine - Poplar Box woodland on footslopes and peneplains" (Benson Community 72) occurs on red and brown loam soils that may be colluvial, on footslopes or flats on low hills or alluvial terraces. It mainly occurs in the eastern half of Cobar Peneplain Bioregion. This community often occurs between Poplar Box and Western Grey Box grassy woodland on finer texture soils on the plains and Dwyer's Red Gum low open woodland on shallow, siliceous soils on hills. It grades into White Cypress Pine on alluvium mainly to the east. The third community "Poplar Box - Gum-barked Coolibah - White Cypress Pine shrubby woodland" (Benson 103) is an open woodland to 25m that occurs on clay loam, sandy loam or lateritic soils on alluvial flats, footslopes and broad ridges of undulating plains mainly in the Cobar Peneplain Bioregion. Cohn (1995) found these communities to be widespread within Nombinnie and Round Hill Nature Reserves and large areas were mapped by Dykes (2002) in the Cobar Shire.

The fourth community "Smooth-barked Coolibah – Mulga open woodland" (Benson Community 108) is a mid-high woodland of up to 15 m high dominated by Gum coolabah (*Eucalyptus intertexta*) in patches with Mulga (*Acacia aneura*) dominating the understorey. This community occurs in areas with the lowest average annual rainfall of any of the semi-arid woodlands, frequently grading into arid shrublands. Other small trees or shrubs include Ironwood (*Acacia excelsa*), Beefwood (*Grevillea striata*), *Senna* spp., Horse Mulga (*Acacia brachystachya*) and Wilga (*Geijera parviflora*). Ground cover is sparse and contains wiregrasses (*Aristida* spp.), *Eragrostis eriopoda*, Mulga Oats (*Monachather paradoxus*), Mulga Grass (*Thyridolepis mitchelliana*) and Box grass (*Paspalidium constrictum*). Small shrubs include copperburrs (*Sclerolaena* spp.) and *Sida* spp. White Cypress Pine (*Callitris glaucophylla*) is mostly absent from this community. This community occupies shallow, red earth soils on low gravel rises and ridge crests with sandstone or quartzite outcrops. Mainly found in the western half of the Cobar Peneplain Bioregion it grades into White Cypress Pine, Mulga and Poplar Box woodlands. Due to its occurrence on shallow gravel soils little has been cleared and the main threat is due to overgrazing by stock or feral animals such as goats (Benson *et al* 2006). Benson *et.al* 2006 note that this community is "*Irregularly burnt and due to a lack of grass cover, fire does not carry easily*" and "*post-fire recovery may be slow due to climatic and grazing pressures*".

MALLEE SYSTEMS

The term Mallee is derived from the Aboriginal word "*mali*", which was used to describe individual eucalypt trees, the water Mallees, whose large roots extend laterally at shallow depths and were prized as sources of pure drinking water (Noble 1984; Noble and Bradstock 1989). Mallee Woodlands are a highly distinctive biome that is characterised by multi-stemmed lignotuberous eucalypts generally no taller than 10m (Noble 1984, Noble and Bradstock 1989). Mallee communities primarily occur on coarse-textured soils, mostly dunes and sandplains in the semi-arid zone (Keith 2004). The understorey varies from dense shrubs to spinifex grass clumps and in some areas chenopod shrubs dominate (Benson *et al* 2006). This vegetation type has been extensively cleared for cereal cropping and is also used for grazing (Myers *et al.* 2004).

Mallee woodlands occupy the extensive sand dune and sand plain systems of south and central western New South Wales in areas receiving between 250 and 400 mm of average annual rainfall (Keith 2004). Two Mallee dominated vegetation classes occur in the Lachlan Valley, Dune Mallee Woodlands and Sand Plain Mallee Woodlands (Keith 2004); both will be addressed in this section of the review. The Mallee systems of the Lachlan Valley are the most extensive remaining within NSW and because of this they are of national conservation value (Benson *et al.* 2006). The mosaic of sand dune and sandplain landforms protected within Nombinnie, Yathong and Round Hill Nature Reserves and those occurring on adjoining freehold lands comprise the largest contiguous stand of Mallee remaining in NSW (NPWS, 1996).

Dune Mallee Woodlands occupy the extensive red sand dunes of south-western NSW and occur on the southern Cobar Peneplain centered on the western Lachlan Valley and between Balranald and Pooncarie and the SA border (Keith 2004). Dune Mallee Woodlands in the Lachlan Valley are dominated by Mallee eucalypts including *E. dumosa* (White Mallee), *E. gracilis* (Snap and Rattle) and *E. socialis* (Red Mallee) often with Mallee Pine (*Callitris verrucosa*). The understorey of these woodlands is characterised by Porcupine Grass (*Triodia scariosa*) and ephemeral shrubs and grasses including *Austrostipa scabra* (Spear Grass) and *Haloragis odontocarpa* (Toothed Raspwort).

Sand Plain Mallee Woodlands occur on the sand plains adjoining Dune Mallee Woodlands. The soils on these landforms have a slightly heavier texture and more fertility than the adjoining dunes. The floristic composition of Sand Plain Mallee Woodlands is distinct from the adjoining dunes with a lack of Mallee Pine and Porcupine Grass and a greater diversity of shrubs including Broom Bush (*Melaleuca uncinata*) and various *Acacias* (Keith 2004).

Mallee communities are the most fire-prone of all plant communities in either the semi-arid or arid zones (Keith 2004). Mallee ecosystems generally have a low canopy (<10m), because of this almost all fires result in the death of above-ground branches and stems (Clarke *et al* 2010). Mallee eucalypts have a striking ability to coppice rapidly after fire, with new stems arising as rapidly as two weeks post-fire from synchronously resprouting lignotubers (Noble 1997; Clarke *et al* 2010). The capacity for rapid post-fire shooting is dependent on the lignotuber being insulated from lethal fire temperatures by surface soil. Dormant buds rapidly reshoot after a warm-season fire to quickly re-establish a canopy that offsets the respiratory demand imposed by the extensive root system (Noble 1997).

There is little known of pre-European fire frequency in mallee systems, but the fact that Edward Eyre and Ernest Giles travelled through dense mallee scrub bordering on the impenetrable, suggests that some of the mallee may have burnt only infrequently (Kerle 2008). The degree of patchiness of mallee systems that was induced by Aboriginal fires is still a matter of conjecture (Noble 1989a). An extensive body of literature has been published on the dynamics of fire within mallee ecosystems, although many facets of fire and biodiversity management in Mallee ecosystems remain poorly-known. Many facets are currently being investigated.

Large mallee fires usually occur after more than a year of above average rainfall. Lightning is the usual source of ignition and the fire regime is dominated by very large fires occurring every 10 to 15 years with few or no intervening fires (Noble 1997). Above-average rainfall promotes prolific growth, particularly of understorey grasses, creating a high degree of fuel continuity (Bradstock and Cohn 2002, Cullenward 1987, Noble and Vines 1990). Noble (1982) suggests that with a 10-15 year interval between serious fires most areas would be burnt approximately once every 50-60 years without any management burning. Fire is less frequent in the lower rainfall Mallee communities fringing the arid rangelands (Noble 1982).

The understorey of Dune Mallee Woodlands is characterised by highly flammable Porcupine Grass (*Triodia scariosa*) and a diversity of annuals including Spear Grass (*Austrostipa scabra*) and various “fire ephemerals”. These species are short lived, heavily-seeding plants that are triggered to germinate by fire (Keith 2004). When Sand Plain Mallee Woodlands burn it is the substantial abundance of shrubs and eucalypt litter that are the greatest contributors to the passage of fire (Keith 2004).

The influence of fire on the demographics of Mallee Pine (*Callitris verrucosa*) in Dune Mallee Woodlands is of great interest due to the sensitivity of the species to fire. This has been the subject of studies by Bradstock and Cohn (2002) and Zimmer *et al* (2010). Mallee Pine seed cones are stimulated to release their seeds when adults are killed by fire. The canopy seed bank is slow to accumulate and is lost when parents die from fire or old age (Keith 2004). Bradstock and Cohn (2002) found that the bark of 60-year-old Mallee Cypress was thick enough to render stems resistant to low-intensity fires. As a result they suggest that the probability of propagation of fire in populations >50 years old may decline and that survival of fire in this age range may increase. Because of the sensitivity of younger plants to fire and the slow accumulation of canopy-stored seed Bradstock and Cohn (2002) found that populations may decline when fires are relatively frequent (<15-year interval). Zimmer *et al*. (2010) examined the dendrochronology of *Callitris verrucosa* in Victoria and found a maximum age of approximately 180 years.

Cohn *et al* (2002) examined species richness and composition in Yathong Nature Reserve; firstly in relation to a range of times since fire (up to 15 years), and secondly in relation to topographic position (lower slope, mid-slope, upper slope), the position of resprouting eucalypts (in gap, under canopy), and their interactions. Cohn *et al* (2002) found that the number of species per transect was inversely proportional to time since fire and that the mean number of species recorded per transect was significantly influenced by time since fire. They also determined that whilst there were no significant differences in species richness between 0.75, 1.75 and 2.5 years post fire, the number of species was significantly higher at 5 years than 15 years after fire. In relation to the influence of topography and eucalypt canopy, Cohn *et al.* (2002) found that lower topographic sites differed from upper topographic sites, but that there were no significant differences between these sites and mid-slope sites. They found that the presence of a eucalypt canopy did not significantly influence species richness. The results from this study indicate a decline in the above ground richness of all understorey species with increasing time since fire. Cohn *et al*. (2002) found that a flush of short-lived species up until 2.5 years after fire resulted in higher richness than five and more years after fire, which was dominated by perennials.

Clarke *et al* (2010) investigated the fire history of six Mallee eucalypts in the Murray Valley. This study found a strong relationship between mean stem diameter and time since fire. This study found that long-unburnt Mallee stands are often well over 50 years old with some areas thought to be considerably older than 100 years post-fire. These findings considerably extend the definition of “old growth mallee” and present challenges to land managers seeking to maintain conservation values.

Many Dune Mallee Woodlands have a ground layer dominated by spinifex/porcupine grass; these areas, when associated with the high essential oil content of mallee foliage, create relatively high fuel loads (for highly climate-limited ecosystems) with extreme flammability (Noble 1997). In addition, variable spear grass grows profusely in response to wet conditions and this provides another highly flammable fuel following drying after strong seasonal growth. In the dune whipstick mallee communities (those with young, dense small-girth stems), the accumulation of up to 13 tonnes per hectare of fuel will allow for canopy ignition (Kerle 2008). In Mallee communities wildfire spotting is common and is enhanced by the ignition of suspended ribbons of bark (Kerle 2008). Kerle (2008) found that spring burning of Mallee systems resulted in significant seedling recruitment, especially when the fire intensity was sufficiently high to maximise the ‘seed rain’ from aerial capsules and offset predation by seed-harvesting ants. Kerle (2008) suggests that fire in autumn may have the opposite effect, reducing mallee density whenever there is sufficient fuel.

Fire regimes in Mallee ecosystems exert a strong influence on the ecology and survival of many fauna species (Cheal *et al* 1979; Gill 1994). For many species endemic to Mallee ecosystems the maintenance of viable populations is entirely dependent upon suitable fire regimes (Olsen and Weston 2005). Bradstock and Cohn (2002) suggest that there is insufficient knowledge to resolve management dilemmas posed by the need to conserve a range of biota with various required fire regimes. Gill (1990) recommends protecting long-unburnt Mallee to conserve species adversely affected by fire.

Watson *et al.* (2012) studied the response of bird species to time-since-fire in Semi-arid Woodland and Shrubland formations. This study analysed data from 499 sites representing a 100 year chronosequence to determine whether post-fire responses of 30 species are idiosyncratic or if multiple species show a limited number of similar responses. The analysis also explored whether these responses can be used to predict the occurrence of species across broad spatial scales. Watson *et al* (2012) found that the occurrence of 16 species was significantly influenced by time-since-fire and that 15 of the 16 species occurred more frequently in mid or older successional vegetation stages (>20 years-since-fire) and only one species in early successional stages (<5 years-since-fire). The study determined that the species studied had a limited range of responses to time-since-fire, concluding that "a small set of fire ages should allow the provision of habitat for most species". The results of this study highlighted the importance of mid- to late- successional vegetation (>30 years) for many bird species inhabiting Mallee ecosystems.

Haslem *et al.* (2012) investigated the availability of hollows across 581 sites in mallee ecosystems, primarily in Victoria. In this study the influences of time-since-fire and inter-fire interval were explored. Predictably the probability of live and dead stems containing hollows increased over time. Haslem *et al* (2012) found that live stems did not provide any hollows before 40 years post-fire and that the probability of dead stems containing hollows peaked at 50-60 years post-fire. They also found that longer inter-fire intervals resulted in more hollows. Because of these findings Haslem *et al*. (2012) recommend that "hollow-dependent fauna will benefit from increased fire-free periods, both in terms of individual fire events and the intervals between repeated fires" and conclude that their results "highlight the complex way in which fire affects the availability of faunal habitat resources, and the extended time periods over which such influences operate".

After fire Mallee Woodlands start accumulating significant quantities of litter fuel (Noble *et al.* 1980). Long unburnt Mallee Woodlands are a critically important habitat for the Endangered and steeply declining Malleefowl, *Leipoa occelata* (Benshemesh 1990, 1992; Arnold *et al.* 1993). This habitat is particularly important for ensuring the availability of the deep litter-layer that the Malleefowl requires for the establishment of nesting mounds (Olsen and Weston 2005). Olsen and Weston (2005) also recognise that mallee fires can create flushes of food (insects, fruit and seeds) for the Malleefowl. For the survival of Malleefowl, the optimal fire frequency needs to be more than 60 years with the availability of adjoining areas of recently burnt habitat (Benshemesh 1992). These factors highlight the complexities of maintaining populations of Mallefowl and at least partially explain why the species has become extinct across a large proportion of its former range.

Long unburnt mallee patches are also recognised as being of great importance for several other threatened species including the Black-eared Miner, *Manorina melanocephala* (Schodde 1981, Starks 1987, 1988), Red-lored Whistler, *Pachycephala rufogularis* (Woinarski 1987; NSW Scientific Committee 2008) and Western Blue-tongued Skink, *Tiliqua occipitalis* (Henley 1987; Sadlier and Pressey 1994; Sass 2006). The Black-eared Miner depends on mallee habitat that has remained unburnt for at least 20 years, with the highest densities occurring 40 years post-fire (Olsen and Weston 2005).

Friend (1993) reviewed the impact of fire on small terrestrial vertebrates in mallee ecosystems. This review found that fire response patterns of small mammal species are closely aligned to their shelter, food and breeding requirements. For reptiles the review found a strong relationship between shelter and foraging requirements of species and their abundance in various successional states. The review also found that the high incidence of burrowing in the mallee amphibian fauna confers considerable resilience to fire and their responses seem more closely linked to moisture regimes than to fire *per se* (Friend 1993).

From the considerable literature published on fire and biodiversity in mallee ecosystems it is clear that they are highly fragmented, but of exceptional conservation value, critically driven by fire and highly limited by climate. It is also apparent that the maintenance of biodiversity in the mallee is dependent upon the retention of areas of long-unburnt vegetation, upon fire regimes that result in ecological conditions conducive to the survival of sensitive species and upon the maintenance of a mosaic of different post-fire stages. To achieve this is a great challenge and a highly complex task that must be undertaken in the context of extremely high levels of fragmentation, making the task of maintaining biodiversity even more difficult.

Conclusions

Fire in semi-arid woodlands is (naturally) strongly related to rainfall patterns, and this largely drives fire frequencies from occurrences of every few years to decades (from north to south). However, productivity of the grass layer is also a critical component to these fire events. The fuel load of the grass layer is driven by rainfall as well as species differences, interactions with shrub and tree species and past management practices (including overgrazing). Overgrazing and less frequent fire are at least two of the main drivers of major shifts in the vegetation of these woodlands.

In the absence of fire grassy semi-arid woodlands may convert to shrublands, however reinstating more frequent fire is not straightforward. Once converted to shrublands, the grass layer may be difficult to recover because of lowered grass productivity, lower seed banks and altered processes such as water flow patterns.

Hodgkinson and Harrington (1985) reviewed the role of prescribed burns in semi-arid woodlands and this article provides an integrated view of the use of prescribed fire in these systems. Fire kills most shrub seedlings and some adult shrubs (but some shrub species have high survival rates). It also reduces the reproductive capacity of shrubs for a number of years. It may promote the germination of some shrubs (hence it can lead to shifts in the dominant shrubs), however on balance, it decreases shrub dominance. Fires of moderate intensity (and even low intensity) can kill shrub species (hence prescribed fire in low danger periods can be utilized). Mass recruitment following rain (of both grass and shrubs) provides the greatest opportunity for successful fire (i.e. high grass biomass) and management of shrub recruitment (particularly through the use of frequent sequences of fires where possible) which will also decrease adult shrub cover. Shrub cover reductions will then benefit growth of the grass layer for ~ 10 – 20 years (generally grass species exhibit low mortality to fire). Grazing post-rain (either by livestock or native species) can significantly reduce grass growth and inhibit fire. Thus, the use of prescribed fire should be structured around opportunity related to rainfall, conservation of fuel loads around proliferating shrubs that need to be controlled and the use of fire when fuel loads can successfully carry fire. Hodgkinson and Harrington (1985) consider alternatives to the use of fire for managing shrub abundance in these semi-arid woodlands to be of limited value (i.e. grazing of shrubs by goats).

In sum, fire is critical to the diversity and managed use of semi-arid woodlands. Fire intervals are largely governed by opportunity (rainfall) and intervals ranging down to every few years can be required to maintain these systems. There is a rich literature on fire, including the socio-economic aspects of its use that supports that fire frequencies have generally been too low. In summary, fire intervals for semi-arid woodlands in Kenny *et al.* (2004) are for a minimum of 6 years and a maximum of 40, with the proviso that "*There was insufficient data to give definite intervals. Available data indicates minimum intervals should be at least 5-10 years, and maximum intervals approximately 40 years*" (Kenny *et al*. 2004). The authors point out that because some birds need areas of old mallee (>30-50 years), some intervals over 40 years are probably appropriate in mallee systems. However other bird species, such as the Chestnut Quail Thrush (*Cinclosoma castanotus*), Shy Heathwren (*Hylacola cauta*) and the Scarlet-chested Parrot (*Neophema splendida*) need, or prefer, mallee less than 10 years old, while Mallee Fowl (*Leipoa occellata*) use adjacent old and young patches. References can be found in Woinarski (1999).

The complete exclusion of fire from areas of Brigalow Clay Plain Woodland within the Lachlan Valley is considered a high priority, because of the known sensitivity of this community to fire, the biogeographic significance of these patches and their limited extent.

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). The Lachlan CMA supports 6 classes of Dry Sclerophyll Forest (Keith 2004). These 6 classes are composed of 19 RVCs (OEH 2011). Research directly addressing the role of fire in the dry sclerophyll forests of the Lachlan Valley has not been extensive. However studies relevant to the vegetation classes occurring in the Lachlan region can be found – though much of this work is from outside the Lachlan region. The four classes of vegetation that are the focus of this review are North-west Slopes Dry Sclerophyll Woodlands, Southern Tableland Dry Sclerophyll Forests, South East Dry Sclerophyll Forests and Western Slopes Dry Sclerophyll Forests. Keith (2004) places the North-west Slopes Dry Sclerophyll Woodlands and Upper Riverina Dry Sclerophyll Forests into the shrub/grass subformation of the dry sclerophyll forest formation, whilst the other four classes that occur within the Lachlan Valley fall within the shrubby subformation: they are Sydney Montane Dry Sclerophyll Forests, South East Dry Sclerophyll Forests, Southern Tableland Dry Sclerophyll Forests and Western Slopes Dry Sclerophyll Forests.

1. Upper Riverina Dry Sclerophyll Forests, 2 RVCs

a) *Red Stringybark - Scribbly Gum - Red Box - Long-leaved Box shrub - tussock grass open forest the NSW South Western Slopes Bioregion (Benson 290)*
b) *Red Stringybark - White Box grassy open forest of the South Western Slopes*

Upper Riverina Dry Sclerophyll Forests are part of the shrub/grass subformation of the Dry Sclerophyll Forest formation. This class of vegetation occurs in the South Western Slopes bioregion on soils of moderate fertility at elevations from 300 – 700m and on a variety of landforms ranging from flat terrain to steep slopes (Keith 2004). Upper Riverina Dry Sclerophyll Forest are the southern analogues of the North-west Slopes Dry Sclerophyll Woodlands, although there is an extensive overlap of these two classes between Orange and Gundagai, where several intermediate communities are found (Keith 2004). This overlap zone essentially covers all of the mid reaches of the Lachlan Valley. In the eastern parts of this zone, Upper Riverina Dry Sclerophyll Forests tend to occupy more elevated areas (Keith 2004).

Upper Riverina Dry Sclerophyll Forests in the Lachlan Valley are a mid-high open forest or woodland dominated by Red Stringybark (*E. macrorhyncha*) with Red Box (*E. polyanthemos*), Long-leaved Box (*E. goniocalyx*) with Scribbly Gum (*E. rossii*) and White Box (*E. albens*) sometimes present. The shrub layer is usually sparse but may be mid-dense where fire has been less frequent (Benson *et al* 2006).

Pulsford (1991) and Pulsford *et al* (1993) explored the history of fire and human activity in Upper Riverina Dry Sclerophyll Forest in Lower Snowy River Valley in Koscuisko National Park. The key finding of this study was that 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 1991; Pulsford *et al* 1993).

2. North-west Slopes Dry Sclerophyll Woodlands, 1 RVC

a) *White Cypress Pine woodland on sandy loam soils on the plains of central NSW (wheatbelt) (Benson 70)*

White Cypress Pine is a widespread species; it is found in all mainland states and is a component of many vegetation classes in the Lachlan Valley (Bowman and Harris 1995, Metcalfe *et al.* 2003, Keith 2004, Thompson and Eldridge 2005b). It is well-known as a dominant tree species across a considerable expanse of dry sclerophyll forests and woodlands. 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 conifers generally 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, welldrained 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.* 2008). 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.* 2008), but increase when stand densities are reduced (Henry 1960, Lacey 1973, Horne 1990a).

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 Lachlan and adjoining regions (Central West, Murrumbidgee, Riverina) 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 NSW 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 Narrowleaved 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.

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 NSW 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.”

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.* (2008). 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 Lachlan Valley. 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) hypothesised, 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 firescarred 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.**

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, 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 below.

** 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).

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 understorey plant species, whether midstorey 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.

The influence of cover and abundance of White Cypress Pine (and to a lesser extent Black Cypress Pine) on native species richness has recently been investigated by Hunter (2011). In this study a total of 809 sites containing *Callitris glaucophylla* and 542 sites containing *Callitris endlicheri* were chosen. Multiple linear regressions of plot data from the 1,347 sites across the New England, Nandewar and Brigalow Belt South Bioregions showed no negative effect of increasing *Callitris* cover on the number of species per site. The strongest influence on species density was found to be altitude, which was far in excess of any other variable analysed. Hunter found a small increase in species density associated with an increase in cover/abundance of *Callitris glaucophylla*. He suggests that “It could potentially be that *Callitris glaucophylla* provides a cover that ameliorates the extremes of climate, allowing species to establish and/or persist”. In this study fire was found to have a significant negative effect on species density in *Callitris glaucophylla* sites but not *Callitris endlicheri*. Conversely grazing had a positive effect on the species density of *Callitris endlicheri* sites but no significant effect on *Callitris glaucophylla* sites.

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** [FOOTNOTE]. 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.

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.

** 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 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 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 ashcovered 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.

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.

Cypress Pine and fire

Research into the impacts of fire on White Cypress Pine is surprisingly limited. No one, to our 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).

- 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 from CW Lit review).
- White Cypress Pine, like several other *Callitris* taxa, is a T species, able to recruit between fires (*page 74*). 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 which again would have provided flammable fuel for the fires that favoured eucalypts over pines.**[FOOTNOTE]

**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. Sydney Montane Dry Sclerophyll Forests

Although not listed as occurring in the Lachlan Valley by OEH (2011), 14 discrete patches of Sydney Montane Dry Sclerophyll Forest are mapped by DEC (2006). No dominant species are detailed for these mapped polygons. Keith (2004) lists Blue Mountains Ash (*E. oreades*), Silvertop Ash (*E. sieberi*), Sydney Peppermint (*E. piperita*) and Hard-leaved Scribbly Gum (*E. sclerophylla*) as the dominant species.

4. South East Dry Sclerophyll Forests

a) Silvertop Ash - Broad-leaved Peppermint dry shrub forest of the South Eastern Highlands

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

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

Fire-related research in South East Dry Sclerophyll Forests has focussed on vertebrate fauna, on species responses to wildfire and on changes with time-since-fire. Effects of fire cycles on plants are being studied in forests near Eden, where replicated trials of low-intensity patchy fuel reduction burning have been underway since 1984 (Penman *et al*. 2007, Penman *et al.* 2009, Penman *et al.* 2009a). Fauna studies come from Nadgee Nature Reserve, which contains large expanses of forest in addition to heathland (eg Newsome *et al.* 1975, Recher *et al.* 1975, Catling *et al.* 2001). They also come from *E. sieberi*-dominated State Forest around Bega where in the early 1980s

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Dan Lunney and his colleagues surveyed a wide range of fauna species. Though theprimary aim of this work was to understand the effects of logging, a wildfire within the sampling period provided the opportunity for observations of the effects of fire (eg Lunney and O'Connell 1988, Lunney and Leary 1989). Work on birds has been carried out just over the border in Victoria by Loyn (1997).

Vegetation change with time-since-fire

Although available studies have not focussed on vegetation, there are indications that shrub dynamics in South East Dry Sclerophyll Forests may have common ground with those in the Southern Tablelands Dry Sclerophyll Forests discussed in the next section. Fire, particularly intense fire, stimulates shrub germination, so that the open environment created by incineration of the understorey is replaced over the next few years with thick regeneration of sclerophyll shrubs. With increasing time-since-fire it appears that many shrub species mature and then start to die off, opening the landscape up once again. This process takes at least two decades.

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

Although there were few signs of a decrease in shrub cover by 18 years post-fire in the data gathered by Catling *et al.* (2001), Loyn (1997) noted that prior to fire in his forest sites shrubs were few. Dense shrub regrowth developed, however, after the fire. In more recent years scientists from the Department of Environment and Conservation have noted the understorey 'thinning out' at Nadgee, where time-since-fire is now mostly >30 years (Ross Bradstock, pers. comm. 2006). Finally, data from the Eden Forestry trials shows a gradual decline in plant species richness over the course of the study across *all* treatments, ie those burnt at low intensity at 2 and 4 year intervals, and those left unburnt since the start of the study in the mid 1980s. The entire study area was burnt in an extensive, intense fire in 1983, and it seems likely that the decline in species richness reflects a time-since-wildfire effect (Penman *et al*. 2007, Penman *et al.* 2009; 2009a)

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

Small mammals

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

At Nadgee, forest plots along the river supported two Antechinus and two native rat species prior to the extensive 1972 wildfire. In the first year after the fire the Swamp Rat *Rattus lutreolus* and the Dusky Antechinus (*A. swansonii*) disappeared completely from this area, while Bush Rats (*Rattus fuscipes*) and Brown Antechinus (*A. stuartii*) were found only in small numbers. All four species, however, occurred in nearby unburnt areas and produced young in the first year after the fire (Recher *et al.* 1975). The rats recovered faster than the Antechinus: according to Recher *et al.* (1975) by two years post-fire Bush Rats were found everywhere in forest where they had been found prior to the fire, while Swamp Rats were also common, even where they had not been found pre-fire. Catling and Newsome (1981) also note that the rodents recovered quickly, and add that this occurred most rapidly in moist habitats. Their data show that native rat numbers generally peaked after five years, and then declined somewhat.

Antechinus stuartii recovered to post-fire levels by the fifth post-fire year, with *A. swainsonii* taking a year or two longer. Recher *et al.* (1975) point out that this difference reflects the ecology of the two animals. *A. stuartii* is noctural, partly arboreal, eats insects on the ground but also on shrubs and trees, and thus is less at risk of predation and less dependent on cover than *A swansonii*, which is active by day, forages on the ground, and likes logs, dense shrubs, and lots of litter. The differential impact of severe fire on the two Antechinus species was also reflected in their condition on the Nadgee River plots in the first post-fire year. *A. swansonii* weighed up to 50% less than in the years before fire, whereas *A. stuartii*, which could find insects on eucalypt shoots, didn’t lose weight at all.

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

In Mumbulla State Forest north-east of Bega studies have included pre- and post-fire surveys after a wildfire in 1980 (Lunney *et al.* 1987), and after the next fire in the area, which was a low-intensity fuel reduction burn in 1999 (Penn *et al.* 2003). As at Nadgee the Dusky Antechinus *Antechinus swansonii* was more affected by fire than other small mammal species: *A. swansonii* was not caught at all over the three years following the wildfire (Lunney *et al.* 1987), and was scarce after the planned burn (Penn *et al.* 2003).

The second Antechinus species in this environment, *A. agilis* (called *A. stuartii* at the time of the earlier study), declined after each fire, particularly where the burn was more intense, but did not disappear. After the planned fire in 2000 this species maintained population numbers on less severely burnt south-east slopes but not on north-west slopes where burn was more intense (Penn *et al.* 2003). Bush Rats (*Rattus fuscipes*) at Mumbulla recovered slowly after wildfire, reaching 31% of pre-fire numbers by 2.5 years post-fire; Lunney *et al.* (1987) attribute this slow recovery to the sustained drought which followed the fire. The later planned burn, however, had little effect on *R. fuscipes*: numbers were slightly greater 10 months after the fuel reduction burn, particularly on the less-affected south-east slopes, while by 20 months post-fire the population was similar to what it had been prior to the fire (Penn *et al.* 2003).

Unlike the rodent and Antechinus species discussed so far, all of which are associated with cover to a greater or lesser extent, the rare and vulnerable White-footed Dunnart in Mumbulla Forest strongly favours open habitat such as logged, burnt ridges, and prefers sites with sparse ground and canopy cover (Lunney and Ashby 1987, Lunney *et al.* 1987, 1989). It survived the wildfire in 1980, reproduced successfully in the first post fire year, but disappeared by the third post-fire winter as vegetation rapidly thickened up (Lunney and Ashby 1987, Lunney *et al.* 1987, 1989; Lunney and Leary 1989).

These small animals vary in how far they travel, with some ‘explorer males’ covering distances of over a kilometre (Lunney *et al.* 1989). Researchers Lunney and Leary (1989) speculate that suitable habitat for this species may occur naturally only as disjunct and temporary patches, making the ability to travel an important attribute for survival.

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

Large and medium-sized mammals

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

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

Kangaroos, wallabies, wombats

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

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

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

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

Bandicoots and potoroos

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

Claridge and Barry (2000), who studied factors associated with diggings of bandicoots and potoroos, found more diggings of both animals in sites unburnt for over 20 years than in sites burnt 0-10, or 10-20 years previously. Bandicoot diggings were also associated with a high density of ground cover, a measure which apparently included shrubs up to 2 m, leading the researchers to suggest that recently burnt habitats may not provide sufficient protection from foxes and dogs. Time-since-fire may also influence food resources, which for bandicoots and potoroos consist primarily of the underground fruiting bodies of fungi with an association with particular plant species. The relationship between fire, these fungi, and their host plants, however, is not at all clear. Claridge and Barry (2000:683) conclude that burning in bandicoot and potoroo habitats "should be done mainly with long intervals."

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

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

Predators

A number of times in the discussion above the importance of dense shrub cover in allowing native animals to minimize the ravages of predators has been noted. How do predators themselves respond to time-since-fire and habitat complexity? Findings for dingos vary. Catling and Burt (1995) found a positive correlation between dingo abundance and habitat complexity, whereas Catling *et al.* (2001), using data from Nadgee and including time-since-fire in their suite of variables, found no relationship with habitat complexity but a strong decline in dingo numbers in later post-fire years. Cats, in this study, increased with both time-since-fire and habitat complexity, while foxes were everywhere, irrespective of these variables. There are no doubt complex relationships between the abundance of predators and prey, which may alter as time-since- fire and/or cover increases.

Arboreal mammals

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

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

Lizards

Lunney *et al.* (1989) studied three skink species in Mumbulla. A survey had just been completed when wildfire burnt through the study area in 1980. A second survey within weeks of the fire showed all three species were still present, in fact one, the Delicate Skink *Lampropholis delicata* was found in greater numbers after the fire than before it – a result put down to the difficulty of finding it in thick vegetation prior to the fire. Grass Skinks (*Lampropholis guichenoti*) were less abundant after than before the fire. Examination of the stomach contents of this species showed little food was available in the post-fire weeks, and the authors surmise that individuals died from starvation together with heat stress and vulnerability to predation resulting from lack of cover. Both *Lampropholis* species need relatively open habitat, however, and were less abundant where post-logging regrowth was thick. A third species, the Water Skink (*Eulamprus heatwolei*), showed no significant changes in numbers as a result of the fire. Almost 20 years later the same area was once again surveyed before and after a low intensity planned burn in 1999; post-fire surveys were conducted at 10 and 20 months post-fire (Penn *et al.* 2003). The Grass Skink again showed a decline in the initial postfire survey, however by 20 months it had recovered to greater than pre-fire levels. Delicate Skinks again showed little effect of fire in the initial survey, but by 20 months were less abundant in burnt than unburnt areas. Water Skinks stayed true to their previous form and were not significantly affected by the 1999 fire.

Birds

Smith (1989) studied bird populations in a 13 ha site in Mimosa Rocks National Park between Bega and the coast, before and after a moderate intensity wildfire. This site, which had a history of frequent burning, had a rich bird fauna, with 96 species recorded over three years. The forest bird community was “surprisingly resilient” to the effects of the fire and of the drought which was associated with it. Smith (1989) puts this down to the role played by gullies: the site encompassed a ridge and two gullies one of which contained some rainforest plants. The number of bird species recorded in the year before the fire, and in the first post-fire year as well as in the second, hardly altered, and changes in species composition were confined to occasional visitors to the site. Effects within the site, however, were noteworthy. Insectivorous species, particularly those which used dense shrubs for feeding, shelter and nesting, generally contracted into the unburnt and lightly burnt vegetation in the gullies; examples included the Brown Thornbill and the White-browed Scrubwren. On the other hand species with plant foods in their diet expanded their distributions and numbers as particular trees and shrubs flowered and fruited – although whether this was in response to the fire is unclear.

Another feature of the year after the fire was a flurry of out-of-season breeding activity,again probably reflecting increased food resources. Recher *et al.* (1975) and Fox (1978) note similar behaviour at Nadgee, where some birds, particularly insect-eaters such as fantails, warblers and wrens, nested two and three times during the spring and summer.

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

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

Canopy insectivores followed the pattern for all birds but changes were not significant. Understorey insectivores as a group also showed few changes, however some individual insect-eating species, particularly those associated with heathlands, showed major declines. Many insectivores that fed from tall shrubs continued to get food from scorched shrub branches and foliage, as did canopy insectivores. By one year post-fire regenerating shrubs and epicormic growth were providing plentiful insect food. Birds which feed from open ground increased significantly in abundance, as a group, after the fire. Some species not recorded pre-fire were found for two or three years post-fire then disappeared again: Flame Robins, Scarlet Robins, and Buff-rumped Thornbills declined as shrub regrowth reduced the availability of open ground. Superb Fairy-wrens continued to increase over the three year post-fire period – they "thrived after fire in all habitats … where they were able to exploit the low shrub regrowth that developed when the canopy was opened" (Loyn 1997:228).

Large hole nesters, mostly seed-eaters, declined after fire to 30% of pre-fire levels, stayed scarce for the first two years post-fire, then increased to pre-fire levels in the third year. Resident carnivores showed few changes as a group, though Laughing Kookaburras were most numerous one and two years post-fire when there was plenty of open ground. There was a "small influx" of birds of prey during and after the fire. Honeyeaters virtually disappeared after fire, due to loss of their normal food sources: concentrations of eucalypt and shrub blossom. Abundance of species in this group increased with time-since-fire, reaching 60% of pre-fire levels by three years post-fire. Some taxa such as Yellow-faced and Lewin's Honeyeaters had fully recovered by this time, while others, including Crescent and New Holland Honeyeaters were still scarce.

These results differ from those of Smith (1989), who found these latter two species in considerably greater abundance in winter and spring surveys in the year after the fire, relative to pre-fire surveys in those seasons, probably due to flowering in the unburnt canopy of trees and gully shrubs.

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

Conclusion

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

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

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

How are the different habitat requirements of the many fauna species in South East Dry Sclerophyll Forests to be accommodated? Here again we see the need for variability in time-since-fire, and probably also in fire frequency. Variable intervals between fires within the statewide thresholds for dry sclerophyll forests, seven and thirty years, should deliver a range of habitats including lots of dense vegetation, areas where new post-fire regeneration is available and areas where fires are more, and less, intense. Where native vegetation covers large areas it is likely that wildfire will fulfil this prescription. Where remnants have been cut off by clearing and urbanisation, or where fire suppression has been unusually effective, some ecological burning may need to occur. Recher *et al.* (1975:163) suggest that when planned fire is added to the wildfire regime, it should be done in a creative, rather than in a “monotonous and unimaginative” way. Hot fires have their place, as well as cool winter burns. A further point which emerges from the South Coast forest studies is the importance of topography in providing refuge areas from which re-colonisation of the post-fire environment can occur. Not only do unburnt areas serve this function, places where fire is less severe also play this role. When thinking about the effects of fire and how best to manage it, it is instructive to consider landscape patterns: how does vegetation change with topography? How does topography affect how fire moves across the landscape? How does fire intensity vary over space and how does this enable plant and animal species to survive and thrive together in a fire-prone environment?

5. Southern Tableland Dry Sclerophyll Forests, comprising 8 RVCs

a) *Apple Box - Broad-leaved Peppermint dry open forest of the South Eastern Highlands*
b) *Apple Box - Yellow Box - Argyle Apple dry open forest of the South Eastern Highlands and South Western Slopes*
c) *Blakely's Red Gum - Black Cypress Pine dry shrub forest of the Lower Abercrombie area of the South Western Slopes*
d) *Blakely's Red Gum - Red Stringybark open forest on slopes and hills of the western slopes*
e) *Broad-leaved Peppermint - Brittle Gum - Red Stringybark dry open forest on the South Eastern Highlands*
f) *Broad-leaved Peppermint - Mountain Gum dry open forest of the Central Tablelands area of the South Eastern Highlands*
g) *Mugga Ironbark - Red Stringybark - Long-leaved Box dry grass forest of the South Western Slopes*
h) *Red Box - Red Stringybark grassy open forest on basalts of the South Eastern Highlands*

Southern Tableland Dry Sclerophyll Forests occur on the Southern and Central Tablelands between 600 and 1100 m above sea level, in rugged, rocky terrain on shallow, infertile soils. This is not productive country: trees reach only 15-20 m in height, sclerophyllous shrubs form an open, species-poor understorey, the open ground cover of tussock grasses contains *Lomandra* species and some forbs but not the wide range of herbs found in adjoining grassy woodlands on more fertile soils. Trees are mostly stringybarks, peppermints and scribbly gums. *Eucalptus macrorhyncha* (Red Stringybark) and *E. rossii* (Scribbly Gum) occur throughout the range of this vegetation class, while other eucalypt species are distributed according to soil type, rainfall and altitude. *E. dives*, for example, co-occurs in elevated, wetter areas. Shrubs are mostly peas, wattles and epacrids; common species include *Daviesia mimosoides*, *Daviesia latifolia*, *Acacia falciformis*, *Hibbertia obtusifolia* and *Melichrus urceolatus*. *Joycea pallida* (Silvertop Wallaby Grass) is commonly found in the ground layer (Keith 2004).

Southern Tablelands Dry Sclerophyll Forest occurs on hills near Bathurst, Goulburn and Yass, and also on Black Mountain near Canberra. Though poor soils discouraged clearing these forests were often selectively logged, and grazed (Moore 1953, Keith 2004). Current distribution includes substantial areas in reserves as well as patches on private property.

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.

The vegetation on Black Mountain falls into the Southern Tableland Dry Sclerophyll Forest class. *Eucalptus macrorhyncha*, *E. rossii* and *E. mannifera* are the dominant tree species. Common shrubs include *Daviesia mimosoides*, *Acacia genistifolia*, *Dillwynia retorta* and *Brachyloma daphnoides*. Herbaceous species include grasses *Joycea pallida* (then *Danthonia pallida*) and *Poa* species, with occasional forbs, particularly orchids, appearing in winter and spring. Elevation is about 650 m above sea level, annual rainfall averages 626 mm (Purdie and Slatyer 1976).

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

Here are some of the findings:

- Almost all shrubs resprouted; the only shrubs which were clearly obligate seeders were *Acacia genistifolia*, *Dillwynia retorta* and *Pimelea linifolia*. *Pimelea linifolia* germinated in the long-unburnt site after fire, though it was not recorded there prior to the burn. All three species flowered within three years of fire (Purdie and Slatyer 1976).
- Most resprouting shrubs also produced seedlings. Some, such as *Daviesia mimosoides*, *Acacia buxifolia* and *Indigofera australis* were able to increase population numbers by root suckering – which was stimulated by fire.

- Flowering on resprouts occurred within two years of fire. Seedlings were slower to reach maturity, with many not flowering by the end of the three year monitoring period (Purdie and Slatyer 1976).
- Shrub and herb seedlings arose from seeds stored in the soil (Purdie 1977b).
- All tree species resprouted and also produced seedlings (Purdie and Slatyer 1976). Tree seedlings developed in both burnt and unburnt plots plots (Purdie 1977b).
- All the resprouters which didn't produce seedlings were geophytic monocots, a category which included many orchids. All orchids flowered in the first year after fire. Several herbaceous species in this category – three Lomandras and *Dianella revoluta* – were able to increase through vegetative regrowth (Purdie and Slatyer 1976).
- Almost all species showed much higher seed germination in burnt compared with unburnt areas (Purdie 1977a, Purdie and Slatyer 1976), and survival rates were also generally higher in burnt vegetation (Purdie 1977b).
- A few species, including the dominant grass *Joycea pallida* and the exotic herb *Hypochoeris radicata*, produced more seedlings in unburnt than in burnt areas (Purdie and Slatyer 1976).
- Obligate seeders had more seedlings and higher seedling survival rates than resprouters (Purdie and Slatyer 1976).
- Many resprouters which weren't able to sucker flowered rapidly after fire on regrowth, and this produced some seedlings in Year 2 (Purdie 1977b, Purdie and Slatyer 1976).
- Resprouter recovery stabilised at a lower level in the previously long-unburnt site (60% of pre-fire numbers – which we have already noted were relatively low), than in the two more recently burnt sites (85-90% of pre-fire numbers) (Purdie 1977a). In the unburnt plots, many species declined with time. Obligate seeder population numbers declined 16-22% in these plots in the first year of monitoring, and an additional 5-10% in the second (Purdie 1977a).

"With the exception of rare species, all the tree, shrub and herb species represented in the quadrats prior to burning … were present in the regenerating communities in the first year after burning" (Purdie and Slatyer 1976:233). Purdie and Slatyer (1976) conclude that changes in floristics as time-since-fire go by are merely changes in dominance due to differences in species growth rates and longevity, and, sometimes, reflect a move from above-ground to presence only in the under-ground seedbank. 'Relay floristics', where additional species join the community as the period without disturbance lengthens, does not appear to characterise Southern Tableland Dry Sclerophyll Forest. This conclusion is consistent with what one might expect from a vegetation type that is, in Bond *et al.* (2003, 2005) terms, primarily climate limited, with some degree of modification by fire (*page 10*).

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??). The small obligate seeder shrub complement and the rapid post-fire flowering, while by no means unique to this ecosystem, distinguishes it from some others such as the Sydney Coastal Heaths.

Post-fire grazing

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

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

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

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

The need to consider the effects of post-fire grazing by native animals may be growing in the Southern Tablelands, particularly as numbers of macropods have built up over time due to landscape and population management changes (reduction or cessation of shooting) (Dr Phil Borchard *pers. comm*.)

Conclusion

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

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

Upper thresholds need to ensure that I species have opportunities to recruit before adult plants and seedbanks decline, and that competitive interactions that occur in the absence of fire are managed (*pages 8-10*).

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

The greater risk, at this end of the fire frequency spectrum, is that I species native to the Southern Tablelands Dry Sclerophyll Forest may decline under very long intervals. The recommended upper threshold of 30 years is several years below the maximum inter-fire 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 (Penny Watson 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 longest inter-fire interval was of the same order as that in the more recently burnt sites (Purdie 1997b), suggesting seedbanks at 30-35 years post-fire are still adequate. However this may not be the case four or five decades post-fire. Grazing before fire may exacerbate the problem by reducing seed input by adult plants, while post-fire grazing may eliminate seedlings and stress resprouting plants, particularly if burns are small (*page 100*). 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.

6. Western Slopes Dry Sclerophyll Woodlands, comprising 7 RVCs

a) *Bulloak - White Cypress Pine woodland mainly in the NSW South Western Slopes Bioregion (Benson 54)*
b) *Dwyer's Red Gum - Currawang grassy mid-high woodland of central NSW (Benson 257)*
c) *Inland Grey Box - Black Cypress Pine shrubby woodland on stony slopes NSW South Western Slopes and Riverina Bioregions (Benson 110)*
d) *Long-leaved Box - Red Box - Red Stringybark sheltered open forest of the NSW South Western Slopes Bioregion (Benson 287)*
e) *Mugga Ironbark - Inland Grey Box - pine tall woodland of the NSW South Western Slopes Bioregion (Benson 217)*

f) *Mugga Ironbark - White Cypress Pine woodland on sedimentary or metamorphic low rises in the temperate (hot summer) climate zone (Benson 243)*
g) *Tumbledown Red Gum - Black Cypress Pine - Red Box low woodland of hills of the South Western Slopes*

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 Lachlan Valley this forest class is particularly prominent within the string of national parks on the north-south ridge systems in the mid reaches of the catchment between Cowra, Forbes and Parkes (e.g Conimbla, Goobang, Weddin Mountains).

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).

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, 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 shrub… 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 *e 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).

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 NSW and are less dependent on fire for regeneration." Ecologist Ross Bradstock suggests that there may be variation in the degree to which *C. endlicheri* 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.

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.

FIRE IN WET SCLEROPHYLL FORESTS

1. Southern Tablelands Wet Sclerophyll Forest, comprising 3 RVCs

a) *Peppermint - Mountain Gum - Brown Barrel moist open forest of the South Eastern Highlands*
b) *Mountain Gum - Manna Gum open forest of the South Eastern Highlands*
c) *Snow Gum - Mountain Gum tussock grass-herb forest of the South Eastern Highlands*

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

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

Much of what we know about fire in wet sclerophyll forests comes from over 50 years of research in high altitude forests of Mountain Ash (*Eucalyptus regnans*), a species which occurs in Victoria and Tasmania but not in NSW.

One class of wet sclerophyll forest occurs in the Lachlan CMA, Southern Tableland Wet Sclerophyll Forest. This grassy subformation Wet Sclerophyll Forest is represented in the Lachlan Valley by three regional vegetation communities (DEC 2006); Peppermint (*E. radiata* ssp *sejuncta*) –Mountain Gum (*E. dalrympleana* ssp *heptantha*) – Brown Barrel (*E. fastigata*) Moist Open Forest, Mountain Gum – Manna Gum (*E.* viminalis) Open Forest and Snow Gum (*E. pauciflora*) - Mountain Gum tussock grass-herb forest. These forest communities extend across the higher plateaux sections in the north of the South Eastern Highlands bioregion (Gellie 2005). Gellie (2005) groups these forests as Montane Tableland Forests. They are dominated by *Eucalyptus pauciflora, Eucalyptus dalrympleana*, and *Eucalyptus dives* and occupy the more productive red earths in the Crookwell and Oberon plateaux with moderate to high rainfall. A higher altitude variant occurs between 900 and 1300 metres, it occupies sites ranging from exposed and rather infertile soils to deeper and moderately fertile soils. The dominant eucalypts are *Eucalyptus pauciflora* and *Eucalyptus dalrympleana subsp. dalrympleana. Eucalyptus dives* and *Eucalyptus rubida* dominate in drier and lower sites, with *Daviesia mimosoides, Persoonia chamaepeuce,* and *Poa sieberiana* subsp. *sieberiana* becoming more dominant in the understorey. On moister sites the understorey changes to a moderate cover of *Poa* spp. and *Daviesia latifolia,* with a greater range of herbs, such as *Stellaria pungens, Asperula scoparia, Viola betonicifolia* subsp. *betonicifolia,* and *Acaena novae-zelandiae* (Gellie 2005).

Wet sclerophyll forests occur across eastern Australia where factors such as higher soil fertility, rainfall and topographic relief (slopes; valleys) provide suitable habitat. Climatic patterns from hot dry summers in the south to wet summers in the north strongly interact with the high fuel loads produced by these productive systems. In the classic studies in the Victorian Ash forests (see Ashton 1981) it has been demonstrated that fire (i.e. hot devastating summer crown fire) is a major ecological event that shapes those forests. McCarthy *et al*. (1999) modeled fire regimes in those forests. The results suggest a mean fire interval in those forests of 35-70 years (with a mean interval between tree killing fires – presumably more intense fires – of about 75 – 150 years). Changes (decreases) in fire intervals predict biodiversity declines in these forests. In northern Australia and along the NSW coast and ranges, fire in wet sclerophyll forests may be different in average intensity and season, none-the-less it is still considered a major determinant of successional change (e.g. Harrington & Sanderson 1994).

A number of canopy species in wet sclerophyll forests are obligate seeders (species that if burnt by intense fire regenerate from seed not lignotubers or basal sprouting). Some canopy dominants in coastal CMAs (Hawkesbury –Nepean, Hunter-Central Rivers, Northern Rivers and Southern Rivers) may be or act as seeder species under certain conditions or in certain locations (e.g. *E. grandis*, *E. pilularis* - Ashton 1981). Hence, based on the Victorian model, intense fire (particularly crown fire) may result in even-aged stands of canopy species that are not capable of recruitment until they age past the primary juvenile stage. Repeat fires within this period (decades for some species) would shift the vegetation towards those species that resprout or with short juvenile periods. This model results in regenerating canopy dominants, or alternatively, thickets of subcanopy species under frequent fire. The floristic and structural similarities of wet sclerophyll forests at regional or national scales, however, mask some significant differences in the fire ecology among these systems.

A fire return intervals of 50 – 150 years can be regarded as a broadly appropriate management target for the Victorian Mountain Ash wet sclerophyll forests. However, in northern New South Wales, there is considerable complexity in the wet sclerophyll forest types and at least some of this relates to differing fire regimes. Tasker (2002), for example, found that fires associated with forest grazing (i.e. high frequency, low intensity ground winter fires) favour diverse open grassy understoreys. Less frequent but more intense fires will generally favour shrub species that recruit from seed in response to fire cues leading to a dense shrub layer of these species. These species typically have juvenile periods that range from a few years to almost a decade. Additionally, many of these understory species from families such as the Mimosaceae respond to fire associated germination cues (Auld & O’Connell 1991; Brown *et al*. 2003). In particular, the temperature reached in the soil as a result of fire is critical in relation to germination for these hard seeded species. This will depend on the characteristics of the fire (intensity; retention time) as well as the soils characteristics (Whelan 1995). These species can therefore be generally be regarded as pyrogenic (i.e. they regenerate in response to fire), but there may also be considerable variability among species in fire response. In contrast to these pyrogenic species, some understorey species of wet sclerophyll forests are not cued to regenerate with fire (Adamson & Fox 1982). Hence long inter-fire intervals can lead to invasion of both rainforest understorey and canopy species and the conversion of the forest.

Achieving ecologically driven fire management can be problematic in urbanized areas (Morrison *et al*. 1996). However, active fire management of wet sclerophyll forests has been undertaken within the urban matrix in the Sydney area for the endangered Blue Gum High Forest (McDonald *et al*. 2002). This work demonstrated the persistence of a recoverable understory seed bank across decades that responded to fire. This recovered understorey was distinct from the pre-burn forest understory. It may be that the wet sclerophyll forests we seek to manage are not representative of forests of the past.

There are likely to be complex relationships among the 36 wet sclerophyll forest vegetation units mapped by Somerville (2009) which transcend the simple division into two wet sclerophyll forest sub-formations. Fine scale mapping on the Central Coast, for example, has mapped significant variability among wet sclerophyll vegetations at scales of ~ 100 m (Bell 2008) which can only partly be resolved to map units given a typical level of survey data (Tierney 2009). This variability is likely to be at least partly a result of fire history and to this extent the wet scleropyll forests represents a “fire modified” system. Thus in north-eastern NSW fire intervals between 2 and 5 years have been associated with open structured wet sclerophyll forests with a diverse flora of tussock grasses, forbs with some shrubs (Stewart 1999; Tasker 2002). Nearby areas which have remained unburnt for periods over 15 or 20 years support higher densities of some shrub and noneucalypt tree species, particularly those able to recruit between fires (Birk & Bridges 1989; Henderson & Keith 2002). It appears these forests can have differing states and fire is associated with at least some of this difference (see below).

Watson (2006b) reviewed the literature on fire regimes for grassy and shrubby wet sclerophyll forests of the Northern Rivers. Sections of this work are relevant to the Lachlan CMA, this is summarized below.

Wet Sclerophyll forests – grassy sub-formation

Birk and Bridges (1989) studied the effect of fire at intervals of two and four years compared to fire exclusion over a 20 year period in blackbutt forest on the lower north coast. They found that burnt plots supported predominantly grasses whilst woody shrubs were dominant in the unburnt plots. York (1999) measured vegetation structure on experimental plots burnt every three years and in plots unburnt for 20 years in blackbutt forest near Port Macquarie. Cover assessments were based on vegetation height classes (not plant life-form), but the “very tall shrub layer” (150-200cm) was significantly greater in unburnt plots. Stewart (1999) investigated the plots of York (1999) and found the seeds of graminoids (sedges and rushes) and shrubs were significantly more abundant in the long-unburnt plots. The seedbank of the shrub species also had a higher species richness in the unburnt plots, although overall seedbank richness was identical in the two treatments. Species found only in unburnt plots included several graminoids, and broad-leafed shrubs and trees. Species found only in burnt plots included grasses, forbs and shrubs whose germination is cued to fire. Doug Binns (pers. comm. 2005) reports that in the study area of York and Stewart where fire had been excluded, a thick subcanopy of *Syncarpia glomulifera* occurred, particularly in wetter areas. Unlike the burnt sites, unburnt sites had very little grass.

Kitchin, Henderson and Tasker have also studied the effects of frequent fire in wet sclerophyll forests. These studies were carried out in wet scleropyll forests of the tablelands in northern NSW. Kitchin (2001) found woody plant species richness was lower in Tablelands sites which had experienced six or more fires in a 25 year period, or where at least one interfire interval of 1-2 years had occurred, than in either long unburnt sites or in vegetation exposed to moderately frequent fire (2-4 fires in 25 years) and relatively long interfire intervals. One woody species (*Pimelea linifolia*) was more frequent in more frequently burnt sites. Two shrubs (*Polyscias sambucifoliis* and *Olearia oppositifolia*) had lower abundance when the shortest interfire interval was lower, while another (*Hibbertia obtusifolia*) had higher abundance when the shortest interval was lower. Multivariate analysis of the shrub data indicated significant effects on community composition for number of fires, length of shortest interfire interval, and time since fire. Total woody plant abundance was considerably higher in recently-burnt sites which had experienced 2-4 fires in 25 years, with no short interfire intervals, than in either long-unburnt sites, or where inter-fire intervals had been short. This pattern held for shrubs 2-10m, and for shrubs under 2m. Very frequently burnt sites had very few shrubs. Multivariate analysis also revealed a cluster of herbaceous species associated with the mid-range on most variables (number of fires; fire interval length; time since fire) but with a tendency towards higher number of fires. Length of shortest interfire interval had a significant association with herb species composition. Grass cover was greatest where fire frequency was high, and/or where at least one very short interfire interval had occurred, and grass species richness was highest in very frequently burnt sites. Sedges and rushes were more abundant where fire had not occurred for a long time and where interfire intervals were relatively long. Tussock grasses *Themeda australis, Poa sieberiana* and *Sorghum leiocladum* dominated high fire frequency sites.

Henderson and Keith (2002) also researched the effects of disturbance in grassy Tablelands forests. Only the shrub component of the vegetation was assessed. While number of fires was used as a variable in multivariate analyses, scarcity of records limited the authors' confidence in its accuracy. Disturbance alone accounted for 15% of the variation in adult shrubs among sites; mostly this was attributed to grazing. Fire frequency alone accounted for only 2.5% of total shrub variation. More disturbed sites had significantly fewer shrub species and shrub density was also lower in more disturbed sites.

Tasker (2002) studied plants, small mammals and invertebrates in Northern Tableland wet sclerophyll forest south and east of Armidale. Some of her work involved a survey of a large number of sites (58) across a 1000km^2 area. More detailed survey work was conducted in 12 sites. Six of these 12 sites had been grazed and burnt in low-intensity 'green pick' fires at approximately 1-5 year intervals, while six were in ungrazed areas which had remained unburnt for at least 15 years. Tasker (2002) found that shrubs and small trees dominated the understorey in ungrazed sites, while grasses dominated in grazed areas (which *includes* frequent burning). Analysis revealed that grazing practices, including burning, were having far the greatest effect than a range of other environmental and disturbance variables (Tasker & Bradstock 2006). Sites on a grazing lease where burning had occurred every 1-2 years had particularly low vegetation complexity scores. However plant species richness was *higher* in the six grazed and burnt sites than in the equivalent unburnt areas in Tasker's 12-site study, at both quadrat and site scale (Tasker 2002). Species composition also differed considerably between the two treatments. Herbaceous species were particularly well-represented in the burnt plots, with many herbs found in these areas absent, or much reduced in abundance, in unburnt areas. Ungrazed/unburnt areas supported many more fern, climber, and small tree species than their frequently burnt counterparts. Many species in these groups were found only, or almost exclusively, in unburnt plots, and many of them had rainforest affinities.

Fauna

York (1999) presents a detailed and rigorous analysis of the effects of two different fire regimes on five groups: ants, beetles, spiders, bugs and flies in grassy wet sclerophyll forest. Overall species richness was identical in the two treatments (burnt and unburnt over 20 years – see above). However individual groups showed a variety of responses to the two treatments, as did subsets of species within them. At subplot level, there were less fly and beetle morphospecies (presumed different species) where burning had occurred, but numbers of bug, spider and ant morphospecies were higher. These results were significant for flies, beetles and ants. Community composition also varied within groups depending on fire treatment, with large numbers of species appearing only, or mostly, in one treatment or the other. In some cases differences in species composition could be linked to habitat features characteristic of the two fire regimes. For example plant-eating bugs, flies and beetles were considerably more abundant on burnt plots, reflecting the dense groundlayer vegetation in these areas, while flies and ants known to feed in the litter layer were more common on unburnt plots. On the other hand, litter-feeding spiders were mostly found on burnt plots.

York (2000) focuses on ants. This article covers both the results for ants of the study reported in York (1999), which used pitfall traps, and further work on the same plots carried out several years later – but also two years post-fire in frequently burnt plots – based on litter samples. Groups of species unique to each habitat were identified in each study, as was a substantial group that occurred on both burnt and unburnt sites. York (2000) concluded that a variety of management strategies, from fire exclusion to frequent burning, would be needed in the forests of the region to maintain the full complement of ant species.

Andrew *et al.* (2000) also studied ants, but two years after the study reported in York (2000) in the same study sites. Burnt plots were four years post-fire. There were no statistical differences between the two burn treatments in either the abundance or the species richness of the ant fauna. Community composition did not differ greatly between habitats; open areas in burnt sites had the most distinctive ant assemblages. Habitats were not, in general, distinguished by differences in the abundance of the various ant functional groups, with one exception: subdominant ant species were only found in the burnt area. The authors concluded that, four years post-fire, no adverse effects of burning on ant diversity could be discerned, that management should aim to maintain a range of burn frequencies, and that retained logs in frequently burnt areas could contribute to invertebrate conservation.

Results for small mammals, from survey work on 12 intensively-studied sites, are reported in Tasker and Dickman (2004). These authors found no difference in species richness among frequently burnt and unburnt sites but there were big differences in species composition. Bush rats (*Rattus fuscipes*) occurred in much greater abundance in the ungrazed and unburnt areas, and Brown Antechinus (*Antechinus stuartii*) also tended to favour these sites. However three species were caught only on the grazed and frequently burnt sites, and another mostly there – and these were rarer species, including the New Holland Mouse (*Pseudomys novaehollandiae*) and the Hastings River Mouse (*Pseudomys oralis*). Swamp Rats (*Rattus lutreolus*) were also more numerous on burnt sites.

Bickel and Tasker (2004) report on differences among invertebrates for the same 12 intensively studied sites reported by Tasker and Dickman (2004). The findings were similar to York's; community composition differed between grazed/burnt and ungrazed/unburnt areas, although there were no significant differences in the overall diversity of invertebrates caught in sticky traps placed on tree trunks. Invertebrates other than flies (Diptera) were significantly more abundant in grazed and burnt sites.

Discussion of studies

In the last decade studies relating to the effects of fire in grassy wet sclerophyll forests have moved understanding forward considerably. The studies described above present a coherent picture of a vegetation type strongly affected by fire. The picture is not, however, simply one of detrimental effects from frequent burning, and unmitigated benefits from long interfire intervals. While some groups of plants and animals do indeed appear to decrease in richness and abundance when burning is frequent, others increase in this situation – and decrease when fire is infrequent or excluded. This picture is familiar from the discussion of 'states' in previous sections. In this wetter, more productive vegetation type, differences between fire frequency categories are more pronounced than in the drier grassy forests and woodlands.

Results from Stewart (1999), York (1999, 2000a) and Andrew *et al.* (2000) are particularly valuable in the current context, as this well-replicated experiment focuses directly on fire frequency without the complication of grazing which appears to have been unavoidable in a number of retrospective studies. Unusually, we know more about the effects of the two fairly extreme fire regime treatments on fauna, than we do about their effects on flora. The results for invertebrates clearly indicate that both very frequently burnt areas (3 year fire cycles), and long unburnt areas (20 years of fire exclusion), support an extremely diverse invertebrate fauna. These faunas differ substantially in composition but are similar in richness.

With respect to vegetation structure, York (1999) found that unburnt plots had significantly higher cover values for shrubs over 100cm. Given that the fire-treated plots had burnt just two years previously, this result is unsurprising. Burnt plots, on the other hand, had greater cover in the three vegetation classes below 100cm. Thus by two years post fire, these plots were by no means devoid of cover. This cover, together with flowering and fruiting grasses, herbs and resprouting shrubs, appears to provide suitable habitat for many invertebrate species.

We know less about plants than about invertebrates from the studies of York, Stewart and Andrew *et al*. However, these studies do suggest a pattern similar to that for invertebrates: both treatments support many species, but composition differs. While Stewart's soil seedbank findings are of interest, they need to be seen in context: only about a quarter of the plant species found in the above-ground vegetation were found in the seedbank (Stewart 1999). This finding is typical of grassy vegetation, as many herb and grass species do not have a permanent store of seeds in the soil (Morgan 1998b, 1997d; Odgers 1999; Hill & French 2003). Adding Stewart's findings to the little we know about the above-ground species complement suggests that plant diversity may be higher where burning has been frequent; that some species, particularly graminoids and soft-leaved shrubs do best where fire has been excluded; and that other species, particularly grasses, forbs and I-species shrubs, may be more diverse and abundant under a regime of fairly frequent fire. These findings are consistent with both the vital attributes and the dynamic equilibrium models.

Dominance of large, long-lived, T-species (the soft-leaved species which have increased in abundance over the years of fire exclusion in unburnt plots must be able to recruit between fires) in the absence of disturbance is predicted by Noble and Slatyer's (1980) model. A concomitant reduction in abundance of small, short-lived, light-loving species in the absence of disturbance is consistent with both models, while Binns' observation that grasses and forbs are affected by shading from thickening shrubs and small trees also points to the existence of competition in the absence of disturbance.

Research into fire in Tablelands wet grassy forests reinforces the picture presented by the previous coastal studies. Rainfall in Tablelands wet sclerophyll forests is similar to that in their coastal counterparts, however temperatures would be lower. Productivity may therefore be somewhat lower, particularly as winter frosts on the Tablelands will reduce the length of the growing season.

Frequent fire on the Tablelands was associated with a diverse grassy understorey, while areas subject to long interfire intervals or long periods without fire tended to carry more multilayered vegetation with a strong mesic/rainforest component. These differences were most apparent in Tasker's work (Tasker 2002), which highlighted both the high diversity of grasses and herbs in sites burnt every 1-5 years, and the relative dearth in these sites of the climbers, ferns and small trees which dominated unburnt areas. Henderson and Keith (2000) found considerably fewer individual shrubs and fewer shrub species in less disturbed sites, although grazing contributed here as well as fire. Kitchin found fire impacts on composition of both shrubs and herbs, with greater numbers of shrubs – which in her study included various sclerophyllous species – in sites which had experienced some fire than in either long unburnt or very frequently burnt sites. Sites exposed to a high fire frequency were dominated by native tussock grasses, and length of shortest inter-fire interval influenced herb species composition. John Hunter (pers. comm. 2005) also studied the effects of fire frequency on herbs in upland forests and also found that herbs declined in long unburnt grassy forest areas, and considers that both shading by shrubs, and a heavy litter layer, may be responsible.

Tasker's results for both small mammals and invertebrates are consistent with the previous coastal studies. Frequently burnt and long unburnt areas supported particular suites of species, though species richness was similar. Tasker and Dickman (2004) point out that this finding "contradicts the prediction of Catling and Burt (1995) that eucalypt forest with fewer understorey shrubs would have fewer species of small mammals. In this study, moderately frequent disturbance resulted in habitat suitable for early-mid successional species" (Tasker & Dickman 2004). In their discussion, Tasker and Dickman (2004) point out that the distribution of the two native mouse species found only in grazed and burnt sites corresponds with that of grazing leases, and that these species forage amongst the diverse herbaceous layer promoted by moderately frequent fire. Although not specifically focused on fire, a local study of *P. oralis* (Townley 2000) also reported that this species was generally found in sites with a predominantly grassy understorey, that grasses and herbs were important food sources, and that plant diversity at a small scale was linked to trap success. This species also appears to need access to dense, low cover (Townley 2000). Tasker and Dickman (2004) conclude that "Management of former grazing leases incorporated into National Park for the conservation of *P. oralis* in our view will require sufficiently frequent fire disturbance to maintain an open and floristically diverse ground cover while still maintaining adequate shelter cover for the species."

Some years previously, Christensen (1998) also argued, on the basis of a review of the literature, that retention of frequent low-intensity fire in the grassy forest landscapes of Northern NSW would be the precautionary approach towards conservation of medium-sized mammals. A number of Australian taxa in this size range use the early post-fire environment and/or grassy areas which are maintained by frequent fire. Vegetation which provides dense cover is also important for species of this size, and fire regimes which promote the juxtaposition of grassy and shrubby patches may be vital. The vulnerable Parma Wallaby (*Macropus parma*) is an example (Maynes 1977; NPWS 2002).

In the upper Clarence River valley (Southwell & Jarman 1987) explored habitat preferences of macropods and found species showed different preferences. Much of the study area was burnt in patchy fires associated with grazing management: the average interval between fires at any one point in the landscape was estimated to be about four years. These areas supported grassy vegetation which was associated particularly with Eastern Gray Kangaroos and Red-necked and Whiptail Wallabies: each of these species was linked with specific features within the grassy landscape. Grey Kangaroos and Red-necked Wallabies preferentially grazed recently burnt areas (Southwell & Jarman 1987). On the other hand, Red-legged pademelons were found only in dense rainforest. Several species, including Red-necked Pademelons and Black-striped Wallabies sheltered in dense forest understorey during the day, but foraged in open forest or pasture at night.

It appears clear, from the range of studies outlined above, that relatively frequent fire in grassy wet sclerophyll forests creates an open landscape in which tussock grasses, forbs and some shrubs thrive, creating habitat which is preferentially utilized by many animals. Vegetation which has not been burnt for some time, or where fire frequency has been low favours some shrub and noneucalypt tree species, particularly those able to recruit between fires. This thicker vegetation has a deep litter layer and is associated with habitat features which are important for a different suite of animal species.

Grassy wet sclerophyll forests can thus exist in at least two 'states.' The dynamic nature of these forests suggests they would fall into either Bond's 'climate limited but fire modified' or his 'fire limited' category (Bond *et al.* 2003, 2005). The extent of successional change in the absence of fire remains to be determined. However as Doug Binns (pers. comm. 2005) points out, the considerable differences between burnt and unburnt plots at Bulls Ground have occurred in a relatively short time-span (20-30 years), emphasising the lability of this vegetation type.

Existing fire regime guidelines

DECC NSW includes a guideline explicitly for the grassy wet sclerophyll forest vegetation type. Suggested intervals are 10 and 50 years, with the proviso that "Occasional intervals greater than 15 years may be desirable. Crown fires should be avoided in the lower end of the interval range (Kenny *et al.* 2004).

Discussion of fire regime guidelines for grassy wet sclerophyll forests: One regime or two?

The NSW guidelines for this vegetation type cover a wide range of intervals. Both refer to fire intensity. Is it possible that both occasional high intensity fire, and more frequent lower intensity burns, play a role in conserving diversity in grassy wet sclerophyll forests? In Victoria, 'tree killing' fires in *Eucalyptus regnans* forests may be interspersed with less intense sub-canopy fires, which regenerate the *Pomaderris aspera* understorey (Ashton 1976; McCarthy *et al.* 1999). Similar dynamics have been recognised in North American pine and mixed conifer forests. There the interplay between different sorts of fires and landscape features are reported to create a shifting mosaic of vegetation structure and composition which provided a wide variety of habitat and help limit forest susceptibility to large-scale insect-related disturbance events (Hessberg *et al.* 2000; Keane *et al.* 2002).

The relationship between fire and eucalypt species in wet sclerophyll forests is not the same in all wet sclerophyll forest types (Florence 1996). While the concept of occasional standreplacing fires fits well for obligate seeder eucalypt species such as *E. regnans* and *E. delegatensis* (the latter species occurs in southern NSW), the relationship between high intensity fire and resprouter wet sclerophyll forest eucalypts is less clear. Keith (2004) lists dominant tree species in each of his vegetation formations, while the NSW Fire Response Database (DEC 2002) summarises what is known about the regeneration mode of individual species. No tree species listed by Keith for grassy wet sclerophyll forest types is unequivocally classed as an obligate seeder, although several act in this manner under certain circumstances. Two grassy subformation eucalypts, *Eucalyptus pilularis* and *Eucalyptus obliqua*, resprout in the drier part of their range, but not in wetter areas (DEC 2002). Thus most if not all grassy wet sclerophyll forest eucalypts are unlikely to die en masse in a fire, and are also unlikely to exhibit the 'wheatfield germination' of their obligate seeder counterparts (Florence 1996). Florence (1996) suggests that high intensity fires in 'fire-tolerant' wet sclerophyll forest may kill individual trees or groups of trees only where they are senescent or weak, creating small patches of even-aged regrowth dispersed though the forest. There is an implication here that intense fire plays a role in providing conditions needed for eucalypt regeneration, however we are not aware of studies addressing this topic directly.

Low to moderate intensity fires may have little effect on the wet sclerophyll forest overstorey; however they may play an important role in understorey dynamics. It is possible that the DEC guidelines pertaining to the forest type under discussion represent something of an uneasy compromise between the fire requirements of overstorey and understorey. This compromise position could fail to deliver either sufficiently frequent low intensity fire to maintain a range of understorey habitats, nor sufficiently infrequent high intensity fire for eucalypt regeneration. The concept of a 'two tier' regime is proposed as a way out of this dilemma.

Uneasy compromises may also be inevitable in any state-wide guideline for grassy wet sclerophyll forests. As noted above, there are likely to be major differences across NSW in the nature of the relationship between wet sclerophyll trees and fire. Understorey dynamics may also differ considerably, reflecting differences in climate from the subtropical north to the subalpine south. These differences highlight the benefits of tailoring regimes for smaller geographic areas.

The upper threshold of 50 years proposed by Kenny *et al.* (2004) may in part reflect the authors' desire to ensure a reasonable life-span for obligate seeder eucalypts, which they note may occur in this vegetation type. This figure was obtained by bypassing 'most sensitive' plant species with lifespans of 20, 30 and 35 years (Kenny *et al.* 2004). In fact, 50 years would be a very short lifespan for obligate seeder eucalypts: for example *Eucalyptus regnans* lives for 350-500 years (McCarthy *et al.* 1990) and only starts producing the tree hollows used by arboreal mammals when it reaches about 120 years of age (Lindenmayer *et al.* 1997); *E. delegatensis,* which occurs in similar habitat, may develop over a similar timeframe. Mackowski (1984) found that hollows in blackbutt trees near Coffs Harbour did not start to form until trees were over 100 years old. Conversely, short intervals (for understory fires) provide habitat for a range of species. It is suggested that by reframing wet sclerophyll fire regimes as having two tiers, and acknowledging the major differences between forest types in the north and south of the State, this paradox can be resolved.

Interval domains for understorey diversity

High intensity fires will occur when weather conditions are extreme, whatever the regime at other times. Of more concern for vegetation managers is the nature of the 'second tier' regime (understorey fire). Here, a range of intervals across both time and space may be desirable. We know that quite frequent firein the Northern Rivers – fire at 1 to 5 year intervals – is associated with diverse ground layer vegetation (Stewart 1999; Tasker 2002) and a high abundance of many invertebrate species (York 1999, 2000a; Andrew *et al.* 2000; Bickel & Tasker 2004). This regime also provides habitat for a number of rare small mammals (Tasker & Dickman 2004). However burning at very short intervals will limit the extent to which vegetation progresses down the path towards shrubbiness and high litter levels (Birk & Bridges 1989; York 1999; Henderson & Keith 2002), features which are important for conservation of another component of forest diversity (Catling *et al.* 2000; York 2000a; Tasker 2002; Tasker & Dickman 2004).

The existence of two understorey 'states' supporting distinct suites of species in the grassy wet forests of North-eastern NSW implies the need for a fire regime which supports the existence of each state somewhere in the landscape. In some places, fire needs to happen often enough to maintain open, grassy forest environments rich in grasses and herbs, where early-successional animal species can thrive. Other places need to support good-sized patches of thicker vegetation where mesophyll shrubs and late-successional fauna can flourish.

This proposal is in line with the recommendations of York (2000a) and Andrew *et al*. (2000). It would also provide the shrubby vegetation advocated by Henderson and Keith (2002), and the "open and diverse ground cover" recommended by Tasker and Dickman (2004), albeit in different parts of the landscape. Both Tasker and Dickman (2004) and Christensen (1998) also point out that some animals need access to both open areas and denser cover; a mosaic of open and shrubby patches should fulfill that requirement.

What will it take to retain significant open areas (State 1) in the grassy mesic forest landscape? These productive landscapes may need more frequent disturbance than their drier counterparts. This reasoning suggests intervals in the 2 to 5 year range. While it is possible that a wider range of intervals (e.g. 2-7 years) may also produce the habitat needed by the suite of species that uses open grassy vegetation, this possibility has not yet been investigated. Two to five years encompasses the regime in coastal studies of York and approximates that in Tasker's burnt/grazed areas, and would allow time for dense grassy vegetation to develop in the later post-fire years.

What regimes might provide habitat for both sclerophyll and mesic shrubs, and mid to late successional fauna (State 2), without setting in train irreversible successional processes? There is a need for relatively frequent disturbance in wetter forests. Intervals in the six to 15 year range, with occasional intervals up to 20 years, would be in line with this thinking. This range is similar to that encountered by Kitchin in her moderate fire frequency sites, and probably encompasses a proportion of Tasker's unburnt areas. In the cooler Tablelands environment, slightly longer intervals would probably be appropriate – see summary guidelines below.

The rider to the DEC guideline – "occasional intervals greater than 15 years may be desirable" – can be read as a recommendation that *most* intervals fall within the 10-15 year range. This brings the guideline closer to the secondary regime recommendation in southeast Queensland (6+ years), and to the State 2 regime proposed above. It is worth noting that Kenny *et al.* (2004) identify only two 'most sensitive' species in grassy subformation wet sclerophyll forest with a minimum time to maturity of greater than one year. This suggests that although the short intervals recommended to maintain open understoreys will undoubtedly disadvantage some plant species, the number disadvantaged may not be high.

The two-tier fire regime concept calls into question how the two regimes might interact in relation to overstorey recruitment. If occasional intense wildfires do indeed play a major role in eucalypt recruitment, would secondary regimes, particularly frequent burning, need to be suspended to allow seedlings to reach the point where they can survive low intensity fires? Or might continued low intensity burning play a positive role in thinning young eucalypt regrowth? Might a cessation in burning result in more intense fire, and more damage to young trees, when the secondary regime is eventually reinstated? Does the nature of the understorey affect eucalypt recruitment? Might either thick grass or thick shrubs pose difficulties for young eucalypts? Might these competitive interactions, if they exist, help regulate sapling density appropriately? We do not have answers to these questions at present.

The concept of a mosaic of states also raises questions. One concerns the scale of the mosaic – should we aim to have large patches in each state, or a fine-scale mosaic of grassy and shrubby areas? A precautionary approach is to build on what already exists, taking into account landscape features. In areas that have been managed on short intervals in the past, it may be appropriate to maintain much of the landscape in an open state, but increase habitat diversity through reducing fire frequency in gullies and in other more mesic areas. Areas where fire has been less frequent and shrubs are thicker could be managed accordingly, with more open areas being introduced into the matrix through more frequent burning in strategic patches. In places where one state or the other dominates most of the landscape, it may be desirable to establish a more equitable distribution of states through greater or lesser use of fire.

Application to the Lachlan CMA grassy wet sclerophyll forests

The work carried out on the north coast and northern tablelands is partly applicable to the grassy wet sclerophyll forests of the Lachlan CMA. It must be noted that there are significant differences in rainfall and climatic factors. Because of this it is suggested that less frequent intervals are appropriate.

Summary recommendations

Wet sclerophyll forests in the Lachlan should be managed for a multilayered understorey: variable intervals between 12 and 25 years are suggested here. Occasional high intensity fire may be important for eucalypt regeneration.

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