HOTSPOTS FIRE PROJECT

Fire and the Vegetation of the Namoi CMA

(Draft)

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Front page: Semi-arid woodland east of Walgett, D. Tierney 2009.

1. INTRODUCTION

Scope of this review

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

Fire affects different plant and animal species differently, and fire regimes compatible with biodiversity conservation vary widely between ecosystems (Bond 1997; Watson 2001; Bradstock *et al.* 2002; Kenny *et al.* 2004). This document explores the role of fire in the vegetation formations of Keith (2004). All vegetation formations covered in this review are found in the Hunter Central Rivers CMA. 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 (it can occur in Forested Wetlands and peat fires are a major factor in some systems), 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 CMA region are discussed (often there is no literature for a class or perhaps only one study).

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 classifications to vegetation formations and classes. Despite these complications, fire is an important driver of biodiversity in many systems in the Namoi 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.

2. 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, and 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. 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 recently been used to develop fire management guidelines for broad vegetation types in NSW (Kenny *et al.* 2004).

The basic idea is that, to keep all species in a community, fire intervals should vary within a lower and an upper threshold. Lower thresholds are set to allow all species vulnerable to frequent fire to reach reproductive maturity, while upper thresholds are determined by the longevity of species vulnerable to lack of burning. Species with similar fire-related characteristics are grouped into functional types (Noble & 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 2003).

Data on plant life history attributes relevant to setting **lower thresholds** – regeneration modes and juvenile periods – are much more readily available than the information needed to set **upper thresholds** – longevity of adults and seeds. Kenny *et al.* (2004) note the lack of quantitative data on these latter attributes, and point out that as a result, upper thresholds in the NSW guidelines are "largely based on assumptions and generalisations" and are therefore surrounded by "considerable uncertainty" (Kenny *et al.* 2004). 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 2005).

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

Landscape productivity, as defined by plant biomass as an example, is likely to increase with rainfall, temperature, season of rainfall – where rainfall and warm temperatures coincide, there is a greater potential for plant growth – and soil fertility (clay soils are often more fertile than sandy soils, however they also tend to support more herbaceous, and fewer shrub, species: 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). 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 non-eucalypt 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.

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2. *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, 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).

3. THE VEGETATION OF THE NAMOI CMA

EA Systems (2008) describe the location, size, bioregions and the broad vegetation (formations) that occur in the Namoi CMA area. The CMA includes part of four bioregions spanning from rainforests to arid shrublands. Eco Logical (2008) have mapped the vegetation of the Namoi CMA and delineated vegetation units at a finer scale than the formations and classes which are recognized for New South Wales (Keith 2004, Table 1). This includes 65 map units (regional vegetation communities- RVCs) and the likely occurrence of these units has been spatially mapped at a recommended scale of usage of 1:50 000. Many Keith classes present in the Namoi CMA are represented by several RVCs with the largest diversity within the dry sclerophyll forests (15), grassy woodlands (13), semi-arid woodlands (13) and grasslands (8).

The literature on fire within different vegetation types is heavily biased towards particular vegetation formations and, in some cases, particular classes within a vegetation formation. Literature is very limited for some formations and classes. This literature review focuses on those vegetation types which occupy the largests areas and are represented by the most RVCs in the CMA. Hence, literature is reviewed for dry sclerophyll forests, grassy woodlands, semi-arid woodlands, grasslands (which together represent a total of 49 RVCs) and also wet sclerophyll forests and heathlands (because the literature is either relative large or particularly informative for these two formations). Rainforests are also briefly considered. The review starts with grasslands and (because the understorey of woodlands often includes many grassland species) then moves to woodlands. Systems that include prominent tree and / or shrub layers are then reviewed (forests and heaths).

Table 1. The Keith Formations and Classes represented by the map units of Eco Logical (2008) within the Namoi CMA. Classes shown as present in the CMA (+) or absent (-----).

Keith Formation	**Keith Class**	**Represented Unit**
Rainforests	Subtropical	+
	Northern Warm Temperate	?
	Southern Warm Temperate	--------
	Cool Temperate	--------
	Dry	+
	Western Vine Thickets	+
	Littoral Rainforests	--------
	Oceanic	--------
	Oceanic Cloud	--------
Wet Sclerophyll Forests	North Coast Wet Sclerophyll	--------
	South Coast Wet Sclerophyll	--------
	Northern Escarpment Wet Sclerophyll	--------
	Southern Escarpment Wet Sclerophyll	--------
	Northern Hinterland Wet Sclerophyll	--------
	Southern Lowland Wet Sclerophyll	--------
	Northern Tableland Wet Sclerophyll	+
	Southern Tableland Wet Sclerophyll	--------
	Montane Wet Sclerophyll	--------
Grassy Woodlands	Coastal Valley Grassy	--------
	Tableland Clay Grassy	+
	New England Grassy	+
	Southern Tableland Grassy	--------
	Subalpine	+
	Western Slopes Grassy	--------
	Floodplain Transition	--------
Grasslands	Maritime	--------
	Temperate Montane	+
	Western Slopes	+
	Riverine Plain	--------
	Semi-arid Floodplain	+
Dry Sclerophyll Forests	Clarence Dry Sclerophyll	--------
	Hunter-Macleay Dry Sclerophyll	--------
	Cumberland Dry Sclerophyll	--------
	Southern Hinterland Dry Sclerophyll	--------
	Northern Gorge Dry Sclerophyll	--------
	Central Gorge Dry Sclerophyll	--------
	New England Dry Sclerophyll	+
	North-west Slopes Dry Sclerophyll	+
	Upper Riverina Dry Sclerophyll	--------
	Pilliga Outwash Dry Sclerophyll	+
	Coastal Dune Dry Sclerophyll	--------

	North Coast Dry Sclerophyll	--------
	Sydney Coastal Dry Sclerophyll	--------
	Sydney Hinterland Dry Sclerophyll	--------
	Sydney Sand Flats Dry Sclerophyll	--------
	South Coast Sands Dry Sclerophyll	--------
	South East Dry Sclerophyll	--------
	Southern Wattle Dry Sclerophyll	--------
	Northern Escarpment Dry Sclerophyll	+
	Sydney Montane Dry Sclerophyll	--------
	Northern Tableland Dry Sclerophyll	+
	Southern Tableland Dry Sclerophyll	--------
	Western Slopes Dry Sclerophyll	+
	Yetman Dry Sclerophyll	+
Heathlands	Coastal Headland	--------
	Wallum Sand	--------
	Sydney Coastal	--------
	South Coast Heaths	--------
	Northern Montane Heaths	+
	Sydney Montane Heaths	--------
	Southern Montane Heaths	--------
Alpine Complex	Alpine Heaths	--------
	Alpine Fjaeldmarks	--------
	Alpine Herbfields	--------
	Alpine Bogs and Fens	--------
Freshwater Wetlands	Coastal Heath Swamps	--------
	Montane Bogs and Fens	+
	Coastal Freshwater Lagoons	--------
	Montane Lakes	--------
	Inland Floodplain Swamps	+
	Inland Floodplain Shrublands	+
Forested Wetlands	Coastal Swamp Forests	--------
	Coastal Floodplain Wetlands	--------
	Eastern Riverine Forests	+
	Inland Riverine Forests	--------
Saline Wetlands	Mangrove Swamps	--------
	Saltmarshes	--------
	Seagrass Meadows	--------
	Inland Saline Lakes	--------
Semi-arid Woodlands	Inland Floodplain Woodlands	+
	North-west Floodplain Woodlands	+
	Riverine Plain Woodlands	+
	Brigalow Clay Plain Woodlands	+
	North-west Alluvial Sand Woodlands	+
	Riverine Sandhill Woodlands	--------
	Inland Rocky Hill Woodlands	--------
	Subtropical Semi-arid	+

	Woodlands	
	Western Peneplain Woodlands	+
	Dune Mallee Woodlands	--------
	Sand Plain Mallee Woodlands	--------
	Semi-arid Sand Plain Woodlands	--------
	Desert Woodlands	--------
Arid Shrublands	Riverine Chenopod Shrublands	+
	Aeolian Chenopod Shrublands	--------
	Gibber Chenopod Shrublands	--------
	North-west Plain Shrublands	--------
	Gibber Transition Shrublands	--------
	Stony Desert Mulga Shrublands	--------
	Sand Plain Mulga Shrublands	--------

4. FIRE IN GRASSLANDS

Eco Logical (2008) have mapped eight regional vegetation classes (RVCs) of grasslands for the Namoi CMA. These comprise three Keith classes (Temperate Montane Grassland; Western Slopes Grassland; Semi-arid Floodplain Grassland – Keith 2004). At the state scale, Keith has mapped relatively small areas of the Temperate Montane Grasslands in northern New South Wales (with a large area in the Monaro). However, within the Namoi CMA, they are mapped as occupying a significant proportion of the north-east of the CMA. The Western Slopes Grasslands are mapped as occupying an extensive area of the central portion of the Namoi CMA, whilst the Semi-arid Floodplain Grasslands occur in patches across the north-west. These grasslands occur across very different landscapes with differing dominant grasses (*Themeda australis* / *Poa labillarderi / Austrodanthonia caespitosa*; *Aristida leptopoda / Austrostipa aristiglumis*; *Astrebla leptopoda* respectively). The fire ecology literature is not evenly spread among these grassland types (see below), thus the fire ecology of Australian grasslands is discussed in general and then considered for these specific grassland classes.

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) to eliminate 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; Morgan 1997; 1998b, 2001). Hence, Lunt and Morgan (1999) document significant vegetation change in a *Themeda* grasslands 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 effects are 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, at least some of the forb species in these grasslands employ a counter-intuitive life history strategy. Rather than being small short lived highly dispersive species, some employ a persistence strategy (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 “C_4 grasses” and grow more effectively in dry conditions than other “C_3 grasses”) may also contribute to different fire responses among grass species. Thus Watson suggested that “summer-growing C_4 grasses such as *Themeda* use water more efficiently and have lower nutrient requirements than all-season and winter-growing C_3 grasses like *Microlaena stipoides*, *Poa* and *Austrodanthonia* species (Ojima *et al.* 1994; Nadolny *et al.* 2003), and these characteristics may give C_4 species a competitive advantage in a frequently-burnt environment (Fynn *et al.* 2003).”

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 regulate (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). Yet there are examples of results (Lunt & Morgan 2000) and applications (Johnson 1999) of integrated strategies that included fire in the management of weed species in grassy systems.

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 lead 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.

The Namoi CMA grasslands

Temperate Montane Grasslands

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

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 in Section 1.2.2 for ensuring seed is available after a fire: creating seeds rapidly. Post-fire rainfall is also almost certainly an important determinant of recruitment success for grassland species (Morgan 1998c, 2001).

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 cuing seed germination in *Themeda* (Baxter *et al.* 1994; Wood 2001), although not all studies have found this to be the case (Clarke *et al.* 2000). Summer-growing C_4 grasses such as *Themeda* use water more efficiently and have lower nutrient requirements than all-season and winter-growing C_3 grasses like *Microlaena stipoides*, *Poa* and *Austrodanthonia* species (Ojima *et al.* 1994; Nadolny *et al.* 2003), and these characteristics may give C_4 species a competitive advantage in a frequently-burnt environment (Fynn *et al.* 2003).

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

An additional factor in the *Themeda* story concerns its response to grazing. Many researchers have noted a decline in *Themeda* with moderate to heavy grazing (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 C_4 species to use nitrogen and water efficiently suggests these species are more likely than C_3 grasses to have a positive response to fire. Some C_3 grasses, particularly *Poa sieberiana*, may also be encouraged, or at least not discouraged, by regular burning. Tasker (2002) found that *Poa sieberiana* was considerably more abundant in frequently burnt sites in the New England Tablelands than in sites which had not had a fire for many years. In Snow Gum woodland in Namadgi National Park Kelly (2004) was unable to detect a trend in the abundance of *Poa sieberiana* in plots subject to between two and 11 fires in a 41-year period.

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 recent test of this proposition (Prober *et al.* 2005) found reductions in levels of both nitrate and exotics in experimental plots seeded with *Themeda*. These reductions were most impressive when *Themeda* seeding was combined with burning (two fires one year apart). Prober *et al.* (2005) 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 aquatica* (Johnson 1999). The extent to which various combinations of fire and other strategies can assist in deterring aggressively invasive weeds in temperate grassy vegetation would be an excellent subject for adaptive management (Bradstock *et al.* 1995; Lunt & Morgan 1999a).

Western Slopes Grasslands

Western Slopes Grasslands occupied an extensive area of the central Namoi CMA area including the Liverpool Plains before European settlement. 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. 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 Namoi 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. 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 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. There are no references to the fire ecology of these grasslands in prominent documents that include the Determination to list this community as an Endangered Ecological Community (http://www.ennvironment.nsw.gov.au/determinations/NativeVegetationCrackingClayEndComListing.htm) under New South Wales legislation, a report on the technical workshop for listing the community as endangered under federal legislation (http://environment.gov.au/biodiversity/threatened/communities/pubs/grassland-bb-south-report.pdf) or published literature which mentions these grasslands (e.g. Prober & Thiele 2005 – who comment that these grasslands are poorly understood).

In the absence of any fire studies in this grassland type, some inferences can be cautiously made about possible effects of fire in this grassland. *Austrostipa* spp. are regarded as a cool season growing grass, generally with low persistence among years. Hence in the west of the CMA and western New South Wales, consecutive good growing seasons (good rainfall) are regarded as a requirement for *Austrostipa* to carry fire (Hodgkinson *et al*. 1984; Noble *et al*. 1986). Thus, overgrazing is regarded as having decreased the cover of perennial grass species to the point where fire can no longer occur as frequently across the landscape and so shrub species now dominate the vegetation (Hodgkinson & Harrington 1985). If Lang (2008) is correct and the *Austrostipa* dominance of these grasslands is an artefact of European settlement, then this dominant may result in altered fire patterns (possibly lower fire frequency). There are no reports of shrub invasion on the clay cracking soils of the Liverpool Plains. However, fire can be important for the recruitment of forb species in grasslands, even for those grassland forbs that are perennial because it induces flowering (Lunt 1990, 1994; Morgan 1996) and provides gaps for recruitment (Hitchmough *et al.* 1996; Morgan 1997, 1998a, 2001) of the often non-dormant seed (Willis & Groves 1991; Lunt 1995, 1996; Morgan 1998b,c). The available studies are from *Themeda* dominated systems and in lower productivity systems that patterns may be somewhat different.

In sum, there are no fire studies in the *Austrostipa aristiglumis* grasslands of the Liverpool Plains area. Inferences from historical records and from other grassland systems suggest that the remnant grasslands are possibly considerably altered in species composition and that fire may act differently in these altered remnants than it did before European settlement. Managing these remnants, as may be required by legislation (as a listed Endangered Ecological Community), is therefore challenged by a lack of known effects of fire (or its exclusion). Generally, the capacity of *Austrostipa* to carry fire may be lower than the inferred grass composition of Lang (2008). This may limit recruitment for some forb species, but this has not been experimentally determined. There is thus a lack of knowledge as to the role of fire in these grasslands and no guidelines can be given as to how fire should be managed specifically for this grassland type.

Semi-arid Floodplain Grasslands

The fire ecology of much of the semi-arid zone is closely tied to fuel loads which are largely driven by episodic rainfall (see detailed analyses in Semi-arid Woodlands below). Thus it is well established that many grass species of the semi-arid zone have their productivity and life cycles closely coupled to rainfall (e.g. Walker *et al*. 1981; Hodgkinson & Muller 2005) and that there is a general relationship among rainfall frequency and fire frequency (Walker *et al*. 1981; Turner *et al*. 2008). Whilst detailed fire ecology studies have been undertaken for a number of prominent grass species of the semi-arid zone including *Spinifex* (see Allan & Southgate 2002), *Themeda australis*; *Monochather paradoxa*, *Aristida* spp. and *Enneapogon* spp. (Walker *et al.* 1981; Daly and Hodgkinson 1996) and *Heteropogon contortus* (Shaw 1957) there are relatively few studies of the dominant grass species of the Semi-arid Floodplain Grasslands, Mitchell Grass (*Astrebla lappacea*).

Mulham (1985) did undertake an informative fire ecology study in these Mitchell Grass grasslands in north-western New South Wales. Following high rainfall in 1974-5 Mitchell Grass in the study area responded (like other dominant grasses of the semi-arid zone) with high growth rates and achieved fuel loads of ~ 3,000 kg / ha. Control burns resulted in: 1) the death of chenopod shrubs, which then grew from seedlings to approximate pre-burn densities after ~ 3 years; 2) an increase in annual forbs (within 12 months), which then declined to pre-burn levels three years later; 3) a decrease in Mitchell Grass cover which then recovered over the following years (but had not reached the cover of the unburnt control sites within the four year study period). A wildfire in a year with low rainfall did not produce similar short-term changes in chenopod shrubs and annuals.

Perennial forbs also were recorded at relatively high levels post-fire (it was presumed they had resprouted). There was a slower recovery after the wildfire of the Mitchell grass which was attributed to a low moisture status of the soil (poor summer rainfall). Scanlan (1980) recorded similar poor growth of Mitchell Grass post-fire in dry conditions. It was concluded that fire was a useful management tool in Mitchell Grass grasslands (after high rainfall events) and potentially of value in reducing shrub invasion. In comparison, other short-lived grasses from a comparative study site recovered poorly post-fire (*Aristida browniana*; *Aristida contorta*; *Enneapogon* spp.) and their recovered was considered to be strongly influenced by rainfall triggering germination from seed.

Other grasslands and variants

The Temperate Montane Grasslands recognised by Eco Logical (2008) included two RVCs (Derived Grasslands, New England Tablelands and Wet Tussock Grasslands of cold air drainage area, New England Tablelands). The Derived Grasslands generally fall within the *Themeda* grassy systems reviewed above. However, there are some likely significant variants in terms of the fire ecology of these systems. Watson (2007) noted that *Austrodanthonia* and *Austrostipa* species can dominate on poorer soils and that these systems of lower productivity may require less frequent fire (cited Stuwe 1994; Johnson 1999; Kenny *et al.* 2004). *Austrostipa aristiglumis* dominated grasslands are, however, reported from the Merriwa Plateau on basalt (high fertility at a high altitude - HCCREMS 2008). Therefore in grasslands dominated by these species the appropriate fire frequency may not necessarily be lower (i.e. productivity may vary with altitude etc.). However, in the west of the CMA *Austrostipa* spp. are regarded as generally of low persistence and low biomass and unable to carry fires except when two consecutive favourable years occur (Hodgkinson *et al.* 1984). Generally, gap closure among grass swards would indicate that a fire event would increase recruitment opportunities among tussocks. Factors such as the level of weed invasion, rainfall, season of burn and grazing pressure will then influence the outcome of this burn.

The Wet Tussock Grasslands of cold air drainage area, New England Tablelands RVC are also likely to have differing fire ecology from other Temperate Montane systems. This RVC would include small frost hollows of *Poa sieberiana* at high altitudes in some areas of the Tablelands.

Regular winter burning would generally be appropriate for these grasslands (e.g. see Tasker 2002). However, *Poa* may persist without high fire frequencies (Prober *et al*. 2007) and these high frequencies (4 to 8 years) may favour *Themeda*.

Derived grasslands also frequently occur on roadsides or in small managed lots (e.g. cemeteries) and are often maintained by slashing. These managed grasslands can be important habitat for threatened grassland species (Prober & Thiele 2005). Generally these grasslands can be maintained by regular slashing (although fire may be required for seedling establishment - Morgan 1998d: Morgan 2001) and the time of slashing should be considered in relation to the life-cycle of these species. The use of fire in managing the diversity of these systems should be considered as one possible approach, dependant upon the specific conditions and the ecological values of individual sites.

Bean and Whalley (2001) recognised a number of grassland types not recognised by Eco Logical (2008). This includes a *Eriochloa pseudoacrotricha* grassland on scree slopes and a *Bothriochloa* spp. grassland on basalt soils on the western edge of the Liverpool Plain. Bean & Whalley (2001) developed a State and Transition Framework for the grasslands of the Liverpool Plains and slopes in their study. This model emphasises the role of soil type and then, secondarily, grazing management in altering the composition and diversity of these grasslands. The role of fire is not elucidated in this scheme. They recognise that many of the grassland variants they describe have interstitial spaces dominated by perennial forbs. Grazing may limit the dominance of the grass species and maintain open spaces for these species so that (as in other grassy systems) the interaction of grazing with fire is likely to be important. Maintaining interstitial spaces (via grazing or fire) is important for forb diversity.

Conclusion

The findings outlined above have led to the conclusion that:

1. Temperate grassland species are well able to cope with frequent fire.

2. Fire or some other disturbance which removes biomass of the dominant grass species is essential if vigorous populations of forbs and less competitive grasses are to persist (Lunt & Morgan 2002).

3. Frequent fire is important for plant recruitment. Fire ensures that two of the three conditions for seedling establishment – gaps in the grass canopy and seed availability – are fulfilled. The third requirement, adequate moisture, may not be met after every fire, but it is argued that with relatively frequent fire, seeds, gaps and rainfall will coincide often enough to maintain forb populations (Morgan 1998a) – but see point 5 below.

4. Relatively frequent fire is needed to ensure the *Themeda australis* matrix which characterises many native temperate grasslands remains healthy.

5. In Mictchell Grass grasslands rainfall is a key trigger for grass growth and fire events that coincide with this high fuel load may result in differing outcomes than when fire occurs with lower fuel loads.

In NSW, the statewide thresholds for interfire intervals in grasslands are currently 2 and 10 years. The upper threshold is based more upon recognition of the need for periodic biomass removal discussed above than on vital attributes data, which was insufficiently detailed (Kenny *et al.* 2004). Intervals as low as 1-3 years have been recommended for Victoria's productive basalt grasslands (Morgan 1998a; Coates *et al.* 2006). Growth rates in the more arid grass systems or high montane grasslands of the Namoi CMA are likely to be slower than those in Victorian lowlands (in the semi-arid zone rainfall will trigger high growth rates). Ecologist John Briggs, who has experimented with various interfire intervals in grasslands around Canberra, suggests 5-8 year intervals may be appropriate in southern tablelands grasslands. However detailed work on the endangered daisy *Rutidosis leptorrhynchoides* (Button Wrinklewort), which is found in grasslands in the Goulburn area (Eddy *et al.* 1998) has led Morgan (1997) to conclude that remnant populations in *Themeda* grasslands “will need to be burnt at a maximum of 3-4-year intervals to ensure that large canopy gaps are regularly created to favour seedling recruitment and to minimize deep shading that will disadvantage the standing population.” In the semi-arid zone fire frequency is complicated by rainfall variability, stocking rates and past management practices (see Semi-arid Woodlands below) and rarely will short fire frequencies be achievable.

Application to the Namoi CMA grasslands

Temperate Montane Grasslands. Fire frequencies towards the upper range of that recommended for grasslands in New South Wales would be recommended (5-10 years).

Western Slopes Grasslands. Whilst there is a considerable body of research into the fire ecology of *Themeda* and *Poa* systems on the ranges and western slopes, there are no studies in the *Austrostipa aristiglumis* systems of the Liverpool Plains. A number of grasslands on fine textured soils also occur in the area. Generally maintenance of an open structure will enhance the species diversity of these grasslands. Fire may increase flowering and recruitment opportunities for forbs but studies are required to provide some definitive answers about the effects of fire in these grasslands.

Semi-arid Floodplain Grasslands. Fire following high rainfall (high growth) is likely to be a valuable management tool for these grasslands. It can reduce cover of chenopod shrubs and induce recruitment of annual forbs.

5. FIRE IN GRASSY WOODLANDS

Eco Logical (2008) have mapped 10 Grassy Woodland Regional Vegetation Communities (RVCs) which comprise three classes of grassy woodlands of Keith (2004 - Tableland Clay Grassy Woodland, New England Grassy Woodland, Western Slopes Grassy Woodland) of the Namoi CMA. 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 Namoi 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 supports that fire plays an important role in regulating these woodlands and this implies that fire was once an important part of these woodlands (Prober & Thiele 1993, 2005). However, the extent of loss and change to these woodlands means that appropriate fire regimes need to be largely implied from experimental studies.

The Ground layer

In the last decade there has been extensive ecological research undertaken in the grassy box woodlands of New South Wales. 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 cover (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.

The question remains: 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 among *Poa sieberiana* and *Microlaena stipoides* with the dominant tree *Eucalyptus laevopinea*. The deep-rooted summer growing *Aristida ramosa*, in contrast, was considered to be favoured 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 are 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 are frequently 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 associated with 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 seedlings). 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 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 HCR 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 Woodlands may fall into this category.

One native shrub species which has been very successful in building up its population in the Central West is *Cassinia arcuata*, or Sifton Bush (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. As noted in Section 2.2.2, repeated biennial burning may produce negative impacts on other ecosystem elements. However this trade-off might be worthwhile in circumstances where dominance of Sifton Bush is causing problems for biodiversity, particularly if only one or two short intervals were applied followed by a more moderate regime. A second or third fire would only be possible where sufficient grass fuel had developed. Strategies to encourage native tussock grasses such as *Themeda australis* might assist here.

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

The situation with fire and herbaceous weeds outlined for grasslands 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 recent article Prober and Thiele (2005) bring together what they have learnt from over a decade of research into grassy white box woodlands, and discuss implications for managers seeking to restore them. 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. As we have seen, Prober *et al.* (tentatively suggest fires at 5-8 year intervals may be compatible with maintaining a balance between the dominant grasses.

Application to the Namoi CMA grassy woodlands

1. Western Slopes Grassy Woodland

Western Slopes Grassy Woodlands have been subject to intensive research over the last 15 years. Prober *et al.* (2007) stress the important regulatory role of fire in these woodlands and a fire frequency of 5-8 years would generally be recommended. These guidelines would be modified by grazing which may decrease the dominance of *Themeda* over forb species. The RVCs of Eco Logical (2008) include two Box Woodlands that generally fit within the Box Woodland types subject to most of this fire research (RVCs 17 and 18). The RVCs of the Namoi CMA, however, have some significant variants in the dominant tree species compared to these Box Woodlands (RVC 19 White Cypress Pine – Silver-leaved Ironbark Grassy Woodland and RVC 20 Rough-barked Apple – Blakely's Red Gum Riparian Grassy Woodland).

White Cypress Pine – Silver-leaved Ironbark Grassy Woodland (RVC 19)

White Cypress Pine has the capacity to dominate sites in the absence of fire (see detailed ecology in the section on Dry Sclerophyll Forests that follows). This dominance results from recruitment in the absence of fire (it is a T species – Noble & Slatyer 1980). Fire potentially has a role in regulating White Cypress Pine dominance with intense fire leading to thinning of stands and greater diversity in ground and shrub species. This should be considered a potential strategy for managing these woodlands where appropriate.

2. New England Grassy Woodland

Three RVCs of Eco Logical (2008) are included in this Keith class (RVCs 15, 16 and 40). Dominant trees include *Eucalyptus blakelyi*, *Eucalyptus bridgesiana*, *Eucalyptus calignosa*, *Eucalyptus laevopinea*, *Eucalyptus melliodora* and *Eucalyptus youmanii*. *Eucalyptus blakelyi* has been the subject of studies by Li *et al.* (2003) which showed increased seedling survival growth rates with fire. Clarke (2002) also found that *E. blakelyi* (and *Eucalyptus melliodora*) seedlings that survived over a five year period had developed lignotubers. Similarly Semple and Koen (2001) report lignotuber resprouting post-fire in both these species. In general, these woodlands have eucalypts that are likely to be resistant to fire intervals of more than ~ 5 years. Knox & Clarke (2004), however, studied the fire ecology of shrub species in two sites in New England Grassy Woodland. They concluded that the primary juvenile period of most shrub species was more than four years, whilst the secondary juvenile for resprouting shrub species was less than four years. They recommend that minimum intervals between fires should be ~ eight years. This is longer than recommended by Prober *et al*. (2007) for Western Slopes Grassy Woodlands and Knox & Clarke caution about using recommended fire intervals from outside the New England region.

3. Tableland Clay Grassy Woodland

Tablelands Clay Grassy Woodland occurs at high altitudes on basalt on the New England Tablelands. Dominant trees are *Eucalyptus pauciflora*, *Eucalyptus stellulata*, *Eucalyptus viminalis* and *Eucalyptus melliodora* (Keith 2004). Eco Logical (2008) recognise three RVCs within this class.

Snow Gum – Black Sallee Grassy Woodlands (RVC 12)

Research in the alpine area of Australia has found high levels of resprouting in tree, shrub and ground layer species (Wahren *et al*. 1999), including *Eucalyptus pauciflora* (Leigh & Noble 1981). Australian alpine grass species in the genus *Poa* resprout and flower in response to fire within 12 months (Wahren *et al*. 1999). In the ACT a study which assessed effects of fire and grazing levels over almost fifty years found only one understorey species, *Daviesia mimosoides*, declined in abundance as fire frequency increased (Kelly 2004). *Daviesia mimosoides* is a resprouter with a short secondary juvenile period (Gill 1975), so it does not fit the model of being a seeder species that is eliminated by short fire frequencies. However other than this study little is known about the effects of fire on subalpine woodland shrubs. Species such as *Tasmannia purpurascens* and *Olearia oppositifolia* occur in this layer. *Olearia oppositifolia* has been recorded as a basal resprouter with no seed recruitment post-fire, as have some *Tasmannia* species (Campbell & Clarke 2006). Obligate seeders may well be comparatively rare in this vegetation however this remains undetermined. Thus there is some uncertainty about fire intervals for this vegetation. Knox & Clarke (2004) found that disturbance effects (e.g. fire) were of lesser importance in more productive systems (i.e. basalt soils) than competitive effects. Hence fire intervals may be less critical in these systems. However, fire will still have a role in regulating grass sward dynamics (i.e. more frequent fire may favour *Themeda australis* over *Poa* spp.) as well as forb diversity. The shrub layer may also provide important nectar resources for fauna, some of which regulate scarab bettles which cause dieback of eucalypts (Smith 1992). Generally the upper range of intervals suggested by Prober *et al*. (2007) should be appropriate for this vegetation type.

The remaining two RVCs (RVCs 13 & 14) within the Tablelands Clay Grassy Woodland are little studied in relation to their fire ecology but are likely to have similar fire requirements to RVC 12.

6. 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 New South Wales from the Darling River east to a line through Wagga Wagga, Dubbo and Narrabri. Rainfall in the semi-arid zone is between 250 and 500 mm per annum, with much variability between years. Hence drought plays a major role in shaping vegetation and water is limiting. The woodlands of the semi-arid zone are dominated by sclerophyll trees (eucalyptus, 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.

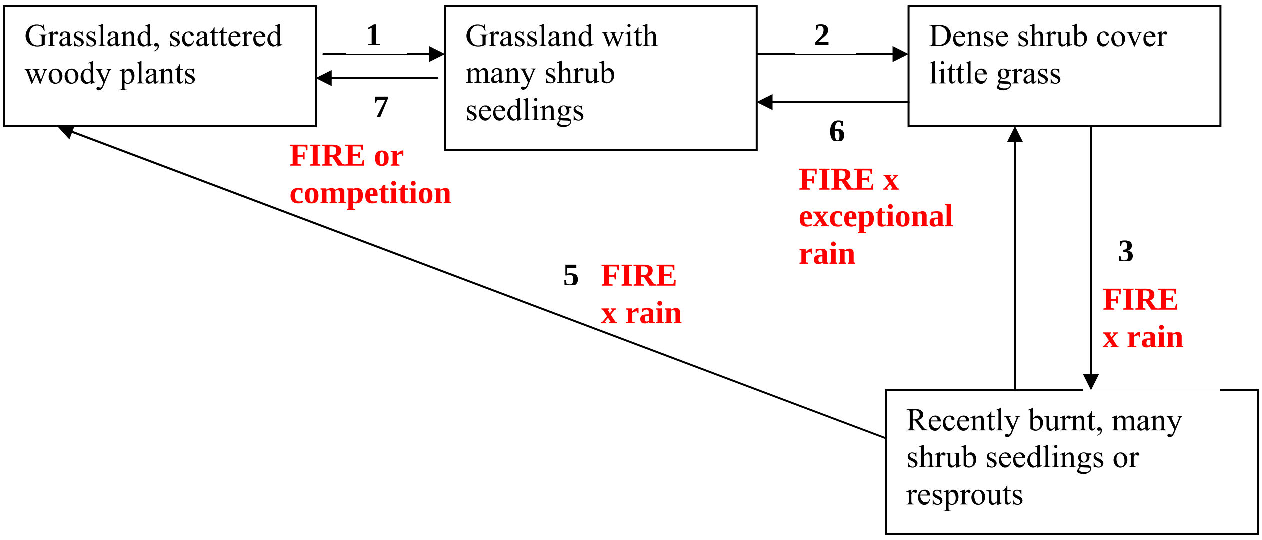
Keith (2004) identifies two subformations within woodlands of the semi-arid zone: grassy and shrubby. Woodlands in the grassy subformation occur on the floodplains in areas subject to occasional inundation. These woodlands intergrade with grasslands and arid shrublands. To the east, grassy subformation 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 classes, disturbance regimes also 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*), and in the east, Yellow Box (*E. melliodora*) and Grey Box (*E. microcarpa*). 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. These woodlands also contain a wide range of native grasses and herbs.

Aboriginal and then 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 Namoi CMA supports fourteen regional vegetation communities (Eco Logical 2008) that combined represent seven state level vegetation classes of semi-arid woodlands defined by Keith (2004 - Inland Floodplain Woodlands; North-west Floodplain Woodlands; North-west Alluvial Sand Plain Woodlands; Sub-tropical Semi-arid Woodlands; Western Peneplain Woodlands; Riverine Plain Woodlands & Brigalow Clay Plain Woodlands).

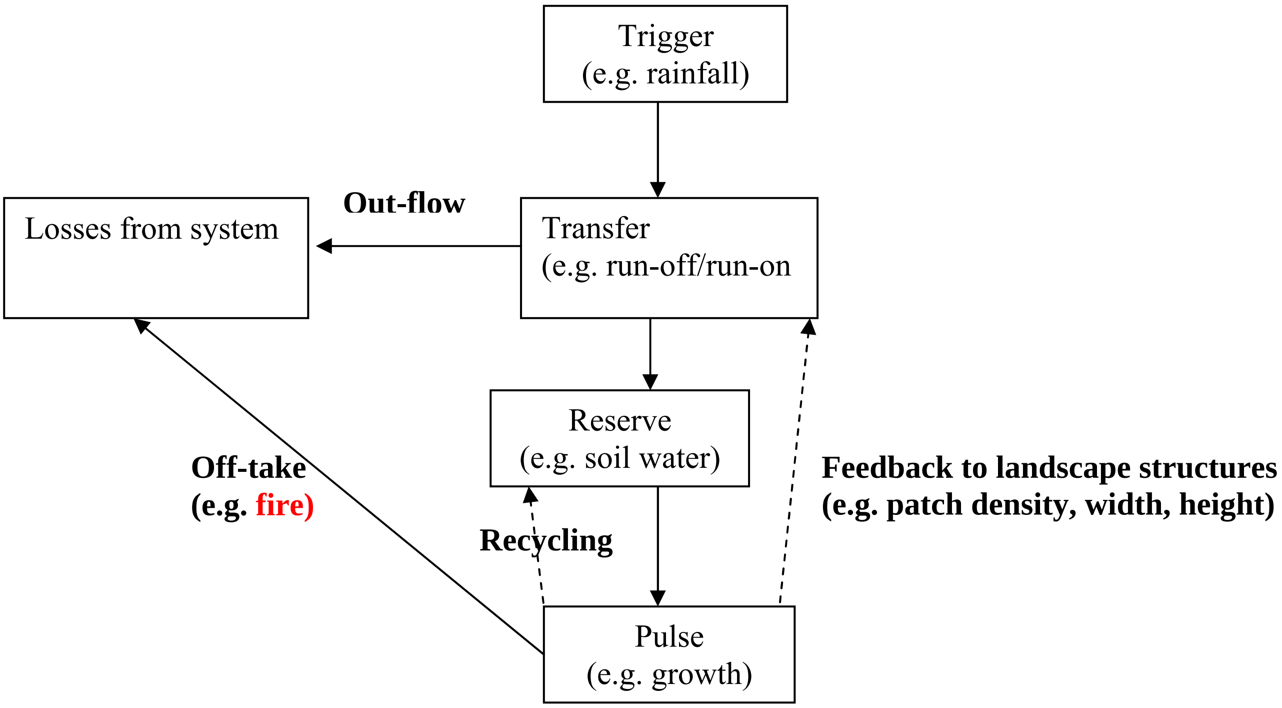
Fire in semi-arid woodlands is a complex issue. Noble & Grice (2002), for example, reviewed the issue of fire in semi-arid lands and it is clear from this review that there are complex interactions of fire with grazing, browsing and rainfall in these systems. It is also true that there is likely to be variability in how fire functions within differing woodland types. This means that reducing fire effects to simplistic fire frequency guidelines is both not possible and potentially counter-productive. 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:



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 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 Namoi CMA have a eucalypt dominated tree layer (Inland Floodplain Woodlands; North-west Floodplain Woodlands; North-west Alluvial Sand Plain Woodlands; Sub-tropical Semi-arid Woodlands; Western Peneplain Woodlands), two classess are dominated by *Acacia* species (Riverine Plain Woodlands & 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:



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 *et al.* (1981) estimated fire intervals of 1 in 25 years for Poplar Box woodlands in the CMA region. 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. 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 New South Wales).

Walker *et al.* (1981) provided a detailed review of fire behavior in Poplar Box Woodlands. Fire frequencies across the *full* area which these woodlands cover (from north of Longreach Qld to south of Hillston NSW) were estimated to vary greatly (down to almost every few years in northern regions). This was attributed to differences in fuel load of the grass species (estimated to vary from 50-600 g per m^2) 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 ~ 100 g per m^2 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 h were required to carry fire in the CMA region, whereas for *Themeda* dominated systems (carrying 300 g per m^2 of grass fuel) fire was reported to carry with wind speeds as low as 1 km per h. 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.

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 vegetations (Mulham 1985). In contrast, Shaw (1957) found that the grass *Heteropogon contortus* had relatively high survival, enhanced germination and was favored by reduced competition by annual burning and 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 – Hodgkinson and Muller 2005), 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 levels (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 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 growth and 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).

Conclusions

The fire regimes of the semi-arid woodlands appear to have undergone significant change post European settlement. Fire 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 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. 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 a*l. 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, the Shy Heathwren and the Scarlet-chested Parrot need, or prefer, mallee less than 10 years old, while Mallee Fowl use adjacent old and young patches. References can be found in Woinarski (1999).

7. FIRE IN HEATHLANDS

Eco Logical (2008) have mapped three heathland types in the Namoi CMA which occur within the Northern Montane Heath class of Keith (2004). There is a rich data set of fire ecology research for heathlands. Fire prone heaths of South Africa (fynbos), North America (chaparral) and Australia have been subject to extensive research. There are consistent patterns in the fire ecology of these heathlands at this global scale (e.g. Keeley & Bond 1997). Locally, a number of classic fire ecology studies have been undertaken on heathlands in New South Wales (see below). These provide a sound basis for understanding the fire ecology of these systems. This system level understanding does not mean that there is a comprehensive understanding of all aspects of the fire ecology of these systems. It is only comparatively recently that fire ecology studies have been undertaken in Northern Montane Heath (e.g. Clarke & Knox 2002; Williams & Clarke 2006). However, broad predictions about relevant fire regimes for these systems can be made with some confidence.

Fire and the standing vegetation

Fire is a dominant force in the ecology of heathlands in Australia and fire events can be intense because the fuel loads are high and the species present produce highly flammable litter (Keith *et al*. 2002). The plant species of these heaths are adapted to fire and there are predictable post-fire shifts in the vegetation over time (e.g. Keith & Bradstock 1994). These changes are largely predicted by the “vital attributes approach” of Noble and Slatyer (1980) and this approach has been specifically applied to heathlands of the CMA (Bradstock & Kenny 2003). Fire frequency is regarded as the predominant driver of plant species diversity and turnover in these heathlands (Cary & Morrison 1995). High fire frequency is a listed key threatening process under the New South Wales Threatened Species Conservation Act (1995) and can result in the local extinction of some plant species (Keith 1996). Resprouting species fit the ecological model of ‘persistent species’ (Bond & Midgely 2001) and these species generally cope with short fire intervals. However, recurrent fire may gradually exhaust the storage reserves of these species (e.g. Knox & Morrison 2005; Knox & Clarke 2006) and may weaken them and lead to their loss. There appears to be some variation in the proportion of resprouters among heaths. Myerscough *et al*. (1995) report that both wet and dry heath on dune ridges of Myall Lakes National Park (wallum heath) are largely dominated by resprouting species. About 27% of heath species in the study by Bradstock & Kenny (2003), which focused on Sydney coastal heath, could be assigned to the non-resprouting category.

Non-resprouting species ('obligate seeders') fit the 'regeneration niche model' (Grubb 1977). These species are predicted to be killed by fire but then regenerate from seed. There has been considerable interest and research into obligate seeder species in heathlands. An informative species that has drawn particular attention is *Banksia ericifolia* (Siddiqi *et al.* 1976; Morris & Mysercough 1983, 1988; Nieuwenhuis 1987; Bradstock & O'Connell 1988; Keith & Bradstock 1994; Bradstock *et al.* 1997). This species has a long juvenile period (i.e. it does not mature and produce seed until up to eight years post-fire). It then retains seed in fire-resistant fruit. If fire events are of high intensity and kill plants and occur at intervals of less than around seven years this species will be eliminated from an area (Siddiqi *et al.* 1976; Bradstock & O'Connell 1988). Alternatively, with long inter-fire intervals this tall species can shade out other species, dominate an area and reduce the diversity of species growing beneath it. This can occur when fire intervals exceed about 12 or 15 years (Keith *et al.* 2006). This species is an example of a species which, with its relatively long juvenile period, can be used to set the recommended minimum fire interval.

The seed bank

The seed bank provides an important buffer for many plant species in heathland systems. *Acacia suaveolens*, for example, is an obligate seeder that occurs in heathlands that (in contrast to *Banksia ericifolia*) produces seed that enters the soil to form a persistent seed bank (Auld 1986). Thus, even if plants of this species are eliminated from an area they may persist in the seed bank and regenerate via seed germination. There is likely to be considerable variability among species in the persistence of seed as a seed bank in the soil of these heath systems. There are few detailed studies, but in a comparison of 18 species Auld *et al.* (2000) reported seed half-lives of between 0.4 and 10.2 years. Auld *et al.* (2000) considered that the seed bank is important for plants with a range of characteristics including: 1) obligate seeders; 2) species that resprout somewhat but which rely on seed for long-term persistence; 3) species that survive fire but which are then competitively eliminated by site dominants; and 4) species in which effective seed production is suppressed by seed predation with time since fire. Species that, for example, are excluded by site dominants are those species that can be used to determine the upper limit of a recommended fire interval for a heathland (though they may persist as a seed bank – see below). Competitive domination of a site can occur within a decade in Sydney coastal heathlands.

The production of seed by a plant does not guarantee reproductive success. There are a range of factors that interact to affect the fate of seed. In the well studied genus *Banksia* these factors combined mean that up to 200 viable seeds may need to be set to guarantee replacement (Lamont *et al.* 2007). Factors that affect seed fate in heathlands include pre-dispersal (e.g. Auld & O'Connell 1988) and post-dispersal (e.g. Auld & Denham 1999) predation, competitive interactions (Lamont *et al.* 2007), weed invasion (Lamont *et al.* 2007), drought (Lamont *et al.* 1989) and herbivory of germinated seeds (Cowling & Lamont 1987). Complex seedling establishment patterns are a likely result of these interacting factors.

Recent work has also demonstrated that heathland seeds have diverse responses to imposed germination cues. This suggests that inherent habitat differences across areas and / or differences in fire intensity may select for the establishment of particular species. *Baeckea imbricata*, for example, has enhanced germination with smoke cues and no heat treatment (Thomas *et al.* 2003), whilst other heathland species have seed germination enhanced by heat treatments (Auld & O'Connell 1991; Auld 1996; Kenny 2000; Brown *et al.* 2003). Similarly, *Kunzea rupestris* has enhanced germination with smoke and no heat (compared to the closely related *Kunzea capitata* where heat and smoke both enhance germination – Tierney & Wardle 2005). These species occur in sites (wet sites and rocky outcrops respectively) that may reduce fire intensity. However, the complexity of interactions in many systems may negate the effect of any given factor. Lamont *et al.* (1997) report that high intensity fire, for example, can increase seed release and seedling establishment, but post-germination competitive effects mean that sites with less intense fire may still provide more favourable population outcomes. There is also diversity in dormancy types present in the seed of heath species that will cue seed germination events to seasons (Ooi *et al.* 2006) or events such as mechanical actions related to soil movements (Morris 2000).

It sum it is clear that fire frequency and intensity are significant factors that interact with a range of other site factors to determine the floristic diversity and structure of heathlands. The season when fire occurs and the long-term fire history of a site will also its species diversity of a site.

Fauna

Whilst fire can cause death, lead to increased predation pressures, or decrease habitat values for some fauna (Fox 1978), even some large animals such as wombats can shelter and then utilize the post-fire environment of heathlands (see below). There is only limited data on fire effects on many faunal groups and the theoretical basis for how different groups respond is still the subject of development (Tasker 2008). Keith *et al.* (2002) summarized the known responses of fauna to fire in heathlands. Salient points are below.

Mortality. Fox (1978) provided an account of the mortality of vertebrates following a large intense fire in Nadgee Nature Reserve. These observations were based on the remains of animals found in the reserve or washed ashore (and therefore would be biased against smaller less easily identified animals; animals killed in burrows etc.) or observed to be active post-fire (i.e. many mobile species would have moved from the burnt areas). Many reptiles were observed to have survived the fire, as did wombats (possibly these groups seek refugee in burrows). Surprisingly, a number of more mobile animals (kangaroos; honeyeater birds) were killed by the fire.

Predation. High levels of predation of lizards (Fox 1978) and birds (Booker & Booker 1994) can occur post-fire. This may relate to less cover (protection) for species from predation. . Recent work by Lindenmayer *et al.* (2009) suggests sensitive bird species, in this case the Eastern Bristlebird, may be able to reoccupy burnt sites more rapidly where feral predator numbers have been reduced by baiting.

Succession. The post-fire environment provides altered conditions that favour different faunal groups. A small mammal post-fire succession is perhaps the best documented change in fauna with time since fire (Fox 1982). Birds also face a high altered environment and the diversity of nectivorous honeyeater will face an initial decrease in food often followed by a major post-fire flowering event whilst quail favour the open early post-fire environment (McFarland 1988). Recher (1981) also found that post-fire changes in nectar resource related to changes in the bird assemblage of a heathland.

Sydney coastal heaths have hosted many fire ecology studies, and the basic dynamics of these heaths are well understood. Fire frequency thresholds for this vegetation type were proposed by Bradstock *et al.* in 1995:

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

Recent analyses using plant species characteristics have reinforced these thresholds (Bradstock & Kenny 2003; Kenny *et al.* 2004) for Sydney Coastal Heathland. Watson (2006) summarised the concepts from which these guidelines are derived. This summary is as below.

Sensitivity to frequent fire

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

What makes these species vulnerable to frequent burning, where others are less sensitive? First, they have relatively long juvenile periods – they can take six to eight years to flower (Benson 1985). As these plants are obligate seeders, a second fire before seedlings germinating post-fire have matured sufficiently to set seed will leave no seeds to establish a new generation. Canopy storage increases this vulnerability; while soil-stored seed may survive through more than one fire ungerminated, and thus ready to burst forth after a second burn, serotinous species do not have this capacity. In addition seeds of these species, although winged, do not generally travel far from the parent plant (Hammill *et al.* 1998), limiting the potential to re-establish from unburnt patches after a second fire. Demographic studies show some Sydney sandstone resprouters are also likely to decline under repeated short interfire intervals, as fire tolerance can take many years to develop (Bradstock & Myerscough 1988; Bradstock 1990).

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

Sensitivity to infrequent fire

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

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

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

Variability within thresholds

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

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

A similar post-fire study by Tozer and Bradstock (2002) which like Keith and Bradstock (1994) took place in Royal National Park south of Sydney, also found many species were less abundant in patches where overstorey had previously been dense. This study added a dimension to the previous work by assessing competitive effects separately in wet and dry heath: the effect was most pronounced in dry heath. Additionally, this study found that some species were *more* abundant in overstorey patches, apparently because of suppression of the grasstree *Xanthorrhoea resinifera* which tended to dominated open patches. “We postulate that full diversity will be maintained when the density of overstorey shrubs fluctuates widely over a relatively short period of time,” the authors conclude. “This is most likely when fire frequency is highly variable” (Tozer & Bradstock 2002).

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

Fire intensity

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

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

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

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

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

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

Application to the Namoi CMA heathlands

Two recent studies into the fire ecology of Northern Montane Heaths have been undertaken (Clarke & Knox 2002; Williams & Clarke 2006). Clarke & Knox (2002) studied the post-fire response of shrubs across vegetation types on the New England Tablelands. Obligate seeders were most common in rocky outcrops (essentially a dry heath) which also recorded the highest post-fire seedling recruitment (75% of species). Wet heaths had a lower proportion of obligate seeders and species recruiting post-fire (31%) as did other vegetation types. These patterns could not be simply explained by tested models, suggesting that a number of factors contribute to these differences.

Williams & Clarke (2006) examined fire history and soil gradients (moisture and fertility) in relation to the distribution and assemblage structure of wet heath and sedgeland communities in Gibraltar Range National Park. Soil moisture status and fertility correlated with differences among these communities (i.e. Tall Heaths on drier sites; sedgelands on wetter sites). Time-since-fire strongly influenced heath structure and species richness (which declined with canopy closure over time). This confirmed that these heaths have a generally consistent time-since-fire pattern to that established for Sydney Coastal Heaths (and other heath systems across the globe). Fire frequency did not affect *shrub* species richness (but some individual species had decreased abundance with higher fire frequencies). An inter-fire interval of less than seven years was considered likely to reduce the abundance of some shrub species and it was concluded that these systems have similar fire response syndromes to other east coast heaths. This includes an overall reduction in floristic composition after ~ 15 years (although there was an overall lower species richness in these heaths compared to studied coastal heath systems). In rocky sites, however, species may be obligate seeders but this may be predominately driven by soil conditions and moisture status variability in some locations. Benwell (2007) concluded this to be the case for granite outcrops on the Tablelands. However, this may not be true in all locations because some species on sandstone outcrops are strong resprouters (Tierney 2003).

In sum, studies to date suggest that the fire regime guidelines developed for Sydney Coastal Heath are generally appropriate for Northern Montane Heaths. There are, however, likely to be different responses among the dry and wet heath communities to fire because of the large number of obligate seeder species in the dry heath types. There are also some species that may require specific management considerations in these rocky systems. At least one species from Gibralter Range heaths has been implicated as possibly at long-term risk from high fire frequency due to population shifts (Gross & Caddy 2006).

In the wetter systems, site dominance and gap closure may be important drivers of reduced diversity. Generalising from coastal systems to montane systems (with differing growing seasons / rates) should be done with caution. However, diversity declines from ~ 15 years are suggested by the studies available and maximum intervals in the more productive systems capable of gap closure should be somewhere in the range from ~ 15-30 years.

8. FIRE IN DRY SCLEROPHYLL FORESTS

Eco Logical (2008) have mapped 19 RVCs that fit within seven Keith classes of Dry Sclerophyll Forest (New England Dry Sclerophyll Forest; North-west Slopes Dry Sclerophyll Forest; Pilliga Outwash Dry Sclerophyll Forest; Northern Escarpment Dry Sclerophyll; Northern Tableland Dry Sclerophyll; Western Slopes Dry Sclerophyll; Yetman Dry Sclerophyll) for the vegetation of the Namoi CMA (Table 1). A number of these forest types intergrade with shrubby woodlands and the distinction among these structural types is likely to be arbitrary in many instances. The discussion below is of relevance to these vegetation units. However, structural variations at the site level will affect fire behavior (i.e. capacity to carry crown fires; the thickness of the understory may be greater with less canopy etc.), these structural variations need to be assessed at the site level.

The fire ecology literature for grasslands, grassy woodlands and heathlands (reviewed above) clearly demonstrates the importance of fire as a disturbance event that drives diversity patterns in these vegetation types. The fire ecology literature for dry sclerophyll forests is not as extensive as it is for those vegetation formations. However, like heathlands, dry scleropyll forests contain a large percentage of the plant species diversity in the sclerophyll shrub layer. In a regional study across vegetation types, Clarke *et al.* (2005) found that mean woody taxa species richness (excluding eucalypts – which mostly form the tree layer) was highest in dry sclerophyll forests (~ 20 species per 0.1 ha) which was significantly higher than heath, wet sclerophyll forests or grassy systems. These shrubs were dominated by species from the Proteaceae and Myrtaceae which were the families that dominated in heath systems and in some instances the same species occur in both systems. This suggests that some of the detailed fire ecology research for heathlands has some application to this shrub layer. However, it also clear that even at the species level, plants can display variability in their response to fire due to intra-species variability (Gill & Bradstock 1992). Secondly, Clarke *et al.* (2005) found significant differences in resprouting capacity among vegetation types which were associated with a model of resource / productivity among systems and that disturbance models were more explanatory for the low productivity systems. Therefore, even where there are floristic similarities, extrapolation among systems needs to be treated with some caution (see also Pausas *et al*. 2004).

Pilliga Outwash Dry Sclerophyll Forests

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

White Cypress Pine *Callitris glaucophylla*

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

1. Vegetation types over the Central West and adjoining regions (Namoi, Lachlan) formed a mosaic, as is commonly found in landscapes elsewhere. Much of the country was open woodland

with a grassy understorey, however there were also areas where shrubs and/or small trees grew in abundance (Benson & Redpath 1997; Croft *et al*. 1997; Allen 1998; Keith 2004).

2. Cypress Pine was a prominent component of many vegetation types in central New South Wales at the time of European settlement. For example in a study south of Tottenham of trees used as markers by 19th century surveyors, Martin (2005) found 19% of trees used were cypress pines; only box eucalypts were used more often. As trunks had to be fairly large to allow room for survey markings, Martin (2005) concluded that "large pines were a prominent component of the vegetation during the 19th century on both the flatter country and the ridges."

3. 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 & Redpath 1997; Martin 2005). Eucalypts appear to have been more abundant, at least in many areas, than cypress pine. Paull (2001), who used stump counts to estimate tree density in 1910 in 20 one hectare plots in the Pilliga West, found that while there was some variation in the relative numbers of the two canopy species, White Cypress Pine and Narrow-leaved Ironbark (*Eucalyptus crebra*), on average plots contained 25 large eucalypts to five large pines. Pine numbers never got above 9 per hectare, and the more ironbarks in a plot, the less pines. Lunt *et al*. (2006) also used stump counts to estimate pre-European tree density across a wide area of central NSW south from Nyngan. The estimated average density of trees with a diameter of 20 cm or more at the time of settlement was 39 per hectare. Seventy-eight percent of these trees were eucalypts, 21% pines. Thus *Callitris glaucophylla* was an important, but sub-dominant, component of the pre-European canopy in these stands. Allen (1995), however, recorded approximately equal numbers of pre-1870 pines and eucalypts in two minimally disturbed state forests south of Cobar, suggesting that White Cypress Pine was a more important component of the canopy in the west than in the east of its distribution in NSW.

4. 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 (e.g. Mitchell 1991; Martin 2005) 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) quotes Cunningham, who reported cypress pines "of various sizes and dimensions from seedlings, generally growing in clumps, to lofty trees of about 60 feet."

5. 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 & Lunt 2000). Strong documentary support for this contention comes from reports requested by the New South Wales Surveyor General in 1880. Land Commissioners in the Lachlan and Murrumbidgee districts described the difficulties caused by pine scrubs which had sprung up over the past 15 years. The problem was less acute around Narromine and Dubbo, although "scrub patches were believed to be increasing in density" (Mitchell 1991). 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 has been 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 & Jennings 1991; Allen 1998; Thompson & 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), 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 profusely.

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 focused 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 (Allen 1995). In the Northern Territory Bowman *et al*. (1988) found reduced growth of *Callitris intratropica* saplings under a eucalypt canopy. In the Snowy River Valley, Clayton-Green and Ashton (1990) found that White Cypress Pine density was lower under White Box (*Eucalyptus albens*) canopies than between them.

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

Changed grazing regimes. Grazing regimes have changed throughout the grassy forests and woodlands of the Central West. Where once kangaroos, wallabies and smaller herbivorous marsupials browsed, sheep, rabbits and cattle have progressively taken their place. There is general agreement that rabbits were a major factor in preventing pine regrowth in first half of the 20th century. Their arrival in the late 1800s is close to the time when the first wave of pine regeneration ceased, and their rapid demise from myxomatosis in the early 1950s matches the commencement of the second wave of pine (Mitchell 1991; Allen 1995). Exclosure experiments have also been conclusive (Lacey 1972). For example in an experiment in south Queensland in the 1950s 27 young pines established in a plot where rabbits and domestic stock had been excluded, while no seedlings at all survived in plots grazed by rabbits and sheep and only one where sheep were excluded but rabbits could get in (Johnston 1969).

The effects of stock grazing are less clear-cut. It has been suggested that grazing may encourage pine regeneration though its influence on the composition and abundance of ground layer plants. A decrease in herbaceous cover might encourage the establishment of *Callitris* seedlings through reduction in competition. A change in composition from deep-rooted perennial grasses and herbs to shallow-rooted annuals might provide more sub-soil water which could assist cypress seedlings to survive (Allen 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) hypothesizes, 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 (Allen 1998). Mitchell recorded Aboriginal fires in 1835 near the cypress pine forests studied by Allen (1995). In a review of the ecology of White Cypress Pine in Queensland, Harris and Lamb (2004) contend that "evidence, in the form of fire-scarred trunks, suggests that the majority of cypress forests incurred regular burning prior to being reserved as State Forests…." In the Northern Territory frequent low-intensity Aboriginal burns were, and in some places continue to be, a feature of the environment where *Callitris intratropica* grows (Bowman *et al.* 2001; Yibarbuk *et al*. 2001). It is likely that the frequency and intensity of fire varied across the landscape due to the effects of natural features (for example soil type would have affected grass growth which would in turn have affected fire). As well, Aboriginal management is unlikely to have been uniform.

European settlement probably decreased fire 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), 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; Allen 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 (Allen 1995). More recently, the potential for fire to help manage White Cypress Pine forests, and particularly its potential to help in reducing the density of young pines in locked stands, has been pointed out, as has the urgent need for research (Johnson & Jennings 1991; Date *et al*. 2002; Andrews 2003).

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.

Plants. It is often said that dense cypress regrowth suppresses grass cover (e.g. 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 (e.g. Belsky *et al*. 1993), many others have found lower herbaceous biomass under trees and/or shrubs (Engle *et al*. 1987; Archer 1990; Scanlan & Burrows 1990), or that ground layer productivity increases when trees and/or shrubs are removed (e.g. Walker *et al*. 1986; Harrington & 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 (1998) 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 & Eldridge 2005) found no association between canopy cover and cover of plant species, whether mid-storey species were included or excluded. There was, however, an east-west trend across the rainfall gradient sampled (530 to 215 mm a year) with significantly more understorey cover in higher rainfall sites. The findings of this study were unexpected, and the authors consider that differences in grazing history, and also lack of rainfall during sampling, may have affected the results.

In these forests, ground cover may develop 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.

A reduction in ground layer plants under cypress pine regrowth does not necessarily mean that species diversity is also reduced. 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). 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). In contrast, Bowman and Latz (1993) found many species, particularly herbs, only in plots where *Callitris glaucophylla* was dead or absent. And after extensive vegetation survey work through the Pilliga forest, Date *et al*. (2002) found the understorey in heavily logged and long unburnt areas with cypress regeneration supported a much lower range of understorey species than lightly logged patches which had burned every 20 years or so.

Fauna. Little research directly addresses the effects of changes in forest and woodland structure on fauna. The most cited work comes from Cape York, where granivorous birds have declined as a result of encroachment by Melaleuca species into open habitats (Crowley & 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. 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). Cameron (nd) points out that the majority of 20 bird species considered 'in decline' across the NSW wheat belt are ground or low shrub feeders unlikely to find suitable habitat in dense stands of regenerating pine.

Birds have also been surveyed in Back Creek State Forest near West Wyalong, in a patch of pure Belah and in adjacent White Cypress Pine-Box woodland with dense pine regrowth (Shelly 1998). 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 1998). 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 (1998) 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 ash-covered ground" (DEC 2005). The better-studied New Holland Mouse frequents areas in the early stages of recovery from fire where plant diversity is high and vegetation height is low (Tasker & 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 & Barrott 2001).

So do dense stands of Cypress Pine regrowth enhance or reduce faunal diversity? Andrews (2003) 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." Major *et al*. (2003) have found that even small patches of White Cypress Pine provide good habitat for arboreal insects. However "pure stands have relatively few fauna habitat resources compared to other native woodlands and forests" (Andrews 2003). As we have seen, thinning of closed dense stands may enhance habitat value for some plant and animal species. Measures which restore eucalypts and/or native perennial grasses whilst retaining some of the mid-story cover are likely to be even more effective.

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

Cypress Pine and fire. Research into the impacts of fire on White Cypress Pine is surprisingly limited. 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:

1. 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).

2. 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 (Allen 1998). In the Snowy River Valley many mature trees survived multiple fires in the early decades of European settlement (Pulsford *et al*. 1993).

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

3. 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 & Wilson 1988). Fourth, dense stands of *Callitris glaucophylla* may naturally discourage fire spread by reducing wind speed (Thompson & Eldridge 2005b).

4. Small *Callitris* plants are more vulnerable to being killed by fire than large plants (Bowman *et al*. 1988; Bowman & Panton 1993; Allen 1998; unpub. data D. Taylor QFRI 2007).

5. 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).

6. White Cypress Pine, like several other *Callitris* taxa, is a T species, able to recruit between fires. 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).

7. 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.

8. Stand density influences seed production considerably: the more open the stand, the more seed is produced (Lacey 1972).

9. 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.

10. Although seed of *Callitris* species does not generally appear to travel far from parent plants (Bowman *et al.* 1988; Bowman & Harris 1995 and references therein), seedlings of *C. glaucophylla* have been found up to 370 m from source trees (Lacey 1973).

11. *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.

12. The life span of White Cypress Pine is at least 200 years (Lacey 1973; Bowman & 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.

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, 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 favored eucalypts over pines. 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 (Allen 1998).

Managing for conservation. Managing Cypress Pine forests for conservation and/or restoration will require intelligence, experimentation, and a number of management tools. While fire is likely to be a useful component of the tool kit, other tools, particularly ones to restore and maintain canopy cover, will also be needed. Thinning alone is unlikely to stop pine recruitment nor restore the forests and woodlands in which it is found to an open structure. Ross *et al*. (2008) found that individual modeled *Callitris glaucophylla* saplings grew less when stands were thinned more, because gaps allowed young pine to recruit. In some thinning treatments, additional recruitment meant that stands approached or exceeded pre-thinning densities in as little as 5 to 10 years. Cameron (nd) reports secondary regrowth after thinning near Dubbo. Approaches for thinning pine are discussed in the Greening Australia landholder guide "Regrowth White Cypress Pine and Natural Resource Management" (Andrews 2003). Given the difficulties of managing secondary regrowth when thinning alone is used, the additional use of fire may be valuable. However, sparse grassy fuels may make it difficult to get a cool fire into dense cypress regrowth, whilst hotter fires which will travel through cypress foliage are likely to destroy or damage most trees, and may be difficult to control (Andrews 2003).

Given the complexity, outlined above, in the relationships between fire frequency, intensity and extent, canopy composition and density, and the effects of grazing, it is impossible to predict exactly what fire regimes would result in stable, healthy populations of White Cypress Pine and other native species. The challenge will be to find regimes which are sufficiently frequent and intense to prevent lock-up, but sufficiently infrequent and/or mild to keep *Callitris glaucophylla* in the landscape. Almost certainly the answer will differ with climate, soil type, nature and abundance of co-occurring species (grasses; shrubs; eucalypts), extent of degradation, and topography. Some general principles for management can, however, be put forward as a basis for experimentation by those wishing to enhance biodiversity values in White Cypress Pine forests and woodlands:

1. Management actions which encourage local eucalypt species back into the canopy should assist in bringing forests into balance.

2. Management actions which encourage deep-rooted native perennial grasses should do the same. Where Cypress Pine is dense, conservative grazing of stock is recommended to allow grasses to recover from defoliation and competition for resources. Native and feral herbivores may also need to be controlled (Andrews 2003).

3. Management actions which encourage both eucalypts and cypress pines to develop into large, old trees should be given a high priority. Where old trees exist, they should be preserved.

4. Fire clearly does have a role in White Cypress Pine forests and woodlands. Periodic fires will thin regeneration and should help establish dominance of smaller numbers of large cypress pines.

5. Fire may encourage eucalypt regeneration, and native grasses and shrubs (Hawkins 1966). After a wildfire in 1957 in Euglo South State Forest "Eucalypts and wattles re-emerged quickly in dense stands … and grass grew particularly well" (Allen 1998).

6. Low intensity fires will kill fewer pines than high intensity fires – and these fires will be easier to control. For this reason it may be best to begin with a cool fire – assuming, of course, that there is sufficient ground fuel to support a relatively low intensity fire. The next fire can always be of higher intensity if the survival rate of pines is higher than desired. Control of subsequent fires may be easier once pine thickets are broken up a bit through an initial cool burn.

7. Where dense stands of small Cypress Pines have established, there may be scope for manual thinning followed by burning once retained trash has dried out. This approach has been recommended in native pine forests in the United States, where similar dynamics operate, as a way to get fire into forests which are otherwise difficult to burn due to lack of grasses (Allen *et al*. 2002; Metlen & Fiedler 2006). This strategy has also been tried at the Zoo near Dubbo (Cameron nd).

Statewide fire frequency guidelines recommend inter-fire intervals of between five and 50 years for shrub-grass subformation dry sclerophyll forests. As recruitment of *Callitris glaucophylla* is episodic and juvenile periods are long, intervals at the lower end of this range should be sufficient to prevent lock-up, and may also discourage recruitment through suppression of seed production. Some longer intervals would allow some young pines to reach reproductive age and grow tall enough to survive fires. Fires may be most effective, and most necessary, after good rains have encouraged both grass growth and pine regeneration. This may also be the time when they are most feasible in terms of both fuel loads and economics.

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

Conclusion

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

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

Western Slopes Dry Sclerophyll Forests

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. Keith (2004) places Western Slopes Dry Sclerophyll Forests in the shrubby subformation of dry sclerophyll forests. Shrubs are mostly sclerophyll species including wattles (e.g. Spur-wing Wattle *Acacia triptera*, Streaked Wattle *Acacia lineata*), heaths (e.g. Urn Heath *Melichrus urceolatus*, Daphne Heath *Brachyloma daphnoides*), daisies (e.g. *Cassinia* species), and members of the Myrtaceae family (e.g. 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 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), “Black cypress pines occur on the most barren soils in these forests and to the east extend to high elevations on the edge of the tablelands. This species regenerates from seedlings in an episodic manner when its woody cones release their seeds *en masse* after fire onto an open bed of ash. White cypress pines, in contrast, extend into the semi-arid interior of New South Wales and are less dependent on fire for regeneration.” Ecologist Ross Bradstock suggests that there may be variation in the degree to which this species retains its fruits locked tight in cones, awaiting the trigger of fire: it may be a serotinous fire-dependent I species in some environments, but release seed and recruit between fires (i.e. act as a T species) in other situations.

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 & 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 & Beckers 2001; Kavanagh & 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

Date *et al.* (2002) studied birds, vegetation, logging and fire history for 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 this 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) consider that there may be "a homogenizing effect on habitat conditions of logging, combined with changed fire regimes and stock grazing, and the dense regeneration of cypress pine, casuarina, and some shrubs… That is, the disturbance history of the Pilliga may have imposed a pattern on the biota which masks the effects of topography, soils and vegetation." As already noted, these authors recommend management strategies to return mature trees to the canopy, and suggest diverse "intermediate fire regimes (2-25 year intervals)," coupled with an adaptive management approach, to increase habitat heterogeneity and benefit avian diversity (Date *et al.* 2002). These authors note the importance of encouraging the relatively open, grassy habitat favoured by woodlands birds as part of the habitat mix.

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

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

Conclusion

Western Slopes Dry Sclerophyll Forests share the ecological processes familiar from moderately productive dry sclerophyll forests elsewhere in NSW. Its sclerophyll shrub component 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 (see above), 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. 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.

Dry Sclerophyll Forests from the Tablelands

A number of studies in recent years have been carried out in Dry Sclerophyll Forests on the New England Tablelands (Clarke & Knox 2002; Clarke *et al*. 2005; Croft *et al*. 2006; Knox & Clarke 2006). The class of dry sclerophyll forests studied is not apparent in all of these studies (either New England Dry Sclerophyll Forest or Northern Escarpment Dry Sclerophyll Forest or perhaps both were included in some of these studies). These studies are briefly summarised below.

Clarke & Knox (2002) studied the post-fire responses and recruitment patterns of shrub species among ecosystems on the tablelands. The shrubby forests supported the highest diversity of shrub species and a relatively large number of these species were obligate seeders. These forests would include dry sclerophyll forests (of indeterminate class). Obligate seeders within these system include *Cassinia quinquefaria, Callitris oblonga ssp. parva, Leucopogon muticus, Melichrus erubescens, Styphelia triflora, Acacia betchei, Acacia burbidgeae, Acacia fimbriata, Acacia obtusifolia, Acacia torringtonennsis, Acacia triptera, Acacia ulicifolia, Acacia venulosa, Bossiaea rhombifolia, Daviesia ulicifolia, Dillwynia sericea, Dillwynia sieberi, Gompholobium huegelii, Pultenaea pycnocepala, Prostanthera scutellarioides, Grevillea beadleana, Grevillea scortechinii ssp. Sarmentosa, Hakea macrorrhyncha, Persoonia rufa, Discaria pubescens* and *Correa reflexa* (Clarke & Knox 2002).

Clarke *et al*. (2005) examined the woody non-eucalypt component among vegetation types on the Tablelands and found that the dry sclerophyll forests were the most species rich *but* with the second least proportion of resprouting species of the vegetation types studied (grassy forests; wet heath; wet sclerophyll forest; dry sclerophyll forest; rocky outcrop heathy eucalypt scrub). This implies that the minimum interval between fires will need to be sufficiently long to allow a range of obligate-seeding woody shrub species to persist (interestingly, however, fire frequency did not correlate strongly with resprouting ability when considered across all these vegetation types). Few species in the shrub layer exhibited bradyspory (had seed stored in fruits) *and* were seeders killed by fire. There was also a pattern for obligate seeders to produce more seedlings post-fire than resprouters: ~ 31 % of species in the dry sclerophyll forests exhibited post-fire seedling recruitment (which is consistent with Purdie's result for Southern Tablelands Dry Sclerophyll Forest – Purdie & Slatyer 1976).

Croft *et al*. (2006) studied the fire response of four rare shrub species from open forest (likely to be Northern Escarpment Dry Sclerophyll Forest) in Gibraltar Range National Park on the northern Tablelands (*Acacia barringtonensis*; *Grevillea rhizomatosa*; *Persoonia rufa*; *Telopea aspera*). Of these species *Persoonia rufa* was an obligate seeder (at least in the fire event observed) and the other three were resprouters. It was suggested that high fire frequency associated with hazard reduction could threaten these species. Thus, for example, a repeat fire killed 65% of resprouting *Grevillea rhizomatosa* plants. However, the interval between fires was three years (well outside the minimum inter-fire interval recommended for dry sclerophyll forests). Primary and secondary juvenile periods were not reported for these species. A minimum inter-fire interval of more than ten years was suggested for the persistence of these species.

Knox & Clarke (2006) is a study carried out in a specific dry sclerophyll forest class of Keith (2004) - Northern Escarpment Dry Sclerophyll Forest. The fire response of three Proteaceae shrubs (*Banksia spinulosa*; *Hakea laevipes*; *Petrophile canescens*) and the grasstree *Xanthorrhoea johnsonii* were studied. Survival ranged from 83 – 99% which occurred with inter-fire intervals from 7-22 years. Of the studied species only *Banksia spinulosa* had a survival rate related to inter-fire interval. This was attributed to loss of the most recently recruited cohort in the more frequently burnt sites for this species. This suggests that the higher fire frequency (7 years if applied without variation) could impact on this species over the long-term as it may restrict population recruitment. The result suggests that this general range of inter-fire intervals is appropriate for these species, however invariant shorter intervals may cause decline in *Banksia spinulosa*.

Watson and Wardell-Johnson (2004) studied fire frequency and time-since-fire effects on the plant composition and species abundance of a dry sclerophyll forest on the Tablelands just across the Queensland border. This study found that species composition varied significantly with both fire frequency and time-since-fire. The study concluded that variable fire regimes (3 to 25 years) were appropriate but that at least some longer intervals (of at least 15 years) should be included.

What is different in the tableland dry sclerophyll forest studies from some other studies in dry sclerophyll forests is the level of obligate seeders reported in some of the tableland studies. In Southern Dry Sclerophyll Forests, Purdie (1977a,b - see below for details) found almost all shrubs were resprouters, while in the New England tableland studies many of the shrubs were found to be obligate seeders (e.g. Clarke & Knox 2002). There may be differences in fire intensity or other differences in fire behaviour that contributed to this observation (Ooi *et al*. 2006b), however it would be precautionary to assume that there are a relatively high number of obligate seeder shrub species in these tableland dry sclerophyll forests.

Tasker and Bradstock (2006) studied the impacts of forest grazing (which includes high frequency winter burns) on species diversity and understorey structure on the edge of the New England escarpment. They found that an open simplified vegetation with a simplified or absent shrub layer was associated with grazing. Grazed sites were typically burnt at ~ 1-5 year intervals and this frequency would eliminate obligate seeder shrub layer with primary juvenile periods of moderate to long length. The practice of grazing and its associated frequent fire was estimated to occur in over ~ 500 000 ha of forest in north-eastern NSW. The findings of Tasker & Bradstock are consistent with the studies of Clarke & Knox (2002) and Clarke *et al*. (2005).

In sum, the tableland dry sclerophyll forests are species rich systems with a large percentage of the plant diversity in the woody shrub layer and a significant number of obligate seeder species in this layer. This focuses attention on managing fire frequency so that these obligate seeder shrubs can reach reproductive maturity between fire events.

Dry Sclerophyll Forests from outside of the region

Studies of dry sclerophyll forests outside of the Namoi CMA are broadly consistent in demonstrating that the sclerophyll shrub layer of these forests is strongly influenced by fire frequency.

Southern Tableland Dry Sclerophyll Forest. Purdie conducted some of Australia's first fire ecology studies in southern tablelands dry sclerophyll forests on the slopes of Canberra's Black Mountain (Purdie & Slatyer 1976; Purdie 1977a,b). Experimental fires were carried out in three sites, which had previously been burnt 9, 11 and 33+ years ago respectively. Purdie and Slatyer (1976) 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 & 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). The key findings from this work were that:

1. 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 & Slatyer 1976).

2. Most resprouting shrubs also produced seedlings. Some, such as *Daviesia mimosoides*, *Acacia buxifolia* and *Indigofera australis* were able to increase population numbers by root suckering – which was stimulated by fire. Flowering on resprouts occurred within two years of fire. Seedlings of resprouting shrub species were slower to reach maturity, with many not flowering by the end of the three year monitoring period (Purdie & Slatyer 1976).

3. Shrub and herb seedlings arose from seeds stored in the soil (Purdie 1977b).

4. All tree species resprouted and also produced seedlings (Purdie & Slatyer 1976). Tree seedlings developed in both burnt and unburnt plots (Purdie 1977b).

5. All the resprouters which didn't produce seedlings were geophytic monocots, a category which includes many orchids. All orchids flowered in the first year after fire. Several herbaceous species in this category – three Lomandras and *Dianella revoluta* – were able to increase through vegetative regrowth (Purdie & Slatyer 1976).

6. Almost all species showed much higher seed germination in burnt compared with unburnt areas (Purdie 1977a; Purdie & Slatyer 1976), and survival rates were also generally higher in burnt vegetation (Purdie 1977b).

7. 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 & Slatyer 1976).

8. Obligate seeders had more seedlings and higher seedling survival rates than resprouters (Purdie & Slatyer 1976).

9. 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 & Slatyer 1976).

10. 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).

11. 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).

12. "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 & Slatyer 1976).

Purdie and Slatyer (1976) conclude that changes in floristics as time-since-fire goes by in this vegetation type are merely changes in dominance due to differences in species growth rates and longevity, and, sometimes, reflect a move from visibility above-ground to presence only in the below-ground seedbank. 'Relay floristics', where additional species join the community as the period without disturbance lengthens, does not appear to characterise Southern Tableland Dry

Sclerophyll Forest on Black Mountain. This conclusion is consistent with what one might expect from a vegetation type that is, in Bond *et al.* (2003; 2005) terms, climate limited but fire modified.

Purdie's findings tell us something about the effects of fire on plants in general, and also specifically about dry sclerophyll forests. 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

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

Observation in long-unburnt sites from the western slopes also suggests that a small number of shrub species may be able to recruit substantially between fires in this environment. Candidates include *Styphelia triflora* (Pink Five Corners), *Cassinia arcuata* (Sifton Bush) and perhaps some other *Cassinia* and *Leptospermum* species. If this is the case, these shrubs are likely to be more abundant in long unburnt areas than where fire has occurred recently, and may become a more prominent component of the shrub layer as short-lived fire-cued shrub species die out. A survey in South Mullion reserve after an extensive fire-free period (up to 100 years) found large populations of Sifton Bush and Five Corners, both obligate seeders (Bower 2005). The two other shrubs which were abundant in this long-unburnt environment, *Monotoca scoparia* and *Melichrus urceolatus*, both resprout in most environments where their regeneration mechanisms have been observed (DEC 2002). These species may survive through long-term persistence of existing individuals. Surprisingly, few researchers have explored whether periodic fire plays a role in allowing long-lived resprouting shrubs to continue to thrive, although the lower levels of post-fire recovery in

Purdie's long unburnt plot (Purdie 1977a) suggests this may be the case. A range of other shrub species were also present in this long-unburnt reserve, suggesting that even fire-cued species have the ability to 'hold on' in the absence of fire, either through persistence of long-lived individuals or through some germination between fires. However as Purdie's work demonstrates, fire is likely to play an important role in maintaining robust populations of quite a few shrubs in Southern Tableland Dry Sclerophyll Forests. Although soil-stored seed can survive for many years, it will not remain viable indefinitely (Floyd 1976; Auld 1986; Auld *et al.* 2000; Moles *et al.* 2003).

Another classic study in Southern Tablelands Dry Sclerophyll Forest is that of Leigh and Holgate (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 Tableland Dry Sclerophyll Forest. Dominant trees were *Eucalyptus rossii*, *E. macrorhyncha* and *E. bridgesiana*. Shrubs included *Daviesia virgata*, *Dillwynia retorta*, *Gompholobium huegelii* and *Melichrus urceolatus*. *Poa sieberiana* dominated the ground layer. The main grazing animals at this site were grey kangaroos, with a few swamp wallabies and rabbits (Leigh & 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 & Jarman 1987). Where burnt areas are extensive, animals may spread out, reducing grazing pressure at any one point (Archibald & 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 & Trollope 2004). The need to consider the effects of post-fire grazing by native animals may be growing in some parts of the central tablelands as numbers of macropods and wombats build up (pers. comm. various land managers in the CW region, 2007).

The species characteristics observed by Purdie and Slatyer (1976) suggest that moderately frequent fire is unlikely to be problematic. All known obligate seeder species on Black Mountain flowered within three years of fire, and all resprouting species flowered, on resprouts, within two years. A seven year minimum is well above these juvenile periods. Seven year intervals, and even occasional intervals down to four or five years are thus unlikely to cause species to be lost from this community due to inability to establish a seedbank. Upper thresholds need to ensure that I species have opportunities to recruit before adult plants and seedbanks decline, and that competitive interactions that occur in the absence of fire are managed.

Competitive exclusion does not appear to be a major issue in Southern Tableland Dry Sclerophyll Forests. Net seedling recruitment in unburnt plots on Black Mountain, and particularly in the plot which had not had a fire for over 30 years, was not high (Purdie 1977b), and species which recruited preferentially in unburnt areas were often herbaceous (Purdie & Slatyer 1976). It appears that the shrub complement in this vegetation type does not contain many T species shrubs that increase when vegetation in some other climates is left unburnt for a long time. Rather, shrubs in Southern Tableland Dry Sclerophyll Forest are generally fire-cued and decrease in abundance as time-since-fire progresses. This vegetation type is thus likely to look more ‘open’ when long unburnt than when a burn has occurred within the last couple of decades. The grass *Joycea pallida* may recruit between fires (Purdie 1977b), and that may assist the development of a grassy understorey in the absence of fire. Whether competitive interactions between dominant grasses and smaller herbaceous species such as orchids exist in long unburnt patches of Southern Tableland Dry Sclerophyll Forest is not known.

The greater risk, at this end of the fire frequency spectrum, is that I species may decline under very long intervals. The recommended upper threshold of 30 years is several years below the maximum 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 & Slatyer 1976). The difference in density was particularly marked for small shrubs and herbs. Although lack of replication of the times-since-fire in this study means these observations should be treated with caution, they are congruent with other findings and observations. Recent inspection of one of Purdie's sites, now unburnt for 34 years, found shrubs were sparse, though cover of *Joycea pallida* was considerable (pers. obs. 2005; pers. comm. Margaret Kitchin, Environment ACT, 2005). Monitoring in unburnt plots over the years of Purdie's study showed declines in plant numbers (Purdie 1977a), and post-fire resprouting was less prolific on the previously long unburnt site than in sites with shorter inter-fire intervals (Purdie 1977a). If numbers of adult plants of many species decline with time-since-fire, after several decades regeneration will increasingly depend on the longevity of seedbanks – an unknown quantity. Seedling regeneration in Purdie's site with the long inter-fire interval was of the same order as that in the more recently burnt sites (Purdie 1977b), suggesting seedbanks at 30-35 years post-fire are still adequate. However this may not be the case four or five decades post-fire, nor for all species. Grazing before fire may exacerbate the problem by reducing seed input by adult plants, while post-fire grazing may eliminate seedlings and stress resprouting plants, particularly if burns are small. 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 inter-fire 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.

North Coast Dry Sclerophyll Forest. Fox *et al*. (1996) studied post-fire recovery (and also recovery from clearing and mining) in coastal sandy dry sclerophyll forests form Tomago (north of Newcastle). These forests had an overstorey of *Angophora costata*, *Eucalyptus gummifera*, *Eucalyptus haemastoma* and *Eucalyptus pilularis* and fit the description of Sydney Dune Dry Sclerophyll Forest. Post-fire the understorey density peaked at 11 years and then decreased for the ground layer (up to 0.5 m height) but the shrub layer (0.5 – 2m height) retained its density until the study conclusion (17 years).

Dry Sclerophyll Forest on Fraser Island. Spencer & Baxter (2006) studied fire frequencies in dry sclerophyll forests on Fraser Island. This study found that high fire frequencies were associated with reduced shrub (0.5-2m height) but increased ground layer cover. The frequently burnt sites were of lower diversity and in some parts of the study bracken (*Pteridium esculentum*) formed a dominant layer below the tree canopy. There was floristic variability among study locations, thus fire had an overriding effect on ground and shrub layers but floristics varied across the study.

Dry Sclerophyll Forest on the New South Wales South Coast. Penman *et al*. (2008) studied the effects of logging and prescribed fire on understorey species richness in dry sclerophyll forests on the south coast of New South Wales. Frequent fire was associated with increased species richness for the ground layer (up to 1m height) but a decline in species in the understorey above this height. This was found at the scale of study plots (0.01 ha). There was also, however, a reported general decline in species richness across the study area that was possibly related to a long term absence of fire (wildfire) at this scale (the area had not had wildfire for 15-29 years).

Application to the Namoi CMA dry sclerophyll forests

Kenny *et al.* (2004) recommend intervals between 7 and 30 years for dry sclerophyll shrub forests. This recommendation is likely to be *generally* appropriate for Dry Sclerophyll Forests of the Namoi CMA. The fire ecology of these dry sclerophyll forests can be broadly inferred from the studies that have occurred in dry sclerophyll forests reported above. However, these broad guidelines need to be varied and sensitively applied.

It is clear that the dry sclerophyll forests that support *Callitris* in the west of the CMA may require careful management that integrates fire with other management practices to produce desired biodiversity or grazing outcomes. The use of a broad frequency range is unlikely to deal with the complexities outlined for Pilliga Outwash Forests or Western Slopes Dry Sclerophyll forests. Specific guidelines have been provided for these forest types above. These are based on the available evidence. It is important to recognise that most of this evidence has some limitation and most studies have been carried out at a specific scale and that these studies are variable in the techniques and in analytical approaches. There is a need for more research and documentation of how fire can be applied to achieve specific outcomes.

There are reported differences in the number of obligate seeder species among the studies carried out in dry sclerophyll forests. The reported relatively high level of obligate shrub species in tableland dry sclerophyll forests suggests that fire frequency is an important consideration in managing these forests. Overall, studies in the tableland dry sclerophyll forests indicate that the shrub layer is responsive to fire and contains a large percentage of the plant diversity in these systems. It is also clear that, in general, high fire frequencies (intervals in the order of 1-3 years) have been associated with reduced shrub layers and dominance by species such as bracken (*Pteridium esculentum*) in dry sclerophyll forests. The shrub layer has important faunal values as a nectar resource and in providing structural complexity and protection, although recently burnt sites are also important for some species. Dry sclerophyll forests of the tablelands should have inter-fire frequencies that are within the 7-30 year threshold. Additionally, there are some species that may be disadvantaged by frequencies that are towards the shorter or longer end of this range and the general principle of frequency variability and patchiness in fires should be applied in the use of prescribed fire.

There are also a number of considerations that should also be attached to the suggested fire domain. These include:

1. There are a number of threatened fauna that are directly dependant upon a nectar rich shrub layer. Maintenance of this shrub layer within important habitat areas should be considered a priority.

2. In relation to the above point, fire patchiness within important faunal habitat areas is likely to be an important goal. This requires forward planning and an ability to adapt fire management plans as wildfire or arson events alter fire histories.

3. Conversely, there are threatened fauna that are favoured by frequently burnt sites. Sites that have a history of high fire frequency may constitute important habitat for some species. Therefore, fire histories should be considered carefully before any decision is made to adopt less frequent fire.

9. FIRE IN WET SCLEROPHYLL FORESTS

Eco Logical (2008) have mapped only one class of wet sclerophyll forest in the Namoi CMA (Northern Tableland Wet Sclerophyll Forest). This forest type has been mapped by Keith (2004) as extending from the tablelands along the Liverpool Range. 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 relation to fire. 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 these forests. McCarthy *et al*. (1999) modeled fire regimes in these 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 New South Wales coasts 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 the HCR CMA 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 sub-canopy 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 Proteaceae and Fabeaceae 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 (see *Fire in Rainforests* below).

The floodplains and coastal lowlands in northern New South Wales have been subject to high levels of clearing and fragmentation, including increasing urbanization. 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.

Two sub-formations of wet sclerophyll forests are recognised: grassy wet scleropyll forests and shrubby wet sclerophyll forests. HCCEMS (2008) recognise 15 vegetation types within the grassy sub-formation and 21 types within the shrubby sub-formation. Grassy wet sclerophyll forests are

“Tall forests dominated by straight-trunked eucalypts, with mixed grassy understories and sparse occurrences of shrubs with broad soft leaves” (RFS 2003). Sometimes called semi-mesic grassy forests, or wet sclerophyll forest (grassy subformation), they typically occur in coastal areas where rainfall is high and soils are moderately fertile. These are forests of high productivity and therefore competition dynamics are important. The tendency for grassy wet forests to succeed to rainforest in North Queensland (Unwin 1989; Harrington & Sanderson 1994; Russell-Smith & Stanton 2002) indicates that major changes can occur in at least some parts of the range of this broad forest type.

Shrubby subformation Wet Sclerophyll Forests are: “Tall forests dominated by straight-trunked eucalypts with dense understories of shrubs with broad soft leaves, ferns and herbs” (RFS 2003). They occur in high rainfall areas near the coast on relatively fertile soils, and have a multilayered understorey of mesic shrubs. Some consider these forests to be a successional stage between open forest and rainforest (Kenny *et al.* 2004), although others believe a stable understorey community can be kept in check by an intact eucalypt canopy (Florence 1996).

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 non-eucalypt 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. Much of this literature is relevant for the HCR CMA and is summarized below.

Wet Sclerophyll forests – grassy sub-formation

Birk and Bridges (1989) studied the effect of fire at two and four years or 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 on unburnt plots 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. 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 Tablelands in northern New South Wales. 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 sambuccafolia* 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 interfire 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 by far the greatest effect (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 ground-layer 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 the 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 few years, 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 interfire 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 here (Maynes 1977; NPWS 2002).

Studies in the upper Clarence River valley explored habitat preferences of macropods and found species showed different habitat preferences (Southwell & Jarman 1987). 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 non-eucalypt 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 New South Wales 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 helped 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 (WSF) types (Florence 1996). While the concept of occasional stand-replacing 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 WSF 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 Northern Rivers wet sclerophyll 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 Northern Rivers grassy WSF 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’ WSF 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 WSF overstorey, however they may play an important role in understorey dynamics. It is possible that both the DEC and the SEQ 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 WSF 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 life-span 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 fire – 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 Northern 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 SEQ (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.

Fire guidelines for south east Queensland exist which recommend a six year minimum for understorey fire (Watson 2001) for two reasons. The first was to ensure young eucalypts had time to reach fire tolerance, either through lignotuber development or through escaping out of the flame zone. The second was a fear that frequent fires would deplete soil nutrient levels, a concern based on a study by Guinto *et al.* (1998) which found that unburnt plots in a south-east Queensland forestry experiment had 40% more topsoil organic carbon and total nitrogen than plots burnt every second year, and 10% more than plots burnt on a four year cycle. Watson (2001a) interpreted these findings on nutrient levels as an indication that frequent fires were problematic. However in Western Slopes woodlands low nutrient levels favour native ground layer species over exotics (Prober *et al.* 2005), while high nutrient levels are linked with degradation (Prober *et al.* 2002b). The relationship between nutrient levels, fire and diversity in grassy WSF requires clarification. Perhaps nutrient levels vary naturally between states.

The need to limit the abundance of lantana and other exotic invaders may have a bearing on the fire regime used. Birk and Bridges (1989) reported an increase in lantana on their 20-year unburnt plots, and anecdotal evidence suggests this shrub is a major problem in some long unburnt Northern Rivers forests (Wardell-Johnson & Lynch 2005). The possibility of a link between eucalypt dieback and fire exclusion has also been suggested (Jurskis & Turner 2002), although researchers are a long way from untangling the web of factors involved (Wardell-Johnson & Lynch 2005). Recent work in the Richmond Range has confirmed an association between dieback severity, bell miner density, shrub cover, lantana abundance, and soil ammonium levels (Stone 2005). Logging, grazing, fragmentation, soil pathogens, and changes in hydrology and nutrient cycles may also be implicated (Wardell-Johnson & Lynch 2005).

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 Namoi CMA grassy wet sclerophyll forests

The work carried out on the north coast and northern tablelands is applicable to the grassy wet sclerophyll forests of the Namoi CMA. Although there are regional differences, many of the uderstorey species are also the same from the two areas. Hence the recommendations of Watson (2005) are adopted for grassy wet sclerophyll forests of the Namoi CMA. These are as below.

Summary recommendations

An interval range between 2 and 25 years is recommended. Some areas should be managed to retain an open, grassy environment: patchy low intensity fires at 2-7 year intervals are suggested. Other areas should be managed for a multilayered understorey: variable intervals between 8 and 25 years are suggested here. Occasional high intensity fire may be important for eucalypt regeneration.

10. FIRE IN RAINFORESTS

Eco Logical have mapped three rainforest classes for the Namoi CMA (Subtropical Rainforest; Dry Rainforest and Western Vine Thickets). Subtropical forests are restricted to the extreme eastern part of the CMA and although developed as a map unit for the CMA, Table 4 of Eco Logical (2008) records no area of Subtropical Rainforest. This may reflect the absence of this class from the CMA or that only a small unmappable portion of the CMA is comprised of this class. Similarly, Dry Rainforests and Western Vine Thickets are recorded as occupying very small areas / proportions of the CMA. None-the-less, these vegetation types are floristically and ecologically distinct from other vegetation types within the CMA and of high conservation significance.

Subtropical and temperate rainforests

Rainforests are generally regarded as fire sensitive and naturally subject to infrequent fires (Bowman 2000; Bradstock *et al*. 2002). Indeed the absence of fire for long time periods is a requirement for ecological succession from eucalypt forest to rainforest. This concept of a post-fire driven succession from eucalypt forest climaxing in rainforest is largely founded on the work of Noble & Slatyer (1980). The empirical basis for their concept of “vital attributes” (attributes that, for example, predict that rainforest species will slowly invade wet sclerophyll forest after fire) were observations of Victorian Mountain Ash – Warm Temperate rainforests. However, it is generally acknowledged that this type of successional sequence occurs across mesic coastal rainforests of eastern Australia (e.g. Unwin 1989; Harrington & Sanderson 1994; Harrington 1995), although rigorous studies of this phenomena are limited (see shrubby wet sclerophyll forests above).

These successional sequences drive rainforest / eucalypt forest boundaries that are highly dynamic in relation to altered fire regimes (Henderson & Wilks 1975). Hence, this model predicts that alteration of the fire regime (predominately the fire frequency) can lead to changes in the areal extent of temperate and subtropical rainforests. Over historical timeframes rainforest distributions therefore vary driven by changing fire regimes. These altered fire regimes are linked to climate and its variability and long-term climate changes (Bradstock *et al*. 2008). However, it is also considered that aboriginal burning practices were previously of a scale that they substantially contributed to the pattern of vegetation in forested landscapes in eastern Australia (Gill & Catling in Bradstock *et al*. 2002; Kershaw *et al*. in Bradstock *et al*. 2002). Post European settlement there is evidence that changes in fire regimes have also occurred leading to altered rainforest distributions and compositions. This can include an expansion of rainforest into eucalypt systems due to local changes in fire regimes (e.g. Harrington 1995).

The broad patterns of this fire response model by subtropical and temperate rainforests can provide useful system level management principles for fire for rainforests. However, in doing so, there needs to be caution in that the empirical basis for a broad application of this model is limited. Generally, rainforests need to be excluded from all prescribed burns since appropriate return intervals exceed the century scale (assuming the goal of maintaining the rainforest pre-burn distribution). However, these system level principles do not necessarily carry over to the management of individual species. The native mint species *P. askania*, for example, can occupy forest gaps within rainforests of the Central Coast and its recruitment may be enhanced by low intensity fire (Tierney 2006). The viability of populations of this species within rainforest systems is however unclear because modelling has demonstrated the importance of recruitment processes to long-term population trends, but there remains uncertainty about realised recruitment levels under field conditions (Tierney unpublished data). Potentially, fire intensity (not just frequency) may contribute to population outcomes for this species.

Weed invasion (e.g. *Lantana camara*) may significantly alter fire intensity on dry rainforest boundaries (see *Dry Rainforests* below), but there appear to be few definitive studies for subtropical and temperate rainforests. This is a question of interest for land managers who work with rainforests in fragmented / disturbed settings. It is clear that small rainforest remnants will be subject to significant alterations in microclimate and weed propagule intensity (Hobbs and Hopkins 1990) that may interact to produce fire events that are different in character and outcome to those of the past.

Conclusions - Subtropical and temperate rainforest

* Subtropical and temperate rainforests are spatially limited by fire.

* Fire should be excluded from sub-tropical and temperate rainforests if these are to be maintained on a given site.

* Alteration of fire regimes of subtropical and temperate rainforests can drive long-term changes in diversity and distribution of these forests. Alteration to these regimes can accompany fragmentation and development.

* There is some evidence that weed invasion may interact with fire to alter fire regimes in these forests but more research is required to understood how these processes interact in these systems.

Dry rainforests

Dry rainforests of the Namoi CMA occur on the slopes of the tablelands and on the Liverpoool Range. Typically these occur in steep often rocky sites on soils of medium to high nutrient status (Keith 2004). Other factors that determine the distribution of these forests may include aspect (Bennett 1989), soil moisture (Mackay 1993) and desiccation due to wind (Seifriz 2004). Because of the strong climatic gradient from east west across the CMA, dry rainforests of the east can generally be considered to support a different vegetation type. This is reflected in the mapping of Eco Logical (2008) which recognizes two dry rainforests RVCs (an eastern RVC which includes species such as Giant Stinging Tree – *Dendrocide excelsa* and an RVC more typical of the western tablelands and ranges).

Floristically these dry rainforests characteristically include canopy species regarded as fire sensitive (e.g. *Acmena smithii*; *Backhousia myrtifolia*; *Pittosporum undulatum*) as well as species that are tolerant of moderate fire frequencies (e.g. a *variety of Eucalyptus* species such as *Eucalyptus tereticornis* and *Eucalyptus melanophloia*). However, species regarded as fire sensitive are often very prominent, implicating fire as a significant factor in limiting the distribution of these dry rainforest systems.

Few detailed fire studies have been undertaken in dry rainforests. The most pertinent studies of fire in dry rainforests include those of Gentle & Duggin (1997) and Russell-Smith & Bowman (1992). Gentle and Duggin found that fire from adjacent open forests can move across the ecotone into dry rainforests of the NSW tablelands and alter the floristic composition of these systems. This was exacerbated where the weed species *Lantana camara* was present, potentially threatening the viability of the generally small patches of dry rainforest present. Peake (2006) reports that the weed *Asparagus asparagoides* (Bridal Creeper) also occurs as a serious weed in at least some of dry rainforests / thickets of the HCR CMA. This species may also interact with fire in these systems (in this case being sensitive to fire) so that fire could potentially be used as a management tool to control this species (e.g. Willis 2003).

The fire response of prominent species in the Namoi CMA dry rainforests was searched in DEC (2002). Species were selected from Group 21 (Rusty Fig Rock Gully Rainforest) of NSW National Parks & Wildlife Service (2002) for species likely to be within western dry rainforests. Species reported to be common in the dry rainforests studied by Gentle & Duggin (1997) were chosen as likely to be representative of common dry rainforest species on the tablelands (eastern dry rainforests). The fire response of these species is shown on Table 2. It can be inferred that a number of species in these thickets can establish without fire (including seeder species) and some may resprout but may be sensitive to short fire intervals. Some species (T species) might potentially be site dominants without fire and the fire ecology of some species is unknown. Thus, it is not possible to confidently prescribe a recommended fire interval on this incomplete dataset. However, fire intervals of < 20 years could eliminate species such as *Acmena smithii* from the dry rainforests whilst longer intervals (of some decades) may allow species such as *Pittosporum undulatum* to recruit and dominate sites. Despite this, no studies are available to determine how these species will function in dry rainforest sites (which are ecologically distinct).

Table 2. Important fire response characteristics of some prominent species in the Namoi CMA Dry Rainforests. The key (below) is from Kenny *et al*. (2004) and describes the symbols used in the table (a detailed explanation is in Noble & Slater 1980).

Species (E =eastern species; W = western species)	**Regeneration** (fire response) S = seed R = resprouts	**Establishment**	**Seed storage**	**VA Group**	**Minimum maturity (years)**	**Maximum maturity (years)**
Acacia diphylla E	-	-	-	-	-	-
Acmena smithii W & E	R (9% mortality) Eliminated by two fires within 20 years	T	transient	VDT	5	30
Alchornea ilicifolia E	R	-	-	V	3	5
Alectryon subcinerus E	Sr (may resprout if low intensity)	T	transient	DT	5	9
Alectryon forsythia W	-	-	-	-	-	-
Backhousia sciadophora E	-	-	-	-	-	-
Backhousia myrtifolia E	R	VCI	transient		-	-
Brachychiton populneus E	-	-	-	-	-	-
Breynia cernua W	-	-	-	-	-	-
Capparis arborea E	-	-	-	-	-	-
Cassinia leptocephala W	S	-	persistent (soil)	G	-	-
Clerodendrum tomentosum E	R	-	-	V	-	-
Coprosma quadrifida W	R & S	-	transient	D	-	-
Dendrocnide excelsa E	-	-	-	-	-	-
Diospyros australis E	-	-	-	-	-	-
Drypetes australasica E	-	-	-	-	-	-
Eucalyptus cypellocarpa W	R	I	canopy storage	VCI	5	10
Eucalyptus elliptica W	R	-	canopy storage	VC	-	-
Ficus rubiginosa W & E	R	I	-	VDI	3	-
Geijera salicifolia E	-	-	-	-	-	-
Mallotus philippensis E	R & S (usually killed)	-	-	-	-	-
Melicope micrococca W	-	-	-	-	-	-
Melia azedarach E	R	-	transient	VD	5	-
Pittosporum undulatum W & E	Sr	T	transient	DT	6	8
Sambucus grandichaudiana W	S	-	-	-	-	-

Key to the symbols used in Table 2 (from DEC 2002).

Persistance

Group:	Persistance attributes:
D	Propagules always available; ie widely dispersed
S	Propagule store long lived (and some seedbank remains after disturbance)
G	Propagule store exhausted (germinates or lost) at first disturbance
C	Propagule store short lived (transient or lost at senescence of plant)
V	Resprout but lose reproductively mature tissue; eg lignotuber shoots
U	Resprout and rapidly reproductively mature; eg pyrogenic flowerers
W	Adults resprout (& reproductively mature) but juveniles die

Establishment

Group:	Tolerance:	Establishment:
I	Intolerant of competion	Establish and grow only after disturbance
T	Tolerant of a wide range of site conditions	Establish and grow both after disturbance and in mature community; eg shade tolerant species
R	Require conditions of mature community	Establish only in mature community, not after disturbance

Transition. Persistance + establishment

Group:	Species type:	Disturbance regime resulting in local extinction:
1	DT, ST, VT	n/a
2	GT, CT	frequent (interval < m)
3	DI	n/a
4	SI	infrequent (interval > l+e)
5	GI	either (m > interval > l+e)
6	CI	either (m > interval > le)
7	VI	infrequent (interval > l)
8	DR, SR	n/a
9	GR, CR, VR	first disturbance
10a	dT, sT, gT, UT, WT	n/a
10b	dR, sR, gR, UR, WR	n/a
11	dI	n/a
12	sI	infrequent (interval > l+e)
13	gI	either (m > interval > l+e)
14	UI, WI	infrequent (interval > l)

Conclusions - Dry rainforest

It is currently not possible to provide guidelines for recommended fire regimes for the dry rainforests. It is likely that frequent fire (repeated short intervals of ~ 5 -7 years or less) is likely to cause the decline and elimination of some prominent species. Inter-fire intervals of several decades are more likely to sustain these species. A number of prominent species will also recruit in the absence of fire, but there is limited empirical evidence that these species become site dominants in these forests in the absence of fire. There is a need for specific research into the fire ecology of these forests.

Western Vine thickets

Western Vine Thickets are floristically similar to dry rainforests, but structurally more open and restricted to the dry hills and flats west of the New England plateau and slopes (Keith 2004). They extend north into Queensland and are alternatively termed Semi-evergreen Vine Thickets (SEVT). They can occur on relatively fertile volcanic soils and these have been subject to significant clearing and fragmentation. Eco Logical (2008) have mapped two RVCs (Ooline Forests and Semi-evergreen Vine Thickets) both from the Brigalow Belt South and Nandewar. Keith (2004) considers that thickets dominated by Ooline (*Cadellia pentastylis*) are floristically very similar to other thickets but differ in the dominance of this species and are structurally more open. Species such as Western Rosewood (*Alectryon oleifolius*) and also a few species widespread in eastern rainforests (e.g. Red Ash *Alphitonia excelsa*; Coffee Bush *Breynia oblongifolia*) occur but there are also many species restricted to west of the ranges.

Fire is considered a threat to these thickets, although there are few studies into their fire ecology. The most relevant studies are those of Fensham (1996) and McDonald (1996). Fensham studied “dry rainforest” across an extensive area of Queensland; some sites of this study would include vine thickets. Fire was found to be relatively scarce in these systems (absent or minor evidence in only ~ 8% of sites). This low occurrence was attributed to factors that have been discussed for Semi-arid Woodlands (i.e. low levels of grass fuel in surrounding grazing lands due to high grazing pressure; possible loss of aboriginal burning practices). Where fire does occur, McDonald (1996) reports that short fire intervals do cause significant decline in these thickets because there is slow recovery (i.e. low levels of growth) post-fire. In Northern Territory Vine Thickets, fire is reported to be more common (perhaps because of high fuel loads due to loss of aboriginal burning coupled with higher more regular rainfall) and is reported to cause severe damage to these systems (Russell-Smith & Bowman 1992). Fensham reports that the invasive Buffel Grass *Cenchrus ciliaris* has produced sufficient biomass to carry fire that killed rainforest trees in some areas of his study. Based on this, and similar observations from the Northern Territory (Panton 1993) it is considered that fuel loads adjacent to Vine Thickets can play a significant role in carrying fire into these thickets at intervals that are potentially too frequent to maintain the integrity of these systems. However, there appear to be no studies carried out in New South Wales that confirm these observations made elsewhere.

The fire response of prominent species in the Namoi CMA Western Vine thickets was searched in DEC (2002). Species were selected from Keith (2004).

Table 3. Important fire response characteristics of some prominent species in the Namoi CMA Western Vine Thickets. The key (above) is from Kenny *et al*. (2004) and describes the symbols used in the table (a detailed explanation is in Noble & Slater 1980).

Species (E =eastern species; W = western species)	**Regeneration** (fire response) S = seed R = resprouts	**Establishment**	**Seed storage**	**VA Group**	**Minimum maturity (years)**	**Maximum maturity (years)**
Abutilon oxycarpum	S (post fire recruitment)	I	Persistent (soil)	SI	-	-
Acacia decora	S	I	Persistent (soil)	SI	-	-
Alectryon oleifolius	R	R	-	VR	-	-
Alectryon subdentatus	-	-	-	-	-	-
Alphitonia excelsa	R	coloniser	Persistent (soil)	VSI	-	8
Alstonia constricta	-	-	-	-	-	-
Beyeria viscosa	S	-	-	C	-	-
Breynia oblongifolia	R (s)	Seedlings shade tolerant	Persistent (soil)	VGT	1	5
Cadellia pentastylis	R	-	-	W	-	-
Capparis lasiantha	-	-	-	-	-	-
Carissa ovata	-	-	-	-	-	-
Cassine australis var angustiflolia	-	-	-	-	-	-
Cassinia laevis	S	-	Persistent (soil)	G	-	-
Dodonea viscosa	S R	I	-	SI	3	-
Eremophila mitchelii	R	R	-	VR	-	-
Geijera parviflora	R	-	Persistent (soil)	V	-	-
Pittosporum angustiflolium	-	-	-	-	-	-
Callitris glaucophylla	See White Cypress Pine review in Dry Sclerophyll Forests (above)					
Casuraina cristata	S	-	Canopy storage	C	-	-
Eucalyptus melanophloia	-	-	-	-	-	-

Conclusions – Western Vine Thickets

The broad conclusions derived for Dry Rainforest are also applicable to the Western Vine thickets. There is limited data on how this vegetation responds to fire. The evidence available suggests that frequent fire may lead to a decline in these thickets. An examination of life history characteristics of prominent tree and shrub species (Table 3) suggests that there is limited data for many species in these thickets, this is not unusual. However, in many systems this lack of life history information is less critical because there are fire ecology studies which can provide an overall guide to how the system responds to fire. This is not the case for Vine Thickets. There is some suggestion in the literature that Vine Thickets recover slowly from fire. Thus, information on primary and secondary juvenile periods would be particularly useful data. This data is relatively simple to collect and would provide a good basis for improved fire ecology assessments. Presently, short fire intervals (of less than a ~ decade) should be considered “high risk”, whilst the potential of species such as White Cypress Pine or Belah to become site dominants in the absence of fire should be kept in mind.

11. SUMMARY

This review has addressed the fire ecology literature for the most widespread vegetation types in the Namoi CMA (Grasslands; Grassy Woodlands; Semi-arid Woodlands; Dry Sclerophyll Forests – Eco Logical 2008) as well as less widespread non–aquatic vegetation types for which significant literature exists (Wet Sclerophyll Forests; Heathlands; Rainforests). The vegetation types recognized in this review are those that fit within the formations and classes of Keith (2004) and their extent and finer scale delineation was referenced to Eco Logical (2008). EA Systems (2008) provide fire interval guidelines for the Namoi CMA. These broadly relate to the vegetation types recognized by Keith and Eco Logical. A general update of these guidelines is provided below (Table 4). However this necessarily oversimplifies the findings of the review and these comments are therefore qualified by the detail provided for each vegetation type in the text. It is also critical to understand that temporal and spatial variability is important: recommended fire frequencies do not represent an invariant regime and that fire patchiness will often be an important goal.

Specific limitations

1. The broad vegetation groups (formations) often represent vegetation types spread across a large spatial extent that encompass significant gradients in altitude, rainfall and soil productivity. The literature suggests that these gradients can correlate with differing fire response characteristics in the region. Clarke *et al.* (unpublished manuscript) reports generally longer primary juvenile periods in Tableland vegetations compared to that reported elsewhere and some variations among Tableland species in primary juvenile periods across from the drier western areas to eastern locations. The number of studies and the spatial extent of the vegetation formation relative to the location of these studies needs to be considered in relation to any specific location of interest. A logical extension of this review would be to map studies against environmental gradients and / or spatially map the study locations against the extent of a given formation.

2. The season of burn and the intensity of the fire need to be carefully considered, these variables may significantly affect the outcome of a given fire (e.g. Knox & Clarke 2006b). It may be that

these are particularly important considerations because planned fires will typically not be hot summer fires and will be therefore different to fires experienced in the past. The ecological impacts of varying intensity may be greatest at lower intensities because some low intensities may kill or reduce the mass of standing plants but not trigger seed recruitment (e.g. *Banksia marginata* – P. Clarke, pers. com.).

3. This review has not considered all vegetation types present in the CMA. Wetlands, for example, can be subject to fire and these can cause major changes to the vegetation of these systems (Fig 2). However, there is a paucity of literature to review spread across a variety of very different wetland types and very little capacity for general patterns to be discerned. In addition, there are likely to be a variety of complex interactions among fire events and hydrological processes that determine outcomes for a given fire. There is an urgent need to understand these interactions with respect to fire and a literature review will currently shed few insights into the general effects of fire in wetlands.

Fig. 2. A *Melaleuca* dominated wetland photographed four years post-fire. The Melaleuca trees no longer dominate the burnt area which is now dominated by weed species (*Typha orientalis*; *Ageratina adenophora*; *Zantedeschia aethiopica*). The developed catchment of this wetland has probably increased run-off and seeds released from the burnt *Melaleuca* trees may be unable to establish in this altered hydrological regime.

Table 4. An updated assessment of fire guidelines for the Namoi CMA (taken from Table 6 EA Systems 2008).

Broad Vegetation Type	**Minimum Interval**	**Maximum Interval**	**Notes**	**Revision**
Saline wetland	NA	NA	Fire should be avoided	
Freshwater wetland	6	NA	No maximum interval. Fire unnecessary.	
Forested wetland	7	35	Fire should be avoided in River red Gum woodlands.	
Rainforest	NA	NA	Fire should be avoided	Subtropical & Temperate. Fire should be avoided. However, these systems can (but will not always) expand into adjacent forests. The rainforest ecotone with wet sclerophyll forest can be a dynamic boundary and this fire driven dynamic can be ecologically important. Dry Rainforest. Insufficient data., research required. Western Vine Thickets. Insufficient data., research required
Alpine complex	NA	NA	Fire should be avoided	
Heathland	7	30		These intervals are generally appropriate.
Rocky outcrop	15	NA	No maximum interval. Fire unnecessary.	More research is required to determine the appropriateness of these intervals. Clarke & Knox (2000) indicate that there are a large number of obligate seeder species on some Tableland outcrops and a number of these may have short juvenile periods. The capacity for species to dominate sites over time also needs to be assessed.
Grasslands	2	10	Insufficient detail to give a definite maximum interval; available evidence indicates maximum intervals should be approximately 10 years. Some intervals greater than 7 years should be included in coastal areas.	*Themeda* / *Poa* dominated systems of the slopes generally are favored by frequent fire (5-10 years) and in high productivity systems diversity would be favored by the shorter intervals. Western Slopes graaslands dominated by *Austrostipa aristiglumis* on the Liverpool Plains are naturally treeless and should be managed to maintain gaps for forbs (no studies available). Semi-arid Floodplain Grasslands (Mitchell Grass) have potential fire events correlated to episodic high rainfall. The Chenopod shrub component recovers from fire within a few years, hence invasive shrub species with long juvenile periods might be managed with fire.
Grassy woodlands	5	40	Minimum interval of 10 years should woodland	The 5-40 interval is recommended by Kenny *et al*. (2004). Prober *et al*. (2007) recommend 5-8 years which would be

			woodland apply in the Southern Tablelands region.	applicable to RVCs 17 & 18 of Eco Logical (2008). This interval could be varied in RVC 19 to deal with White Cypress Pine dominance (as appropriate). New England Grassy Woodland (RVCs 15, 16 & 40) may have lower growth rates and a minimum of 8 years is recommended by Knox & Clarke (2004). Tableland Clay Grassy Woodland (RVC 12). Towards the upper range recommended by Prober *et al.* (2007) may be appropriate (fire may be less ecologically important on richer soils). RVCs 13 & 14 are poorly known. Adopt general grassy woodland recommendations (research is recommended).
Arid shrubland (Acacia subformation)	6	40	A minimum of 10-15 years should apply to communities containing *Callitris*. Fire should be avoided in Chenopod srublands.	
Semi-arid woodland (shrubby subformation)	6	40	Insufficient detail to give a definite maximum interval. Available data minimum intervals should be at least 5-10 years and maximum intervals approximately 40 years. No fire necessary for communities of subtropical and tropical origin (e.g. rosewoods, wilga).	The overriding factors for consideration are the interactions of fire with rainfall and grazing and the potential for fire to be utilized for management purposes given these constraints. There are also complex social and economic factors in using fire as a management tool in these woodlands. High fuel loads in the grass layer present an opportunity to disadvantage (in general) invasive shrub species, but this requires a sustained commitment (more than a single fire) and a possible short-term loss of grazing opportunity. Hodgkinson *et al.* (1984) recommend 10-50 years.
Semi-arid woodland (grassy subformation)	6	40	Insufficient detail to give a definite maximum interval. Available data minimum intervals should be at least 5-10 years and maximum intervals approximately 40 years.	As above.
Dry slcerophyll forest (shrub/grass subformation)	5	50		Generally 7-30 years is an appropriate fire interval range. In Pilliga Outwash Forests this range could be opportunistically varied in attempting to manage dominance of White Cypress Pine. Tableland Dry Sclerophyll Forests have a relatively high number of obligate seeder species and this implies

				sensitivity to fire. Knox & Clarke (2006) implies some risk for threatened shrub species in Northern Escarpment DSF with more frequent fire within this range.
Wet sclerophyll forest (shrubby formation)	25	NA	No maximum interval. Fire unnecessary. Crown fires should be avoided in the lower end of the interval range.	Not mapped by Eco Logical (2008).
Wet sclerophyll forest (grassy formation)	10	NA	No maximum interval. Fire unnecessary. Crown fires should be avoided in the lower end of the interval range.	One Class (Northern Tableland Wet Sclerophyll Forests) mapped by Eco Logical (2008). Crown Fires should be avoided (> 100 year intervals). Two regimes for ground fires should be maintained (in the landscape – not necessarily on all sites): 2- 5 years for open grassy sites and 15-20 years for less open sites (more towards the shrubby range). See *Rainforests* for issues about rainforest invasion with low frequency fire.

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