HOTSPOTS FIRE PROJECT

Fire and the Vegetation of the Hunter Central Rivers CMA

(Draft)

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Front page: A high intensity burn of the threatened species *Angophora inopina* in a coastal woodland of the Central Coast, D. Tierney 2000.

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 Hunter 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 HCR 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, 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 and Pate 1996; Benwell 1998; Bell 2001).

These categories are not invariant. Survival rates in the field for both resprouters and obligate seeders change with fire intensity (Morrison & 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; Clarke 2000), while smoke plays a role for other species (Dixon *et al.* 1995; Roche *et al.* 1998; Flematti *et al.* 2004). Some taxa respond best to a combination of these two fire-related cues (Morris 2000; Thomas *et al.* 2003).

A third strategy is to create seeds rapidly after a fire, through fire-cued flowering. *Xanthorrhoea* species are a well-known example of this phenomenon (Harrold 1979; McFarland 1990), however shrubs such as the Crinkle Bush *Lomatia silaifolia* (Denham & 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 1986a; Keith 1996; Hammill *et al.* 1998). However some wind and vertebrate-dispersed species do occur in these environments; examples include plants with fleshy fruits such as *Persoonia* species and some epacrids (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.* 2006).

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 1987b; 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 1995; Morrison *et al.* 1996). Species also vary in whether seedbanks are exhausted by a single fire (either through germination or destruction). Species which store seed in the canopy, and species whose soil-stored seeds are relatively permeable (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 and 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; Specht 1970; Prober 1996; Clarke 2003). Relatively frequent fire may thus be more appropriate in wet, warm, productive fire-prone systems than in those whose productivity is limited by poor soils, low rainfall or a short growing season.

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

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

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

- **Fire-limited**. These vegetation types will succeed to a different community in the absence of fire. In South Africa, these communities occur in higher rainfall areas, and include both 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 (see Noble & Grice 2002).

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 (York 1999, 2000a; Andrew *et al.* 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. Increasing fire frequency and intensity have been predicted to lead to major fire driven shifts in vegetation patterns. There are, however, a number 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 the direction and magnitude of fire regime changes are by no means certain. 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 HUNTER CENTRAL RIVERS CMA

Somerville (2009) describe the location, size, bioregions and the broad vegetation (formations) that occur in the Hunter Central Rivers CMA area (HCR CMA). This comprises approximately 4,000,000 ha and extends from the Central Coast to the lower north coast in the north to the top of the Hunter Valley in the west and encompasses 14 local government areas. Two IBRA bioregions are represented in the HCR CMA, the North Coast and Sydney Basin bioregions.

A vegetation classification scheme for the entire Hunter Rivers CMA was produced in 2009 (Somerville 2009). This classification scheme recognizes 210 vegetation units. These vegetation units are a fine scale delineation of the vegetation compared to the formations and classes which are recognized for New South Wales (Keith 2004, Table 1).

Table 1. The Keith formations and classes represented by the map units of Somerville (2009). Vegetation Classes shown as present in the CMA (+) were interpreted from descriptions and maps in Keith (2004) and vegetation descriptions from Somerville (2009). Vegetation classes absent from the CMA are shown as -----.

Keith Formation	Keith Class	Represented by HCCREMS 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	--------
	Northern 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

Somerville (2009) has included two grassland communities in the classification of the vegetation of the HCR CMA. These communities are a *Themeda australis* dominated coastal headland community (MU 162 – which would approximately equate to the Endangered Ecological Community Themeda grasslands on seacliffs and coastal headlands in the NSW north coast, Sydney basin and south east corner bioregions) and a *Austrostipa aristiglumis* dominated grassland on basalt from the Merriwa Plateau. A number of other unsampled grasslands may occur across the study area; up to 13 are included as potential grassland communities in the CMA (Somerville 2009). This includes a number of “derived” grasslands (i.e. grasslands that have been created by human activity). Despite an ongoing vegetation mapping program, some of these grasslands may remain unmapped, but are none-the-less ecologically significant. At least two threatened species known from coastal areas appear to utilize derived roadside *Themeda australis* grasslands as primary habitat (Bell & Driscoll 2004; Tierney 2008 – Fig. 1 & 2). Roadside grasslands are highly susceptible to disturbance and how fire interacts with these disturbances is an important issue.

Fig. 1. The previously assumed extinct *Diuris bracteata* occurs predominately in derived roadside *Themeda australis* grasslands on plateaus of the Central Coast.

Fig. 2. The threatened *Rutidosis heterogama* which favours the unmapped derived grasslands on roadsides and railway reserves of the Central Coast and Hunter Valley as well as grasslands on coastal seacliffs.

A substantial literature on the fire ecology of south-east Australian grasslands exists (e.g. Stuwe & Parsons in 1977; Morgan 1995 a,b; Lunt 1995; Morgan 1997; Lunt 1997a; Lunt & Morgan 1999;

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; Lunt 1994; Morgan 1996). These species then produce seed which is generally non-dormant (Willis & Groves 1991; Lunt 1995; Lunt 1996, Morgan 1998, 1998) and they colonise available gaps, provided post-fire conditions such as rainfall are favourable (Hitchmough *et al.* 1996; Morgan 1997; 1998; 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).

These 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 those 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 & 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 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 include 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 HCR CMA grasslands

Limited fire ecology research has been undertaken directly in the identified grasslands of the HCR CMA (a *Themeda australis* dominated coastal headland community and an *Austrostipa aristiglumis* dominated grassland on basalt from the Merriwa Plateau).

1. Themeda coastal headland grassland

The *Themeda* coastal headland grassland is a naturally fragmented restricted community comprising a distinct set of species to those of other studied *Themeda* grasslands (Adam *et al*. 1989). The ecology of this system is likely to differ substantially from the other well studied *Themeda* grasslands. These headland grasslands occur on raised and frequently steep slopes that are subject to strong salt laden winds and the *Themeda australis* present may be genetically distinct (www.environment.nsw.gov.au/determinations/ThemedaGrasslandSeaclffsEndSpListing.htm). Bond (1997) has previously noted that distinct forms of *Themeda* do exist. *Themeda* on seacliffs is generally more compact in habit and in these harsh conditions growth rates may be lower. Cohn (2004) studied the effects of slashing and burning on these grasslands and found significant differences in the effect of slashing and burning on populations of the forb *Thesium australe* which grows among the grass tussocks. In exposed headlands the applied treatments had no significant effect on the survival, density and vigour of this forb. However, in protected sites the treatments resulted in increased recruitment of this species. It was concluded that exposed sites may require less frequent fire events to maintain diversity in the ground layer than protected sites. Shrub invasion is considered a threat to this grassland (www.environment.nsw.gov.au/determinations/ThemedaGrasslandSeaclffsEndSpListing.htm) and Cohn cites suggested fire intervals of 2-10 years as appropriate to reduce the dominance of these shrubs. A number of threatened species (i.e. *Diuris praecox*; *Rutidosis heterogama*; *Thesium australe*; *Zieria prostrata*) occur in these grasslands and any managed fire events should consider the effects on these species. Generally, tussock growth rate will determine the rate of gap closure and this will depend on the inherent growth rate of the species and the environmental conditions (soil type; grazing pressure; rainfall etc.).

2. The *Austrostipa aristiglumis* dominated grassland on the Merriwa Plateau

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). The *Austrostipa aristiglumis* dominated grassland on the Merriwa Plateau is reported to be on basalt (high fertility at a high altitude). Therefore in this system the appropriate fire frequency may not necessarily be lower. Generally, in this system it would be recommended that gap closure 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.

Other grasslands

Small frost hollows of *Poa sieberiana* may occur in some areas across the HCR CMA. 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 (4 to 8 years) may favour *Themeda*.

Derived grasslands frequently occur on roadsides and are often maintained by roadside slashing. In at least one site, this managed grassland can be considered important habitat for threatened grassland species (Tierney 2008). Generally these grasslands will be maintained by regular slashing (not fire) and the time of slashing should be considered in relation to the life-cycle of these species.

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 and 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).

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

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 grass swards in the *Austrostipa aristiglumis* dominated grassland on the Merriwa Plateau and *Themeda* coastal headland grasslands under consideration here may be slower than those in the Victorian lowlands. This means that gaps between *Themeda* clumps may close more slowly, leaving more opportunities for forb growth and recruitment, and also that *Themeda* collapse may be delayed. Ecologist John Briggs, who has experimented with various interfire intervals in grasslands around Canberra, suggests 5-8 year intervals may be appropriate in southern tablelands grasslands. However detailed work on the endangered daisy *Rutidosis leptorrhynchoides* (Button Wrinklewort), which is found in grasslands in the Goulburn area (Eddy *et al.* 1998) has led 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."

Application to the HCR CMA grasslands

Fie frequencies towards the upper range of that recommended for grasslands in New South Wales would be recommended (5-10 years). This assumes a relatively slow growth rate of the *Themeda* swards which may be variable among sites. Where threatened species are present (likely to occur on seacliffs and some roadsides) detailed work should be undertaken to manage both disturbance and fire. This recommendation would be altered if *Themeda* growth rates are faster than expected.

5. FIRE IN GRASSY WOODLANDS

Somerville (2009) has mapped 14 grassy woodland types (map units 148 to 161) that equate to the grassy woodland formation of Keith (2004). This includes a number of communities with dominants or co-dominants in the upper strata of *Eucalyptus laevopinea* (Silvertop Stringybark - upper Hunter Valley to Liverpool Ranges), *Eucalyptus blakelyi* (Blakley's Red Gum - upper Hunter Valley), *Eucalyptus melliodora* and *Eucalyptus albens* (Yellow Box and White Box) from the Merriwa Plateau, *Eucalyptus pauciflora* (Snow Gums) from elevated sites from the Barrington Tops area and *Allocasuarina leumannii* from the central Hunter Valley.

A considerable body of fire ecology research now exists for the Grassy Box Woodlands of the western slopes and for 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 Grassy Box Woodlands of the western slopes directly relates to the grassy woodlands of the HCR CMA and research from Cumberland Plain Woodland is also highly informative for the HCR CMA woodlands.

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. The discontinuity of the Great Dividing Range in the upper Hunter Valley supported a large area of this vegetation that connected to the western slopes woodlands. Remnants of 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 and Thiele 1993; Prober & Thiele 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 the dominant tree *Eucalyptus laevopinea* had an association with *Poa sieberiana* and *Microlaena stipoides*. 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 for resprouting 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 2006a).

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 HCR CMA grassy woodlands

1. *Eucalyptus melliodora* and *Eucalyptus albens* (Yellow Box and White Box) from the Merriwa Plateau.

These woodlands are part of the grassy box woodlands of the western slopes that 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 for these woodlands. These guidelines would be modified by grazing which may decrease the dominance of *Themeda* over forb species and intervals of up to 15 years may be appropriate in some circumstances.

2. *Eucalyptus laevopinea* woodlands (Silvertop Stringybark - upper Hunter Valley to Liverpool Ranges).

Somerville (2009) have mapped a number of woodlands from the upper Hunter Valley to the Liverpool Ranges that include *Eucalyptus laevopinea* as a prominent tree, shrubs such as *Bursaria spinosa*, *Cassinia quiquefaria* and *Pittosporum undulatum* and a ground layer typically containing *Microlaena stipoides*, *Echinopogon ovatus*, *Aristida ramosa* and *Poa sieberiana* (map units 148-151). The eucalypts present in these woodlands are likely to be resistant to the fire that this vegetation would generally support (ground fire, even if reasonably frequent). This tree cover interacts with the grass layer so that summer growing *Aristida ramosa* is favoured in open sites while *Microlaena stipoides* and *Poa siberiana* are favored by tree cover (Gibbs *et al.* 1999). This may have as strong an influence on ground layer dynamics as any variations to fire regime. However, it is suggested that the productivity of these grasses in these woodlands is unlikely to be greater than that found for *Themeda australis* in other woodlands, although the ground layer dynamics in relation to fire have not been determined. Fire intervals of 5-10 years are likely to provide opportunity for forbs to persist in the ground layer.

The shrubs *Bursaria spinosa*, *Cassinia quinquefaria* and *Pittosporum undulatum* may potentially grow to form a dominant thicket that reduces the ground layer diversity of these woodlands. The low abundance of obligate seeder shrubs in low fire frequency sites may be due 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. However, 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 Cumberland Plain obligate seeders. If this is so, the absence of these species in long unburnt remnants may partially reflect lack of fire-related recruitment opportunities. Conversely *Pittosporum undulatum* is a shrub species found in these woodlands which has been found to be dominant with low fire frequency elsewhere (McDonald *et al*. 2003). This species is killed by fire and therefore can be controlled with a fire frequency less than its juvenile period. Whilst its growth rate and juvenile period in woodlands is not recorded, observations elsewhere suggest that it should require ~ 5 years to reach maturity (D. Tierney – pers. obs.). Similarly, *Cassinia arcuata* (Sifton Bush) may partly be regulated by fire in these woodlands (Semple 1990).

It is also true that 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).

Fire frequencies within the range of 5-10 years are likely to be generally appropriate for these woodlands. However, Knox and Clarke (2004) report that shrub species of the tablelands appear to have long juvenile periods relative to studies from other regions and to preserve the shrub layer recommend intervals of greater than 8 years. It is suggested that intervals > 8 years may be appropriate for sites where the shrub layer needs to be maintained.

3. *Eucalyptus blakelyi* (Blakley's Red Gum - upper Hunter Valley)

Eucalyptus blakelyi has been the subject of studies by Li *et al.* (2003) which showed increased seedling survival growth rates with fire. Clarke (2002) 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 (like the *Eucalyptus laevopinea* woodlands**)** also have eucalypts that are likely to be resistant to fire intervals of more than about 5 years. There is however a shrubby understorey of *Melaleuca* and *Acacia* reported in these woodlands that would generally be favoured by fire frequencies greater than ~ 8 years (Knox & Clarke 2004) up to about 15 years. Although it is not possible to be precise, fire regimes within these ranges are likely to be favourable. The relatively more dominant shrub layer reported for these woodlands should be protected by fire interval > 8 years.

4. *Eucalyptus pauciflora* (Snow Gums) in elevated sites from the Barrington Tops area.

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* have also been found to resprout and flower in response to fire within 12 months (Wahren *et al*. 1999). However, the effects of fire on the shrub layer are not well known. 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). Oligate seeders may well be comparatively rare in this vegetation, however, this remains undetermined. Thus there is some uncertainty about fire intervals for this vegetation.

5. *Allocasuarina leumannii* woodland from the central Hunter Valley.

The *Allocasuarina leumannii* woodland from the central Hunter Valley belongs to the Coastal Valley Grassy Woodland class. However, this woodland is recorded as having a dominant layer of *Allocasuarina leumannii* with few other shrubs over a predominately grassy ground layer with scattered emergent *Eucalyptus creba* and *Eucalyptus moluccana* (Somerville 2009). A key consideration for this vegetation type is the fire dynamics of the *Allocasuarina leumannii* shrub layer. The importance of fire in regulating the shrub layer is emphasized by the study of Watson (2005) in Cumberland Plain Woodland of *Bursaria spinosa* (see above summary). Watson concluded that *Bursaria spinosa* has the capacity to dominant a site in the absence of fire.

Allocasuarina leumannii has been reported to be a species that can form dense thickets in the absence of fire (A. Deane pers. comm. 2005). Although this is an observation from the central west of New South Wales, it is likely to be also relevant to the HCR CMA. The question remains open as to the extent to which this vegetation represents a relatively recent change in fire regime in recent decades. Fire interval of > 30 years are likely to retain large areas of *Allocasuarina leumannii* understory. More frequent fire may reduce the dominance of this species, increase the diversity of grasses and forbs and encourage recruitment of any hard seeded species that have persisted in the seedbank.

6. FIRE IN HEATHLANDS

Somerville (2009) has mapped 12 heathland and 5 heathy woodland types in the HCR CMA region. These cover three heath classes (Coastal Headland, Wallum Heath and Sydney Coastal Heath) of Keith (2004). There is a rich fire ecology literature for heathlands: fire prone heaths of South Africa (fynbos), North America (chaparral) and Australia have been subject to extensive research. This work has demonstrated consistent patterns in the fire ecology of these heathlands at this global scale (e.g. Keeley & Bond 1997). Locally, a number of classic Australian fire ecology studies have been undertaken on heathland species that occur in the HCR CMA heaths (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. 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 2006a) and may weaken them and lead to their loss. There appears to be some variation in the proportion of resprouters among heaths within the HCR CMA region. 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 influence its species diversity.

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 highly 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 resources related to changes in the bird assemblage of a heathland.

Application to the HCR CMA heathlands

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

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

Recent analyses using plant species characteristics has reinforced these thresholds (Bradstock & Kenny 2003; Kenny *et al.* 2004). They apply directly to the Sydney coastal heathland classes mapped in the HCR CMA. 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 & 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 (1999) and Morris (2000) found smoke increased germination of several *Grevillea* species found on sandstone, with some species also responding positively to heat. Thomas *et al.* (2003)

documented a range of responses to heat and smoke cues amongst a group of Sydney sandstone species, including interactions between cues. Relatively moderate heat shock produced maximum germination in several species, leading these authors to conclude that “low-intensity fire or patches within fire” may be important for recruitment of some plants.

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

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

Somerville (2009) map units 164 and 172 are examples of Sydney coastal heath mapped in the HCR CMA.

Other heathlands

The guidelines for Sydney coastal heath are based on common widespread species that potentially occur in heathlands outside of the Sydney area. Do they apply to the two other classes of heathland mapped for the HCR CMA (headland and wallum heathlands?).

Wallum heath

Somerville (2009) map units 168-171 represent heathlands that occur at least partly on sand dunes and barriers of the coastal strip. These include true wallum heath (e.g. unit 168) and a range of variants that probably reflect different soil, exposure and fire histories. Myerscough *et al*. (1995) report that very few non-resprouters occur in the wallum heathlands of Myall Lakes National Park, suggesting that this system may therefore be tolerant (generally) of more frequent fire than Sydney coastal heathland. However, a single study will not include all species that occur within these heaths. In a later study which reported time-since-fire effects, Myerscough and Clarke (2007) found that up to 25% of cover was provided by obligate seeder species 10 years post-fire and that an interval of 3.75 years decreased the cover of these species. Griffith *et al*. (2004) report that the seeder species *Banksia ericifolia macrantha* is a significant component of wallum heath at Crowdy Bay National Park. Another species in this heath which is of interest is *Baeckea imbricata* which may be sensitive to fire intensity (Thomas *et al*. 2003). Many of the species present in Griffith's study are in fact common species from Sydney coastal heathlands.

The unit descriptions for map units 168-171 include frequent references across these units to species that are both strong resprouters in the region (e.g. *Banksia oblongifolia* – D. Tierney pers. obs.) and species that are favoured by frequent fire (e.g. *Blandfordia grandiflora* - D. Tierney pers. obs.) as well as the potentially dominant *Banksia ericifolia macrantha* which has been reported to be in at least some of these heaths. It is therefore recommended that the guidelines for Sydney coastal heath be adopted for wallum heath in HCR CMA as these are representative of the major life-history types for which the Sydney coastal heath guidelines were designed.

Headland heaths

Somerville (2009) map units 166 and 167 are examples of headland heaths from the HCR CMA. These heathlands (and also map unit 165) contain the potentially dominant *Allocasuarina distyla*. This species is an obligate seeder that can dominate a site subject to a low fire frequency and reduce site diversity (see discussion of Sydney coastal heath above). Headland heaths also contain the strongly resprouting *Banksia oblongifolia* as well as species which will be excluded by the long lived site dominants but which recruit strongly with fire (e.g. *Actinotus helianthi*). This again supports that the Sydney coastal heath guidelines are broadly applicable to these headland heathlands.

A particular issue for headland vegetation is that the heaths and *Themeda* grasslands that occur on these headlands (see Grasslands above) are often both patchy and interspersed. Managing fire on these headlands is also challenged by the presence of threatened species and highly invasive weeds. These headlands present a set of challenging taxing problems for management and no one vegetation type or issue can be considered in isolation.

Fig 3. Post-fire regeneration in a wet coastal heath.

7. FIRE IN DRY SCLEROPHYLL FORESTS

Somerville (2009) has recognized 14 dry sclerophyll forest types in the classification of the vegetation of the HCR CMA (Table 1). This includes both communities with a shrub/grass understorey (many units from units 58 – 83) as well as shrubby understorey types (many units from units 84 – 147). 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. probability of carrying 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. 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) in dry sclerophyll forests, at about 20 species per 0.1 ha, was significantly higher than in heath, wet sclerophyll forest 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 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).

Southern Tablelands 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). A small amount of this vegetation class occurs in the HCR CMA. This careful work demonstrated the varied responses of plant species to fire, and is still highly relevant today.

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 and 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 of New South Wales 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).

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

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

Post-fire grazing in Southern Tablelands Dry Sclerophyll Forest

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

Conclusion

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

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

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 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 interfire interval involved in Purdie’s study (33 years). Shrubs in this long unburnt site were senescent prior to the experimental fire, and total plant density, at 8.0 plants/m^2, was considerably lower than in the more recently burnt sites, which averaged 30.4 and 19.2 plants/m^2 respectively (Purdie & Slatyer 1976). The difference in density was particularly marked for small shrubs and herbs. Although lack of replication of the times-since-fire in this study means these observations should be treated with caution, they are congruent with other findings and observations. Recent inspection of one of Purdie’s sites, now unburnt for 34 years, found shrubs were sparse, though cover of *Joycea pallida* was considerable (pers. obs. 2005; pers. comm. Margaret Kitchin, Environment ACT, 2005). Monitoring in unburnt plots over the years of Purdie’s study showed declines in plant numbers (Purdie 1977a), and post-fire resprouting was less prolific on the previously long unburnt site than in sites with shorter interfire intervals (Purdie 1977a). If numbers of adult plants of many species decline with time-since-fire, after several decades regeneration will increasingly depend on the longevity of seedbanks – an unknown quantity. Seedling regeneration in Purdie's site with the long interfire interval was of the same order as that in the more recently burnt sites (Purdie 1977b), suggesting seedbanks at 30-35 years post-fire are still adequate. However this may not be the case four or five decades post-fire, nor for all species. Grazing before fire may exacerbate the problem by reducing seed input by adult plants, while post-fire grazing may eliminate seedlings and stress resprouting plants, particularly if burns are small. In very long unburnt sites with few shrubs it may be necessary to supplement seedbanks with seed from nearby more frequently burnt areas if the aim is to restore the pre-existing species complement.

An additional issue for fire frequency concerns how long the seedlings of the many resprouting species take to become fire-tolerant. We do know that they grow slowly relative to their obligate seeder counterparts, and Purdie (1977a) suggests that, from her observations, even after a 9 to 11 year interfire interval young plants may be vulnerable. Intervals above 15 years within a variable regime of intervals between seven and 30 years should allow opportunities for these seedlings to fully develop, while still allowing species which produce most of their seed-crop in the early post-fire years the opportunity to build up population numbers through some short intervals now and then.

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 White Cypress Pine (*Callitris glaucophylla*) and Black Cypress Pine (*C. endlicheri*) occur in these woodlands. Herbs and grasses grow in the ground layer. These forests have been, and in some cases continue to be, a major source of timber (Metcalfe *et al.* 2003; Keith 2004).

Western Slopes Dry Sclerophyll Forests are prone to wildfire. In 2006 major fires occurred in both the Pilliga and Goonoo State Forests, while much of the eastern Pilliga also burnt in 1997 (Kavanagh & 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." 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. Whilst many faunal ecologists emphasise the impact of loss of treed landscapes on fauna, Date *et al.* 2002 note the importance of encouraging the relatively open, grassy habitat favoured by woodlands birds as part of the habitat mix.

Shelly (1998) 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).

Black Cypress Pine, *Callitris endlicheri*

Both Black Cypress Pine (*Callitris endlicheri)* and White Cypress Pine (*C. glaucophylla*) are found in Western Slopes Dry Sclerophyll Forests. According to Binns and Beckers (2001), the two are negatively associated, with Black Cypress more common in heathy areas, while White Cypress is more abundant in grassy vegetation.

Black Cypress Pine, like *C. glaucophylla,* is an obligate seeder which has the ability to survive fire when scorch is not too severe (DEC 2002). Whether Black Cypress recruits primarily after fire, or between fires as is the case with White Cypress, is unclear. According to Keith (2004), "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.

Conclusion

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

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

New England Dry Sclerophyll Forests

Clarke & Knox (2002) and Clarke *et al.* (2005) carried out fire ecology research on the New England Tablelands which included New England Dry Sclerophyll Forest. Clarke *et al.* (2005) examined the woody non-eucalypt component among vegetation types 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). What is different from Purdie's studies is that Purdie found almost all shrubs were resprouters, while in these New England Dry Sclerophyll Forests many of the shrubs were found to be obligate seeders (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 number of obligate seeder shrub species in these New England Dry Sclerophyll Forests.

In sum, the New England 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. Obligate seeders within this 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).

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 about 1-5 year intervals and this frequency would eliminate the obligate seeder shrub layer. This practice 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).

Sydney Dune 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 Forests from outside of the region

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

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).

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 from south-east Queensland. 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 in the regime for this tableland dry sclerophyll forest.

Application to the HCR CMA dry sclerophyll forests

A large diversity of dry sclerophyll forest types are recognised in the HCR CMA. The fire ecology of these dry sclerophyll forests can be broadly inferred from the few studies that have occurred in dry sclerophyll forests reported above. However, these studies do not cover many of the specific classes of dry sclerophyll forests present. It is also important to recognise that most of these studies have limitations and have been carried out at specific scales and are variable in the techniques and in analytical approaches.

There are reported differences in the number of obligate seeder species among the studies carried out in dry sclerophyll forests. However, overall, it is clear that the shrub layer is very responsive to fire, contains a large percentage of the plant diversity that is affected by fire and in the absence of fire that diversity declines (at least as standing plants and possibly some of the diversity in the seedbank). It is also clear that very 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 some studies. 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.

Kenny *et al.* (2004) recommend intervals between 7 and 30 years for dry sclerophyll shrub forests. This recommendation is reinforced by this review of the literature as broadly applicable. There are a number of considerations that should also be attached to this suggested fire domain. These include:

* There are a number of threatened fauna (Pygmy Possum; Squirrel Glider) that are directly dependant upon a nectar rich shrub layer. Maintenance of this shrub layer within important habitat areas should be considered a priority.

* 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.

* Conversely, there are threatened fauna that are favoured by frequently burnt sites (e.g. Hastings River Mouse). 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.

* There is significant variability in the shrub layer of dry sclerophyll forests within the HCR CMA. Somerville (2009), for example, reports the presence of map unit 59 which includes a number of small trees or shrubs that are non-sclerophyllous and commonly associated with dry rainforests. This is an example where lower fire frequencies may be considered appropriate.

Fig. 4. Dry sclerophyll forest, Towarri National Park, Upper Hunter Valley.

8. FIRE IN WET SCLEROPHYLL FORESTS

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 Mountain 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 Mountain Ash 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, frequency and season; none-the-less it is still considered a major determinant of successional change (e.g. Harrington & Sanderson 1994).

Like Mountain Ash, 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 interval 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. In some situations, 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 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 soil characteristics (Whelan 1995). These species can therefore 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 the 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 return intervals of 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 whose seeds were 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 appear[ed] to result 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 guideline for this vegetation type covers a wide range of intervals, and refers 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 once created 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 the DEC guidelines pertaining to the forest type under discussion represent something of an uneasy compromise between the fire requirements of overstorey and understorey. This compromise position could fail to deliver either sufficiently frequent low intensity fire to maintain a range of understorey habitats, nor sufficiently infrequent high intensity fire for eucalypt regeneration. The concept of a 'two tier' regime is proposed as a way out of this dilemma.

Uneasy compromises may also be inevitable in any state-wide guideline for grassy wet sclerophyll forests. As noted above, there are likely to be major differences across NSW in the nature of the relationship between 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 the fire frequency guidelines currently in use in south-east Queensland (6+ years – Watson 2001), 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 tomaintain open understoreys will undoubtedly disadvantage some plant species, the number disadvantaged may not be high.

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 HCR CMA grassy wet sclerophyll forests

The work carried out on the north coast and northern tablelands is very applicable to the grassy wet sclerophyll forests of the HCR CMA. Many of the rare and threatened taxa (e.g. Parma Wallaby; New Holland Mouse and the Hastings River Mouse) that favour open more frequently burnt sites in this vegetation type are also recorded from the HCR 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 HCR CMA. These are as below.

Summary recommendations

Grassy wet sclerophyll forests of the coast. An interval range between 2 and 20 years is recommended. Some areas should be managed to retain an open, grassy environment: patchy low intensity fires at 2-5 year intervals are suggested. Other areas should be managed for a multilayered understorey: variable intervals between 5 and 20 years are suggested here. Occasional high intensity fire may be important for eucalypt regeneration.

Grassy wet sclerophyll forests of the upper Hunter or from high elevations. 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.

Wet sclerophyll forests - shrubby sub-formation

In the 1960s and 70s, Alex Floyd conducted some of the earliest fire ecology experiments in Australia in wet forest near Coffs Harbour. One study (Floyd 1966) sought to understand the germination of shrub species which compete with eucalypt seedlings attempting to establish after logging. ‘Weeds’ considered particularly problematic were the fire-cued native shrubs *Dodonaea triquetra*, *Indigofera australis*, *Kennedia rubicunda*, *Acacia binervata* and *Acacia irrorata*. All these species germinated in greater abundance when heated. In a later field study in *E. pilularis/E. grandis* forest, Floyd (1976) found less seed of two common rainforest pioneer species (*Callicoma serratifolia* and *Piptocalyx moorei*) in a site which had had two fires at approximately 15 year intervals, than in a matched site which had burned after 30 years without fire. The short-lived, shade-intolerant species *Acacia binervata* had more seed in the more frequently burnt site; in fact no seed of this species was found in the soil from the long unburnt site.

Smith and Guyer (1983) surveyed a rainforest/WSF ecotone in Giraud State Forest east of Tenterfield, using a series of 100m transects. Soil profiles were similar in both vegetation types. Charcoal was present in the upper layers of the eucalypt forest soils, but not in the rainforest. Charred logs, stumps and fire-damaged trees were also limited to eucalypt and ecotone areas. Large, old eucalypts (*E. saligna*, *E. microcorys*) penetrated furthest through the ecotone towards rainforest, while small eucalypts were comparatively rare. Conversely, the smallest rainforest trees penetrated furthest towards the eucalypt forest. From this study and a companion one at Barrington Tops, Smith and Guyer (1983) concluded that the rainforest was advancing, and the eucalypt forest retreating, across the ecotone in these sites, and that fire delineated the boundary. While several fires had occurred in the vicinity over past decades, the exact extent and frequency of fires at survey sites was not known.

Turner (1984) used carbon dating to study the frequency of ‘severe fires’ in a coastal valley in Whian Whian State Forest. He counted layers of charcoal in the soil of three vegetation types: gully rainforest, adjacent *Lophostemon confertus*-dominated wet sclerophyll forest, and blackbutt forest on the slopes above the brush box. Estimated average fire return intervals were: rainforest, 1100 years; brush box, 325-380 years; blackbutt, 280 years. It was not possible to tell whether the fires detected in this study were interspersed with fires of lower intensity which did not show up in the charcoal record.

Campbell and Clarke (2006) studied the fire response of shrubs and subcanopy species in shrubby wet sclerophyll forests on the Northern Tablelands after a low intensity fire. In contrast to previous Victorian studies (with high intensity fire where 30% of species resprouted), > 80% of these species resprouted after a single fire. Campbell and Clarke (2006) considered their findings consistent with a previous study in northern New South Wales where 60% of species resprouted (cited by Ashton 1981). There was also an associated low variability within a species in fire response (i.e. the species were well classified as either resprouting or not resprouting). Campbell and Clarke (2006) hypothesize that resprouting also has selective advantages where mechanical damage (e.g. cyclones or strong storms) occurs. They propose that a gradient from the wetter more rainforest understorey species (that generally resprout) to the sclerophyllous understory species (that typically occur in the drier more fire prone forest areas) occurs, the sclerophyllous species being more prone to seeding in response to fire. In contrast, Tierney (unpublished data) investigated the species *Prostanthera askania* in a study which included aspects of its fire ecology and population modeling in response to fire and varied disturbances to the forest. This species occurs on the Central Coast from rainforest margins to shrubby wet sclerophyll forests. The species is killed by fire and seeds prolifically post-fire in eucalypt forest, but also probably colonises canopy gaps in the absence of fire. This is a species that has both fire and disturbance driven recruitment.

Discussion of studies

Research into the role of fire in wet sclerophyll forest with a mesic shrub understorey has been sparse in the subtropics. As noted in the previous section, work from Victoria's Mountain Ash (*Eucalyptus regnans*) forests is not necessarily relevant. These Victorian studies, along with those from North Queensland, are reviewed by Watson (2001), who concludes that there is still much to be understood in relation to this forest type, and cautions that suggested fire regimes for WSF may well need modification as research findings come to hand. This situation has not changed.

Floyd's first study (Floyd 1966) documented the now well-established effect of heat on legume germination. It is interesting that these fire-cued species had soil-stored seed in a forest type that is generally considered to have a mesophyllous understorey: a comparison of tree species placed this site in Keith's North Coast Wet Sclerophyll Forest category. Floyd's second study (Floyd 1976) showed a shift in the seedbank away from early-maturing, shade-intolerant species toward species that are slow to mature but tolerant of shade. Presumably this is succession in action. The lack of *any Acacia binervata* seed in the site unburnt for 30 years suggests this species needs intervals well below this level to persist in the community.

Smith and Guyer (1983) also document succession in action. It is interesting that the rainforest advance was occurring despite some fire in the ecotone at both study sites, and despite the high altitude in Barrington Tops (1340m): it suggests that succession can proceed not only where fire is excluded, but also where it is infrequent. Unfortunately, we do not know the fire history of Smith and Guyer's sites.

Turner's estimated fire return interval for blackbutt forest – 280 years (Turner 1984) – implies either that fires were extremely far apart prior to European settlement, or that if fires were more frequent, they were generally not intense. It seems unlikely that wildfires in landscape-wide old fuels would be so far apart, given the fire-prone nature of Australia's eucalypt forests. This suggests that these forests had some protection through reduction of fuel in low intensity understorey fires. Also, the studies discussed above, together with those detailed in the section on grassy WSF, strongly suggest that eucalypt forest would succeed to rainforest during a 280 year interfire interval – although whether and how fast this would actually occur at any particular site is unknown.

Existing fire regime guidelines

Guidelines for south-east Queensland do not clearly distinguish between wet sclerophyll forest subformations. The relevant parts of the guideline reads: "high intensity fire catalyses regeneration in eucalypt-dominated wet sclerophyll forests. Wildfire in its natural season should adequately fulfill this role in most cases. Intervals are likely to range between 20 and 100+ years." "Less intense understorey fires may also play a role in some tall eucalypt forests… Intervals below ….12 years for shrubby systems, are not recommended." "*Lophostemon confertus* (brush box) dominated forests are probably adapted to very long interfire intervals, and may not need fire at all." (Watson 2001a).

Kenny *et al.* (2004) recommend intervals between 25 to 60 years, with the proviso that "Crown fires should be avoided in the lower end of the interval range."

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

This question is also relevant for this vegetation type. Shrubby formation wet sclerophyll forests may be more likely to host obligate seeder eucalypts than their grassy counterparts; *Eucalyptus grandis* is a candidate. This species lacks a lignotuber but may coppice, according to the NSW Flora Fire Response database (DEC 2002). Kenny *et al.* (2004) note that dominant WSF eucalypts may be obligate seeders, do not have soil-stored seed, and thus will be vulnerable to local extinction in the face of a single short inter-fire interval.

Again, current guidelines may represent an uneasy compromise between the occasional high intensity fire needed to ensure successful generational change in eucalypts, and a regime appropriate for the understorey. Kenny *et al.* (2004) bypassed lifespan figures for 'most sensitive' species of 15, 20, 30, 35, 40 and 50 years before settling on a maximum threshold of 60 years. These lower figures may represent more than inaccuracies; they may flag the need for a secondary regime of understorey fires. And again, even a 60 year interfire interval would be extremely short for an obligate seeder eucalypt.

Turner's findings, although limited to a single site, imply that the time between intense wildfires in the past, may have been very long indeed. An average interval of 280 years is, however, compatible with what we know about the lifespan of obligate seeder eucalypts.

Interval domains for understorey diversity

There is a paucity of data to determine appropriate fire regimes for shrubby wet sclerophyll forests. The vital attributes analysis provided in Kenny *et al.* (2004) provides some clues. This sets a minimum fire frequency of 25 years. This assessment assumes that obligate seeder eucalypts are present, but there remains some uncertainty both about the distribution of obligate seeding eucalypts in central and northern NSW (there may be no obligate seeder eucalypts in many areas), and about the length of juvenile periods. Apart from the species with a reputed 25-year juvenile period, the most sensitive species with the next longest time to maturity in the Kenny *et al.* analysis has a juvenile period of about 10 years. Taking this figure and the second-lowest figure in the lifespan chart (which sets maximum intervals) would give a range of 10-20 years. This range would be compatible with retention of *Acacia binervata*, the species Floyd (1976) found missing from the seedbank of a site with a 30 year interval. Understorey fires at a frequency of 10-20 years may, however, generally reduce the density of the shrub layer relative to understorey fires at 25-60 year intervals. This may have effects (both positive and negative) for fauna, for example nest sites for bell miners can be located in this shrub layer. Shorter intervals might also, however, encourage fire-cued shrubs at the expense of T-species and thus maintain a dense shrub layer, albeit one of somewhat different composition.

These forests probably fall into Bond's 'fire-limited' category, implying the potential for at least two states to exist at any one site. Kenny *et al.* (2004) discuss this possibility explicitly: "Wet sclerophyll forests are considered to be a successional stage between open forest and rainforest, leading to differences of opinion regarding management. Frequent fires (c.15-20 years) will favour the sclerophyllous species over the rainforest elements, with the forest tending towards dry sclerophyll forest or even scrub. Conversely, long fire intervals (c. 100 years) allow encroachment of more rainforest species while suppressing establishment of sclerophyll species, resulting in 'expansion' of rainforest into wet sclerophyll forests (Ashton 1981)." The implication here is that possible states may include not only rainforest and WSF with a range of understorey types, but also other sclerophyll vegetation formations. Of course, this statement and the figures in it are a generalization covering a diverse range of forests across the state. Juvenile periods of obligate seeders may also be shorter in warmer areas (Knox & Clarke 2004).

Application to the HCR CMA shrubby wet sclerophyll forests

Watson (2005) could not recommend an explicit fire regime for shrubby wet sclerophyll forests of the Northern Rivers of New South Wales. These shrubby wet sclerophyll forests (and those of the HCR CMA) are likely to be successional in many locations and replaced by rainforest over time and this is driven (at least in part) by absence of fire. Watson hypothesized that the Northern Rivers wet sclerophyll forest understorey can exist in three states whose maintenance is mediated by the frequency of understorey fires:

- **State 1**. Understorey dominated by grasses, herbs and ferns with occasional, mostly I-species shrubs. It is hypothesized that this state is maintained by fires at approximately 2-6 year intervals.
- **State 2**. Understorey dominated by I-species shrubs. T-species shrubs present in low to moderate abundance. Grasses, herbs and ferns present but in lower abundance than in State 1. It is hypothesized that this state is maintained by fires at variable intervals between 7 and 20 years. Note that this state may be more viable in some types of wet sclerophyll forest than others: in those with a relatively thick overstorey canopy, low light levels may limit the distribution of sclerophyll shrubs.
- **State 3**. Understorey dominated by T-species shrubs, and vines. It is hypothesized that this state develops in the absence of fire and is not reversed by fires at intervals above 20 years. If fire continues to be excluded this state transitions to rainforest.

Shrubby wet sclerophyll forests with a mesic understorey (state 3) are likely to be maintained by fire intervals greater than 20 years and less than 50 to 100 years. This broadly accords with Kenny *et al.* (2004), however, there remains a high degree of uncertainly around these fire frequencies. More research is needed that addresses our limited understanding of fire in these forests. Watson (2005) suggests a number of research approaches to deal with this uncertainty. These included fully replicated, long-term field experiments or the use of comparative fire histories from equivalent sites to determine if these drive site differences. Additionally, population modeling of sensitive species (Tierney 2004; Menges *et al.* 2006; Tierney unpublished data) or at the system level (McCarthy *et al.* 1999) can provide insights that will help establish credible fire regimes.

a.

b.

Fig. 5. Following a low intensity fire in wet sclerophyll forest (a) standing plants of the endangered species *Prostanthera askania* have been killed but regeneration via seed is occurring (b) one year post-fire.

9. FIRE IN RAINFORESTS

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) was 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 phenomenon are limited (see shrubby wet sclerophyll forests above).

These successional sequences drive rainforest / eucalypt forest boundaries that are highly dynamic in relation to 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 these 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 & 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 rainforests

* The subtropical and temperate rainforests of the Hunter Central Rivers CMA 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 occur across the HCR CMA and include a range of structural types (here extended to *include* vine thickets to closed forests) and floristic associations (Peake 2006; Somerville 2009). Floristically they characteristically include canopy species that are unlikely to persist with frequent to moderate fire frequencies (e.g. *Acmena smithii*; *Backhousia myrtifolia*; *Pittosporum undulatum*) as well as species that are tolerant of moderate fire frequencies (*Angophora floribunda*; *Eucalyptus punctata*). Species regarded as fire sensitive are often 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 and 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.

The fire response of prominent species in the HCR CMA dry rainforests likely to be fire sensitive was searched in DEC (2002). This is shown on Table 2 and can be used to infer that a number of species 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. It is not possible to confidently prescribe a recommended fire interval on this incomplete dataset. However, fire interval of < 20 years could eliminate *Acmena smithii* form the coastal dry rainforests whilst longer intervals (of some decades) may allow the vigorous shrub *Pittosporum undulatum* to dominate in some areas.

Table 2. Important fire response characteristics of some prominent species considered likely to be fire sensitive in HCR CMA Dry Rainforests.

Species	**Regeneration** (fire response) S = seed R = resprouts	**Establishment** T= without fire	**Seed storage**	**VA Group**	**Minimum maturity (years)**	**Maximum maturity (years)**
Acmena smithii	R (9% mortality) Eliminated by two fires within 20 years	T	transient	VDT	5	30
Alectryon subcinerus	Sr (may resprout if low intensity)	T	transient	DT	5	9
Backhousia sciadophora	-	-	-	-	-	-
Backhousia myrtifolia	R	VCI	transient		-	-
Brachychiton populneus	-	-	-	-	-	-
Clerodendrum tomentosum	R	-	-	V	-	-
Ficus rubiginosa	R	I	-	VDI	3	-
Melia azedarach	R	-	transient	VD	5	-
Pittosporum undulatum	Sr	T	transient	DT	6	8

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.

Littoral rainforests

There are few if any studies of fire in littoral rainforests. Thus appropriate fire regimes can only be applied from a consideration of the recorded fire ecology of the component species of these systems. Prominent species likely to be intolerant of fire include those in Table 3 (below).

Table 3. Important fire response characteristics of some prominent species considered likely to be fire sensitive in HCR CMA Dry Rainforests (see Key above).

Species	**Regeneration** (fire response) S = seed R = resprouts	**Establishment** T= without fire	**Seed storage**	**VA Group**	**Minimum maturity (years)**	**Maximum maturity (years)**
Acmena smithii	R (9% mortality) Eliminated by two fires within 20 years	T	transient	VDT	5	30
Cupaniopsis anacardioides	R s	T	persistent (soil)	VDT	5	-
Endiandra sieberi	R	-	-	VD	-	-
Guioa semiglauca	-	-	-	-	-	-
Alphitonia excelsa	R		Persistent (soil)	VSI	-	8
Syzgium paniculatum	SR	I	transient?	CI	-	-
Syzgium oleosum	-	-	-	-	-	-
Elaeocarpus obovatus	-	-	-	-	-	-

There is a paucity of data for a number of these prominent species. Some species are likely to recruit without fire, but little has been documented about many species. *Acmena smithii* is one species that short fire intervals (< 20 years) will eliminate from a site.

Conclusions – Littoral rainforest

It is currently not possible to provide guidelines for recommended fire regimes for the littoral rainforests. Inter-fire intervals of several decades are more likely to sustain a number of the important tree species of these forests. However, there is a critical need for research.

10. SUMMARY

This review has addressed the fire ecology literature for the most widespread vegetation types in the HCR CMA (Grassy Woodlands; Heathlands; Dry Sclerophyll Forests; Wet Sclerophyll Forests; Rainforests) as well as Grasslands (not widespread but of high conservation significance). The vegetation types recognized in this review are those that fit within the formations and classes of Keith (2004) and their local representation was referenced to Somerville (2009). A general summary of this review 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 formation 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 vegetation types compared to that reported elsewhere and some variations among Tableland species in primary juvenile periods 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. comm.).

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 6). 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. 6. 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. Summary fire guidelines for the HCR CMA.

Broad Vegetation Type	**State Guideline (Kenny *et al*. 2004)**	**Summary points from this review**
Grasslands	2- 10 years Occasional intervals greater than 7 years in coastal areas.	Generally: Statewide guidelines are supported. Exposed sites and sites on poorer soils may require less frequent fire. Grazing may maintain open structures. Site-by-site assessments required for some threatened species in these grasslands *Themeda* coastal headland grasslands – Cohn (2004) recommends 2-10 years as generally appropriate. Intervals need to be managed to address specific threatened species. Growth rates of *Themeda* may be limited in exposed situations and this would reduce fire intervals to maintain diversity. *Austrostipa aristiglumis* grasslands are likely to be of lower productivity than *Themeda* grasslands and may require less frequent fire (than the 2- 10 year interval). *Themeda* / *Poa* dominated systems of the slopes and montane areas are likely to require inter-fire intervals towards the upper end of the state-wide recommended intervals (perhaps 5-10 years).
Grassy woodlands	5-40 years Occasional intervals greater than 15 years	Research in sub-coastal and western slopes woodlands suggests intervals in the lower part of statewide range, ie 5-20 years. New England Grassy Woodlands may have lower growth rates and a minimum of 8 years is recommended by Knox & Clarke (2004). *Eucalyptus laevopinea* woodlands from the upper Hunter would generally fit in the 8-15 year range. *Eucalyptus blakelyi* woodlands from the upper Hunter are also likely to fit this range (i.e. less frequent fire would favour retention of *Melaleuca* & *Acacia* understorey). *Eucalyptus melliodora* & *Eucalyptus albens* from the Merriwa Plateau – Prober *et al.* (2007) recommendation of 5-8 years is generally appropriate. *Allocasuriana leumannii* woodland from the central Hunter Valley are likely to have reduced dominance of *A. leumannii* with shorter intervals within this range.
Heathland	7-30 years Occasional intervals greater than 20 years	Guidelines for Sydney coastal heathlands are generally applicable (page 48):these involve variable intervals within the statewide range, including some 7-15 year intevals to limit competitive exclusion of smaller species by dominant shrubs. However, headland heaths often form a matrix with grassy headland vegetation and these communities have specific issues (see text) that require management. Growth rates in exposed & high altitude sites may be slower and there are likely to be large variations in the level of obligate seeding among sites .
Dry slcerophyll forest	5- 50 (shrub/grass) 7-30 (shrub) Occasional intervals greater than 25 years	Generally 7-30 years is an appropriate fire interval range. However, the comments on page 70-71 are important and fire interval diversity is an appropriate management strategy for biodiversity conservation.
Wet sclerophyll forest (shrubby formation)	25-60 Crown fire avoided in the lower interval range	Crown fires should be avoided (> 100 year intervals). There is some uncertainty about recommended intervals for the understorey (see pages 94-95). Research required. Short intervals will reduce shrub cover and probably >20-year intervals would be needed to maintain a mesic shrubby understorey, whereas intervals between 10 and 20 years may tip shrub composition more towards sclerophyll species in some more open-canopied WSF types. Managing the ecotone and the expansion of rainforest in the

		absence fire oresents challenges.
Wet sclerophyll forest (grassy formation)	10-50 Crown fire avoided in the lower interval range	Crown fires should be avoided (> 100 year intervals). Two regimes for ground fires should be maintained (in the landscape – not on all sites): 2- 5 years for open grassy sites and 6-20 years for less open sites (more towards the shrubby range). See *Rainforests* for issues about rainforest invasion..
Rainforest	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. Littoral Rainforest. Insufficient data., research required

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