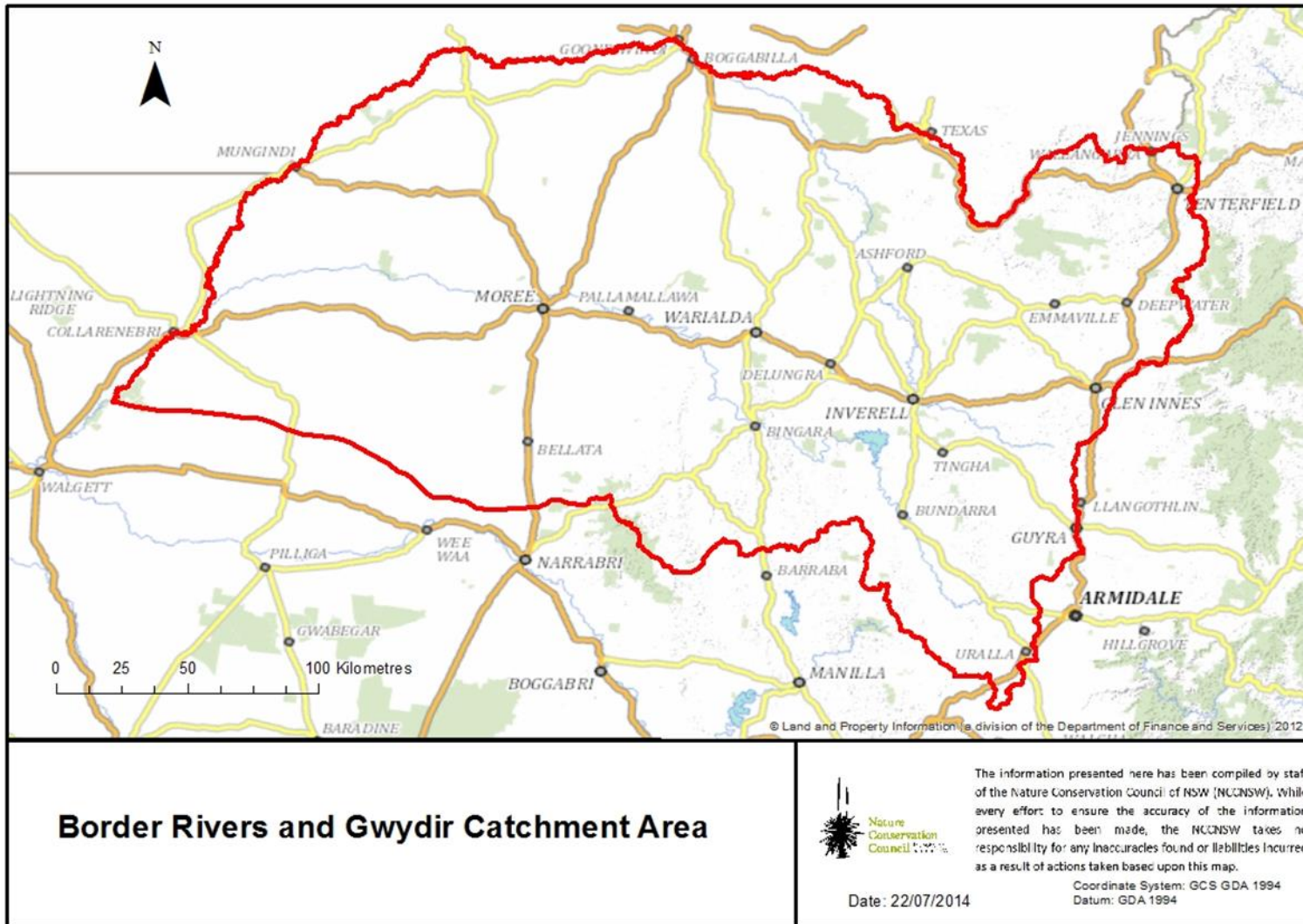# HOTSPOTS FIRE PROJECT

Fire and the Vegetation of the Border Rivers-Gwydir Region

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Map: Border Rivers and Gwydir catchment area covered by literature review

Scope of this review

This literature review forms part of a suite of materials that Hotspots aims to produce in each 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 Border Rivers-Gwydir region of New South Wales (NSW). It aims to help land and fire managers not only to understand the impacts of fire in the region, but also to place that understanding in a wider ecological context. Companion documents covering the Central West, Hunter, Lachlan, Namoi, Northern Rivers and Southern Rivers regions are also available (Watson 2006 a, b 2007; Watson and Tierney 2009a,b; Graham *et al.* 2012).

Fire affects different plant and animal species differently, and fire regimes compatible with biodiversity conservation vary widely between ecosystems (Bond 1997; Watson 2001; Bradstock *et al.* 2002; Kenny *et al.* 2004). This document explores the role of fire in the vegetation formations of Keith (2004). All vegetation formations covered in this review also occur in the Namoi region. However the literature is limited for some vegetation formations and this is reflected in this review. Fire is also of limited occurrence in most wetland types (though it can occur in Forested Wetlands), therefore wetlands are also not considered in this review. The broad vegetation formations of Keith (2004) can be further subdivided into classes (Table 1). Where literature permits, the fire ecology of classes that occur in the Border Rivers-Gwydir region are discussed (often there is no literature available at the class level or limited to only one study).

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Table 1: Vegetation Formations and Classes (Keith) occurring in the Border Rivers – Gwydir region

VEGETATION FORMATION	KEITH VEGETATION CLASS
ARID SHRUBLANDS *ACACIA SUBFORMATION* **1 vegetation community**	• North-west Plain Shrublands
CHENOPOD SUBFORMATION **4 vegetation communities**	• Riverine Chenopod Shrublands
SEMI-ARID WOODLANDS *GRASSY SUBFORMATION* **9 vegetation communities**	• North-west Floodplain Woodlands • Riverine Plain Woodlands • Brigalow Clay Plain Woodlands
SHRUBBY SUBFORMATION **7 vegetation communities**	• Western Peneplain Woodlands • Subtropical Semi-arid Woodlands • North-west Alluvial Sand Woodlands
DRY SCLEROPHYLL FORESTS *SHRUB/GRASS SUBFORMATION* **24 vegetation communities**	• New England Dry Sclerophyll Forests • Northern Gorge Dry Sclerophyll Forests • North-west Slopes Dry Sclerophyll Woodlands • Pilliga Outwash Dry Sclerophyll Forests
SHRUBBY SUBFORMATION **35 vegetation communities**	• Northern Escarpment Dry Sclerophyll Forests • Northern Tableland Dry Sclerophyll Forests • Western Slopes Dry Sclerophyll Forests • Yetman Dry Sclerophyll Forests
HEATHLANDS **3 vegetation communities**	• Northern Montane Heaths
FORESTED WETLANDS **5 vegetation communities**	• Eastern Riverine Forests • Inland Riverine Forests
FRESHWATER WETLANDS **14 vegetation communities**	• Montane Bog and Fens • Montane Lakes • Inland Floodplain Shrublands • Inland Floodplain Swamps
GRASSLANDS **13 vegetation communities**	• Semi-arid Floodplain Grasslands • Temperate Montane Grasslands • Western Slopes Grasslands
GRASSY WOODLANDS **27 vegetation communities**	• Western Slopes Grassy Woodlands • Subalpine Woodlands • New England Grassy Woodlands • Tableland Clay Grassy Woodlands • Floodplain Transition Woodlands
WET SCLEROPHYLL FORESTS *SHRUBBY SUBFORMATION* **4 vegetation communities**	• Northern Escarpment Wet Sclerophyll Forests
GRASSY SUBFORMATION **7 vegetation communities**	• Northern Tableland Wet Sclerophyll Forests • Northern Hinterland Wet Sclerophyll Forests
RAINFORESTS **6 vegetation communities**	• Northern Warm Temperate Rainforests • Dry Rainforests • Western Vine Thickets

(OEH, nd; Benson et al. 2010)

The guidelines – a word of caution

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

1. FIRE ECOLOGY: SOME IMPORTANT CONCEPTS

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

Species responses to fire

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

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

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

Regeneration modes

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

Non-sprouting species, or '**obligate seeders**', are plants that die when their leaves are all scorched in a fire, these species rely on regeneration from seed. Obligate seeder species generally produce more seed (Lamont *et al.* 1991), and greater numbers of seedlings (Wark *et al.* 1987; Benwell 1998) than resprouters, and seedling growth rates tend to be more rapid (Bell & Pate 1996; Benwell 1998; Bell 2001a).

These categories are not invariant. Survival rates in the field for both resprouters and obligate seeders change with fire intensity (Morrison & Renwick 2000). Some species exhibit different regeneration strategies in different environments (Williams *et al.* 1994; Benwell 1998; DEC 2002).

Seed storage and dispersal

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

Some species hold their seeds in on-plant storage organs such as cones, and release them after a fire. These 'serotinous' taxa include species in the Proteaceae and Cupressaceae families, for example *Banksia*, *Hakea* and *Callitris*. Some eucalypts release seed in response to fire (Noble 1982; Gill 1997). The degree to which seed release also occurs in the absence of fire varies between species (Ladd 1988; Enright *et al.* 1998).

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

A third strategy is to create seeds rapidly after a fire, through fire-cued flowering. *Xanthorrhoea* species are a well-known example of this phenomenon (Harrold 1979; McFarland 1990); however shrubs such as the Crinkle Bush *Lomatia silaifolia)* (Denham & Whelan 2000) and the Waratah *(Telopea speciossima)* (Bradstock 1995) also flower almost exclusively in the years after a fire. Many Grassland forbs (herbaceous plants which are not a grass, sedge or rush) exhibit this characteristic (Lunt 1994).

Finally, some species rely to a greater or lesser extent on seed coming in from outside the burnt area. This strategy is not common in very fire-prone environments such as coastal heaths, where seed dispersal distances seem to be limited to tens of metres or less in most species (Auld 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.* 2006b).

Recruitment relative to fire

Species also differ in when they establish new plants relative to fire. For many species in fire-prone environments, recruitment is confined to the immediate post-fire period (Auld 1987a,b: 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; Clarke *et al.* 2009). 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; Clarke *et al* 2009).

The length of time seed remains viable is another important variable, but one about which not a great deal is known. It is clear, however, that species vary greatly (Keith 1996). The seedbanks of serotinous species are likely to be depleted more quickly than those of species with soil-stored seed, although much variation exists even here (Gill & Bradstock 1992; Morrison *et al.* 1996; Clarke *et al.* 2009). Species also vary in whether seedbanks are exhausted by a single fire (either through germination or destruction). Species which store seed in the canopy, and species whose soil-stored seeds are relatively permeable (e.g. *Grevillea* species - Morris 2000), are unlikely to retain a residual store of pre-fire seeds through the next interfire interval. However some species, generally those with hard, soil-stored seeds such as peas and wattles, retain viable ungerminated seed through more than one fire; *Bossiaea laidlawiana*, from south-west Western Australia, is an example (Christensen & Kimber 1975). This gives these species a 'hedge' against a second fire within the juvenile period.

Disturbance, succession and a paradigm shift

Disturbance and succession are basic concepts in ecology. Advances in understanding of these processes has informed several theories and models which are useful for understanding the way plant communities – groupings of species with different fire-related attributes – respond to fire. Fire is a **disturbance**. A disturbance can be defined as "any relatively discrete event in time that removes organisms and opens up space which can be colonised by individuals of the same or different species" (Begon *et al.* 1990). The concept encompasses recurring discrete events such as storms, floods and fires, as well as on-going processes like grazing. Disturbance may stem from natural phenomena or human activities (Hobbs & Huenneke 1992), and is ubiquitous throughout the world's ecosystems (Sousa 1984).

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

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

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

This **non-equilibrium paradigm** can be encapsulated by the phrase 'the flux of nature'. **Scale** is important in this paradigm: equilibrium at a landscape scale may be the product of a distribution of **states** or **patches** in flux (Wu & Loucks 1995). Implications include a legitimate – or even vital – role for people in ecosystem management, and a focus on the conservation of processes rather than objects. This does *not*, of course, imply that all human-generated change is okay; it does mean

human beings must take responsibility for maintaining the integrity of natural ecosystem processes (Pickett *et al.* 1992; Partridge 2005). Fire fits much more comfortably into the non-equilibrium paradigm, where it takes its place as a process integral to many of the world's ecosystems.

Theory into thresholds

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

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

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

Species whose establishment is keyed to fire (Noble and Slatyer call these 'I-species') are highly sensitive to **long interfire intervals** (infrequent fire): they are liable to local extinction if fire does not occur within the lifespan of established plants and/or seedbanks (Noble & Slatyer 1980). The maximum interval (upper threshold) therefore needs to accommodate the taxon in this category with the shortest lifespan, seedbank included (DEC 2002; Bradstock & Kenny 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 *et.al* 2009).

Bursaria has the advantage of being able to recruit between fires, whereas most sclerophyllous (hard-leaved) shrub species recruit almost exclusively after a fire (Purdie & Slatyer 1976; Cowling *et al.* 1990; Keith *et al.* 2002a). The vital attributes model explicitly identifies species able to recruit between fires – Noble and Slatyer call them 'T-species' – and their propensity to dominate in the absence of disturbance is also explicitly noted (Noble & Slatyer 1980). However to date little emphasis has been placed on the role of T-species when determining fire frequency guidelines. T-species are almost certainly more prevalent in some environments than in others, for example in rainforests, and conversely in arid areas where recruitment may be geared more to periods of unusually high rainfall or flooding than to fire.

The importance of competition between plant species, and thus the importance of disturbance to disrupt competitive exclusion, is likely to vary with **landscape productivity**. A second 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, abiotic factors such as low rainfall, heavy frosts and infertile soils limit the number of plant species

able to grow, and also limit their growth rates. The need for disturbance to reduce competitive superiority is therefore minimal. In fact, a high disturbance frequency is predicted to reduce diversity in these ecosystems, as organisms will be unable to grow fast enough to recover between disturbances. In highly productive, resource-rich environments, however, competition is likely to be much more intense, as many species can grow in these areas, and they grow quickly. Here, diversity is predicted to decline where disturbance frequency is low, as some species will out-compete others, excluding them from the community.

Landscape productivity, as defined by plant biomass as an example, is likely to increase with rainfall, temperature, season of rainfall – where rainfall and warm temperatures coincide, there may be a greater potential for plant growth – and soil fertility (clay soils are often more fertile than sandy soils, however they also tend to support more herbaceous, and fewer shrub, species (Prober 1996; Clarke & Knox 2002). Relatively frequent fire may thus be more appropriate in wet, warm, productive fire-prone systems than in those whose productivity is limited by poor soils, low rainfall or a short growing season.

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

- **Climate-limited systems**. These communities are not prone to either major structural change, nor to succeeding to another vegetation type in the absence of fire, although fire frequency may influence species composition to some extent. In South Africa these communities occur in arid environments, and also in areas nearer the coast where rainfall is moderate but occurs in winter.
- **Climate-limited but fire modified systems**. These vegetation types do not succeed to another vegetation type in the absence of fire, but their structure may alter from grassy to shrubby. The Cumberland Plain Woodland described above fits into this category.
- **Fire-limited**. These vegetation types will succeed to a different community in the absence of fire. In South Africa, these communities occur in higher rainfall areas, and include both savannah and heath.

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

Climate-limited but fire-modified systems can occur in at least two 'states', for example Grassy Woodland and *Bursaria*-dominated shrub thicket woodland on the Cumberland Plain (Watson 2005). Fire-limited vegetation types could also be said to be able to exist in different states, although the differences between them are so great that they are rarely thought about in this way. For example, in north Queensland, *Eucalyptus grandis* grassy Wet Sclerophyll Forest is succeeding to rainforest, probably due to a reduction in fire frequency and/or intensity (Unwin 1989; Harrington & Sanderson 1994; Williams *et al.* 2003). 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 thrive (Stewart 1999; Tasker 2002). Nearby areas which have remained unburnt for periods over 15 or 20 years support higher densities of some shrub and non-eucalypt tree species, particularly those able to recruit between fires (Birk & Bridges 1989; Henderson & Keith 2002). Each regime provides habitat for an equally diverse, but substantially different, array of invertebrates and small mammals (Andrew *et al.* 2000; York 2000; Bickel & Tasker 2004; Tasker & Dickman 2004).

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

Fire and climate change

Climate change is a challenging example of the non-equilibrium paradigm in operation. Altered fire regimes have been predicted over ensuing decades and are considered to be one of the major ecological management challenges for Australia (Morton *et al*. 2009). This presents a significant issue for fire ecologists who (like most ecologists) have operated under an assumption of an overall steady state (albeit with large variability) in fire regimes. General predictions of increasing fire frequency and intensity have been predicted to lead to major fire driven shifts in vegetation patterns. There are, however, a range of uncertainties around these predictions. Firstly, it is by no means certain what (if any) changes will occur. Bradstock *et al*. (2008) report that the two major drivers of fire are "fire weather" (hot, dry conditions; ignition rates) and fuel load. Whilst increases in fire weather may occur, drier conditions could lead to less fuel (Bradstock 2010; Penman & York 2010), so that fire regime changes are by no means understood (Matthews *et al.* 2013). 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).

2. THE LANDSCAPES, CULTURES AND CLIMATE OF THE BORDER RIVERS–GWYDIR REGION

The Border Rivers-Gwydir (BRG) region is composed of three distinct landscapes (*Figure 1*): tablelands (part of the New England Tableland bioregion), slopes (containing parts of the Nandewar and Brigalow Belt South bioregions) and plains (containing parts of the Brigalow Belt South and Darling Riverine Plains bioregion). All are part of the upper reaches of the Murray–Darling Basin, encompassing an area of approximately 50,000 square kilometres. The "Border Rivers" component of the catchment extends into Queensland, with the Macintyre River forming the State border (NSW Office of Water, nd).

Elevation in the BRG region varies from just over 1500 m above sea level (asl) on the crest of the Great Dividing Range between Tenterfield and Guyra, as well as on the peaks of Mt Kaputar, to approximately 140 m asl at the confluence of the Gwydir and Barwon River in the far west of the region. The BRG region contains a substantial proportion of the New England Tableland, the largest highland landscape in Australia (Barnes *et al.* 1988).

The BRG region contains all or part of 11 Local Government Areas (LGAs) with a collective population of approximately 65,000 residents. Major population centres include Glen Innes, Tenterfield and Uralla on the Tablelands, Inverell and Bingara on the Slopes, and Moree and Mungindi on the Plains. The region is heavily agriculturally developed and the majority of pre-European ecosystems across the more fertile and productive parts of the catchment have been degraded or removed by over a century and a half of agricultural landuse (OEH VIS spatial data). Synergistic pressures that have led to the extinction of numerous native mammals and other fauna include invasion and occupation by feral animals and weeds, serious degradation to land and water resources and the imposition of inappropriate fire regimes. This has resulted in the decline and loss of functionality of many high-productivity ecosystems, a process that continues to the present day.

Because of the presence of large areas of highly productive land including extensive fertile floodplains and basalt enriched landscapes, a relatively small proportion of the BRG region is Public Land (Conservation Reserves, State Forests and Crown Lands), but the majority of Public Land is vegetated. There are several landscapes in the BRG region where relatively large and well-connected areas of native vegetation remain (*Figure 2*). Examples of extensive areas of native vegetation include Torrington State Conservation Area, Dthinna Dthinnawan National Park (near Yetman and Texas) and the northern and eastern parts of Mt Kaputar National Park. Moderately large and connected areas of native vegetation exist along the crest of the Great Dividing Range (Bald Rock, Bolivia Hill, Butterleaf and Capoompeta National Parks). On the western slopes relatively large and connected areas of native vegetation exist along the Nandewar Range and the gorges draining the New England Tableland, reserves in these landscapes include Gwydir River, Kwiambal, King Plains,

Indawarra and Single National Parks and Arakoola, The Basin, Ironbark and Severn River Nature Reserves (Benson *et al*. 2010).

Many landscapes in the BRG region have been entirely cleared; as a result they have little biodiversity or conservation value. Notably the fertile floodplains of the main rivers and the extensive basaltic landscapes of the BRG region have very little native vegetation remaining. Only steep, relatively infertile and rocky sandstone, granite and leucogranite landscapes still support extensive areas of native vegetation (Benson *et al.* 2010).

The Gwydir River rises in the New England Tablelands near Uralla and flows north-west through steep valleys until it reaches the flat plains near Gravesend. Upstream of Moree the valley widens and the Gwydir River breaks into a complex pattern of creeks, anabranches and wetlands. The Gwydir River's main tributaries are Copes, Moredun, Georges and Laura Creeks, and the Horton River. The wetlands at the end of the catchment soak up much of the Gwydir River's flow. During flood events water spills out across the floodplain and enters the Barwon River at numerous points (NSW Office of Water, nd).

The major river systems comprising the Border Rivers are the Dumaresq, Severn, Macintyre and Mole Rivers. The western half of the catchment comprises flat alluvial plains drained by intermittent watercourses – mainly Croppa, Whalan and Gil Gil Creeks. At the lower end of the catchment the Macintyre River is characterised by a complex series of anabranching channels. The Weir River in Queensland is the Macintyre River's largest tributary, flowing into the Macintyre River around 25 kilometres upstream of Mungindi and forming the start of the Barwon River.

The BRG region has a temperate to sub-tropical climate, with a considerable graduation from east (cooler and wetter) to west (hotter and drier). Since 1950, the region has experienced warming of around 0.8–1.3º C. This is likely to be partly due to human activities. The future climate of the Border Rivers-Gwydir catchment is likely to be warmer and drier. Such trends would also increase evaporation, heat waves, extreme winds and fire risk. Nevertheless, despite this trend toward drier conditions, there is also potential for increases in extreme rainfall events (CSIRO, 2007).

The flows of the Darling River catchment great variability, responding strongly to El Nino – La Niña cycles and deriving major flood pulses from southern Queensland and the western New England Tableland (NSW Office of Water). CSIRO (2007) predicts that rainfall patterns and hydrological regimes across the BRG region will be strongly influenced by global warming. Changes in rainfall and higher evaporation rates are likely to lead to less water for streams and rivers in the BRG region.
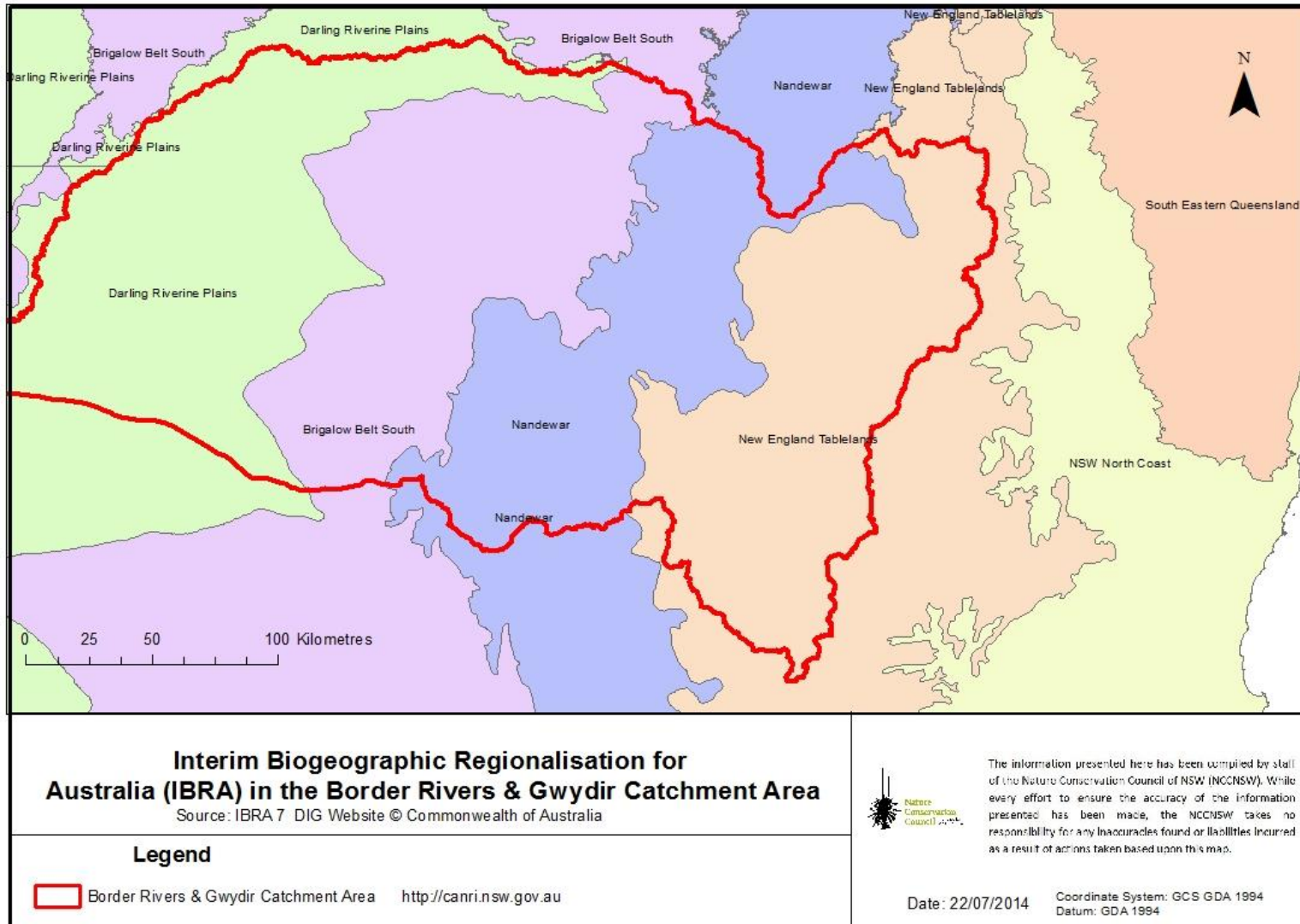
Figure 1: Bioregions in the BRG region

Extant Vegetation of the Border Rivers and Gwydir Catchment Area

Legend

- Border Rivers Gwydir Catchment Area   http://canri.nsw.gov.au
- NSW Extant Native Vegetation (Keith&Simpson, 2006) V2
- State Forests (Forests NSW)
- Office of Environment and Heritage Estate http://mapdata.environment.nsw.gov.au

Date: 22/07/2014

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

Coordinate System: GCS GDA 1994
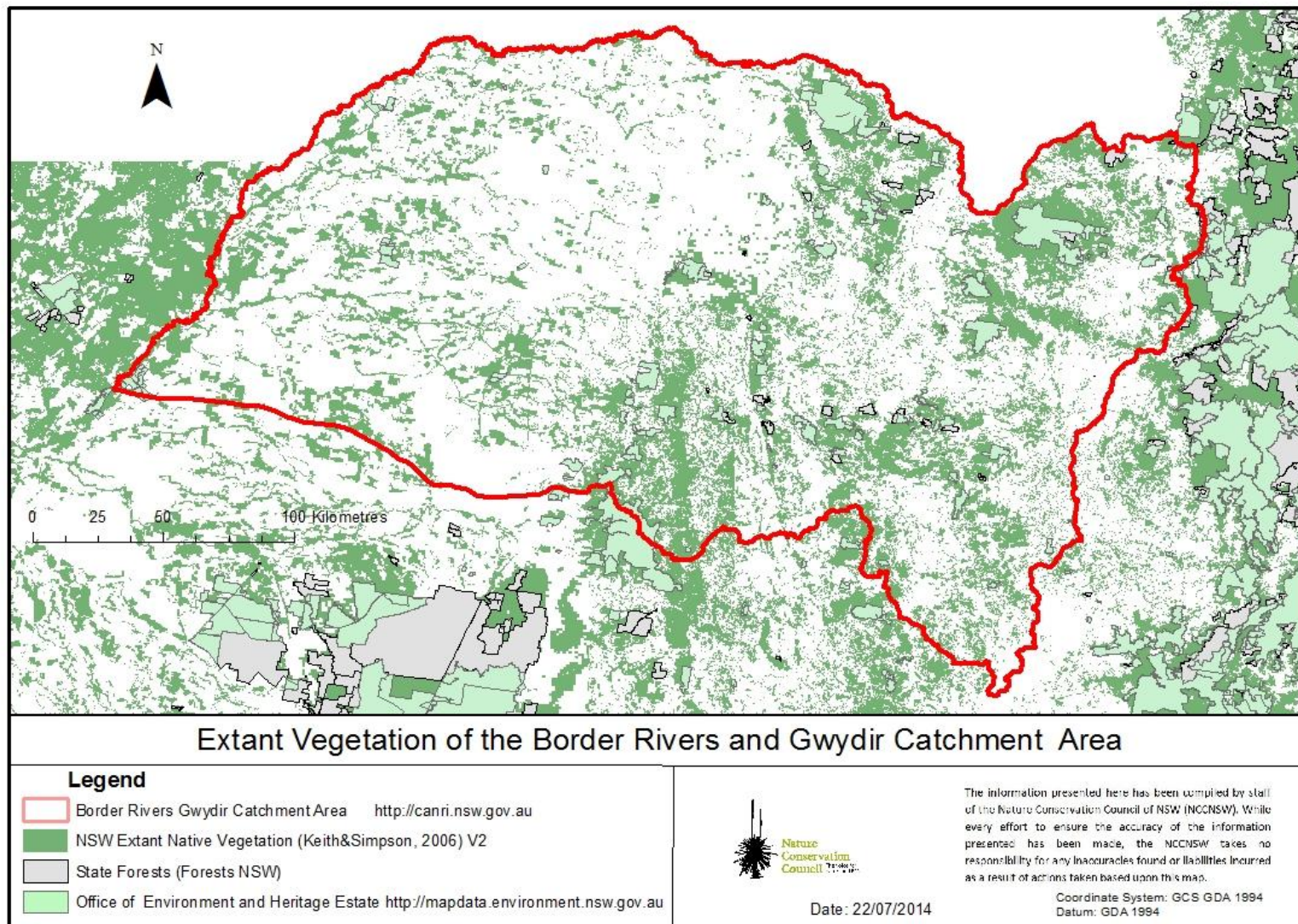Datum: GDA 1994

Figure 2: Remnant native vegetation and conservation reserves in the BRG region

The BRG region is rich in Aboriginal cultural heritage, with many Aboriginal nations living in the area for over thousands of years and still living in the area today. Aboriginal nations of the Border Rivers-Gwydir region include the Anawaan, Banbai, Bigambul, Kamilaroi (Gomeroi) and Ngarrabal. Within the region the land of the Kamilaroi nation stretches from the western area of the catchment to the border with the Banbai nation near Guyra in the east of the catchment. The Anawaan nation's land extends south from the border with the Banbai nation (near Guyra) towards Uralla and northwest towards the Tingha district (Connah *et al.* 1977). The land of the Ngarrabal nation is around the Glen Innes area. The Bigambul people occupied lands around Bonshaw and Yetman and northwards into Queensland.

There are a number of Aboriginal cultural heritage sites in the Border Rivers-Gwydir region including: artefact scatters and rock art, burial sites, scarred and carved trees, grinding grooves and middens, and mission and reserve sites and contemporary Aboriginal people maintain an association with their lands. There are two Indigenous Protected Areas (IPA) within the BRG region, The Willows-Boorabee near Emmaville and Tarriwa-Kurrukun near Tingha (Commonwealth of Australia, nd).

3. LANDSCAPE GRADIENTS AND DRIVERS OF BIODIVERSITY IN THE BRG REGION

The substantial rainfall gradient across the BRG region from east (>1400 mm p.a.) to west (<350 mm p.a.), creates substantial variation in vegetation systems. These range from small areas of Rainforest and shrubby Wet Sclerophyll Forests on the high rainfall north-eastern edge of the catchment along the crest of the Great Dividing Range (Benson & Ashby 1996,2000), to Grassy Woodlands and a wide variety of Dry Sclerophyll Forests in the mid-reaches of the catchment, Subalpine Woodlands above 1300 m the high elevation at Mt Kaputar and arid shrublands dominated by leopardwood (*Flindersia maculosa*), Mitchell Grass dominated Grasslands and chenopod shrublands in the far west of the catchment (Benson *et al*. 2010).

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

The vegetation communities of the BRG region have evolved with fire as a prominent driver during the increasing aridity of the Tertiary and Quaternary periods (White, 1994), and many communities, populations and species are critically reliant upon fire to maintain essential life cycle components. Fire intensity and frequency vary greatly across the vegetation communities occurring within the BRG region, primarily as a function of variation in rainfall and site productivity. The Rainforests and shrubby Wet Sclerophyll Forests in the far east of the catchment have the greatest biomass of any vegetation community in the catchment, whilst the Grasslands and Chenopod Shrublands on the western plains have the least.

4. FAUNA EXTINCTION IN THE BRG REGION

Most Critical Weight Range (CWR) mammals are extinct in the BRG region. Species such as the Bilby (*Macrotis lagotis*), Eastern hare-wallaby (*Lagorchestes leporides*) and Gould's Mouse (*Pseudomys gouldii*) were known from the BRG region historically (*Table 2*), the latter two species are extinct nationally, the former remains in scattered small colonies from western Queensland to the Kimberley coast. The extinction of populations of bandicoots, small wallabies and potoroids as well as native rodents and carnivores (Burbidge & Mackenzie, 1989) and replacement with sheep, cattle, foxes, rabbits and cats has led to a fundamental shift in ecosystem dynamics across the catchment. This shift in mammal community composition has resulted in a myriad of deleterious impacts on the diversity, structure and function of remaining native species. The consequences of such a large shift in the fauna community will likely resonate for many generations to come. Further extinctions of fauna species seem likely because of the highly fragmented nature of native vegetation across most of the region.

Extinction rates of woodland and forest bird species have historically been low, however in recent decades many species have experienced major declines, and this rate is accelerating (Reid 1999, 2000). Although few, if any, viable populations of bush stone curlew and barking owl remain within the BRG region, it is renowned as a national stronghold for declining woodlands birds. In particular the area around Bingara and Bundarra is known to be one of the last breeding strongholds of the Critically Endangered regent honeyeater (*Anthochaera phrygia)* (Menkhorst *et al.* 1999). There are also significant populations of the eastern subspecies of the grey-crowned babbler (*Pomatostomus temporalis temporalis)*, diamond firetail (*Stagonopleura guttata*), hooded robin (*Melanodryas cucullata cucullata*) and brown treecreeper (*Climacteris picumnus victoriae*) known from the BRG region.

5. ENDEMIC SPECIES, THREATENED SPECIES, ENDANGERED POPULATIONS AND ENDANGERED ECOLOGICAL COMMUNITIES OF THE BRG REGION

The BRG region contains a rich diversity of native flora and fauna species including numerous species listed as Vulnerable, Endangered or Critically Endangered (*Table 2*), as well as two Endangered Populations of fauna, namely populations of the Tusked Frog (*Adelotus brevis*) and Brush Turkey *(Alectura lathami)* (*Table 3*).

A considerable number of native plant species are endemic to the BRG region, including (amongst many notable examples) a major evolutionary radiation of the heathy myrtle shrubs (*Homoranthus)*, on rocky outcrops across the region. *Homoranthus* species that are endemic to the BRG region and areas immediately adjoining include *H.bebo, H. biflorus*, *H. binghiensis, H. bruhlii*, H*. croftianus*, *H. elusus, H. lunatus* and *H. melanostictus* (Williams & Wissmann 1991; Hunter 1998; Copeland *et al.* 2011). The Torrington and Bolivia Hill districts are recognised as being amongst the most floristically rich landscapes in NSW with many endemic, rare and threatened species known. At Torrington over forty five rare and threatened species are known including the Torrington beard-heath (*Leucopogon confertus*) including Torrington mint-bush (*Prostanthera staurophylla sensu stricto*) and the Torrington Pea (*Almaleea cambagei*). Endemic species know from Bolivia Hill include Bolivia stringybark (*Eucalyptus boliviana*), Bolivia wattle (*Acacia pycnostachya*), Bolivia Hill boronia (*Boronia boliviensis*), Bolivia homoranthus (*H. croftianus*), Bolivia Hill riceflower (*Pimelea venosa*) (Clarke *et al.* 1998a; OEH 2011). Other Threatened Species restricted to the BRG region include the Endangered Rock of Gibraltar apple (*Angophora exul*) and the braid fern (*Platyzoma microphylla*). Many of these plant species occur in small populations or occupy very narrow ranges; as a result they are vulnerable to many threatening processes including vertebrate pests, inappropriate fire regimes, stochastic events and climate change.

Within the BRG region there are fourteen Endangered Ecological Communities and two Critically Endangered Ecological Communities listed under the NSW Threatened Species Conservation Act 1995 (TSC Act, 1995) that are known or predicted to occur (OEH data), as well as six Endangered Ecological Communities and three Critically Endangered Ecological Communities (Table 3) listed under the Commonwealth Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act, 1999).

Table 2: NSW and Commonwealth listed Threatened Species known and predicted to occur in the BRG region (NSW TSC Act, 1995 & Commonwealth EPBC Act, 1999)

Threatened Plants of the Border Rivers-Gwydir Region

Common name	Scientific Name	Conservation Status		Occurrence
		NSW Status	Commonwealth Status	
Acalypha	*Acalypha eremorum*	Endangered		Known
Austral toadflax	*Thesium australe*	Vulnerable	Vulnerable	Known
Backwater grevillea	*Grevillea scortechinii subsp. sarmentosa*	Vulnerable		Known
Bailey's indigo	*Indigofera baileyi*	Endangered		Known
Barrington tops ant orchid	*Chiloglottis platyptera*	Vulnerable		Known
Beadle's grevillea	*Grevillea beadleana*	Endangered	Endangered	Known
Belson's panic	*Homopholis belsonii*	Endangered	Vulnerable	Known
Binghi homoranthus	*Homoranthus binghiensis*	Endangered		Known
Blackbutt candlebark	*Eucalyptus rubida subsp. barbigerorum*	Vulnerable	Vulnerable	Known
Bluegrass	*Dichanthium setosum*	Vulnerable	Vulnerable	Known
Bolivia hill boronia	*Boronia boliviensis*	Endangered		Known
Bolivia hill pimelea	*Pimelea venosa*	Endangered	Endangered	Known
Bolivia homoranthus	*Homoranthus croftianus*	Endangered		Known
Bolivia stringybark	*Eucalyptus boliviana*	Vulnerable		Known
Bolivia wattle	*Acacia pycnostachya*	Vulnerable	Vulnerable	Known
Braid fern	*Platyzoma microphyllum*	Endangered		Known
Callistemon pungens	*Callistemon pungens*		Vulnerable	Known
Creeping tick-trefoil	*Desmodium campylocaulon*	Endangered		Known
Crescent-leaved homoranthus	*Homoranthus lunatus*	Vulnerable	Vulnerable	Known
Cyperus conicus	*Cyperus conicus*	Endangered		Known
Desert cow-vine	*Ipomoea diamantinensis*	Endangered		Known
Finger panic grass	*Digitaria porrecta*	Endangered		Known
Gibraltar rock apple	*Angophora exul*	Endangered		Known
Granite boronia	*Boronia granitica*	Vulnerable	Endangered	Known
Granite homoranthus	*Homoranthus prolixus*	Vulnerable	Vulnerable	Known
Grove's paperbark	*Melaleuca groveana*	Vulnerable		Known
Hairy jointgrass	*Arthraxon hispidus*	Vulnerable	Vulnerable	Known
Hawkweed	*Picris evae*	Vulnerable	Vulnerable	Known
Heath wrinklewort	*Rutidosis heterogama*	Vulnerable	Vulnerable	Known

Inverell cycad	*Macrozamia humilis*	Endangered		Known
Keith's zieria	*Zieria ingramii*	Endangered	Endangered	Known
Lake keepit hakea	*Hakea pulvinifera*	Endangered	Endangered	Known
Large-leafed Monotaxis	*Monotaxis macrophylla*	Endangered		Known
Macnutt's wattle	*Acacia macnuttiana*	Vulnerable	Vulnerable	Known
Mckie's stringybark	*Eucalyptus mckieana*	Vulnerable	Vulnerable	Known
Myall creek wattle	*Acacia atrox*	Endangered		Known
Narrow-leaved black peppermint	*Eucalyptus nicholii*	Vulnerable	Vulnerable	Known
Narrow-leaved Guinea flower	*Hibbertia tenuifolia*	Endangered		Predicted
Native milkwort	*Polygala linariifolia*	Endangered		Known
Northern blue box	*Eucalyptus magnificata*	Endangered		Known
Ooline	*Cadellia pentastylis*	Vulnerable	Vulnerable	Known
Ovenden's ironbark	*Eucalyptus caleyi subsp. ovendenii*	Vulnerable	Vulnerable	Known
Pindari wattle	*Acacia acrionastes*	Endangered		Known
Pine donkey orchid	*Diuris tricolor*	Vulnerable		Known
Polblue eyebright	*Euphrasia ciliolata*	Vulnerable		Known
Prasophyllum sp. Wybong	*Prasophyllum sp. Wybong*		Critically Endangered	Known
Rodd's star hair	*Astrotricha roddii*	Endangered	Endangered	Known
Rulingia procumbens	*Rulingia procumbens*	Vulnerable	Vulnerable	Known
Rupp's boronia	*Boronia ruppii*	Endangered		Predicted
Rusty desert phebalium	*Phebalium glandulosum subsp. eglandulosum*	Endangered	Vulnerable	Known
Scant pomaderris	*Pomaderris queenslandica*	Endangered		Known
Scrambling lignum	*Muehlenbeckia costata*	Vulnerable		Known
Severn river heath-myrtle	*Micromyrtus grandis*	Endangered	Endangered	Known
Shrub sida	*Sida rohlenae*	Endangered		Known
Silky Swainson-pea	*Swainsona sericea*	Vulnerable		Known
Slender darling pea	*Swainsona murrayana*	Vulnerable	Vulnerable	Known
Small snake orchid	*Diuris pedunculata*	Endangered	Endangered	Known
Spiny peppercress	*Lepidium aschersonii*	Vulnerable	Vulnerable	Known
Tenterfield eyebright	*Euphrasia orthocheila subsp. peraspera*	Endangered		Known
Torrington beard-heath	*Leucopogon confertus*	Endangered	Endangered	Known
Torrington mint-bush	*Prostanthera staurophylla sensu stricto*	Endangered	Vulnerable	Known

Torrington pea	Almaleea cambagei	Endangered	Vulnerable	Known
Tylophora linearis	Tylophora linearis	Vulnerable	Endangered	Known
Velvet wattle	Acacia pubifolia	Endangered	Vulnerable	Known
Wallangarra white gum	Eucalyptus scoparia	Endangered	Vulnerable	Known
Wandering pepper cress	Lepidium peregrinum	Endangered	Endangered	Known
Warra broad-leaved Sally	Eucalyptus camphora subsp. relicta	Endangered		Known
Wild orange	Capparis canescens	Endangered		Known
Wollemi mint-bush	Prostanthera cryptandroides subsp. cryptandroides	Vulnerable	Vulnerable	Predicted
Yetman wattle	Acacia jucunda	Endangered		Known

Threatened Animals of the Border Rivers-Gwydir Region

Common Name	Scientific Name	Conservation Status		Occurrence
		NSW Status	Commonwealth Status	
Australasian bittern	Botaurus poiciloptilus	Endangered	Endangered	Known
Australian brush-turkey	Alectura lathami - endangered population	Endangered Population		Known
Australian bustard	Ardeotis australis	Endangered		Known
Australian painted snipe	Rostratula australis	Endangered	Endangered	Known
Barking owl	Ninox connivens	Vulnerable		Known
Beccari's freetail-bat	Mormopterus beccarii	Vulnerable		Known
Bell's turtle	Elseya belli	Vulnerable	Vulnerable	Known
Bilby	Macrotis lagotis	Presumed Extinct	Vulnerable	Known
Black falcon	Falco subniger	Vulnerable		Known
Black-breasted Buzzard	Hamirostra melanosternon	Vulnerable		Known
Black-chinned Honeyeater (eastern subspecies)	Melithreptus gularis gularis	Vulnerable		Known
Black-necked stork	Ephippiorhynchus asiaticus	Endangered		Known
Black-striped wallaby	Macropus dorsalis	Endangered		Known
Black-throated finch (southern subspecies)	Poephila cincta cincta	Endangered	Endangered	Known
Blue-billed duck	Oxyura australis	Vulnerable		Known
Booroolong frog	Litoria booroolongensis	Endangered	Endangered	Known
Border thick-tailed	Underwoodisaurus sphyrurus	Vulnerable	Vulnerable	Known

Gecko				
Bristle-faced free-tailed bat, hairy-nosed freetail bat	*Mormopterus eleryi*	Endangered		Known
Brolga	*Grus rubicunda*	Vulnerable		Known
Brown treecreeper (eastern subspecies)	*Climacteris picumnus victoriae*	Vulnerable		Known
Brush-tailed phascogale	*Phascogale tapoatafa*	Vulnerable		Known
Brush-tailed rock-wallaby	*Petrogale penicillata*	Endangered	Vulnerable	Known
Bush stone-curlew	*Burhinus grallarius*	Endangered		Known
Comb-crested jacana	*Irediparra gallinacea*	Vulnerable		Known
Corben's long-eared bat	*Nyctophilus corbeni*	Vulnerable	Vulnerable	Known
Cotton pygmy-goose	*Nettapus coromandelianus*	Endangered		Known
Curlew sandpiper	*Calidris ferruginea*	Endangered		Known
Delicate mouse	*Pseudomys delicatulus*	Endangered		Known
Diamond firetail	*Stagonopleura guttata*	Vulnerable		Known
Dunmall's snake	*Furina dunmalli*		Vulnerable	Known
Eastern bentwing-bat	*Miniopterus schreibersii oceanensis*	Vulnerable		Known
Eastern cave bat	*Vespadelus troughtoni*	Vulnerable		Known
Eastern false pipistrelle	*Falsistrellus tasmaniensis*	Vulnerable		Known
Eastern grass owl	*Tyto longimembris*	Vulnerable		Known
Eastern hare-wallaby	*Lagorchestes leporides*	Presumed Extinct	Extinct	Known
Eastern pygmy-possum	*Cercartetus nanus*	Vulnerable		Predicted
Five-clawed worm-skink	*Anomalopus mackayi*	Endangered	Vulnerable	Known
Flame robin	*Petroica phoenicea*	Vulnerable		Known
Flock bronzewing	*Phaps histrionica*	Endangered		Known
Freckled duck	*Stictonetta naevosa*	Vulnerable		Known
Giant dragonfly	*Petalura gigantea*	Endangered		Predicted
Glandular frog	*Litoria subglandulosa*	Vulnerable		Known
Glossy black-cockatoo	*Calyptorhynchus lathami*	Vulnerable		Known
Golden-tipped bat	*Kerivoula papuensis*	Vulnerable		Known
Gould's mouse	*Pseudomys gouldii*	Presumed Extinct	Extinct	Known
Greater broad-nosed Bat	*Scoteanax rueppellii*	Vulnerable		Known
Green and golden bell frog	*Litoria aurea*	Endangered	Vulnerable	Known

Grey falcon	*Falco hypoleucos*	Endangered		Known
Grey-crowned babbler (eastern subspecies)	*Pomatostomus temporalis temporalis*	Vulnerable		Known
Grey-headed flying-fox	*Pteropus poliocephalus*	Vulnerable	Vulnerable	Known
Hoary wattled bat	*Chalinolobus nigrogriseus*	Vulnerable		Known
Hooded robin (south-eastern form)	*Melanodryas cucullata cucullata*	Vulnerable		Known
Koala	*Phascolarctos cinereus*	Vulnerable	Vulnerable	Known
Large-eared pied bat	*Chalinolobus dwyeri*	Vulnerable	Vulnerable	Known
Little eagle	*Hieraaetus morphnoides*	Vulnerable		Known
Little lorikeet	*Glossopsitta pusilla*	Vulnerable		Known
Little pied bat	*Chalinolobus picatus*	Vulnerable		Known
Magpie goose	*Anseranas semipalmata*	Vulnerable		Known
Masked owl	*Tyto novaehollandiae*	Vulnerable		Known
New Holland mouse	*Pseudomys novaehollandiae*		Vulnerable	Known
Painted honeyeater	*Grantiella picta*	Vulnerable		Known
Pale imperial hairstreak	*Jalmenus eubulus*	Critically Endangered		Known
Pale-headed snake	*Hoplocephalus bitorquatus*	Vulnerable		Known
Powerful owl	*Ninox strenua*	Vulnerable		Known
Red-tailed black-cockatoo (inland subspecies)	*Calyptorhynchus banksii samueli*	Vulnerable		Known
Regent honeyeater	*Anthochaera phrygia*	Critically Endangered	Endangered	Known
Rufous bettong	*Aepyprymnus rufescens*	Vulnerable		Predicted
Scarlet robin	*Petroica boodang*	Vulnerable		Known
Sloane's froglet	*Crinia sloanei*	Vulnerable		Known
Sooty owl	*Tyto tenebricosa*	Vulnerable		Known
Southern myotis	*Myotis macropus*	Vulnerable		Known
Speckled warbler	*Chthonicola sagittata*	Vulnerable		Known
Spotted harrier	*Circus assimilis*	Vulnerable		Known
Spotted-tailed quoll	*Dasyurus maculatus*	Vulnerable	Endangered	Known
Square-tailed kite	*Lophoictinia isura*	Vulnerable		Known
Squatter pigeon	*Geophaps scripta*	Endangered	Vulnerable	Known
Squirrel glider	*Petaurus norfolcensis*	Vulnerable		Known
Stripe-faced dunnart	*Sminthopsis macroura*	Vulnerable		Known
Stuttering frog	*Mixophyes balbus*	Endangered	Vulnerable	Predicted
Swift parrot	*Lathamus discolor*	Endangered	Endangered	Known

Turquoise parrot	*Neophema pulchella*	Vulnerable		Known
Tusked frog	*Adelotus brevis - endangered population*	Endangered Population		Known
Varied sittella	*Daphoenositta chrysoptera*	Vulnerable		Known
White-fronted chat	*Epthianura albifrons*	Vulnerable		Known
Wompoo fruit-dove	*Ptilinopus magnificus*	Vulnerable		Predicted
Yellow-bellied glider	*Petaurus australis*	Vulnerable		Known
Yellow-bellied sheathtail-bat	*Saccolaimus flaviventris*	Vulnerable		Known
Yellow-spotted tree frog	*Litoria castanea*	Critically Endangered	Endangered	Known
Zigzag velvet gecko	*Oedura rhombifer*	Endangered		Known

Table 3: NSW and Commonwealth listed Ecological Communities known and predicted to occur in the BRG region (NSW TSC Act, 1995 & Commonwealth EPBC Act, 1999)

Endangered Ecological Communities of the Border Rivers-Gwydir Region

Scientific Name	Conservation Status		Occurrence
	NSW Status	Commonwealth (EPBC) Status	
Brigalow	Endangered Ecological Community	Endangered	Known
Cadellia pentastylis (Ooline)	Endangered Ecological Community		Known
Carbeen Open Forest	Endangered Ecological Community		Known
Carex Sedgeland	Endangered Ecological Community		Known
Coolibah-Black Box Woodland	Endangered Ecological Community	Endangered	Known
Fuzzy Box Woodland	Endangered Ecological Community		Known
Howell Shrublands	Endangered Ecological Community		Known
Inland Grey Box Woodland	Endangered Ecological Community	Endangered	Known
Marsh Club-rush Sedgeland	Critically Endangered Ecological Community		Known
McKies Stringybark/Blackbutt Open Forest	Endangered Ecological Community		Known
Myall Woodland	Endangered Ecological Community	Endangered	Known
Natural grasslands on basalt and fine-textured alluvial plains of northern New South Wales and southern Queensland		Critically Endangered	Known
New England Peppermint (Eucalyptus nova-anglica) Woodland	Critically Endangered Ecological Community	Critically Endangered	Known
Ribbon Gum-Mountain Gum-Snow Gum	Endangered Ecological		Known

Grassy Forest/Woodland	Community		
Semi-evergreen Vine Thicket	Endangered Ecological Community	Endangered	Known
Upland Wetlands	Endangered Ecological Community	Endangered	Known
White Box Yellow Box Blakely's Red Gum Woodland	Endangered Ecological Community	Critically Endangered	Known

6. VEGETATION COMMUNITY AND FLORA SURVEYS OF THE BRG REGION

The first recorded European to traverse the Northern Tablelands was the Surveyor General John Oxley on his expedition of 1818 The botanist Alan Cunningham accompanied Oxley on this expedition, travelling from the Warrumbungle Range through the Pilliga Scrub, across the Liverpool Plains and crossing the New England Tableland to arrive at Port Macquarie on the coast. Oxley described the landscapes and vegetation he encountered on this expedition (Oxley 1820).

Later, in 1827, Cunningham returned to the region, traversing the Liverpool Plans and travelling through the Bingara, Warialda and Yetman districts. His notes on the fertility and agricultural potential of the landscape lead to rapid European colonisation and the introduction of domestic stock (Cunningham 1825). This led to rapid degradation and massive erosion of many landscapes and caused substantial early modification of native vegetation communities across the region.

A limited number of botanical explorations and publications relating to the BRG region were made during the early 1900s (Maiden 1906; Maiden & Betche 1908) and the botanical surveys of Cambage across the region (Cambage 1904, 1908, 1912). Publications containing mapping and detailed floristic descriptions of the vegetation of the BRG region did not commence until the 1960s, when botanists and students from the University of New England (UNE) began undertaking detailed botanical investigations of many parts of the region. These early botanical investigations were led by Professor J.B. Williams, who continued his detailed botanical investigations until the turn of the millennium (Williams 1969, 1972, 1991, 1992, 1994, 1998; Williams & Wissmann 1991). Professor Williams taught and inspired the work of many of the contemporary botanical experts studying the vegetation of the BRG region.

In more recent times botanical investigations have been completed in various conservation reserves (publicly and privately owned as well as Indigenous Protected Area) as well as Crown Lands, State Forests and privately-owned native vegetation (Roberts 1983,1992; Hunter & Williams 1994; Hunter 1995, 1997a, b, c, 1998a, b, 2003, 2006a, b, 2009a, b, c, d, e, f, g, h, I, j, k, 2010, 2011b, 2012a, b, c; Hunter & Bruhl 1996; Hunter *et al.* 1996, 1998; Richards 1996; Richards & Hunter 1997). Many of these investigations have been completed by academics associated with UNE, (notably Dr John Hunter) and plant ecologists, botanists and other researchers associated with the Royal Botanic Gardens (RBG), notably Dr John Benson and results published across several volumes of Cunninghamia. The vegetation of the Guyra 1:100000 mapsheet was surveyed and the conservation status of vegetation communities elucidated by Benson and Ashby (1999); this work forming a precursor and important component in the BRG region of the NSW Statewide Vegetation Classification and Assessment Database VCA project managed by the RBG (Benson 2006, 2008; Benson *et al.* 2006, 2010)

In 1999 the NSW Government commenced a bioregional assessment of the Brigalow Belt South and Nandewar Bioregions involving extensive and intensive plot-based sampling of native vegetation within most areas of public land across the BRG region. This culminated in the Western Regional Forest Agreement and resulted in the dedication of many areas of public land as State Conservation Areas (RACAC, 2004).

In still more recent times a number of statewide and regionally vegetation classifications have been adopted (Keith 2004; Benson *et al.* 2010). These vegetation classifications describe distinct vegetation formations, subformations, classes, communities and associations that occur within the BRG region. From this work it is clear that the BRG region supports an incredible diversity of vegetation communities, many either limited to the region, or as broader vegetation classes occurring exclusively in NSW within the region. Examples of vegetation classes that are endemic, mostly confined to, or with the largest remaining areas in NSW occurring in the BRG region include Brigalow Clay Plain Woodlands, North-west Alluvial Sand Woodlands, Semi-arid Floodplain Grasslands and Yetman Dry Sclerophyll Forests. Many finer-scale vegetation communities and associations are known to be restricted or mostly limited to the BRG region for example Carbeen Open Forest, Ooline Dry Rainforest, Howell Shrublands and Marsh Club-rush Sedgeland (Table 3). The conservation and future of these communities depends entirely on sympathetic and appropriate management within the BRG region (Benson *et al.* 2010).

Table 1: Vegetation Formations and Classes (Keith) occurring in the Border Rivers–Gwydir region

VEGETATION FORMATION	KEITH VEGETATION CLASS
ARID SHRUBLANDS *ACACIA SUBFORMATION* **1 vegetation community**	• North-west Plain Shrublands
CHENOPOD SUBFORMATION **4 vegetation communities**	• Riverine Chenopod Shrublands
SEMI-ARID WOODLANDS *GRASSY SUBFORMATION* **9 vegetation communities**	• North-west Floodplain Woodlands • Riverine Plain Woodlands • Brigalow Clay Plain Woodlands
SHRUBBY SUBFORMATION **7 vegetation communities**	• Western Peneplain Woodlands • Subtropical Semi-arid Woodlands • North-west Alluvial Sand Woodlands
DRY SCLEROPHYLL FORESTS *SHRUB/GRASS SUBFORMATION* **24 vegetation communities**	• New England Dry Sclerophyll Forests • Northern Gorge Dry Sclerophyll Forests • North-west Slopes Dry Sclerophyll Woodlands • Pilliga Outwash Dry Sclerophyll Forests
SHRUBBY SUBFORMATION **35 vegetation communities**	• Northern Escarpment Dry Sclerophyll Forests • Northern Tableland Dry Sclerophyll Forests • Western Slopes Dry Sclerophyll Forests • Yetman Dry Sclerophyll Forests
HEATHLANDS **3 vegetation communities**	• Northern Montane Heaths
FORESTED WETLANDS **5 vegetation communities**	• Eastern Riverine Forests • Inland Riverine Forests
FRESHWATER WETLANDS **14 vegetation communities**	• Montane Bog and Fens • Montane Lakes • Inland Floodplain Shrublands • Inland Floodplain Swamps
GRASSLANDS **13 vegetation communities**	• Semi-arid Floodplain Grasslands • Temperate Montane Grasslands • Western Slopes Grasslands
GRASSY WOODLANDS **27 vegetation communities**	• Western Slopes Grassy Woodlands • Subalpine Woodlands • New England Grassy Woodlands • Tableland Clay Grassy Woodlands • Floodplain Transition Woodlands
WET SCLEROPHYLL FORESTS *SHRUBBY SUBFORMATION* **4 vegetation communities**	• Northern Escarpment Wet Sclerophyll Forests
GRASSY SUBFORMATION **7 vegetation communities**	• Northern Tableland Wet Sclerophyll Forests • Northern Hinterland Wet Sclerophyll Forests
RAINFORESTS **6 vegetation communities**	• Northern Warm Temperate Rainforests • Dry Rainforests • Western Vine Thickets

(OEH, nd; Benson et al. *2010)*

7. VEGETATION COMMUNITIES OF THE BRG REGION

The Border Rivers and Gwydir Rivers arise at an elevation of approximately 1400 m and 1500 m respectively at the crest of the Great Dividing Range (upper tributaries of the Horton River also begin at elevations in excess of 1400 m near the peak of Mt Kaputar). The Gwydir River then flows almost 700 km to the confluence with the Barwon River to the east of Collarenebri (*Figure 2*). The catchment is composed of three distinct landscapes: tablelands, slopes and plains. Sections of four bioregions are contained within the BRG region. From east to west they are New England Tableland, Nandewar, Brigalow Belt South and Darling Riverine Plains. The great diversity of landscapes, substrates, soils types and climatic conditions in the BRG region generates the conditions for the existence of an exceptional diversity of native vegetation communities.

The NSW Office of Environment and Heritage (OEH, nd) determined that 157 Regional Vegetation Communities (RVCs) occur within the BRG region. These 157 RVCs align with 37 State (Keith) vegetation classes and 14 formations or subformations (*Table 1*). Of these 37 classes, 16 align with Final Determinations for listing as Endangered Ecological Communities (*Table 2*) under the NSW Threatened Species Conservation Act (1995), a further 8 are listed as Threatened Ecological Communities (*Table 3*) under the Commonwealth Environmental Protection and Biodiversity Conservation Act (1999).

VEGETATION CLASSES OF LIMITED EXTENT OF OCCURRENCE IN THE BRG REGION

Several classes of vegetation identified as occurring within the BRG region by OEH (2011) are of limited extent. This includes Northern Gorge Dry Sclerophyll Forests and Northern Hinterland Wet Sclerophyll Forests (both possibly limited to very small areas east and southeast of Tenterfield), Northern Escarpment Wet Sclerophyll Forests (limited to relatively small areas in the Northeast Forest Lands subregion), Northern Warm Temperate Rainforests (limited to several gullies at the headwaters of the Deepwater River within Capoompeta National Park, Dry Rainforest (limited to small patches in ravines along the Nandewar and Kaputar Ranges), North-west Plain Shrublands (limited to small areas dominated by leopardwood on the Barwon and Gwydir plains) and Western Slopes Grassland (limited to small patches in the southern central sections of the BRG region).

8. THE VEGETATION OF THE BRG REGION

Benson *et al.* (2010) describe the location, size and the vegetation communities that occur in the BRG region, in particular in the Brigalow Belt South, Nandewar and western New England Tableland bioregions. The BRG region includes part of four bioregions spanning from rainforests to Arid Shrublands. Benson *et al.* (2010) and OEH (nd) have described the vegetation of the BRG region and delineated vegetation units at a finer scale than the formations and classes which are recognised for New South Wales (Keith 2004, Table 1).

This includes 157 regional vegetation communities (RVCs). Many Keith classes present in the BRG region are represented by numerous RVCs with the largest diversity within the Dry Sclerophyll Forests (59), Grassy Woodlands (27), Semi-arid Woodlands (16), Freshwater Wetlands (14) and Grasslands (13).

The literature on fire within different vegetation types is heavily biased towards particular vegetation formations and, in some cases, particular classes within a vegetation formation. Literature is very limited for some formations and classes. This literature review focuses on those vegetation types which occupy the largest areas and are represented by the most RVCs in the region. Hence, literature is reviewed for Dry Sclerophyll Forests, Grassy Woodlands, Semi-arid Woodlands, Grasslands (which together represent a total of 115 RVCs) and also Wet Sclerophyll Forests and Heathlands (because the literature is either relative large or particularly informative for these two formations). Rainforests are also briefly considered. The review starts with Grasslands (because the understorey of Grassy Woodlands often includes many Grassland species) and then moves to Grassy Woodlands. Systems that include prominent tree and/or shrub layers are then reviewed (Forests and Heaths).

i. FIRE IN GRASSLANDS

Extensive natural Grasslands were a prominent feature of the pre-European vegetation of the BRG region (Keith 2004; Benson *et al.* 2010). These vegetation communities are considered to have contained isolated, or no, trees and shrubs (Benson 1994; Keith 2004). Natural Grasslands mostly occupied higher fertility and flatter landscapes that have been extensively converted to cropping or grazing uses (Keith 2004; Prober & Thiele 2006).

The structural and floristic composition of the natural Grasslands of the BRG region varies widely as a result of the interplay between climate, fire, landscape productivity and altitudinal influences. Most natural Grasslands in the BRG region have a high proportion of exotic plants and few, if any, are free of weeds. Within the BRG region all the Temperate Montane Grassland communities and most Western Slopes Grassland communities are critically fragmented and reduced to highly degraded and widely scattered remnants (Benson 1997; Benson & Ashby 2000; Benson *et al.* 2010). Whilst Semi-arid Floodplain Grasslands and variants in the BRG region are slightly more intact and extensive than the more easterly and higher elevation Grassland types, they are still critically fragmented and very poorly reserved. Most native Grasslands in the BRG fall under the definition of the Critically Endangered Ecological Community "Natural Grasslands on basalt and fine-textured alluvial plains of northern New South Wales and southern Queensland" listed under the EPBC Act. This determination includes elements of all 3 native Grassland (Keith) classes in the BRG region.

Because of their extensive use for livestock grazing purposes, native Grasslands in the BRG region have been the subject of research focusing on these management outcomes. Notable examples of management-focused research for the BRG region include Lodge & Whalley (1989), who provided a description, review and bibliography of native Grasslands of the tablelands and slopes, and Nadolny *et al.* (2009), who provided advice on conservation and grazing management issues associated with native Grasslands to the BRG Catchment Management Authority. The Grasslands of the Moree Plains have been studied by McGann & Earl (1999) who sampled, classified and described several Grassland communities occurring in this landscape, all variants of the Semi-arid Floodplain Grassland class, whilst Clarke *et al.* (1998b) undertook a detailed survey of native Grasslands within Kirramingly Nature Reserve, the largest (and only) reserved patch of the Mitchell grass/Queensland bluegrass community in NSW (NPWS, 2003).

OEH (nd) and Benson *et al.* 2010 identified 13 regional vegetation classes (RVCs) of Grasslands for the BRG region. These comprise three Keith classes (Temperate Montane Grassland; Western Slopes Grassland; Semi-arid Floodplain Grassland – Keith 2004). At the state scale, Keith has mapped relatively small areas of the Temperate Montane Grasslands in northern New South Wales between Guyra and Tingha (with a large area in the Monaro). The Western Slopes Grasslands are mapped as occupying a limited area of the southern central portion of the region, whilst the Semi-arid Floodplain Grasslands occur more extensively across the north-west. These Grasslands occur across very different landscapes with differing dominant grasses (*Themeda australis/Poa labillarderi/Austrodanthonia caespitosa*; *Aristida leptopoda/Austrostipa aristiglumis*; *Astrebla leptopoda* respectively). The fire ecology literature is not evenly spread among these Grassland types (see below), thus the fire ecology of Australian Grasslands is discussed in general and then considered for these specific Grassland classes.

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

O'Connor 1991) and the fire ecology of these southern *Themeda* Grasslands is therefore an important body of literature.

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

This time since fire effects are also likely to interact with the spatial scale of a fire because seed dispersal capacity may be a limiting factor for some species (Lunt & Morgan 1999a). Hence completely unburnt sites (i.e. large sites completely unburnt or small isolated long unburnt sites) appear difficult to restore from the seed bank by burning alone (Lunt & Morgan 1999a; Morgan 1999). These sites are likely to have exhausted the seed bank of non-dormant seeds. However, at least some of the forb species in these Grasslands employ a counter-intuitive life history strategy. Rather than being small short lived highly dispersive species, some employ a persistence strategy (Bond & Midgley 2001). Seedling establishment appears to be a relatively rare event in these *Themeda* Grasslands but it is generally associated with fire (Morgan 1998d: Morgan 2001). Thus, life time fitness (total off-spring from an individual) is related to survivorship through time to maximize reproductive opportunity when a fire event occurs.

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

Watson (2007) also found compelling evidence that the competitiveness of *Themeda* may provide resilience to the Grassland systems it dominates in relation to weed invasion. This evidence comes from negative associations of weediness with *Themeda* (based on Morgan 1998d; Lunt and Morgan

1999b; Cole & Lunt 2005; Watson 2005) and also from studies that have investigated nitrogen levels and regulation in woodlands with a *Themeda* dominated ground layer (Prober *et al.* 2002b; Prober *et al.* 2005). Regeneration of *Themeda* combined with high frequency fire may thus regulate (reduce) the elevated nutrient status of weed infested sites and promote system resilience. However, there needs to be some caution attached to its widespread application as "other weed species may be well equipped to take advantage of the 'stable invasion window' provided by frequent fire (Morgan 1998d; Setterfield *et al.* 2005). Concerns that this might apply to perennial grass species with high invasive potential such as *Eragrostis curvula* (African lovegrass), *Nassella neesiana* (Chilean Needle Grass) and Serrated Tussock (*Nassella trichotoma*), have been expressed (Stuwe 1994; Nadolny *et al.* 2003, Marshall & Miles 2005). Yet there are examples of results (Lunt & Morgan 2000) and applications (Johnson 1999) of integrated strategies that included fire in the management of weed species in grassy systems.

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

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

Grasslands of the BRG region

Temperate Montane Grasslands

Temperate Montane Grasslands occur between 600 and 1500 m above sea level, and receive between 500 and 750 mm of rainfall each year. Composition varies with altitude, topography and soil type. In montane valleys on the New England Tableland small areas of snow grass-dominated Grassland occur in valleys subject to cold air drainage. These are now mainly cleared or heavily modified and are dominated by *Poa labillardierei* var. *labillardierei*, *Pennisetum alopecuroides*, *Carex appressa* and *Poa sieberiana* (Benson *et al.* 2010).

Clay soils are generally dominated by the tussock grasses *Themeda australis* (Kangaroo Grass) and *Poa sieberiana* (Snowgrass), drainage lines are often filled with dense *Poa labillardieri* (Tussock), while the sandy clay loams formed from granite tend to be dominated by *Austrostipa* (Speargrass), *Austrodanthonia* (Wallaby Grass) or *Bothriochloa* (Red Grass) (Keith 2004). This latter community, or parts of it, may be a degraded form of the former, having lost its *Themeda* to grazing over time (Benson 1994; Marshall & Miles 2005; Wong *et al.* 2006). Amongst the tussock grasses which dominate Temperate Montane Grasslands grow forbs, including daisies, lilies, peas and orchids (Lunt *et al.* 1998; Keith 2004). Unfortunately, exotic grasses and forbs are also common (Benson 1994; Dorrough *et al.* 2004). Modification through the use of fertilizer and sowing of exotic herbs and

grasses has also been widespread (Benson 1994; Semple 1997). Remaining native species-dominated remnants are therefore a valuable conservation resource (Eddy *et al.* 1998).

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

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

More recent work has confirmed aspects of Stuwe and Parson's theory. *Themeda australis* does indeed grow rapidly after fire (Morgan 1996; Lunt 1997b), so that by three years post-fire, gaps between *Themeda* tussocks in Victoria's lowland Grasslands have mostly disappeared (Morgan 1998a). A study by Lunt and Morgan (1999a) confirmed that species richness is significantly reduced in patches where *Themeda* is dense. Studies have found that forb seedlings need gaps to survive and grow (Hitchmough *et al.* 1996; Morgan 1997, 1998a), that regular burns can increase gap size (Henderson & Hocking in prep.), and that short inter-fire intervals are important for maintaining populations of adult interstitial species (Coates *et al.* 2006).

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

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

Other studies point to an important characteristic of the great majority of native species in these Grasslands: they tend to rely on persistence of existing individuals, rather than on recruitment of

new individuals. Although there are some annual and biannual species, most are perennials, and *all* perennial species resprout after fire (Lunt 1990; Morgan 1996). Unlike many Heathland species, Grassland perennial forbs tend *not* to have a large permanent store of seed in the soil (Morgan 1995a,b; 1998b). Many species germinate easily and rapidly, and are not inhibited by darkness (Willis & Groves 1991; Lunt 1995, 1996; Morgan 1998c), characteristics which imply that seedbanks will be rapidly depleted by germination. On the other hand, Grassland species almost all flower within the first year after a fire (Lunt 1990; Morgan 1996, 1999), and flowering effort for many forbs is concentrated in the first post-fire year, dropping considerably in year two (Lunt 1994). These species are therefore using the third strategy discussed in Section 1.2.2 for ensuring seed is available after a fire: creating seeds rapidly. Post-fire rainfall is also almost certainly an important determinant of recruitment success for Grassland species (Morgan 1998c, 2001).

Maintaining matrix grass vigour. Research has shown that in many situations, fire plays a vital role in maintaining the vigour of the Grassland matrix species *Themeda australis*. *Themeda* dominated large areas of temperate Australia prior to European settlement (Prober & Thiele 1993; Nadolny *et al.* 2003; Prober & Thiele 2004), and in some places, continues to do so today. Research has also found that a healthy *Themeda* sward can, in turn, limit or reduce weed invasion; this is discussed in the next section. Studies from Grasslands and Grassy Woodlands are included in this and the following section.

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

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

Why does fire maintain *Themeda* vigour? Periodic defoliation appears to prevent the self-shading which suppresses tiller production (Bond 1997; Morgan & Lunt 1999). Fire removes the thick mulch of dead material which prevents seedling regeneration (Morgan & Lunt 1999). Smoke may play a role in cuing seed germination in *Themeda* (Baxter *et al.* 1994; Wood 2001), although not all studies have found this to be the case (Clarke *et al.* 2000). Summer-growing C4 grasses such as *Themeda* use water more efficiently and have lower nutrient requirements than all-season and winter-growing C3 grasses like *Microlaena stipoides*, *Poa* and *Austrodanthonia* species (Ojima *et al.* 1994; Nadolny *et al.*

2003), and these characteristics may give C4 species a competitive advantage in a frequently-burnt environment (Fynn *et al.* 2003).

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

An additional factor in the *Themeda* story concerns its response to grazing. Many researchers have noted a decline in *Themeda* with moderate to heavy grazing (Prober & Thiele 1995; Chilcott *et al.* 1997; Fensham 1998; McIntyre *et al.* 2003; Dorrough *et al.* 2004). Where grazing has reduced but not eliminated *Themeda* from the grass sward, fire may play a useful role in its retention and regeneration (Prober & Thiele 2005).

Are other matrix grasses likely to respond in a similar fashion to fire? The ability of C4 species to use nitrogen and water efficiently suggests these species are more likely than C3 grasses to have a positive response to fire. Some C3 grasses, particularly *Poa sieberiana*, may also be encouraged, or at least not discouraged, by regular burning. Tasker (2002) found that *Poa sieberiana* was considerably more abundant in frequently burnt sites in the New England Tablelands than in sites which had not had a fire for many years. In Snow Gum woodland in Namadgi National Park Kelly (2004) was unable to detect a trend in the abundance of *Poa sieberiana* in plots subject to between 2 and 11 fires in a 41-year period.

Themeda and weeds. *Themeda australis* is of particular interest because it may be one of a small number of native grass species able to compete successfully against exotics (Cole & Lunt 2005). Morgan (1998d), and Lunt and Morgan (1999b) reported a significant negative correlation between *Themeda* abundance and species richness of exotic herbs in Victorian Grasslands, while Watson (2005) reported similar findings in Grassy Woodland near Sydney.

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

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

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

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

Integrated weed management, combining fire with other strategies, may provide answers in some situations. The post-fire environment presents opportunities to target weed species while in an active growth phase, before they flower, and while they do not overlap native species. Henderson & Hocking (nd) recommends a combination of periodic spot spraying, late spring burning, and *Themeda* establishment for managing CNG. In Grassland remnants in north-east Victoria, managers follow ecological burning in early summer with herbicide applications to post-fire regrowth of perennial exotic grasses, particularly *Paspalum dilatatum* and *Phalaris aquatic* (Johnson 1999). The extent to which various combinations of fire and other strategies can assist in deterring aggressively invasive weeds in temperate grassy vegetation would be an excellent subject for adaptive management (Bradstock *et al.* 1995; Lunt & Morgan 1999a).

Western Slopes Grasslands

Western Slopes Grasslands occupied an extensive area across the northwest of NSW including the Liverpool Plains and adjoining landscapes before European settlement (Yates & Hobbs 2000). The heavy basalt derived clay soils of the plains are regarded as having supported botanically distinct Grassland within an area of generally lower rainfall (400-650 mm) compared to the Temperate Montane Grasslands to the east and south (Keith 2004). Today, these Grasslands have generally been heavily modified by agriculture and remnants often occupy small roadside strips or fragmented patches in the landscape. This Grassland class is distinguished by the dominance of *Austrostipa aristiglumis* (Plains Grass) and common grass genera of the Temperate Montane Grasslands such as *Poa* are generally absent. *Dichanthium sericeum* subsp. *sericeum, Austrodanthonia bipartita* and *Aristida leptopoda* and the daisy *Leiocarpa panaetioides* are also dominant in contemporary Western Slopes. Prior to European settlement and the arrival of domestic stock this Grassland probably contained a higher cover of palatable grass species such as *Themeda avenacea* and *Eulalia aurea* (Lang 2008).

The Liverpool Plains are regarded as naturally treeless and historical records compiled by Lang (2008) support this view. However, although fire may be important in restricting woody species dominance in Grassy Woodlands and Semi-arid Woodlands of the BRG region (see reviews of these vegetation formations that follow), this may not be the case on the *Austrostipa aristiglumis* dominated plains. Lang (2008) reviewed the factors that drive Grassland occurrence and concluded that fine textured soils, topography and climate commonly interact to restrict water availability to

the sub-soil which then restricts woody vegetation growth in many Grassland systems. The heavy clay soils of the plains are thus considered to be a primary driver of the Grassland occurrence.

Lang (2008) also reports on early records that mention high levels of native mammals with numerous diggings that would have provided natural soil disturbance to the plains Grasslands and thus created colonisation opportunities for forbs. The dominance of *Austrostipa aristiglumis* is regarded by Lang to be an artefact of high grazing pressure and other changes post European settlement. There is limited historical evidence of aboriginal burning on these plains, however, given the high productivity of the landscape and reported extensive use of fire by Aboriginal people elsewhere, this remains an open question.

There appear to be no studies into the fire ecology of the *Austrostipa aristiglumis* Grasslands of the plains. These Grasslands are ecologically distinct from other Grasslands of the BRG region (i.e. they may not be "fire modified" where fire has removed woody vegetation, but may lack woody vegetation because of soil/water characteristics). Thus, it is problematic to infer appropriate fire regimes from other Grasslands. There are no references to the fire ecology of these Grasslands in prominent documents that include the Determination to list this community as an Endangered EcologicalCommunity
(*http://www.environment.nsw.gov.au/determinations/NativeVegetationCrackingClayEndComListing.htm*) under New South Wales legislation, a report on the technical workshop for listing the community as endangered under federal legislation
(*http://environment.gov.au/biodiversity/threatened/communities/pubs/Grassland-bbsouth-report.pdf*) or published literature which mentions these Grasslands (e.g. Prober & Thiele 2005 – who comment that these Grasslands are poorly understood).

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

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

lack of knowledge as to the role of fire in these Grasslands and no guidelines can be given as to how fire should be managed specifically for this Grassland type.

Semi-arid Floodplain Grasslands

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

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

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

Lewis *et al.* (2010) conducted experiments in Kirramingly Nature Reserve (near Moree in the BRG region) to investigate the influence of kangaroo grazing, fire, mowing and disc-ploughing on grassland species richness and composition. Variability due to rainfall between season and years was found to have a greater influence upon species composition than grazing, mowing, ploughing or fire. Recovery of species composition and richness after fire and mowing was found to be rapid because of the prevalence of resprouting perennial species. Lewis *et al (*2010) determined that no native species required frequent disturbance to persist within this grassland community. The findings of this study suggest that the high capacity of Mitchell Grass and Queensland Blue Grass to recover quickly after disturbances means that these grasslands are well adapted to the natural disturbances of fire, grazing by native species, drought and flooding.

Other Grasslands and variants

When Grassy Woodlands are cleared of their woody vegetation, leaving a native grassy understorey, derived Grasslands are created. Because of agricultural clearance and ongoing degradation of Grassy Woodlands, derived Grasslands are extensive across the BRG region (Benson *et al.* 2010). The Derived Grasslands generally fall within the *Themeda* grassy systems reviewed above. However,

there are some likely significant variants in terms of the fire ecology of these systems. Watson (2007) noted that *Austrodanthonia* and *Austrostipa* species can dominate on poorer soils and that these systems of lower productivity may require less frequent fire (cited Stuwe 1994; Johnson 1999; Kenny *et al.* 2004). *Austrostipa aristiglumis* dominated Grasslands are, however, reported from the Merriwa Plateau on basalt (high fertility at a high altitude - HCCREMS 2008). Therefore in Grasslands dominated by these species the appropriate fire frequency may not necessarily be richne (i.e. productivity may vary with altitude etc.). However, in the west of the BRG region *Austrostipa* spp. are regarded as generally of low persistence and low biomass and unable to carry fires except when two consecutive favourable years occur (Hodgkinson *et al.* 1984). Generally, gap closure among grass swards would indicate that a fire event would increase recruitment opportunities among tussocks. Factors such as the level of weed invasion, particularly of Coolatai Grass (*Hyparrhena hirta*), rainfall, season of burn and grazing pressure will then influence the outcome of this burn.

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

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

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

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

Conclusion

The findings outlined above have led to the conclusion that:

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

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

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

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

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

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

Application to the Grasslands of the BRG region

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

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

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

ii.   FIRE IN GRASSY WOODLANDS

Prior to European settlement of the BRG region, Grassy *Eucalyptus*-dominated woodlands dominated the extensive areas of nutrient-rich deep soils, including alluvial and volcanic soils derived from basalt, across the plains, lower slopes and gentle relief landforms from near the crest of the Great Dividing Range to the Barwon River floodplain (Benson & Ashby 2000; Benson *et al.* 2010). Because of their high productivity Grassy Woodlands have mostly been cleared for the development of cropping and grazing industries (Keith 2004), as a result most of the Grassy Woodland communities of the BRG region are listed as Endangered Ecological Communities under both the TSC Act (1995) and the EPBC Act (1999), see Table 3.

The BRG region supports a high diversity of Grassy Woodlands (OEH, nd). This diversity is primarily the result of the substantial variation in rainfall across the east-west gradient and the complex interactions of elevation, fire, topography, geology and soil formation processes. The climatic gradient present across the BRG region allows for the existence of a wide spectrum of Grassy Woodland classes from New England and Tableland Clay Grassy Woodlands in the east to Floodplain Transition Woodlands in the west.

OEH (nd) have identified 27 Grassy Woodland Regional Vegetation Communities (RVCs) which comprise four classes of Grassy Woodlands of Keith (2004 - Tableland Clay Grassy Woodland, New England Grassy Woodland, Western Slopes Grassy Woodland and Floodplain Transition Woodlands) of the BRG region, whilst Porteners (1997) has identified small areas of a fifth class of Grassy Woodlands, Subalpine Woodlands, as occurring on the highest plateaux of Mt Kaputar. These areas fall just within the south-western boundary of the BRG region. Because of the highly limited distribution of Subalpine Woodlands in the BRG region, they will not be addressed further in this review.

The floristic composition of Grassy Woodland habitats in the BRG is strongly driven by climatic and altitudinal gradients that range from an average annual rainfall exceeding 1200 mm per annum at the eastern edge of the New England Tableland to less than 400mm in the far west of the catchment and vary from in excess of 1500 m asl near Tenterfield, Guyra and at Mt Kaputar to less than 200 m asl on the far western plains near Collarenebri.

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

A considerable body of fire ecology research now exists for the Grassy Woodlands. This research has predominately focused on the Western Slopes Grassy Box Woodlands of New South Wales and on Cumberland Plain Woodland from western Sydney. These woodlands are structurally similar and share a number of species from the lower strata but have distinct tree species. Thus the research in

these Grassy Woodlands is of considerable value in relation to the Grassy Woodlands of the BRG region.

Western Slopes Grassy Woodland

OEH (nd) lists six vegetation communities within the Western Slopes Grassy Woodland class. In the higher rainfall and more productive parts of the BRG region these communities are dominated by White Box (*E. albens*), Yellow Box (*E. melliodora)*, Silver Ironbark (*Eucalyptus melanophloia*), Apple Box (*E. bridgesiana*), Blakelys Red Gum (*E. blakelyi*) and Rough-barked Apple (*Angophora floribunda*). On floodplains with lower rainfall (500-600 mm) further to the west, Fuzzy Box and Inland Grey Box dominate and in lower productivity sites and on slopes and ridges, Mugga (*E. sideroxylon*), Tumbledown Red Gum (*E. dealbata)* and White Cypress (*Callitris glaucophylla*) dominate these woodland communities.

The Ground layer

In the last decade there has been extensive ecological research undertaken in the grassy box woodlands of 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 savannah, for example, tree clearing can lead to undesired shifts in the grass composition of the ground layer (Barnes 1979). Given that tree loss from the Grassy Woodlands has been ongoing until at least the recent past (Fisher & Harris 1999), this is an issue to consider in these woodland systems. Gibbs *et al*. (1999) studied tree and grass interactions on the New England Tablelands and found evidence that there was an association among *Poa sieberiana* and *Microlaena stipoides* with the dominant tree *Eucalyptus laevopinea*. The deep-rooted summer growing *Aristida ramosa*, in contrast, was considered to be favoured away from tree canopies and root zones. This suggests that *Themeda* (also a summer growing grass) may be favoured away from tree canopies. This was confirmed in the study of Prober *et al*. (2002a) in which *Themeda* dominated in gaps and *Poa* dominated under tree canopy in box woodland. Watson (2005) also found significant differences in grass and forb species composition among open patches, patches around trees and patches under *Bursaria* bushes in Cumberland Plain Woodland. Similar interactions among ground layer species and shrubs have been found in other systems and nutrient patterns, water relations, herbivory levels and seed distributions are associated with these patterns (e.g. Facelli & Temby 2002).

Because overstorey cover and fire can both influence grass dominance there are likely to be complex interactions of tree and shrub cover with fire and the ground layer. For example, tree cover may favour *Poa* which appears to be more resilient to longer inter-fire periods. Tree cover in grassy box

woodlands was also associated with higher species richness. This could relate to altered nutrient status, water status or changes in the sward structure (i.e. recruitment opportunity) but it is also likely these factors interact. Watson (2005) did not find increased species richness under canopies in Cumberland Plain Woodland (CPW). 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 (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 (see Windsor 1999 for example). 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 1980) 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 *E. albens* has recently been reported as both a basal and epicormic resprouter following wildfire in the Warrumbungles National Park (Penny Watson *pers.obs 2014*). Additionally, seed characteristics of the eucalypts are generally considered consistent within the group. Eucalypt seed generally lacks dormancy (Grose & Zimmer 1958; Langkamp 1987) and does not form persistent soil seed banks (Ashton 1976; Vlahos and Bell 1986; Myerscough 1998; Read *et al.* 2000; Hill & French 2003) although some aerial seed storage does occur (House 1997). Eucalypt seed also has morphological features associated with low dispersal capacity (small size; high density; no dispersal appendages – see van der Pijl 1982) and related species with these features do have limited dispersal abilities (Tierney 2003). Of these attributes, the capacity to resprout post-fire is perhaps the major driver of eucalypt fire response. Yates *et al*. (2007) modelled population persistence in the rare Myrtaceous species *Verticordia staminosa* and found that adult survival contributed the most to long-term population persistence. Similarly, *Eucalyptus caesia* populations appear to be most buffered from decline because this species resprouts (Yates *et al*. 2003). These species occur in vegetation with an open structure. The coastal woodland species *Angophora inopina* is another resprouting species from the Myrtaceae that has been modelled to be highly resilient to altered fire frequencies in the short to medium term (Tierney 2004). In contrast to the general trend amongst woodland eucalypts, a number of forest eucalypts are seeder species and allocate more resources 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 CPW. Hill and French (2004) found both species richness and abundance of shrubs was significantly greater in plots burnt 18 months earlier, whether by planned or unplanned fire, than in unburnt plots. Similar findings have been obtained in Grassy Woodlands on the northern tablelands (Knox & Clarke 2006).

More recently, a survey in nine CPW remnants with differing fire histories found a high abundance of *Bursaria spinosa* in sites where fire frequency was low (these sites had been unburnt for at least 20 years prior to a recent fire), to the point where this species dominated much of the landscape. This finding accords with those from productive grassy ecosystems around the world, where increases in the density of woody plants in the absence of fire have been observed (Lunt 1998a, b; Roques *et al.* 2001; Allen *et al.* 2002; Bond *et al.* 2005). Patterns for native shrubs other than *Bursaria*, however, were different: these shrubs were more abundant in sites burnt once or twice a decade than in either low, or high, fire frequency sites. Obligate seeder shrubs were particularly influenced by fire

cycles: the abundance of these species was lowest in sites whose last interfere 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 BRG region?

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 – Redgum – 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.* 2006a). 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; Semple & Koen 1993), a relatively short juvenile period. As noted above, 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 Boxthorn, 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. (*2007) tentatively suggest fires at 5-8 year intervals may be compatible with maintaining a balance between the dominant grasses.

Application to Grassy Woodlands in the BRG region

1. Western Slopes Grassy Woodland

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

White Cypress Pine has the capacity to dominate sites in the absence of fire (see detailed ecology in the section on Dry Sclerophyll Forests that follows). This dominance results from recruitment in the absence of fire (it is a T-species – Noble & Slatyer 1980). Fire potentially has a role in regulating

White Cypress Pine dominance with intense fire leading to thinning of stands. This should be considered a potential strategy for managing these woodlands where appropriate.

2. New England Grassy Woodland

Eleven RVCs are included in this Keith class in the BRG region. Dominant trees include *Eucalyptus blakelyi*, *Eucalyptus bridgesiana*, *Eucalyptus caliginosa*, *Eucalyptus laevopinea*, *Eucalyptus melliodora* and *Eucalyptus youmanii*. *Eucalyptus blakelyi* has been the subject of studies by Li *et al.* (2003) which showed increased seedling survival growth rates with fire. Clarke (2000) also found that *Eucalyptus blakelyi* (and *Eucalyptus melliodora*) seedlings that survived over a five year period had developed lignotubers. Similarly Semple and Koen (2001) report lignotuber resprouting post-fire in both these species. In general, these woodlands have eucalypts that are likely to be resistant to fire intervals of more than ~5 years. Knox & Clarke (2004), however, studied the fire ecology of shrub species in two sites in New England Grassy Woodland. They concluded that the primary juvenile period of most shrub species was more than four years, whilst the secondary juvenile for resprouting shrub species was less than four years. They recommend that minimum intervals between fires should be ~8 years. This is longer than recommended by Prober *et al*. (2007) for Western Slopes Grassy Woodlands and Knox & Clarke (2004) caution about using recommended fire intervals from outside the New England region.

3. Tableland Clay Grassy Woodland

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

Snow Gum – Black Sallee Grassy Woodlands

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

4. Floodplain Transition Woodlands

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

The BRG region supports 3 Floodplain Transition Woodland communities (OEH, nd). These communities occupy several distinct landforms, including undulating peneplains atop sandstone, sandy alluvial rises, clay loam flood terraces and riverbanks. Species dominance varies accordingly, from Inland Grey Box (*E. microcarpa*) and Poplar Box (*E. populnea* ssp. *bimbil) to* White Cypress Pine (*Callitris glaucophylla*), Belah (*Casuarina cristata*) and Yellow Box (*E.melliodora*) (Austin *et al.* 2000; Prober and Thiele 2004). Small variations in topography as well as subtle changes in fluvial and aeolian processes are the main drivers of variation in floristic composition and structure within the Floodplain Transition Woodland vegetation class (Austin *et al.* 2000; Metcalfe *et al.* 2003).

Floodplain Transition Woodland communities that include Fuzzy Box (*E. conica*) Woodland on alluvial soils (*Table 3*) are gazetted as an Endangered Ecological Community under the NSW TC Act (1995). This gazettal recognises the very poor condition and reduced extent of these woodlands as well as their ongoing loss (NSW Scientific Committee, 2004). Remaining areas are highly fragmented and suffer the continued effects of grazing and invasion by agricultural weeds (Keith 2004). In many landscapes supporting floodplain transition woodlands, large "scalds" are frequent in areas where overgrazing has been severe (Beeston 1980; Keith 2004; Prober and Thiele 2004).

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

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

iii.   FIRE IN SEMI-ARID WOODLANDS

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

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

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

The BRG region supports sixteen regional vegetation communities (OEH nd) that combined represent six classes of Semi-arid Woodlands defined by Keith (2004); North-west Floodplain Woodlands; North-west Alluvial Sand Plain Woodlands; Sub-tropical Semi-arid Woodlands; Western Peneplain Woodlands; Riverine Plain Woodlands & Brigalow Clay Plain Woodlands). The BRG region supports the majority of the Brigalow Clay Plain Woodlands in NSW including the largest, best condition and best-connected areas. Brigalow Woodlands have been extensively cleared for cropping, with very little of a previously extensive expanse remaining. Brigalow Woodlands were

previously such a characteristic and abundant vegetation class that entire bioregions were named for it, those parts of the Brigalow Belt South bioregion within the BRG region well-illustrating this point. Because of the major loss of Brigalow Woodlands, they are regarded as Endangered Ecological Communities under State and Commonwealth legislation.

Fire in Semi-arid Woodlands is a complex issue. Noble & Grice (2002), for example, reviewed the issue of fire in semi-arid lands and it is clear from this review that there are complex interactions of fire with grazing, browsing and rainfall in these systems. It is also true that there is likely to be variability in how fire functions within differing woodland types. This means that reducing fire effects to simplistic fire frequency guidelines is both not possible and potentially counterproductive. This review concentrates on the literature for fire in these systems and what it indicates. Firstly, however, broad ecosystem models that are used to understand these woodlands are explained. Then the contribution of the literature to understanding these woodlands in relation to fire can be elucidated.

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

Figure 3
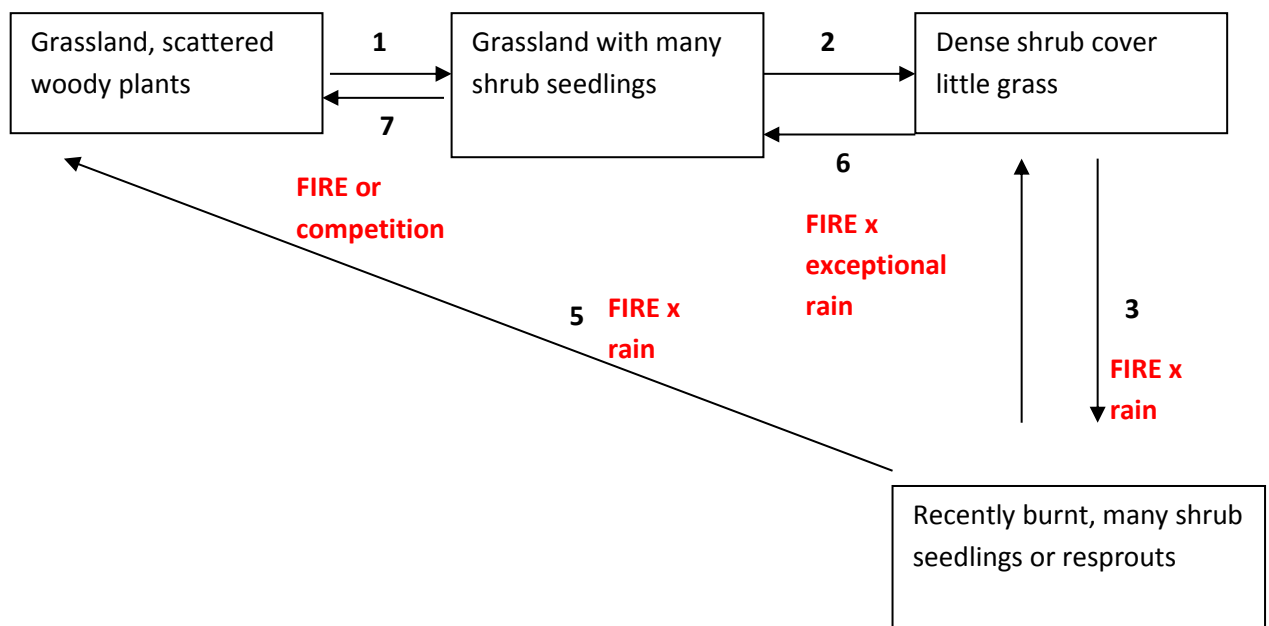
Fire is required for four of the seven transitions (shown as numbers), but it must interact with rainfall for three of these transitions to occur and the other transition (transition 7) can occur alternatively via grass competition with shrub seedlings (or perhaps browsing?). The requirement that fire must occur with other variables to cause ecosystem change is not unique to these Semi-arid Woodlands, however the variability in rainfall in the semi-arid zone means that there are likely to be high levels of stochasticity in these transitions. "Rainfall variability is typically between 25 and 30% of the mean, but can exceed 100%, and severe droughts have a frequency of recurrence of about 1:8 to 1:10." (Mitchell 1991).

Whilst the majority of Semi-arid Woodland classes in the BRG region have a eucalypt dominated tree layer (North-west Floodplain Woodlands; North-west Alluvial Sand Plain Woodlands; Sub-tropical Semi-arid Woodlands; Western Peneplain Woodlands), two classes are dominated by *Acacia* species (Riverine Plain Woodlands & Brigalow Clay Plain Woodlands). Hodgkinson (2002) used a different model: the trigger-transfer-reserve-pulse model (Ludwig *et al.* 1997) to explain the functioning of *Acacia* dominated woodlands:

Figure 4
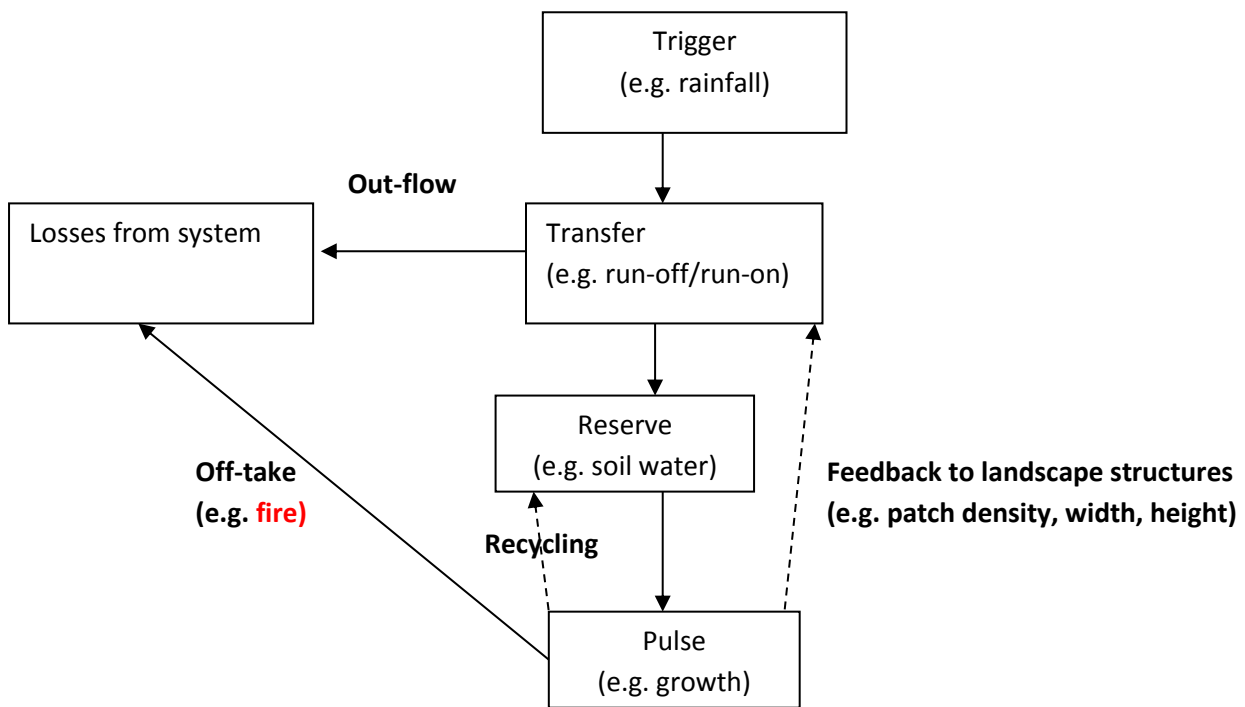
This model emphasizes the role of rainfall as the master trigger; the event that exerts the major influence on the system. It is then the transfer or storage of water that exerts the predominant control of the system (and consequently alterations to water flow/storage can produce major system dysfunction). Fire, along with grazing, are secondary mechanisms (removal of plant biomass – off-take). A range of fire adaptations and life history strategies in the plant species of these *Acacia* systems are reported. Many of these studies, however, are in Mulga (*Acacia aneura*) shrublands. This review covers Semi-arid Woodlands, including *Acacia* dominated woodlands (see below). Overall, Hodgkinson (2002) suggests that fire (or the use of prescribed fire) has not been proven by studies to be a tool to meet general biodiversity objectives in these *Acacia* systems.

Studies generally support a consistent story of fire in Semi-arid Woodlands. Perhaps the most difficult aspect of this is the pre-European fire regimes. Although caution needs to be attached to accounts of early European settlers (Benson & Redpath 1997), settlers and explorers did report an open Grassy Woodland structure across much of the area (Rothery 1970) and frequent burning by Aboriginal people (Nicholson 1981). Fire frequency is considered to have been a factor in

maintaining an open grassy structure before European settlement (Hodgkinson & Harrington 1985). This concept is supported by numerous studies that have demonstrated that many woody shrub species are susceptible as seedlings to fire (e.g. Hodgkinson & Griffin 1982) and that grass competition can restrict woody shrub establishment (e.g. Noble *et al*. 1986). However, drought coupled with overgrazing in the decades before 1900 dramatically altered the rangelands (e.g. Griffiths 2001; Lunney 2001) and extrapolating from present day studies is therefore problematic. Hence imposing pre-European settlement fire regimes is untenable and fire needs to be managed to a set of currently defined objectives (Hodgkinson 2002; Hodgkinson *et al*. 1984).

Fire regime

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

Walker *et al*. (1981) provided a detailed review of fire behaviour in Poplar Box Woodlands. Fire frequencies across the *full* area which these woodlands cover (from north of Longreach Qld to south of Hillston NSW) were estimated to vary greatly (down to almost every few years in northern regions). This was attributed to differences in fuel load of the grass species (estimated to vary from 50-600 g/m^2) and also differing rainfall and degree of shrub and tree cover. Native perennial grass species potentially accumulate fuel over years (depending on grazing etc.) and produce high fuel loads, compared to the annual species present. The invasive *Cenchrus cilliaris* (buffel grass) was noted as producing very high fuel loads (Clarke *et al.* 2005) leading to the potential for more frequent fire (potentially in most years). The minimum fuel to carry a fire depends on the grass distribution and moisture content, but was estimated to be ~100 g per m^2 in the Border Rivers – Gwydir region. Wind speed, temperature, humidity and fuel moisture content also influence the rate of fire spread. Thus, for *Eragrostis – Aristida* dominated systems (carrying 150-300 g per m^2 of fuel) minimum wind speeds of 7 km/h were required to carry fire in the BRG region, whereas for *Themeda* dominated systems (carrying 300 g per m2 of grass fuel) fire was reported to carry with wind speeds as low as 1 km/h. Graphs provided by Walker *et al*. (1981) show the relationship among fire intensity, flame length and fire spread and also fire risk (according to fuel moisture content, wind and rate of spread) for these woodlands. Walker *et al*. (1981) also report a trend for fire to occur after high rainfall (which triggers plant growth) and hence tabulate large wildfire events (1 million hectares or more) in relation to years (i.e. fires in 1957 followed high rainfall in 1956; fires in 1974-75 followed high rainfall in 1974). Fires in the Western Division were mostly started by lightning (77%) compared to estimates of 9-10% in other areas.

Fire ecology

The grass layer. A number of authors have investigated the fire ecology of plant species within these woodlands and the grass layer has drawn attention because of its importance in carrying fire. Walker *et al.* (1981) note the importance of the dichotomy among perennial species (e.g. *Themeda australis*; *Monochather paradoxa*) that persist (and therefore potentially can carry fire across long time periods) verses short-lived species that are only present for periods after rainfall. Other researchers report similar findings. Harrington *et al*. 1984, for example, found that the main fuel in Semi-arid Woodland was the cool season speargrass (*Stipa* spp.) which has low persistence year-to-year. These

short-lived species may also not persist as a seed bank through drought periods (Anderson *et al*. 1996). Other short-lived species in Semi-arid Woodlands include *Aristida* spp. and *Enneapogon* spp. (Daly & Hodgkinson 1996) and these have been recorded as recovering poorly after fire, at least in some western vegetations (Mulham 1985). In contrast, Shaw (1957) found that the grass *Heteropogon contortus* had relatively high survival, enhanced germination and was favoured by reduced competition by annual burning and increased in density in northern semi-arid (Poplar Box) woodlands. Tothill (1969) attributed the germination success of *Heteropogon contortus* to its seed's capacity to bury itself in soil and to germinate with increased soil temperatures which occur post-fire.

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

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

The grass x shrub/tree interaction. The interaction among the grass layer and shrubs is well established at the landscape level (Daly & Hodgkinson 1996). There is a negative relationship at the landscape level among shrub cover and grass cover. Shrub recruitment can be decreased by grass cover (Harrington 1991), but there are seasonal and species differences so that for some species in some seasons grass cover has been found to facilitate shrub seedlings (Booth *et al*. 1996 b). Thus there are potentially complex interactions among grasses and shrubs (Booth *et al*. 1996 a, b). Scanlan (2002) reviewed the literature on trees and their relationships with the grass layer in semiarid systems. The main points of this review were that: 1) most tree species reduce grass production, but some introduced leguminous species increase grass production; 2) grass can reduce tree seedling survival but have little impact on mature trees except via influencing fire events; 3) modeling indicated that differing outcomes can be produced from the tree/grass interaction and that tree clumping can significantly increase grass production. There is also evidence that trees are an important part of water dynamics and make a significant contribution to "water catching" rather than water shedding at the landscape level. These water dynamics are ecologically significant (Ludwig *et al*. 1997; Eldridge & Freudenberger 2005). Thus water flow/infiltration is considered a critical component to the functioning of the Semi-arid Woodlands (Ludwig *et al*. 1997) and this is therefore an important ecological role of trees (i.e. for rainfall driven growth and drives fire).

The shrub layer. In the absence of fire shrub cover increases (Hodgkinson & Harrington 1985). Since European settlement the evidence that there is suggests that decreased fire frequency has favoured shrub species (Hodgkinson 1986), whilst increased grazing has decreased grass cover (Harrington 1991). These two changes strongly interact. These are, however, broad patterns and there are also a range of differing/competing responses (e.g. grazing may also reduce shrub establishment – Auld

1995; grazing effects on shrubs can vary according to the shrub species, its life-cycle stage and the grazer species – Harrington *et al.* 1979).

The delicate management needs of Brigalow Clay Plain Woodlands: an evolutionary link from the wet Rainforests of the coast to the Dry Rainforests of the interior.

Stands of Brigalow have various rainforest elements and across the extensive Brigalow Belt are numerous small and scattered patches of Western Vine Thicket (a dry variant of Rainforest) with bright green leaves that contrast strongly with the silvery leaves of Brigalow stands. These are relictual patches that represent the evolutionary adaptations of Subtropical Rainforests genera to the increasing aridity experienced by Australia throughout the Tertiary period (White 1994).

In NSW Brigalow is known to be sensitive to fire (Benson *et al.* 2006) and in Queensland high intensity fires are known to kill the entire root system of mature trees, thus leading to loss of Brigalow stands (Johnson 1964). The pre-European fire regime of Brigalow Clay Plain Woodland is not known. Benson *et al.* (2006) highlight the presence of Brigalow between the Culgoa and Warrego rivers and the existence of associated rainforest genera, suggesting this may be the result of infrequent fires historically. Butler (2007) documents the low density of fine fuels in intact Brigalow stands and suggests that, historically, fire was likely to have been rare in this ecosystem.

Although many of the shrub and tree species associated with Brigalow are capable of resprouting after damage to their above ground parts (Butler 2007). Johnson and Purdie (1981) noted that in Brigalow open forests, six of the nine most common canopy trees and six of the eight most common understorey trees sucker freely from butts or horizontal roots and many survive fires of low to moderate intensity.

Native tree or shrub species known to form dense regrowth after clearing and burning of Brigalow vegetation include, *Alectryon diversifolius* (Scrub Boonaree), *Carissa ovata* (Blackberry, Currantbush, Kunker Berry), *Citrus glauca* (Limebush), *Eremophila mitchellii* (False Sandalwood) (Johnson 1964; Anderson 1984). Pressland (1982) noted that *Carissa ovata*, *Eremophila deserti* (Ellangowan Poison Bush) and *Eremophila mitchellii* are all susceptible to hot fires.

From these findings it can be concluded that, although some species inhabiting Brigalow Woodlands are capable of resprouting following fire, for the purposes of maintaining viable and ecologically functional areas of Brigalow Clay Plain Woodlands across the BRG region, fire should be excluded from this class of vegetation.

Conclusions

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

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

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

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

iv.   FIRE IN HEATHLANDS

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

Fire and the standing vegetation

Fire is a dominant force in the ecology of Heathlands in Australia and fire events can be intense because the fuel loads are high and the species present produce highly flammable litter (Keith *et al*. 2002). The plant species of these heaths are adapted to fire and there are predictable post-fire shifts in the vegetation over time (e.g. Keith & Bradstock 1994). These changes are largely predicted by the "vital attributes approach" of Noble and Slatyer (1980) and this approach has been specifically applied to Heathlands (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 & Midgley 2001) and these species generally cope with short fire intervals. However, recurrent fire may gradually exhaust the storage reserves of these species (e.g. Knox & Morrison 2005; Knox & Clarke 2006) and may weaken them and lead to their loss. There appears to be some variation in the proportion of resprouters among heaths. Myerscough *et al.* (1995) report that both wet and dry heath on dune ridges of Myall Lakes National Park (wallum heath) are largely dominated by resprouting species. About 27% of heath species in the study by Bradstock & Kenny (2003), which focused on Sydney coastal heath, could be assigned to the non-resprouting category.

Non-resprouting species ('obligate seeders') fit the 'regeneration niche model' (Grubb 1977). These species are predicted to be killed by fire but then regenerate from seed. There has been considerable interest and research into obligate seeder species in Heathlands. An informative species that has drawn particular attention is *Banksia ericifolia* (Siddiqi *et al*. 1976; Morris & Mysercough 1983, 1988; Nieuwenhuis 1987; Bradstock & O'Connell 1988; Keith & Bradstock 1994; Bradstock *et al*. 1997). This species has a long juvenile period (i.e. it may take up to eight years post-fire to mature and produce seed). 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 1986b). 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 (Bradstock & Auld 1995). 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; Denham & Auld 2002) 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.* 2006a) or events such as mechanical actions related to soil movements (Morris 2000).

In 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 determine the species diversity of a site.

Fauna

Whilst fire can cause death, lead to increased predation pressures, or decrease habitat values for some fauna (Fox 1978), even some large animals such as wombats can shelter and then utilize the post-fire environment of Heathlands (see below). There is only limited data on fire effects on many faunal groups and the theoretical basis for how different groups respond is still the subject of development (Tasker *et al.* 2008). Keith *et al*. (2002) summarised 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 refuge in burrows). Surprisingly, a number of more mobile animals (kangaroos; honeyeater birds) were killed by the fire.

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

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

Sydney coastal heaths have hosted many fire ecology studies, and the basic dynamics of these heaths are well understood. Fire frequency thresholds for this vegetation type were proposed by Bradstock *et al.* in 1995:
"A decline in populations of plant species can be expected when:
- there are more than two consecutive fires less than 6-8 years apart (fire-sensitive shrubs decline);
- intervals between fires exceed 30 years (herbs and shrubs with short-lived individuals and seedbanks decline);
- three or more consecutive fires occur at intervals of 15-30 years (sub-dominant herbs and shrubs decline);
- more than two consecutive fires occur which consume less than 8-10 tonnes/ha of surface fuel (species with heat-stimulated seedbanks in the soil decline)"

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

Sensitivity to frequent fire

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

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

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

Sensitivity to infrequent fire

Field research around Sydney has identified shrub species which are disadvantaged if fire is too *infrequent* (Fox & 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 1987a), 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 1996). 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 5 or 6 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 multisite study by Morrison *et al.* (1995) in Ku-ring-gai Chase National Park in Sydney's north. Here increased variability in interfire intervals was associated with an increase in the species richness of both obligate seeders and resprouters.

Fire intensity

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

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

We have already noted that the majority of species in this vegetation type do all, or most, of their recruitment in the months following a fire. Studies which confirm this include Auld and Tozer (1995) for *Acacia suaveolens*, *Grevillea buxifolia* and *Grevillea speciosa* and Vaughton (1998) for *Grevillea barklyana*, a rare obligate seeder which occurs mostly near Jervis Bay. This is not simply a matter of fire cues promoting germination however. Studies have found that seedlings of serotinous species survive better in burnt areas. In fact, all seedlings of *Banksia ericifolia* and *Banksia oblongifolia* which

germinated from seed planted out by Zammit and Westoby (1988) into sites burnt 3, 7 and 17 years previously died within six months of germination, whereas seeds placed into recently burnt sites had much higher rates of both germination and survival. Bradstock (1991) found seedlings of four Proteaceous species placed into unburnt sites completely failed to survive due to predation, while in burnt areas the majority of seedlings survived, possibly due to reduced densities of small mammals.

Fire-related germination cues which operate across a wide variety of species found in Sydney Coastal Heath include heat and smoke. In the mid-1980s Auld (1986b) showed that heat broke seed dormancy in the obligate seeder legume *Acacia suaveolens*. A larger study of 35 species from the Fabaceae and Mimosaceae families confirmed the ubiquitous nature of heat cues in these families (Auld & O'Connell 1991), although species varied somewhat in the temperatures which were associated with a maximal response. These data, in conjunction with information on soil temperatures relative to fire intensities, led to the conclusion that repeated low intensity fire should be avoided. More recent studies have demonstrated the dormancy-breaking properties of smoke. Kenny (2000) and Morris (2000) found smoke increased germination of several *Grevillea* species found on sandstone, with some species also responding positively to heat. Thomas *et al.* (2003) documented a range of responses to heat and smoke cues amongst a group of Sydney sandstone species, including interactions between cues. Relatively moderate heat shock produced maximum germination in several species, leading these authors to conclude that "low-intensity fire or patches within fire" may be important for recruitment of some plants.
At a community level, a study by Morrison (2002) found that floristic composition in Sydney's sandstone country varied with fire intensity. Species favoured by relatively high intensity fire included peas and monocots. Species from the Proteaceae and Rutaceae families were most abundant where fire had been of low to medium intensity, while low intensity burns favoured species in the Epacridaceae family. Recent work by Ooi *et al.* (2006) shows that a number of obligate seeder *Leucopogon* species appear to rely at least in part on persistence of adult plants in unburnt patches and places where fire intensity is low.

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

Application to Heathlands in the BRG region

Three recent studies into the fire ecology of Northern Montane Heaths have been undertaken (Clarke & Knox 2002; Williams & Clarke 2006; Clarke *et al.* 2009). Clarke & Knox (2002) studied the post-fire response of shrubs across vegetation types on the New England Tablelands. Obligate seeders were most common in rocky outcrops (essentially a dry heath) which also recorded the highest post-fire seedling recruitment (75% of species). Wet heaths had a lower proportion of obligate seeders and species recruiting post-fire (31%) as did other vegetation types. These patterns could not be simply explained by tested models, suggesting that a number of factors contribute to these differences.

Williams & Clarke (2006) examined fire history and soil gradients (moisture and fertility) in relation to the distribution and assemblage structure of wet heath and sedgeland communities in Gibraltar Range National Park. Soil moisture status and fertility correlated with differences among these communities (i.e. tall heaths on drier sites; sedgelands on wetter sites). Time-since-fire strongly influenced heath structure and species richness (which declined with canopy closure over time). This confirmed that these heaths have a generally consistent time-since-fire pattern to that established for Sydney Coastal Heaths (and other heath systems across the globe). Fire frequency did not affect *shrub* species richness (but some individual species had decreased abundance with higher fire

frequencies). An inter-fire interval of less than seven years was considered likely to reduce the abundance of some shrub species and it was concluded that these systems have similar fire response syndromes to other east coast heaths. This includes an overall reduction in floristic composition after ~15 years (although there was an overall lower species richness in these heaths compared to studied coastal heath systems). In rocky sites, however, species may be obligate seeders but this may be predominately driven by soil conditions and moisture status variability in some locations. Benwell (2007) concluded this to be the case for granite outcrops on the Tablelands. However, this may not be true in all locations because some species on sandstone outcrops are strong resprouters (Tierney 2003).

Clarke *et al.* (2009) documented the post-fire response of 489 woody species in six vegetation formations across the New England Tableland, including rocky outcrops supporting various Northern Montane Heath communities. Significantly, 62 taxa occurring within Northern Montane Heaths were found to have an obligate seeding (Type II) response (killed with a soil seed bank), the largest number of the six vegetation formations investigated. Conversely, only 18 taxa were found to have a resprouting Type V (basal resprouting) response, the lowest of any of the vegetation formations investigated. This study strongly indicates an immaturity risk threshold of 12 year for western variants of Northern Montane Heaths, such as those occurring within the BRG. Clarke *et al.* (2009) caution that "minimum thresholds need to be slightly more conservative in comparison to similar vegetation formations in other areas of NSW as many New England Tableland species take longer to reach maturity and therefore require a longer minimum fire interval".

In sum, studies to date suggest that the fire regime guidelines developed for Sydney Coastal Heath may not be entirely appropriate for Northern Montane Heaths. At least one species from Gibraltar Range heaths has been implicated as possibly at long-term risk from high fire frequency due to population shifts (Gross & Caddy 2006). The caveats of Clarke *et al.* (2012) regarding the lower productivity of Northern Montane Heathlands in the BRG region and the immaturity risk threshold of 12 years should be heeded when actively managing these diverse vegetation communities.

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

v. FIRE IN DRY SCLEROPHYLL FORESTS

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

"Bushfires play a vital role in the dynamic ecology of the Dry Sclerophyll Forests … provid[ing] a critically important cue for regeneration by periodically stimulating the release of seeds, liberating resources such as nutrients and light, and by creating the open space essential for slow-growing sclerophyllous seedlings to survive" (Keith 2004).

Dry Sclerophyll Forest communities in the BRG region possess the greatest diversity of any of the vegetation formations within the region, with OEH (nd) identifying 59 RVCs as occurring within eight Keith classes (New England Dry Sclerophyll Forest; Northern Gorge Dry Sclerophyll Forest; North-west Slopes Dry Sclerophyll Forest; Pilliga Outwash Dry Sclerophyll Forest; Northern Escarpment Dry Sclerophyll Forest; Northern Tableland Dry Sclerophyll; Western Slopes Dry Sclerophyll and Yetman Dry Sclerophyll). 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.

One class, Yetman Dry Sclerophyll Forest, is essentially restricted to the sandstone landscapes between Warialda and Yetman. These forests are at the southern edge of an extensive band that extends a considerable distance into southeast Queensland. Smooth-barked Apple (*Angophora leiocarpa)* and Long-fruited Bloodwood (*Corymbia dolichocarpa*) are prominent elements of these forests, giving them an appearance similar to Dry Sclerophyll Forests on the sandstones of the Sydney Basin. They occupy a substantially more subtropical climatic domain than any other class of Dry Sclerophyll Forest on the western slopes of NSW.

The discussion below is of relevance to these vegetation communities. However, structural variations at the site level will affect fire behaviour (i.e. capacity to carry crown fires; the thickness of the understory may be greater with less canopy etc.), these structural variations need to be assessed at the site level.

The fire ecology literature for Grasslands, Grassy Woodlands and Heathlands (reviewed above) clearly demonstrates the importance of fire as a disturbance event that drives diversity patterns in these vegetation types. The fire ecology literature for Dry Sclerophyll Forests is not as extensive as it is for those vegetation formations. However, like Heathlands, Dry Sclerophyll Forests contain a large percentage of the plant species diversity in the sclerophyll shrub layer. In a regional study across vegetation types, Clarke *et al*. (2005) found that mean woody taxa species richness (excluding eucalypts – which mostly form the tree layer) was highest in Dry Sclerophyll Forests (~20 species per 0.1 ha) which was significantly higher than heath, Wet Sclerophyll Forests or grassy systems. These shrubs were dominated by species from the Proteaceae and Myrtaceae which were the families that dominated in heath systems and in some instances the same species occur in both systems. This suggests that some of the detailed fire ecology research for Heathlands has some application to this shrub layer. However, it also clear that even at the species level, plants can display variability in their response to fire due to intra-species variability (Gill and Bradstock 1992). Secondly, Clarke *et al*.

(2005) found significant differences in resprouting capacity among vegetation types which were associated with a model of resource/productivity among systems and that disturbance models were more explanatory for the low productivity systems. Therefore, even where there are floristic similarities, extrapolation among systems needs to be treated with some caution (see also Pausas *et al*. 2004).

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

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

White Cypress Pine recruits episodically, in years of good rainfall (Ross *et al.* 2008). Rain in autumn and winter encourages germination, while follow-up rain the subsequent summer appears to be essential for seedlings to establish (Lacey 1972). When a recruitment event does occur, the density of young pines can be quite stunning: Lacey (1972) reports rates of up to 2.5 million seedlings per hectare. Unlike eucalypts, White Cypress Pine does not easily self-thin: thick stands of young spindly trees can persist for decades. Parker and Lunt (2000), for example, report densities of 92,500 plants per hectare in 40-year-old stands in fenced plots west of Echuca in Victoria, while in state forests south of Cobar Allen (1995) recorded many stands from the late 1800s which were still severely 'locked' 100 years later. Growth rates in these locked stands are very low (Horne 1990b, Ross *et al.* 2008), but increase when stand densities are reduced (Henry 1960; Lacey 1973; Horne 1990a).

Forests of the past

What did the White Cypress Pine forests and woodlands of NSW look like 200 years ago? A number of authors have used historical records to develop a picture of the landscape prior to European settlement. Many have focussed on vegetation structure: the density of trees and shrubs, and the balance between grassy and woody plants. Other researchers have used counts of old trees and tree stumps to reconstruct the forests and woodlands of the past. Here are some of their findings:

- Vegetation types over the BRG and nearby regions (Namoi and Central West) formed a mosaic (Eco Logical 2008), as is commonly found in landscapes elsewhere. Much of the country was open woodland with a grassy understorey, however there were also areas where shrubs and/or small trees grew in abundance (Benson & Redpath 1997; Croft *et al.* 1997; Allen 1998; Keith 2004).

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

- In many places, big, mature cypress pines co-occurred with big, mature eucalypts. The density of large trees appears to have been of the order of 20 to 40 per hectare (Benson and Redpath 1997;, Martin 2005). Eucalypts appear to have been more abundant, at least in many areas, than cypress pine. Paull (2001), who used stump counts to estimate tree density in 1910 in 20 one-hectare plots in the Pilliga West, found that while there was some variation in the relative numbers of the two canopy species, White Cypress Pine and Narrowleaved Ironbark (*Eucalyptus crebra*), on average plots contained 25 large eucalypts to 5 large pines. Pine numbers never got above 9 per hectare, and the more ironbarks in a plot, the less pines. Lunt *et al.* (2006) also used stump counts to estimate pre-European tree density across a wide area of central NSW south from Nyngan. The estimated average density of trees with a diameter of 20 cm or more at the time of settlement was 39 per hectare. Seventy-eight percent of these trees were eucalypts, 21% pines. Thus *Callitris glaucophylla* was an important, but sub-dominant, component of the pre-European canopy in these stands. Allen (1995), however, recorded approximately equal numbers of pre-1870 pines and eucalypts in two minimally disturbed state forests south of Cobar, suggesting that White Cypress Pine was a more important component of the canopy in the west than in the east of its distribution in NSW.

- Thickets of small cypress pines, and multi-aged pine stands, were a feature of the landscapes traversed by the first Europeans to explore the Central West. Several authors (eg Mitchell 1991 Martin 2005 cite extracts from Oxley's journals describing his 1817 exploration of the Lachlan River valley, which included phrases such as " …. after going through about eight miles of very thick cypress scrub" and "it was a continued scrub, and where there was timber it chiefly consisted of small cypress…" Allen (1998) quotes Cunningham, who reported cypress pines "of various sizes and dimensions from seedlings, generally growing in clumps, to lofty trees of about 60 feet."

- Allen (1998) points out that pre-European cypress pine forests were dynamic, with *Callitris* cohorts going through a cycle from dense seedlings to scattered mature veterans. He suggests that the cycle would have played out more rapidly in some times and places than others, and that the forest in different parts of the landscape would have been in different stages of the cycle. Patch sizes no doubt also differed across the landscape. The patchy landscape encountered by the explorers, Allen points out, would have been a snapshot of this dynamic landscape at a particular point in time.

Changed forests

How have White Cypress Pine forests changed over the last 200 years? Environmental historians agree that by the 1870s, extensive regrowth of cypress pine was occurring in what had formerly been relatively open woodland areas (Mitchell 1991; Norris *et al.* 1991; Allen 1998; Parker and Lunt 2000). Strong documentary support for this contention comes from reports requested by the NSW Surveyor General in 1880. Land Commissioners in the Lachlan and Murrumbidgee districts described the difficulties caused by pine scrubs which had sprung up over the past 15 years. The problem was less acute around Narromine and Dubbo, although "scrub patches were believed to be increasing in density" (Mitchell 1991). Pine regeneration in the Central West (Allen 1995) and the Pilliga appears to have occurred somewhat later than that further south. Rolls (1981) contends higher than average rainfall in several years between 1879 and 1887 was important for pine regrowth in the Pilliga, while Norris *et al.* (1991) argues that wet years between 1889 to 1894 are more likely to have formed the precursor.

There is also general agreement that very little cypress pine recruitment occurred in NSW during the first half of the 20th century (Lacey 1972; Rolls 1981; Norris *et al.* 1991; Allen 1998). In the 1950s, however, a series of wet years coincided with dramatic rates of pine seedling establishment (Mitchell 1991; Allen 1998). These waves of regeneration were clearly distinguished by Allen (1995), who surveyed pine age and size classes in Cumbine and Nangerybone State Forests south of Cobar. These forests were unusual in having been subject to only limited logging and thinning. Pines haled either from before 1870 ('old greys', with an average density of about 29/ha), from 1870 to 1900 (mostly in dense, but patchy, locked stands), or from post 1950 (scattered individuals to dense clumps).

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

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

Causes of change

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

Logging

The loss of many large eucalypts appears to have been almost ubiquitous across the forest and woodland landscapes west of the Divide, even in the least disturbed patches (Lunt *et al.* 2006). Large cypress pines have also been logged out. Thus reduction in competition from large, mature trees is one possible explanation for the massive establishment of pine seedlings. In a paper prepared for the then Forestry Commission of NSW, Lacey (1972) summarises the relationship as follows: "In the absence of grazing there is an obvious relationship between stand density and the amount of regeneration which becomes established. Overstocked stands have very little regeneration while understocked stands are seldom without it." He attributes this relationship to both lower competition in understocked stands, and to the fact that seed set is higher when pine density is low.

More recently, strong support for the ability of fully stocked stands to limit regeneration has been provided through modelling work by Ross *et al.* (2008). These researchers used data from thinning trials in state forests to develop and test their model, which simulates the dynamics of *Callitris glaucophylla* in Central Western NSW. Work to date has focussed on monospecific stands, over the period 1948 to 1998, in the absence of fire. Besides confirming the importance of high rainfall years for recruitment, both actual and modelled recruitment clearly decreased with increasing stand basal area. By 1998, recruits from the 1950s into understocked stands had only grown to between 2.5 and 5.5 m, and stand densities had stabilised at around 20 000 stems per hectare. This research team plans to add eucalypts and fire to their model in the future.

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

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

Changed grazing regimes

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

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

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

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

Changed fire regimes
While we can never be sure what fire regimes pertained prior to European settlement, there is no doubt fire was a component of the landscapes where White Cypress Pine is found. Ignition sources were almost certain two-fold: lightning strikes and Aboriginal burning (Allen 1998). Mitchell recorded Aboriginal fires in 1835 near the cypress pine forests studied by Allen (1995). In a review of the ecology of White Cypress Pine in Queensland, Harris and Lamb (2004) contend that "evidence, in the form of firescarred trunks, suggests that the majority of cypress forests incurred regular burning prior to being reserved as State Forests…." In the Northern Territory frequent low intensity Aboriginal burns were, and in some places continue to be, a feature of the environment where *Callitris intratropica* grows (Bowman *et al.* 2001; Yibarbuk *et al.* 2001). It is likely that the frequency and intensity of fire varied across the landscape due to the effects of natural features (for example soil type would have affected grass growth which would in turn have affected fire). As well, Aboriginal management is unlikely to have been uniform. Although again we can never know in detail, it appears that with European settlement, fire became increasingly less common in the White Cypress Pine landscapes of NSW.[1]

We have already noted that stock grazing may have reduced fuels, and thus the intensity and frequency of fire. The need for grass to feed sheep and cattle probably also encouraged graziers to suppress fires wherever possible. Reduced fuel loads due to grazing, and fragmentation due to clearing, almost certainly made fire suppression increasingly viable. Thompson and Eldridge (2005b),

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

in a review of the literature on White Cypress Pine, assert that "burning cycles ceased within the cypress pine zone with the expansion of agriculture and pastoralism by the early Europeans." Fire exclusion has been normal practice in cypress pine forests managed for forestry (Henry 1960; Lacey 1973; Johnson and Jennings 1991; Allen 1998).

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

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

Flora
It is often said that dense cypress regrowth suppresses grass cover (eg Lacey 1973; Harris *et al.* 2003). Logically this makes sense, as grasses are in competition with the young trees. It also fits with the findings of many studies of effects of trees in general on grass production. Trees affect the growth of understorey plants by altering the availability of resources, particularly light, water and nutrients.

While some studies have documented increased productivity under trees (eg Belsky *et al.* 1993), many others have found lower herbaceous biomass under trees and/or shrubs (Engle *et al.* 1987; Archer 1990; Scanlan and Burrows 1990), or that ground layer productivity increases when trees and/or shrubs are removed (eg Walker *et al.* 1986a, b; Harrington and Johns 1990).

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

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

However a study designed to assess trends in understorey vegetation with varying density of cypress pine across a wide geographic area (Thompson and Eldridge 2005a) found no association between canopy cover and cover of understorey plant species, whether midstorey species were included or excluded. There was, however, an east-west trend across the rainfall gradient sampled (530 to 215 mm/year) with significantly more understorey cover in higher rainfall sites. The findings of this study

were unexpected, and the authors consider that differences in grazing history, and also lack of rainfall during sampling, may have affected the results.

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

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

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

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

Hunter (2013) completed a far more detailed examination of the influence of the above ground biomass of both Black and White Cypress on species density. Completing detailed analyses (canonical correspondence) of 997 survey site across the western New England Tableland, Nandewar and Brigalow Belt South bioregions, Hunter (2013) found that increasing above ground biomass of *C. glaucopylla* had a significantly positive effect on native species density. He concluded that "there is no reason to thin dense *Callitris* stands to increase local species richness" and suggested that dense

stands of *Callitris* are important, when associated with a range of stand densities, to ensure the highest regional diversity.

The findings of Hunter (2011; 2013) are compelling in that they have analysed and reported upon a large data set derived from a substantial number of floristic survey sites in the BRG region. These findings clearly demonstrate the value of maintaining landscape heterogeneity of density in cypress pine stands in these heavily cleared bioregions for the maximisation of biodiversity. As such, they are likely to stand contrary to the assumptions held by many landowners, land managers and others, about the lack of floristic diversity (and hence conservation value) of dense stands of cypress within these landscapes.

Fauna

Little research directly addresses the effects of changes in forest and woodland structure on fauna. The most cited work comes from Cape York, where granivorous birds have declined as a result of encroachment by *Melaleuca* species into open habitats (Crowley and Garnett 1998; Crowley 2001). A study of "Woody Weeds and Biodiversity in Western NSW" by Ayers *et al.* (2001) also found that the abundance of ground feeding birds decreased as shrub cover increased.[2]

Changes in forest composition and structure have been invoked by Date *et al.* (2002) to explain their findings in the Pilliga. In the early 1990s these researchers surveyed 568 transects right across the forest, for birds. On 230 of these transects vegetation structure and plant species were also recorded, along with disturbance history. They found a much greater diversity of birds in patches along creeks where old trees were still to be found. These patches had a diverse shrub understorey, low cover in the subcanopy, and had experienced moderately frequent wildfires. The majority of forest, however, supported a smaller number of generalist bird species, which also occurred in the creeklines. These "non creek" patches had less shrub cover, more sub-canopy cover, had been more heavily logged and had experienced very little fire over the last 40 years.

Many of the species found only in the creeklines were either ground-foraging and/or ground nesting species, or species dependent on mature trees for food and nest sites. Date *et al.* (2002) believe that logging, grazing and fire exclusion have created a more homogeneous environment than once existed in the Pilliga, leading to a simplification of vegetation and bird communities. They recommend management strategies to bring back mature trees, and "intermediate fire regimes" which they suggest would involve variable intervals between 2 and 25 years.

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

The Ayers *et al.* (2001) study, which involved a survey of 35 sites with varying degrees of shrub cover up to about 40%, found no overall effects of shrubs on species richness or abundance, although some individual species and species groups responded positively or negatively to increasing shrub

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[2] While fauna research in White Cypress Pine communities is not extensive, there are suggestions that some birds and other animal species may be similarly affected.

cover. This research focussed on *Eremophila*, *Dodonaea* and *Senna* species. Localities with shrub cover over 40% were not included because it was not possible to find open areas for comparison.

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

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

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

So do dense stands of Cypress Pine regrowth enhance or reduce faunal diversity? Andrews (2003 suggests the answer depends on their location: "Where they occur in heavily cleared landscapes for instance, patches of regrowth cypress forest may be important as dense foliage refuge habitat for birds and other fauna." In other words, these patches are likely to be an advance, in biodiversity terms, on completely cleared land. Major *et al.* (2003) have found that even small patches of White Cypress Pine provide good habitat for arboreal insects.

Erosion

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

Cypress Pine and fire

Research into the impacts of fire on White Cypress Pine is surprisingly limited. No one, to our knowledge, has experimented with the effects of different fire intervals or intensities in *Callitris glaucophylla* forests or woodlands, and even the effects of a single fire have rarely been documented. Two sources, however, provide useful information on the fire-related attributes of the species. The first is the extensive research undertaken in NSW and Queensland with a view to

informing forestry management of White Cypress Pine; the second is a series of studies on *Callitris intratropica*, a closely-related species in the Northern Territory. Here are some of the findings:

- White Cypress Pine is classified as an obligate seeder (NSW Fire Response Database – DEC 2002). Individual pines are certainly more likely to die in a fire than the eucalypts that grow with them (Lacey 1972).
- This does not mean, however, that all *C. glaucophylla* plants are killed in every fire. Even in hot, extensive wildfires, a proportion survives (Hawkins 1966). For example Mitchell (1991) cites the NSW Forestry Commission Annual Report for 1951/2, which indicates that many mature cypress pines survived the 1951 wildfires in the Pilliga Forest. Surviving veterans from a fire in 1926 could still be seen in Strahorn State Forest 70 years later (Allan 1998). In the Snowy River Valley many mature trees survived multiple fires in the early decades of European settlement (Pulsford *et al.* 1993).
- Several characteristics of the species encourage survival in the face of fire. First, some plants are tall enough to get above the flames (Figure 1). Second, even after complete leaf scorch, some individuals survive through production of epicormic shoots (Lacey 1973). Third, White Cypress Pine litter is probably less flammable than that of co-occurring eucalypts and grasses (Lacey 1973): fuel loads in *Callitris intratropica* stands are lower, moister, and contain much less grass than those under nearby eucalypts (Bowman and Wilson 1988). Fourth, dense stands of *Callitris glaucophylla* may naturally discourage fire spread by reducing wind speed (Thompson and Eldridge 2005b).
- Small *Callitris* plants are more vulnerable to being killed by fire than large plants (Bowman *et al.* 1988; Bowman and Panton 1993; Allen 1998; unpub data D. Taylor QFRI 2007).
- A fire shortly after a regeneration event can eliminate an entire cohort of young seedlings. Allen (1998) reports that this is definitely known to have happened in Euglo South State Forest in 1957. Once saplings have managed to get a few meters above grass height, however, if a fire occurs some generally survive (see *Figure 5*).
- White Cypress Pine, like several other *Callitris* taxa, is a T-species, able to recruit between fires. As we have already seen, recruitment occurs episodically, in periods of higher than average rainfall, irrespective of fire. It may be some years after fire before any recruitment occurs (Allen 1998).
- White Cypress Pine seeds develop in cones and are released in November and December (Hawkins 1966). Once released seed remains viable for only a short time. Seeds have "negligible dormancy" and there is no long term seed store either in the soil or on the plant (Lacey 1972, 1973).
- Seed crops vary considerably between years, although even in poor years some seed is produced (Hawkins 1966). It is unclear whether reduced seed supply in poor years limits regeneration; Bowman and Harris (1995) conclude that it does not. As already noted when conditions are right rates of seed germination can be extremely high.
- Stand density influences seed production considerably: the more open the stand, the more seed is produced (Lacey 1972).
- Fire inhibits seed production. Hawkins (1966), who compared seed fall in burnt and unburnt patches, found a considerable reduction in the quantity and viability of seed produced in moderately burnt areas for five years post-fire, after which levels returned to normal. Almost no seed was produced for at least seven years post-fire in a patch where intense fire had killed most cypress pine trees.
- Although seed of *Callitris* species does not generally appear to travel far from parent plants (Bowman *et al.* 1988; Bowman and Harris 1995 and references therein), seedlings of *C. glaucophylla* have been found up to 370 m from source trees (Lacey 1973).
- *Callitris glaucophylla* reaches sexual maturity at between 6 and 12 years of age (Bowman and Harris 1995). Lacey (1972) reports individuals in the field producing cones at 6 years old when less than 1.2 m in height.

- The life span of White Cypress Pine is at least 200 years (Lacey 1973; Bowman and Harris 1995).

Taken together, these characteristics paint a picture of a species whose abundance is likely to be readily regulated by fire. Elements of the life history of White Cypress Pine suggest extreme vulnerability to burning. These elements include its obligate seeder status, its lack of soil-stored seed which could potentially survive through more than one fire, the suppressive effects of fire on seed production and its relatively long juvenile period.

On the other hand, elements of its life history indicate an ability to survive in a fire prone environment, particularly where fire intensity is low. These elements include the capacity to create a local environment where fire may be precluded or at least reduced in intensity, the ability of larger individuals to survive fire while smaller competitors are eliminated, the capacity for seed dispersal over moderate distances, the ability to recruit massively outside the window of opportunity provided by fire, and its longevity.

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

In fact, as already noted, long fire-free periods have seen the development of dense, locked stands of young cypress. In pre-European forests and woodlands, fire – along with competition from large adult trees and perhaps also predation by native animals – probably limited sapling abundance, with fire selectively killing many small pines while leaving others to survive and dominate. The presence of eucalypts is likely to have both limited pine recruitment through direct competition for resources, and also encouraged fire through the production of flammable litter – which in turn would have discouraged the pine. Periodic fire may well have encouraged native tussock grasses which again would have provided flammable fuel for the fires that favoured eucalypts over pines.[3]

Western Slopes Dry Sclerophyll Forests

As their name implies, this vegetation class occurs on the western side of the Great Dividing Range, on sandy, infertile soils – often derived from sandstone – where rainfall exceeds 500 mm. Keith (2004) places Western Slopes Dry Sclerophyll Forests in the shrubby subformation of Dry Sclerophyll Forests. Shrubs are mostly sclerophyll species including wattles (e.g. Spur-wing Wattle *Acacia triptera*, Streaked Wattle *Acacia lineata*), heaths (e.g. Urn Heath *Melichrus urceolatus*, Daphne Heath *Brachyloma daphnoides*), daisies (e.g. *Cassinia* species), and members of the Myrtaceae family (e.g. Broombush *Melaleuca uncinata*, Fringe Myrtle *Calytrix tetragona*). Trees are mostly eucalypts, particularly ironbarks, and cypress pines. Eucalypts include Narrow-leafed Ironbark (*Eucalyptus crebra*), Mugga Ironbark (*E. sideroxylon*) and Tumbledown Red Gum (*E. dealbata*). Both Black Cypress Pine (*Callitris endlicheri)* and White Cypress Pine (*C. glaucophylla*) are found in Western Slopes Dry Sclerophyll Forests. According to Binns and Beckers (2001), the two are negatively associated, with Black Cypress more common in heathy areas, while White Cypress is more abundant in grassy vegetation. Black Cypress Pine, like *C. glaucophylla*, is an obligate seeder which

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

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

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

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

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

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

Conclusion

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

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

Dry Sclerophyll Forests from the Tablelands

A number of studies in recent years have been carried out in Dry Sclerophyll Forests on the New England Tablelands (Clarke & Knox 2002; Clarke *et al*. 2005; Croft *et al*. 2006; Knox & Clarke 2006; Clarke *et al*. 2009). The class of Dry Sclerophyll Forests studied is not apparent in all of these studies (either New England Dry Sclerophyll Forest or Northern Escarpment Dry Sclerophyll Forest or perhaps both were included in some of these studies). These studies are briefly summarised below. Clarke & Knox (2002) studied the post-fire responses and recruitment patterns of shrub species among ecosystems on the tablelands. The shrubby forests supported the highest diversity of shrub species and a relatively large number of these species were obligate seeders. These forests would include Dry Sclerophyll Forests (of indeterminate class). Obligate seeders within these system include *Cassinia quinquefaria, Callitris oblonga ssp. parva, Leucopogon muticus, Melichrus erubescens, Styphelia triflora, Acacia betchei, Acacia burbidgeae, Acacia fimbriata, Acacia obtusifolia, Acacia torringtonennsis, Acacia triptera, Acacia ulicifolia, Acacia venulosa, Bossiaea rhombifolia, Daviesia ulicifolia, Dillwynia sericea, Dillwynia sieberi, Gompholobium huegelii, Pultenaea pycnocepala, Prostanthera scutellarioides, Grevillea beadleana, Grevillea scortechinii ssp.*

sarmentosa, Hakea macrorrhyncha, Persoonia rufa, Discaria pubescens and *Correa reflexa* (Clarke & Knox 2002).

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

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

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

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

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

would be precautionary to assume that there are a relatively high number of obligate seeder shrub species in these tableland Dry Sclerophyll Forests.

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

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

Dry Sclerophyll Forests from outside of the region

Studies of Dry Sclerophyll Forests outside of the Border Rivers –Gwydir region are broadly consistent in demonstrating that the sclerophyll shrub layer of these forests is strongly influenced by fire frequency.

Southern Tableland Dry Sclerophyll Forest. Purdie conducted some of Australia's first fire ecology studies in southern tablelands Dry Sclerophyll Forests on the slopes of Canberra's Black Mountain (Purdie & Slatyer 1976; Purdie 1977a, b). Experimental fires were carried out in three sites, which had previously been burnt 9, 11 and 33+ years ago respectively. Purdie and Slatyer (1976) note that prior to the experimental fire in the long unburnt site, "many of the larger shrubs were senescent or in varying stages of decay," whereas shrub density at the other two sites was considerable. At each site 60 small quadrats were set up, 30 in an area which was then burnt (in summer), and 30 in an area which was left unburnt. Post-fire regeneration was monitored over several years (Purdie & Slatyer 1976). Species were classified as 'obligate seed regenerators' (obligate seeders), 'obligate root resprouters' (resprouters which didn't produce seedlings), or 'facultative root resprouters' (resprouters which did produce seedlings). The key findings from this work were that:

1. Almost all shrubs resprouted; the only shrubs which were clearly obligate seeders were *Acacia genistifolia*, *Dillwynia retorta* and *Pimelea linifolia*. *Pimelea linifolia* germinated in the long unburnt site after fire, though it was not recorded there prior to the burn. All three species flowered within three years of fire (Purdie & Slatyer 1976).

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

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

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

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

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

7. A few species, including the dominant grass *Joycea pallida* and the exotic herb *Hypochoeris radicata*, produced more seedlings in unburnt than in burnt areas (Purdie & Slatyer 1976).

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

9. Many resprouters which weren't able to sucker flowered rapidly after fire on regrowth, and this produced some seedlings in Year 2 (Purdie 1977b; Purdie & Slatyer 1976).

10. Resprouter recovery stabilised at a lower level in the previously long-unburnt site (60% of pre-fire numbers – which we have already noted were relatively low), than in the two more recently burnt sites (85-90% of pre-fire numbers) (Purdie 1977a).

11. In the unburnt plots, many species declined with time. Obligate seeder population numbers declined 16-22% in these plots in the first year of monitoring, and an additional 5-10% in the second (Purdie 1977a).

12. "With the exception of rare species, all the tree, shrub and herb species represented in the quadrats prior to burning … were present in the regenerating communities in the first year after burning" (Purdie & Slatyer 1976).

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

Purdie's findings tell us something about the effects of fire on plants in general, and also specifically about Dry Sclerophyll Forests. Higher germination and seedling survival rates in burnt than in unburnt patches is a common finding in fire-prone vegetation, as is the tendency for obligate seeders to outdo resprouters in numbers of post-fire seedlings.

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

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

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

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

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

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

growing in some parts of the central tablelands as numbers of macropods and wombats build up (pers. comm. various land managers in the CW region, 2007).

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

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

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

An additional issue for fire frequency concerns how long the seedlings of the many resprouting species take to become fire-tolerant. We do know that they grow slowly relative to their obligate

seeder counterparts, and Purdie (1977a) suggests that, from her observations, even after a 9 to 11 year inter-fire interval young plants may be vulnerable. Intervals above 15 years within a variable regime of intervals between 7 and 30 years should allow opportunities for these seedlings to fully develop, while still allowing species which produce most of their seed-crop in the early post-fire years the opportunity to build up population numbers through some short intervals now and then.

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

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

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

Application to Dry Sclerophyll Forests of the BRG region

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

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

There are reported differences in the number of obligate seeder species among the studies carried out in Dry Sclerophyll Forests. The reported relatively high level of obligate shrub species in tableland Dry Sclerophyll Forests suggests that fire frequency is an important consideration in managing these forests. Overall, studies in the tableland Dry Sclerophyll Forests indicate that the shrub layer is responsive to fire and contains a large percentage of the plant diversity in these

systems. It is also clear that, in general, high fire frequencies (intervals in the order of 1-3 years) have been associated with reduced shrub layers and dominance by species such as bracken (*Pteridium esculentum*) in Dry Sclerophyll Forests. The shrub layer has important faunal values as a nectar resource and in providing structural complexity and protection, although recently burnt sites are also important for some species. Dry Sclerophyll Forests of the tablelands should have inter-fire frequencies that are within the 7-30 year threshold. Additionally, there are some species that may be disadvantaged by frequencies that are towards the shorter or longer end of this range and the general principle of frequency variability and patchiness in fires should be applied in the use of prescribed fire.

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

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

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

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

vi.   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 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) modelled fire regimes in these forests. The results suggest a mean fire interval in those forests of 35-70 years (with a mean interval between tree killing fires – presumably more intense fires – of about 75–150 years). Changes (decreases) in fire intervals predict biodiversity declines in these forests. In northern Australia and along the New South Wales coasts and ranges, fire in Wet Sclerophyll Forests may be different in average intensity and season, none-the-less it is still considered a major determinant of successional change (e.g. Harrington & Sanderson 1994).

OEH (nd) identify three classes of Wet Sclerophyll Forest as occurring within the BRG region, one class within the shrubby subformation of Wet Sclerophyll Forest (Northern Escarpment Wet Sclerophyll Forest) and two classes within the grassy subformation of Keith (2004), namely Northern Hinterland Wet Sclerophyll Forest and Northern Tableland Wet Sclerophyll Forest.

Of the three classes known to occur within the BRG region, two are highly restricted, namely grassy Northern Hinterland Wet Sclerophyll Forest and shrubby Northern Escarpment Wet Sclerophyll Forest. Both are limited to the extreme eastern periphery of the catchment to the southeast of Tenterfield at the crest of the Great Dividing Range and along the upper reaches of the Deepwater River. The Northern Escarpment Forests of the BRG region comprise four closely related RVCs. Growing on fertile soils with high rainfall (Keith 2004) these forests are characterised by the dominance of very large Messmate (*Eucalyptus obliqua*), New England Blackbutt (*E. campanulata*) and Sydney Blue Gum (*E. saligna*) with a dense mesic understorey. Such forests are restricted to public lands in Campoompeta and Butterleaf National Parks and adjoining State Forests. The single RVC constituting Grassy Northern Hinterland Wet Sclerophyll Forest in the BRG region is listed by OEH (nd) as dominated by Silvertop Stringybark (*E. laevopinea*) and Tallowwood (*E. microcorys*), although no records of Tallowwood exist within the BRG region and Keith (2004) suggests that the altitudinal limit of this vegetation class is 600 m asl. Because of these factors the existence of grassy Northern Hinterland Wet Sclerophyll Forest within the BRG region is questionable. Because of their limited occurrence on public lands and dubious occurrence respectively, these forests will not be considered further in this review,

Grassy Northern Tableland Wet Sclerophyll Forests are relatively extensive within the BRG region, extending across the crest of the Great Dividing Range, much of it on private property. Significant outliers of these forests occur along the Nandewar Range, in higher elevation and wetter parts of Mt Kaputar National Park and on the Torrington Plateau (Benson & Ashby 2000; Benson *et al.* 2010; OEH nd).

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). Hence, based on the Victorian model, intense fire (particularly crown fire) may result in even-aged stands of canopy species that are not capable of recruitment until they age past the primary juvenile stage. Repeat fires within this period (decades for some species) would shift the vegetation towards those species that resprout or with short juvenile periods. This model results in regenerating canopy dominants, or alternatively, thickets of subcanopy species under frequent fire. The floristic and structural similarities of Wet

Sclerophyll Forests at regional or national scales, however, mask some significant differences in the fire ecology among these systems.

A fire return 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. Less frequent but more intense fires will generally favour shrub species that recruit from seed in response to fire cues leading to a dense shrub layer of these species. These species typically have juvenile periods that range from a few years to almost a decade. Additionally, many of these understory species from families such as the Proteaceae and Fabeaceae respond to fire associated germination cues (Auld & O'Connell 1991; Brown *et al*. 2003). In particular, the temperature reached in the soil as a result of fire is critical in relation to germination for these hard seeded species. This will depend on the characteristics of the fire (intensity; retention time) as well as the soils characteristics (Whelan 1995). These species can therefore be generally 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 lowlands of coastal catchments in northern New South Wales have been subject to high levels of clearing and fragmentation, including increasing urbanisation. Achieving ecologically driven fire management can be problematic in urbanised areas (Morrison *et al*. 1996). However, active fire management of Wet Sclerophyll Forests has been undertaken within the urban matrix in the Sydney area for the endangered Blue Gum High Forest (McDonald *et al*. 2002). This work demonstrated the persistence of a recoverable understory seed bank across decades that responded to fire. This recovered understorey was distinct from the pre-burn forest understory. It may be that the Wet Sclerophyll Forests we seek to manage are not representative of forests of the past.

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" (Keith 2004). 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" (Keith 2004). 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 communities 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 vegetation 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 Sclerophyll 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. Sections of this literature are relevant to the BRG region and summarised below.

Wet Sclerophyll Forests – grassy subformation

Birk and Bridges (1989) studied the effect of fire at two and four years or fire exclusion over a 20 year period in blackbutt forest on the lower north coast. They found that burnt plots supported predominantly grasses whilst woody shrubs were dominant in the unburnt plots. York (1999) measured vegetation structure on experimental plots burnt every three years and on unburnt plots for 20 years in blackbutt forest near Port Macquarie. Cover assessments were based on vegetation height classes (not plant life-form), but the "very tall shrub layer" (150-200 cm) was significantly greater in unburnt plots. Stewart (1999) investigated the plots of York (1999) and found the seeds of graminoids (sedges and rushes) and shrubs were significantly more abundant in the long-unburnt plots. The seedbank of the shrub species also had a higher species richness in the unburnt plots. Species found only in unburnt plots included several graminoids, and broad-leafed shrubs and trees. Species found only in burnt plots included grasses, forbs and shrubs whose germination is cued to fire. Doug Binns (pers. comm. 2005) reports that in the study area of York and Stewart where fire had been excluded, a thick sub-canopy 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 Sclerophyll 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 2-10 m, and for shrubs under 2 m. 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 1000 km^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 *include* 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 sub-plot level, there were less fly and beetle morphospecies (presumed different species) where burning had occurred, but numbers of bug, spider and ant morphospecies were higher. These results were significant for flies, beetles and ants. Community composition also varied within groups depending on fire treatment, with large numbers of species appearing only, or mostly, in one treatment or the other. In some cases differences in species composition could be linked to habitat features characteristic of the two fire regimes. For example plant-eating bugs, flies and beetles were considerably more abundant on burnt plots, reflecting the dense groundlayer vegetation in these areas, while flies and ants known to feed in the litter layer were more common on unburnt plots. On the other hand, litter-feeding spiders were mostly found on burnt plots.

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

fire exclusion to frequent burning, would be needed in the forests of the region to maintain the full complement of ant species.

Andrew *et al.* (2000) also studied ants, but two years after the study reported in York (2000) in the same study sites. Burnt plots were four years post-fire. There were no statistical differences between the two burn treatments in either the abundance or the species richness of the ant fauna. Community composition did not differ greatly between habitats; open areas in burnt sites had the most distinctive ant assemblages. Habitats were not, in general, distinguished by differences in the abundance of the various ant functional groups, with one exception: sub-dominant 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 species mostly on these sites. These were rare species including 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 100 cm. 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 100 cm. 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 multi-layered vegetation with a strong mesic/rainforest component. These differences were most apparent in Tasker's work (Tasker 2002), which highlighted both the high diversity of grasses and herbs in sites burnt every 1-5 years, and the relative dearth in these sites of the climbers, ferns and small trees which dominated unburnt areas. Henderson and Keith (2000) found considerably fewer individual shrubs and fewer shrub species in less disturbed sites, although grazing contributed here as well as fire. Kitchin found fire impacts on composition of both shrubs and herbs, with greater numbers of shrubs – which in her study included various sclerophyllous species – in sites which had experienced some fire than in either long unburnt or very frequently burnt sites. Sites exposed to a high fire frequency were dominated by native tussock grasses, and length of shortest inter-fire interval influenced herb species composition. John Hunter (pers. comm. 2005) also studied the effects of fire frequency on herbs in upland forests and also found that herbs declined in long unburnt grassy forest areas, and considers that both shading by shrubs, and a heavy litter layer, may be responsible.

Tasker's results for both small mammals and invertebrates are consistent with the previous coastal studies. Frequently burnt and long unburnt areas supported particular suites of species, though species richness was similar. Tasker and Dickman (2004) point out that this finding "contradicts the

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

Some years previously, Christensen (1998) also argued, on the basis of a review of the literature, that retention of frequent low-intensity fire in the grassy forest landscapes of Northern NSW would be the precautionary approach towards conservation of medium-sized mammals. A number of Australian taxa in this size range use the early post-fire environment and/or grassy areas which are maintained by frequent fire. Vegetation which provides dense cover is also important for species of this size, and fire regimes which promote the juxtaposition of grassy and shrubby patches may be vital. The vulnerable Parma Wallaby (*Macropus parma*) is an example 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 utilised by many animals. Vegetation which has not been burnt for some time, or where fire frequency has been low favours some shrub and noneucalypt tree species, particularly those able to recruit between fires. This thicker vegetation has a deep litter layer and is associated with habitat features which are important for a different suite of animal species.

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

Existing fire regime guidelines

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

Discussion of fire regime guidelines for grassy Wet Sclerophyll Forests: One regime or two?

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

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

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

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

The upper threshold of 50 years proposed by Kenny *et al.* (2004) may in part reflect the authors' desire to ensure a reasonable life-span for obligate seeder eucalypts, which they note may occur in this vegetation type. This figure was obtained by bypassing 'most sensitive' plant species with lifespans of 20, 30 and 35 years (Kenny *et al.* 2004). In fact, 50 years would be a very short lifespan for obligate seeder eucalypts: for example *Eucalyptus regnans* lives for 350-500 years (McCarthy *et al.* 1990) and only starts producing the tree hollows used by arboreal mammals when it reaches about 120 years of age (Lindenmayer *et al.* 1997); *E. delegatensis,* which occurs in similar habitat, may develop over a similar timeframe. Mackowski (1984) found that hollows in blackbutt trees near Coffs Harbour did not start to form until trees were over 100 years old.

Conversely, short intervals (for understory fires) provide habitat for a range of species. It is suggested that by reframing wet sclerophyll fire regimes as having two tiers, and acknowledging the major differences between forest types in the north and south of the State, this paradox can be resolved.

Interval domains for understorey diversity

High intensity fires will occur when weather conditions are extreme, whatever the regime at other times. Of more concern for vegetation managers is the nature of the 'second tier' regime (understorey fire). Here, a range of intervals across both time and space may be desirable.

We know that quite frequent 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 motion irreversible successional processes? There is a need for relatively frequent disturbance in wetter forests. Intervals in the six to 15 year range, with occasional intervals up to 20 years, would be in line with this thinking. This range is similar to that encountered by Kitchin in her moderate fire frequency sites, and probably encompasses a proportion of Tasker's unburnt areas. In the cooler Tablelands environment, slightly longer intervals would probably be appropriate – see summary guidelines below.

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

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

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

The two-tier fire regime concept calls into question how the two regimes might interact in relation to overstorey recruitment. If occasional intense wildfires do indeed play a major role in eucalypt recruitment, would secondary regimes, particularly frequent burning, need to be suspended to allow seedlings to reach the point where they can survive low intensity fires? Or might continued low intensity burning play a positive role in thinning young eucalypt regrowth? Might a cessation in burning result in more intense fire, and more damage to young trees, when the secondary regime is eventually reinstated? Does the nature of the understorey affect eucalypt recruitment? Might either thick grass or thick shrubs pose difficulties for young eucalypts? Might these competitive

interactions, if they exist, help regulate sapling density appropriately? We do not have answers to these questions at present.

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

Application to the grassy Wet Sclerophyll Forests of the BRG region

The work carried out on the North Coast and the more southerly and south-easterly parts of the New England Tableland is applicable to the grassy Wet Sclerophyll Forests of the BRG region. Although there are regional differences, many of the understorey species are also the same from these two areas. Hence the recommendations of Watson (2005) are adopted for grassy Wet Sclerophyll Forests of the BRG region. These are as below.

Summary recommendations

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

vii.   FIRE IN RAINFORESTS

Three rainforest classes are known from the BRG region (Northern Warm Temperate Rainforest; Dry Rainforest and Western Vine Thickets). Northern Warm Temperate Rainforests are restricted to several small gullies at the headwaters of the Deepwater River within Capoompeta National Park in the high elevation (>1200 m) areas at the extreme eastern edge of the region. Dry Rainforests and Western Vine Thickets occur as small patches and occupy only a very small proportion of the BRG region. Despite their small size, the BRG region is recognised as supporting the majority of Western Vine Thicket in NSW. This includes the largest and most significant patches, namely Planchonella Nature Reserve and Derra Derra Ridge (Pulsford 1983; Benson *et.al.* 1996; Williams 1998; Curran *et al.* 2008). Furthermore, these vegetation types are floristically and ecologically distinct from other vegetation types within the BRG region and of high conservation significance.

Northern Warm Temperate Rainforests

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

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

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

field conditions (Tierney unpublished data). Potentially, fire intensity (not just frequency) may contribute to population outcomes for this species.

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

Conclusions – Warm Temperate (and Subtropical) Rainforests

* Warm Temperate (and Subtropical) Rainforests are spatially limited by fire. Fire should be excluded from these Rainforests if these are to be maintained on a given site.
* Because of the exceptionally small area of this vegetation class and the narrow linear patches present within the BRG region, there is a very high risk of loss to wildfire. For obligate rainforest fauna species known from these patches and occurring in small populations, such as the nationally Endangered Stuttering Frog (*Mixophyes balbus*), this presents a high risk of regional extinction.
* Alteration of fire regimes of Warm Temperate and Subtropical Rainforests can drive long-term changes in diversity and distribution of these forests. Alteration to these regimes can accompany fragmentation and development.
* There is some evidence that weed invasion may interact with fire to alter fire regimes in these forests but more research is required to understood how these processes interact in these systems.

Dry rainforests

Dry Rainforests of the BRG region occur on the western slopes of the New England Tableland, along the Nandewar Range and in the gorges and scree slopes radiating off Mt Kaputar (Curran *et al.* 2008; Benson *et al.* 2010). These are typically steep and often rocky sites on soils of medium to high nutrient status (Keith 2004). Other factors that determine the distribution of these forests may include aspect (Bennett 1989), soil moisture (Mackey 1993) and desiccation due to wind (Seifriz 2004).

Floristically these Dry Rainforests characteristically include canopy species regarded as fire sensitive (e.g. *Alectryon subdentatus; Alphitonia exccelsa; Dendrocnide excels; Ficus rubiginosa*; *Pittosporum undulatum*) as well as species that are tolerant of moderate fire frequencies (e.g. a variety of *Eucalyptus* species such as *Eucalyptus albens* and *Eucalyptus melanophloia*). However, species regarded as fire sensitive are often very prominent, implicating fire as a significant factor in limiting the distribution of these Dry Rainforest systems.

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

case being sensitive to fire) so that fire could potentially be used as a management tool to control this species (e.g. Willis 2003).

The fire response of prominent species in the Dry Rainforests of the BRG region was searched in DEC (2002). Species were selected from Group 21 (Rusty Fig Rock Gully Rainforest) of NSW National Parks & Wildlife Service (2002) for species likely to be within western Dry Rainforests. Species reported to be common in the Dry Rainforests studied by Gentle & Duggin (1997) were chosen as likely to be representative of common Dry Rainforest species on the tablelands (eastern Dry Rainforests). The fire response of these species is shown on Table 4. It can be inferred that a number of species in these thickets can establish without fire (including seeder species) and some may resprout but may be sensitive to short fire intervals. Some species (T-species) might potentially be site dominants without fire and the fire ecology of some species is unknown. Thus, it is not possible to confidently prescribe a recommended fire interval on this incomplete dataset. However, fire intervals of <20 years could eliminate species such as *Alphitonia excelsa* from Dry Rainforests whilst longer intervals (of some decades) may allow species such as *Pittosporum undulatum* to recruit and dominate sites. Despite this, no studies are available to determine how these species will function in Dry Rainforest sites (which are ecologically distinct).

Conclusions - Dry Rainforest

It is currently not possible to provide guidelines for recommended fire regimes for the Dry Rainforests. It is probable 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.

Table 4: Fire responses of species known from Dry Rainforests in the BRG region

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

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

Persistance		
Group:		Persistance attributes:
D		Propagules always available; ie widely dispersed
S		Propagule store long lived (and some seedbank remains after disturbance)
G		Propagule store exhausted (germinates or lost) at first disturbance
C		Propagule store short lived (transient or lost at senescence of plant)
V		Resprout but lose reproductively mature tissue; eg lignotuber shoots
U		Resprout and rapidly reproductively mature; eg pyrogenic flowerers
W		Adults resprout (& reproductively mature) but juveniles die
Establishment		
Group:	Tolerance:	Establishment:
I	Intolerant of competion	Establish and grow only after disturbance
T	Tolerant of a wide range of site conditions	Establish and grow both after disturbance and in mature community; eg shade tolerant species
R	Require conditions of mature community	Establish only in mature community, not after disturbance
Transition. Persistance + establishment		
Group:	Species type:	Disturbance regime resulting in local extinction:
1	DT, ST, VT	n/a
2	GT, CT	frequent (interval < m)
3	DI	n/a
4	SI	infrequent (interval > l+e)
5	GI	either (m > interval > l+e)
6	CI	either (m > interval > le)
7	VI	infrequent (interval > l)
8	DR, SR	n/a
9	GR, CR, VR	first disturbance
10a	dT, sT, gT, UT, WT	n/a
10b	dR, sR, gR, UR, WR	n/a
11	dI	n/a
12	sI	infrequent (interval > l+e)
13	gI	either (m > interval > l+e)
14	UI, WI	infrequent (interval > l)

Western Vine thickets

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

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

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

Conclusions – Western Vine Thickets

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

Table 5 Important fire response characteristics of some prominent species in the Western Vine Thickets of the BRG region. The key (above) is from Kenny et al. (2004) and describes the symbols used in the table (a detailed explanation is in Noble & Slater 1980).

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

9. SUMMARY

This review has addressed the fire ecology literature for the most widespread vegetation types in the Border Rivers–Gwydir region (Grasslands; Grassy Woodlands; Semi-arid Woodlands; Dry Sclerophyll Forests), as well as less widespread non–aquatic vegetation types for which significant literature exists (Wet Sclerophyll Forests; Heathlands; Rainforests). The vegetation types recognised in this review fall within the formations and classes of Keith (2004) and their extent and finer-scale delineation was referenced to Benson *et al.* 2009, OEH (nd) and various other locally or regionally relevant sources. Statewide recommended thresholds (Kenny *et al.* 2004) have been used as a foundation for this review, with various regionally and locally specific fire management factors explored and reported herein.

It is a critical need 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*. (2009) report generally longer primary juvenile periods in New England Tableland vegetation when compared to that reported elsewhere and some variations among Tableland species in primary juvenile periods across from the drier western areas when compared to more easterly locations. The number of studies and the spatial extent of the vegetation formation relative to the location of these studies need to be considered in relation to any specific location of interest. A logical extension of this review would be to map studies against environmental gradients and/or spatially map the study locations against the extent of a given formation.
2. The season of burn and the intensity of the fire need to be carefully considered, these variables may significantly affect the outcome of a given fire (e.g. Knox & Clarke 2006b). It may be that these are particularly important considerations because planned fires will typically not be hot summer fires and will be therefore different to fires experienced in the past. The ecological impacts of varying intensity may be greatest at lower intensities because some low intensities may kill or reduce the mass of standing plants but not trigger seed recruitment (e.g. *Banksia marginata* – P. Clarke, pers. com.).
3. This review has not considered all vegetation types present in the BRG region. Wetlands, for example, can be subject to fire and these can cause major changes to the vegetation of these systems. 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.

10.   REFERENCES

Adamson DA & Fox MD (1982) Changes in Australian vegetation since European settlement. In *A history of Australasian vegetation*. (Ed JMB Smith), pp. 109-146. McGraw-Hill: Sydney.

Allan GE & Southgate RI (2002) Fire regimes in the spinifex landscapes of Australia. In *Flammable Australia: the fire regimes and biodiversity of a continent*. (eds R.A. Bradstock, J.E. Williams, A.M. Gill), pp. 145-176. Cambridge Press: Cambridge.

Allen MR (1995) *'In a Writing Unexpected': the Landscape Archaeology of Two Western New South Wales Forests*. Master of Letters thesis, University of New England.

Allen MR (1998) *Forest History Projects for State Forests of New South Wales. Case Studies of Three Cypress Pine Forests in the Lachlan and Bogan River Catchments, Forbes Forestry District, on Back Yamma, Euglo South and Strahorn State Forests*. State Forests of New South Wales, NSW.

Allen CD, Savage M, Falk DA, Suckling KF, Swetnam TW, Schulke T, Stacey PB, Morgan P, Hoffman M & Klingel JT (2002) Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. *Ecological Applications* 12: 1418-1433.

Andersen AN (1988) Immediate and longer-term effects of fire on seed predation by ants in sclerophyllous vegetation in south-eastern Australia. *Australian Journal of Ecology* 13: 285-293

Anderson, ER (1984) The native woody weed problem following Brigalow development in Australia. In A Bailey (ed.), *The Brigalow Belt of Australia*, The Royal Society of Queensland, Brisbane, pp 183-92.

Anderson VJ, Hodgkinson KC & Grice AC (1996) The influence of recent grazing pressure and landscape position on grass recruitment in a semi-arid woldland of eastern Australia. *The Rangeland Journal* 18: 3-9.

Andrew MH (1986) Use of fire for spelling monsoon tallgrass pasture grazed by cattle. *Tropical Grasslands* 20: 69-78.

Andrew N, Rodgerson L & York A (2000) Frequent fuel-reduction burning: the role of logs and associated leaf litter in the conservation of ant biodiversity. *Austral Ecology* 25: 99-107.

Andrews S (2003) *Regrowth White Cypress Pine and Natural Resource Management*. Greening Australia NSW, Armidale, NSW.

Archer S (1990) Development and stability of grass/woody mosaics in a subtropical savannah parkland, Texas, U.S.A. *Journal of Biogeography* 17: 453-462.

Archibald S & Bond WJ (2004) Grazer movements: spatial and temporal responses to burning in a tall-grass African savanna. *International Journal of Wildland Fire* 13: 377-385.

Ashton DH (1976) The development of even-aged stands of *Eucalyptus regnans* F. Muell. In Central Victoria. *Australian Journal of Botany* 24: 397-414.

Ashton DH (1981) Fire in tall open forests (Wet sclerophyll forests). Pp. 339-366 in *Fire and the Australian Biota* (ed. by A.M. Gill, R.H. Groves and I.R. Noble). Australian Academy of Science, Canberra.

Auld TD (1986a) Population dynamics of the shrub *Acacia suaveolens* (Sm.) Willd: dispersal and the dynamics of the soil seed bank. *Australian Journal of Ecology* 11: 235-254.

Auld TD (1986b) Population dynamics of the shrub *Acacia suaveolens* (Sm.) Willd: Fire and the transition to seedlings. *Australian Journal of Ecology* 11: 373-385.

Auld TD (1987a) Population dynamics of the shrub *Acacia suaveolens* (Sm.) Willd: survivorship throughout the life cycle, a synthesis. *Australian Journal of Ecology* 12: 139-151.

Auld TD (1987b) Post-fire demography in the resprouting shrub *Angophora hispida* (Sm). Blaxell: flowering, seed production, dispersal, seedling establishment and survival. *Proceedings of the Linnean Society of New South Wales,* 109: 259-269.

Auld TD (1995) The impacts of herbivores on regeneration in four trees from arid Australia. The *Rangeland Journal* 17: 213-27.

Auld TD (1996) Ecology of the Fabaceae in the Sydney region: Fire, ants, and the soil seedbank. *Cunninghamia* 4: 531-551.

Auld TD, Bradstock RA, Keith DA (1993) Fire as a threat to populations of rare plants. Australian National Parks and Wildlife Service Endangered Species Program Project no.251. New South Wales National Parks and Wildlife Service, Hurstville, New South Wales, Australia.

Auld TD & Denham AJ (1999) The role of ants and mammals in dispersal and post-dispersal seed predation of the shrubs *Grevillea* (Proteaceae). *Plant Ecology* 144: 201-213.

Auld TD, Keith, DA & Bradstock RA (2000) Patterns of longevity of soil seedbanks in fire-prone communities of south-eastern Australia. *Australian Journal of Botany* 48: 539-548.

Auld TD & O'Connell MA (1991) Predicting patterns of post-fire germination in 35 eastern Australian Fabaceae. *Australian Journal of Ecology* 16: 53-70.

Auld TD & Tozer M. (1995) Patterns in emergence of *Acacia* and *Grevillea* seedlings after fire. *Proceedings of the Linnean Society of NSW* 115: 5-15.

Austin MP, Cawsey EM, Baker BL, Yialeloglou MM, Grice DJ, Briggs SV (2000) *Predicted vegetation over in the Central Lachlan Region*. CSIRO, Canberra.

Ayers D, Melville G, Szigethy-Gyula J, Read D, Rees R & Atkinson A (2001) *Woody Weeds and Biodiversity in Western New South Wales*. WEST 2000, Dubbo, NSW.

Barnes DL (1979) Cattle ranching in the semi-arid savannas of east and southern Africa. In Management of semi-arid ecosystems'. (Ed BH Walker) pp. 9-54. (Elsevier: Amsterdan).

Barnes, R.G., Brown, R.E., Brownlow, J.W., Gilligan, L.B., Krynen, J. and Willis, I.L (1988) A review of the mineral deposits of the New England Orogen in New South Wales. Pp. 211–227. In J.D. Kleeman (ed.) *New England Orogen Tectonics and Metallogenesis*. (University of New England: Armidale).

Baxter BJM., van Staden J, Granger JE & Brown NAC (1994) Plant-derived smoke and smoke extracts stimulate seed germination of the fire-climax grass *Themeda triandra*. *Environmental and Experimental Botany* 34: 217-223.

Bean JM & Whalley RDB (2001) Native Grasslands on non-arable slopes of the Garrawillie Creek sub-catchment, western Liverpool Plains, New South Wales. *The Rangeland Journal* 23: 119-47.

Beeston GR, Walker PJ, Purdie R & Pickard J (1980) Plant communities of the Poplar Box (*Eucalyptus populnea*) lands of eastern Australia*. Australian Rangelands* 2: 1-30

Begon M, Harper JL & Townsend CR (1990) *Ecology: Individuals, Populations and Communities*. Blackwell, Oxford.

Bell DT (2001a) Ecological response syndromes in the flora of southwestern Western Australia: fire resprouters versus reseeders. *Botanical Review* 67: 417-440.

Bell SJA (2008) Rare or threatened vascular plant species of Wollemi National Park, central eastern New South Wales. *Cunninghamia* 10 (3),331-371

Bell TL and Pate JS (1996) Growth and fire response of selected Epacridaceae of south-western Australia. *Australian Journal of Botany* 44: 509-526.

Belsky AJ, Mwonga SM, Amundson RG, Duxbury JM & Ali AR (1993) Comparative effects of isolated trees on their undercanopy environments in high- and low-rainfall savannas. *Journal of Applied Ecology* 30: 143-155.

Bennett RJ (1989) Dry rainforest-fire interactions in the Apsley-Macleay gorges: Implications for management. M. Nat. Res. thesis, University of New England, Armidale.

Benson DH (1985) Maturation periods for fire-sensitive shrub species in Hawkesbury sandstone vegetation. *Cunninghamia* 1: 339-349.

Benson JS (1994) The native Grasslands of the Monaro region: Southern Tablelands of NSW. *Cunninghamia* 3: 609-650.

Benson J (1997) Conservation of native lowland grassland remnants in south-eastern Australia, *in* P Hale and D Lamb (eds), Conservation outside nature reserves, Centre for Conservation Biology, The University of Queensland, Brisbane, pp 424-28.

Benson JS 2006, 'New South Wales Vegetation Classification and Assessment: Introduction - the classification, database, assessment of protected areas and threat status of plant communities. *Cunninghamia* 9 (3) 331-381

Benson JS 2008, 'New South Wales Vegetation Classification and Assessment: Part 2 Plant communities in the NSW South-western Slopes Bioregion and update of NSW Western Plains plant communities. Version 2 of the NSWVCA database'. *Cunninghamia* 10 (4) 599-673

Benson, J.B. & Ashby, E. (1996) *Vegetation of the Guyra 1: 100 000 Map Sheet.* Unpublished report to the Australian Heritage Commission. (Royal Botanic Gardens: Sydney).

Benson, J.S., Dick, R. & Zubovic, A. (1996) Semi-evergreen Vine Thicket vegetation at Derra Derra Ridge, Bingara, New South Wales. *Cunninghamia* 4: 497–510.

Benson, J.S. & Ashby, E.M. (2000) Vegetation of the Guyra 1:100 000 map sheet New England Bioregion, New South Wales: *Cunninghamia* 6(3): 747-872.

Benson JS & Redpath PA (1997) The nature of pre-European native vegetation in south-eastern Australia: a critique of Ryan, D.G., Ryan, J.R. and Starr, B.J. (1995) *The Australian Landscape - Observations of Explorers and Early Settlers*. *Cunninghamia* 5: 285-328.

Benson JS, Allen CB, Togher C, Lemmon J. 2006. New South Wales Vegetation Classification and Assessment: Part 1 Plant communities of the NSW Western Plains. *Cunninghamia. 9(3)*: 383-450.

Benson JS, Richards PG, Waller S & Allen CB (2010) New South Wales Vegetation classification and Assessment: Part 3 Plant communities of the 457 NSW Brigalow Belt South, Nandewar and west New England Bioregions and update of NSW Western Plains and South-western Slopes plant communities, Version 3 of the NSWVCA database. *Cunninghamia 11(4):* 457-579.

Benwell AS (1998) Post-fire seedling recruitment in coastal Heathland in relation to regeneration strategy and habitat. *Australian Journal of Botany* 46: 75-101.

Benwell AS (2007) Response of rocky-outcrop and fringing vegetation to disturbance by fire and drought. *Australian Journal of Botany* 55: 736-748.

Bickel DJ & Tasker EM (2004) Tree trunk invertebrates in Australian forests: conserving unknown species and complex processes. Pp. 888-898 in *The Conservation of Australia's Forest Fauna (2nd edition)* (ed. by D. Lunney). Royal Zoological Society of New South Wales, Mosman, NSW.

Binns D & Beckers D (2001) Floristic patterns in the Pilliga. Pp. 104-110 in *Perfumed Pineries: Environmental History of Australia's Callitris Forests* (ed. by J. Dargavel, D. Hart and B. Libbis). Centre for Resource and Environmental Studies, Australian National University, Canberra.

Birk EM & Bridges RG (1989) Recurrent fires and fuel accumulation in even-aged Blackbutt (*Eucalyptus pilularis*) forests. *Forest Ecology and Management* 29: 59-79.

Bond WJ (1980) Fire and senescent fynbos in the Swartberg, Southern Cape. *South African Forestry Journal* 114: 68-71.

Bond WJ (1997) Fire. Pp. 421-446 in *Vegetation of Southern Africa* (ed. by R.M. Cowling, D.M. Richardson and S.M. Pierce). Cambridge University Press, Cambridge.

Bond WJ & Ladd PG (2001) Dynamics of the overstorey and species richness in Australian Heathlands. *Journal of Mediterranean Ecology* 2: 247-257.

Bond WJ & Midgley JM (2001) The persistence niche: the ecology of sprouting in woody plants. *Trends in Ecology and Evolution,* 16: 45-51.

Bond WJ, Midgley GF & Woodward FI (2003) What controls South African vegetation - climate or fire? *South African Journal of Botany* 69: 79-91.

Bond WJ, Woodward FI & Midgley GF (2005) The global distribution of ecosystems in a world without fire. *New Phytologist* 165: 525-538.

Booker LC & Booker MG (1994) A model for the effects of fire and fragmentation on the population viability of the Splendid Fairy-wren. *Pacific Conservation Biology* 1: 334-358.

Booth CA, King GW & Sanchez-Bayo F (1996a) Establishment of woody weeds in western New South Wales. 1. Seedling emergence and phenology. *The Rangeland Journal* 18: 58-79.

Booth CA, King GW & Sanchez-Bayo F (1996b) Establishment of woody weeds in western New South Wales. 2. Growth and Competition Potential. *The Rangeland Journal* 18: 80-98.

Bower C (2005) *NPA Biodiversity Survey, South Mullion Reserve, 22-25 April 2005. Flora.* Unpublished manuscript.

Bowen BJ & Pate JS (1993) The significance of root starch in post-fire shoot recovery of the resprouter *Stirlingia latifolia* R.Br. (Proteaceae). *Annals of Botany* 72: 7-16.

Bowman DMJS (2000) *Australian rainforests:islands of green in a land of fire.* Cambridge University Press.

Bowman DMJS & Harris S (1995) Conifers of Australia's dry forests and open woodlands. Pp. 252-270 in *Ecology of the Southern Conifers* (ed. by N.J. Enright and R.S. Hill). Melbourne University Press, Melbourne.

Bowman DMJS & Latz PK (1993) Ecology of *Callitris glaucophylla* (Cupressaceae) on the MacDonnell Ranges, Central Australia. *Australian Journal of Botany* 41: 217-225.

Bowman DMJS & Panton WJ (1993) Decline of *Callitris intratropica* R.T. Baker & H.G. Smith in the Northern Territory: implications for pre- and post- European colonization fire regimes. *Journal of Biogeography* 20: 373-381.

Bowman DMJS & Wilson BA (1988) Fuel characteristics of coastal monsoon forests, Northern Territory, Australia. *Journal of Biogeography* 15: 807-817.

Bowman DMJS, Wilson BA & Davis GW (1988) Response of *Callitris intratropica* R.T.Baker and H.G.Smith to fire protection, Murgenella, Northern Australia. *Australian Journal of Ecology* 13: 147-159.

Bowman D. M. J. S., Price O, Whitehead P. J. and Walsh A. (2001) The 'wilderness effect' and the decline of *Callitris intratropica* on the Arnhem Land Plateau, northern Australia. Australian Journal of Botany, 49, 665–672

Bradstock RA (1990) Demography of woody plants in relation to fire: *Banksia serrata* Lf. and *Isopogon anemonifolius* (Salisb.) Knight. *Australian Journal of Ecology* 15: 117-132.

Bradstock RA (1991) The role of fire in establishment of seedlings of serotinous species from the Sydney Region. *Australian Journal of Botany* 39: 347-356.

Bradstock RA (1995) Demography of woody plants in relation to fire: *Telopea speciosissima*. *Proceedings of the Linnean Society of New South Wales* 115: 25-33.

Bradstock RA & Auld TD (1995) Soil temperatures during experimental bushfires in relation to fire intensity: consequences for legume germination and fire management in south-eastern Australia. *Journal of Applied Ecology* 32: 76-84.

Bradstock R, Davies I, Price O & Cary G (2008) Climate change and fire: a lay-down misere? Abstracts of the Ecological Society of Australia Annual Conference, Sydney, New South Wales. The Ecological Society of Australia, Alice Springs, NT.

Bradstock RA, Keith DA & Auld TD (1995) Fire and conservation: imperatives and constraints on managing for diversity. Pp. 323-333 in *Conserving Biodiversity: Threats and Solutions* (ed. By R.A. Bradstock, T.D. Auld, D.A. Keith, R.T. Kingsford, D. Lunney and D.P. Sivertsen). Surrey Beatty and Sons, Chipping Norton, NSW.

Bradstock RA & Kenny BJ (2003) An application of plant functional types to fire management in a conservation reserve in southeastern Australia. *Journal of Vegetation Science* 14: 345-354.

Bradstock RA & Myerscough PJ (1988) The survival and population response to frequent fires of two woody resprouters *Banksia serrata* and *Isopogon anemonifolius*. *Australian Journal of Botany* 36: 415-431.

Bradstock RA & O'Connell MA (1988) Demography of woody plants in relation to fire: *Banksia ericifolia* L.f. and *Petrophile pulchella* (Shrad.) R.Br. *Australian Journal of Ecology* 13: 505-518.

Bradstock RA, Tozer MG & Keith DA (1997) Effects of high frequency fire on floristic composition and abundance in a fire-prone Heathland near Sydney. *Australian Journal of Botany* 45: 641-655.

Bradstock RA, Williams JE, Gill MA (Ed) (2002) *Flammable Australia: the fire regimes and biodiversity of a continent*, Cambridge University Press, Cambridge.

Bradstock, R. A. (2010), A biogeographic model of fire regimes in Australia: current and future implications. *Global Ecology and Biogeography,* 19: 145–158.

Brown J, Enright, NJ & Miller BP (2003) Seed production and germination in two rare and three common co-occurring *Acacia* species from south-east Australia. *Austral Ecology* 28: 271-280.

Burbidge AA & McKenzie NL (1989) Patterns in the modern decline of Western Australia's vertebrate fauna: causes and conservation implications. *Biological Conservation* 50: 143-198

Butler, DW (2007) *Recovery plan for the "Brigalow (Acacia harpophylla dominant and co-dominant" endangered ecological community* (draft of 1 May 2007). Report to the Department of the Environment and Water Resources, Canberra. Queensland National Parks and Wildlife Service, Brisbane.

Cambage, R.H. (1904) Notes on the Native Flora of New South Wales. Part ii. Western Slopes of New England. *Proc. Linn. Soc. NSW*, Part 4, October 26th, pp 781–797 +3 plates.

Cambage, R.H. (1908) Notes on the Native Flora of New South Wales. Part vi. Deepwater to Torrington and Emmaville. *Proc. Linn. Soc. NSW* 33: 45–65 +2 plates.

Cambage, R.H. (1912) Notes on the Native Flora of New South Wales. Part ix. Barraba to Nandewar Mountains and Boggabri. *Proc. Linn. Soc. NSW* 37: 622–651 +2 plates.

Cameron, P. undated. *Actively managing for biodiversity: some observations of the benefits of thinning cypress stands and discussion of management options for western forests and woodlands.* Document. Western Plains Zoo, Dubbo.

Campbell ML & Clarke PJ (2006) Response of montane Wet sclerophyll forest understorey species to fire: evidence from high and low intensity fires. *Proceedings of the Linnean Society of New South Wales* 127: 63-73.

Cary GJ & Morrison DA (1995) Effects of fire frequency on plant species composition of sandstone communities in the Sydney region: combinations of inter-fire intervals. *Australian Journal of Ecology* 20: 418-426.

Catling PC & Burt RJ. (1995). Studies of the ground-dwelling mammals of eucalypt forests in south-eastern New South Wales: the effect of habitat variables on distribution and abundance. *Wildlife Research* **22**, 271–288.

Catling PC, Burt RJ & Forrester RI (2000) Models of the distribution and abundance of ground dwelling mammals in the eucalypt forests of north-eastern New South Wales in relation to habitat variables. *Wildlife Research* 27: 639-654.

Chilcott C, Reid NCH & King K (1997) Impact of trees on the diversity of pasture species and soil biota in grazed landscapes on the Northern Tablelands, NSW. Pp. 378-386 in *Conservation Outside Nature Reserves* (ed. by P. Hale and D. Lamb). Centre for Conservation Biology, University of Queensland, Brisbane.

Christensen PE & Kimber PC (1975) Effect of prescribed burning on the flora and fauna of southwest Australian forests. *Proceedings of the Ecological Society of Australia* 9: 85-106.

Christensen P (1998) The precautionary principle and grazing, burning and medium sized mammals in northern New South Wales. *Australian Forestry* 61: 195-203.

Clarke, P.J., Copeland, L.M., Hunter, J.T., Nano, C.E., Williams, J.B. and Wills, K.E. (1998a) *The Vegetation and Plant Species of Torrington State Recreation Area*. (Division of Botany, University of New England: Armidale).

Clarke, P.J., Gardener, M.R., Nano, C.E. & Whalley, R.D.B. (1998b) *The vegetation and plant species of Kirramingly* (Division of Botany, University of New England: Armidale).

Clarke PJ (2000) Plant population processes in temperate woodlands of eastern Australia – premises for management. Pp. 248-270 in *Temperate Eucalypt Woodlands in Australia: Biology, Conservation, Management and Restoration* (ed. by R.J. Hobbs and C.J. Yates). Surrey Beatty and Sons, Chipping Norton, NSW.

Clarke PJ & Davison EA (2001) Experiments on the mechanism of tree and shrub establishment in temperate Grassy woodlands: seedling emergence. *Austral Ecology* 26: 400-412.

Clarke PJ & Knox KJE (2002) Post-fire response of shrubs in the tablelands of eastern Australia: do existing models explain habitat differences? *Australian Journal of Botany* 50: 53-62.

Clarke PJ, Knox KJE, Wills KE & Campbell M (2005) Landscape patterns of woody plant to crown fire: disturbance and productivity influence sprouting ability. *Journal of Ecology* 93: 544-555.

Clarke PJ, Latz PK & Albrecht DE (2005) Long-term changes in semi-arid vegetation: Invasion of an exotic perennial grass has larger effects than rainfall variability. *Journal of Vegetation Science* 16: 237-248.

Clarke PJ, Knox, KJE, Campbell ML & Copeland LM. (2009) Post-fire recovery of woody plants in the New England Tableland Bioregion. *Cunninghamia 11(2): 221-239*.

Clarke S (2003) *The Effect of Fire and Grazing on Cumberland Plain Woodlands*. Master of Science (Research) thesis, University of Wollongong.

Clayton-Greene KA & Ashton DH (1990) The dynamics of *Callitris columellaris/Eucalyptus albens* communities along the Snowy River and its tributaries in south-eastern Australia. *Australian Journal of Botany* 38: 403-432.

Clements, FE (1916) *Plant succession: An analysis of the development of vegetation*. Carnegie Institution of Washington Publication 242. Washington, DC: Carnegie Institution of Washington.

Coates F, Lunt ID & Tremblay RL (2006) Effects of disturbance on population dynamics of the threatened orchid *Prasophyllum correctum* D.L. Jones and implications for Grassland management in south-eastern Australia. *Biological Conservation* 129: 59-69.

Cole I & Lunt ID (2005) Restoring Kangaroo Grass (*Themeda triandra*) to Grassland and woodland understorys: a review of establishment requirements and restoration exercises in south-east Australia. *Ecological Management and Restoration* 6: 28-33.

Connah, G.E., Davidson, I. & Rowland, M.J. (1977) Prehistoric settlement. Pp 127–136 in Lea, D.A.M., Pigram, J.J. & Greenwood, L. (eds.) *An atlas of New England*: *vol 2 The Commentaries* (Department of Geography, University of New England: Armidale).

Connell JH (1978) Diversity in tropical rain forests and coral reefs. *Science* 199: 1302-1310.

Connell JH & Slatyer RO (1977) Mechanisms of succession in natural communities and their role in community stability and organization. *American Naturalist* 111: 1119-1144.

Copeland LM., Craven LA & Bruhl JJ. (2011) A taxonomic review of *Homoranthus* (Myrtaceae: Chamelaucieae). *Australian Systematic Botany* **24**, 351–374.

Cowling RM & Gxaba T (1990) Effects of a fynbos overstorey shrub on understorey community structure: implications for the maintenance of community-wide species richness. *South African Journal of Ecology* 1: 1-7.

Cowling RM & Lamont BB (1987) Post-fire recruitment of four co-occurring *Banksia* species. *Journal of Applied Ecology* 24: 645-658.

Cowling RM, Lamont BB & Enright NJ (1990) Fire and management of south-western Australian banksias. *Proceedings of the Ecological Society of Australia* 16: 177-183.

Croft M, Goldney, D & Cardale, S (1997) Forest and woodland cover in the Central Western Region of New South Wales prior to Euopean settlement. Pp. 394-406 in *Conservation Outside Nature Reserves* (ed. by P. Hale and D. Lamb). Centre for Conservation Biology, University of Queensland, Brisbane.

Crowley GM (2001) Grasslands of Cape York Peninsula - a fire-dependent habitat. P. 34 in *Savanna Burning: Understanding and Using Fire in Northern Australia* (ed. by R. Dyer, P. Jacklyn, I. Partridge, J. Russell-Smith and R. Williams). Tropical Savannas CRC, Darwin.

Crowley GM & Garnett S (1998) Vegetation change in the Grasslands and Grassy woodlands of east-central Cape York Peninsula, Australia. *Pacific Conservation Biology* 4: 132-148.

CSIRO (2007) *Climate change in the Border Rivers – Gwydir catchment.* Prepared for the NSW Government. CSIRO, Canberra.

Cunningham, A. (1825) *Journal of the route from Bathurst to Liverpool Plains.* In: Geographical Memoirs on New South Wales. Barron Field, ESQ. F.L.S. (Ed.), John Murray, London. pp. 131.

Cunningham GM & Milthorpe PL (1976) *Twenty-five Years with Exclosures*. Soils Conservation Service of New South Wales, Western District, Condobolin.

Cunningham GM, Mulham WE, Milthorpe PL & Leigh JH (1992) *Plants of Western New South Wales*. Inkata Press, Sydney.

Curran, T.J., Clarke, P.J. & Bruhl, J.J. (2008) A broad typology of dry rainforests on the western slopes of New South Wales. *Cunninghamia* 10: 381–405.

Curtis D (1990) Natural regeneration of eucalypts in the New England region. Pp. 7-16 in *'Sowing the Seeds': Direct Seeding and Natural Regeneration Conference* . Greening Australia, Canberra.

Curtin, FK (1987). *Notes on establishment of Cypress Pine regeneration*. For. Comm. NSW report, 9pp.

Daly RL & Hodgkinson KC (1996) Relationship between grass, shrub and tree cover on four landforms of semi-arid eastern Australia. *The Rangeland Journal* 18: 104-117.

Date EM, Ford HA & Recher HF (2002) Impacts of logging, fire and grazing regimes on bird species assemblages of the Pilliga woodlands of New South Wales. *Pacific Conservation Biology* 8: 177-195.

DEC (2002) *NSW Flora Fire Response* DEC (Department of Environment and Conservation NSW) *Database. Version 1.3a.* Department of Environment and Conservation, Hurstville, NSW.

DEC (2005) NSW Department of Environment and Climate Change (NSW DECC) (2005). *NSW Threatened Species - Pilliga Mouse - profile*.

Denham AJ & Auld TD (2002) Flowering, seed dispersal and seedling recruitment in two pyrogenic flowering resprouters. *Australian Journal of Botan*y 50: 545-557.

Denham AJ & Whelan RJ (2000) Reproductive ecology and breeding system of *Lomatia silaifolia* (Proteaceae) following a fire. *Australian Journal of Botany* 48: 261-269.

Department of the Environment (undated) Indigenous Protected Areas www.environment.gov.au/indigenous/ipa/index.html Accessed May 2014

Dixon KW, Roche S & Pate JS (1995) The promotive effect of smoke derived from burnt native vegetation on seed germination of Western Australian plants. *Oecologia* 101: 185-192.

Dorrough J, Ash J & McIntyre S (2004) Plant responses to livestock grazing frequency in an Australian temperate grassland. *Ecography* 27: 798-810.

Eco Logical (2008) A vegetation map for the Namoi Catchment Management Authority. Unpublished report to Namoi Catchment Management Authority by Eco Logical Australia, Coffs Harbour, New South Wales.

Eddy D, Mallinson D, Rehwinkel R & Sharp S (1998) *Grassland Flora: a Field Guide for the Southern Tablelands (NSW & ACT)*. Environment ACT, Canberra.

Eldridge DJ & Freudenberger D (2005) Ecosystem wicks: Woodland trees enhance water infiltration in a fragmented agricultural landscape in eastern Australia. *Austral Ecology* 30: 336-347.

Eldridge DJ, Wilson BR & Oliver I (2003) *Regrowth and Soil Erosion in the Semi-arid woodlands of New South Wales: a Report to the Native Vegetation Advisory Council*. NSW Department of Land and Water Conservation, Sydney.

Engle DM, Stritzke JF & Claypool PL (1987) Herbage standing crop around eastern red cedar trees. *Journal of Range Management* 40: 237-239.

Enright NJ & Goldblum D (1999) Demography of a non-sprouting and resprouting *Hakea* species (Proteaceae) in fire-prone *Eucalyptus* woodlands of southeastern Australia in relation to stand age, drought and disease. *Plant Ecology* 144: 71-82.

Enright NJ, Marsula R, Lamont BB, Wissel C (1998) The ecological significance of canopy seed storage in fire-prone environments: a model for non-sprouting shrubs. *Journal of Ecology* 86: 946-959.

Facelli JM & Temby AM (2002) Multiple effects of shrubs on annual plant communities in arid lands of South Australia. *Austral Ecology* 27: 422-432.

Fensham RJ (1996) Land clearance and conservation of inland dry rainforest in North Queensland, Australia. *Biological Conservation* 75: 289-298.

Fensham RJ (1998) The grassy vegetation of the Darling Downs, south-eastern Queensland, Australia. Floristics and grazing effects. *Biological Conservation* 84: 301-310.

Fisher AM & Harris SJ (1999) The dynamics of tree cover change in a rural Australian landscape. *Landscape and Urban Planning* 45: 193-207.

Flematti GR, Ghisalberti EL, Dixon KW & Trengove RD (2004) A compound from smoke that promotes seed germination. *Science* 305: 977.

Florence RG (1996) *Ecology and Silviculture of Eucalypt Forests*. CSIRO Publishing, Collingwood.

Floyd AG (1976) Effect of burning on regeneration from seeds in Wet sclerophyll forest. *Australian Forestry* 39: 210-220.

Ford HA, Barrett GW, Saunders DA & Recher HF (2001) Why have birds in the woodlands of Southern Australia ddeclined? *Biological Conservation* 97: 71-88.

Fordyce IR, Eamus D & Duff GA (2000) Episodic seedling growth in *Allosyncarpia ternata*, a lignotuberous, monsoon rainforest tree in northern Australia. *Austral Ecology* 25: 25035.

Fox, A.M. (1978) The '72 fire of Nadgee Nature Reserve. *Parks and Wildlife* **2**, 5-24.

Fox BJ (1982) The '72 fire of Nadgee Nature Reserve. *Parks and Wildlife* 2: 5-24.

Fox MD & Fox BJ (1986) The effect of fire frequency on the structure and floristic composition of a woodland understorey. *Australian Journal of Ecology* 11: 77-85.

Fox BJ, Fox MD, Taylor JE, Jackson GP, Simpson J, Higgs P, Rebec L & Avery R (1996) Comparison of regeneration following burning, clearing or mineral sand mining at Tomago, NSW: Structure and growth of the vegetation. *Australian Journal of Ecology* 21: 184-199.

French K & Westoby M (1996) Vertebrate-dispersed species in a fire-prone environment. *Australian Journal of Ecology* 21: 379-385.

Friedel M.H.& James C. D. (1995) How grazing of native pastures affect their biodiversity? In: *Conserving biodiversity threats and solutions.* (Ed. R.A. Bradstock, T.D. Auld, D.A. Keith, R.T. Kingsford, D. Lunney & D. Siversten) pp. 250-259 Surrey Beatty and Sons Pty Ltd, Chipping Norton.

Fynn RWS, Haynes RJ & O'Connor TG (2003) Burning causes long-term changes in soil organic matter content of a South African Grassland. *Soil Biology and Biochemistry* 35: 677-687.

Gellie NJH (2005) Native vegetation of southern forests: South-east Highlands, Australian Alps, South-west Slopes, and SE Corner bioregions. *Cunninghamia* 9: 219-254.

Gentle CB & Duggin JA (1997) *Lantana camara* L. invasions in dry rainforest-open forest ecotones: The role of disturbances associated with fire and cattle grazing. *Australian Journal of Ecology* 22: 298-306.

Gibbs L, Reid N & Whalley RDB (1999) Relationships between tree cover and grass dominance in a grazed temperate stringybark (*Eucalyptus laevopinea*) open-forest. *Australian Journal of Botany* 47: 49-60.

Gill AM (1975) Fire and the Australian flora: a review. *Australian Forestry* 38: 4-25.

Gill AM (1981) Adaptive responses Australian vascular responses to fire. In *Fire and the Australian Biota*. (ed. A M Gill, R H Groves & I R Noble) pp. 243-72. Australian Academy of Science, Canberra.

Gill AM (1997) Eucalypts and fires: interdependent or independent? Pp. 151-167 in *Eucalypt Ecology: Individuals to Ecosystems* (ed. by J. Williams and J. Woinarski). Cambridge University Press, Cambridge.

Gill AM & Bradstock R (1992) A national register for the fire responses of plant species. *Cunnninghamia* 2: 653-660.

Gill AM & Catling PC (2002) Fire regimes and biodiversity of forested landscapes of southern Australia. *Flammable Australia: the fire regimes and biodiversity of a continent*. (eds R.A. Bradstock, J.E. Williams, A.M. Gill), pp. 351-369. Cambridge Press: Cambridge.

Goldney DC & Bowie IJS (1990) Some management implications for the conservation of vegetation remnants and associated fauna in the central western region of New South Wales. *Proceedings of the Ecological Society of Australia* 16, 427-440.

Graham, MS, Watson P., and Tierney, D. (2012) *Fire and the Vegetation of the Lachlan Region.* Hotspots Fire Project, Nature Conservation Council of NSW, Sydney.

Greenslade P (1997) Short term effects of a prescribed burn on invertebrates in grassy woodland in south-eastern Australia. *Memoirs of the Museum of Victoria* 56: 305-312.

Grice AC & Barchia I (1992) Does grazing reduce survival of indigenous perennial grasses of the Semi-arid woodlands of western New South Wales? *Australian Journal of Ecology* 17: 195-205.

Griffiths T (2001) One hundred years of environmental crisis. *The Rangeland Journal* 23: 5-14.

Griffiths T (2002) How many trees make a forest? Cultural debates about vegetation change in Australia. *Australian Journal of Botany* 50:375-389.

Grose E & Zimmer WJ (1958) Preliminary laboratory studies on light requirements for the germination of some eucalypt seeds. *Australian Forestry* 21: 76-80.

Gross CL & Caddy HAR (2006) Are differences in breeding mechanisms and fertility among populations contributing to rarity in *Grevillea rhizomatosa* (Proteaceae)? *American Journal of Botany* 93: 1791-9.

Grubb P (1977) The maintenance of species richness in plant communities: the importance of the regeneration niche. *Biological Review,* 52: 107-145.

Guinto DF, House APN, Xu ZH, Saffigna PG (1998) Changes in soil chemical properties and forest floor nutrients under repeated prescribed burning in eucalypt forests of south-east Queensland, Brisbane, Queensland Forestry Research Institute.

Hammill KA, Bradstock RA & Allaway WG (1998) Post-fire seed dispersal and species reestablishment in proteaceous heath. *Australian Journal of Botany* 46: 407-419.

Hannah D, Woinarski JCZ, Catterall CP, McCo sker JC, Thurgate NY, Fensham RJ (2007). Impact of clearing, fragmentation and disturbance on the bird fauna of eucalypt savanna woodland in central Queensland, Australia*. Austral Ecology* 32,261-276.

Harrington GN (1991) Effects of soil moisture on shrub seedling survival in a semi-arid grassland. *Ecology* 72: 1138 –1149

Harrington GN (1995) Should we play God with rainforest? *Wildlife Australia* Winter: 8-11.

Harrington GN, Oxley RE & Tongway DJ (1979) The effects of European settlement and domestic livestock on the biological system in poplar box (*Eucalyptus populnea*) lands. *The Rangeland Journal* 1: 271-279.

Harrington GN & Johns GG (1990) Herbaceous biomass in a *Eucalyptus* savanna woodland after removing trees and/or shrubs. *Journal of Applied Ecology* 27: 775-787.

Harrington GN & Sanderson KD (1994) Recent contraction of wet sclerophyll forest in the Wet Tropics of Queensland due to invasion by rainforest. *Pacific Conservation Biology* 1: 319-327.

Harris MR & Lamb D (2004) A history of recent disturbances in white cypress pine forests in southern Queensland. *Proceedings of the Royal Society of Queensland* 111: 19-30.

Harris MR, Lamb D, Erskine PD (2003) An investigation into the possible inhibitory effects of white cypress pine (*Callitris glaucophylla*) litter on the germination and growth of associated ground cover species. *Australian Journal of Botany* 51: 93-102.

Harrold A (1979) Heathland regeneration after fire at Noosa. *Queensland Naturalist* 22: 88-96.

Hawkins PJ (1966) Seed production and litter fall studies of *Callitris collumellaris*. *Australian Forest Research* 2: 3-16.

HCCREMS (2008) Hunter, Central & Lower North Coast vegetation classification & mapping project; Volume 1 vegetation classification technical report.

Henderson MK & Hocking C (undated) *Vegetation dynamics in response to fire and slashing in remnants of Western Basalt Plains grasslands: preliminary results*. Unpublished report, Environment Australia.

Henderson MK & Keith DA (2002) Correlation of burning and grazing indicators with composition of woody understorey flora of dells in a temperate eucalypt forest. *Austral Ecology* 27: 121-131.

Henderson W & Wilkins GW (1975) The interaction of bushfire and vegetation. *Search* 6: 130-133.

Henry NB (1960) The effect of silvicultural treatment on the production from native forests. *Australian Forestry* 24: 30-45.

Henry NB & Florence RG (1966) Establishment and development of regeneration in Spotted Gum - Ironbark forests. *Australian Forestry* 30: 304-316.

Hessburg PF, Smith BG, Salter RB, Ottmar RD & Alvarado E (2000) Recent changes (1930s - 1990s) in spatial patterns of interior northwest forests, USA. *Forest Ecology and Management* 136: 53-83.

Hill SJ & French K (2003) Response of the soil seed-bank of Cumberland Plain Woodland to heating. *Austral Ecology* 28: 14-22.

Hitchmough JD, Curtain H., Hammersley L and Kellow J (1996) Effect of gap width and turf type on the establishment of the Australian forb *Bulbine bulbosa*. *Restoration Ecology* 4: 25-32.

Hobbs RJ & Hopkins AJM (1990) From frontier to fragments: European impact on Australia's vegetation. *Proceedings of the Ecological Society of Australia* 16: 93-114.

Hobbs RJ & Huenneke LF (1992) Disturbance, diversity, and invasion: implications for conservation. *Conservation Biology* 6: 324-337.

Hodgkinson KC (1986) Responses of rangeland plants to fire in water-limited environments. In *Rangelands: A resource under siege*. (eds. P.J.Joss, P.W. Lynch and O.B. Williams) pp 437-441. Proceedings of the 2nd International Rangeland Conference, Adelaide. Australian Academy of Science, Canberra/ Cambridge University Press, Melbourne

Hodgkinson KC (1993) Identification of critical thresholds to opportunistic management of rangeland vegetation. Pp.127-9 in: *Proceedings IVth International Rangeland Congress*. Montepellier, France, 1991.

Hodgkinson KC (2002) Fire regimes in *Acacia* wooded landscapes: effects on functional processes and biological diversity. Pp. 257-277 in *Flammable Australia: The Fire Regimes and Biodiversity of a Continent* (ed. by R.A. Bradstock, J.E. Williams and A.M. Gill). Cambridge University Press, Cambridge.

Hodgkinson KC & Griffin GF (1982) Adaptation of shrub species to fire in the arid zone. In *Evolution of the flora and fauna of Arid Australia.* (eds W.R. Barker and P.J.M. Greenslade) pp 145-152. (Peacock Publications: Adelaide

Hodgkinson KC & Harrington GN (1985) The case for prescribed burning to control shrubs in eastern Semi-arid woodlands. *The Rangeland Journal* 7: 64-74.

Hodgkinson KC, Harrington GN, Griffin GF, Noble JC & Young MD (1984) Management of vegetation with fire. Pp. 141-156 in *Management of Australia's Rangelands:* (ed. by G.N. Harrington, A.D. Wilson and M.D. Young). CSIRO, Canberra.

Hodgkinson KC & Muller WJ (2005) Death model for tussock perennial grasses: a rainfall threshold for survival and evidence for landscape control of death in drought. *The Rangeland Journal* 27: 105-115.

Horne R (1990a) Early espacement of wheatfield white cypress pine regeneration: the effect on secondary regeneration, limb size, and stand merchantability. *Australian Forestry* 53: 160-167.

Horne R (1990b) Stand height response following variable spacing of wheatfield white cypress pine regeneration in New South Wales. *Australian Forestry* 53: 47-54.

House S (1997) Reproductive biology of eucalypts. In 'Eucalypt ecology: individuals to ecosystems'. (Ed J Williams, JWoiarski) pp.30-55. (Cambridge , UK).

Hunter, J.T. (1995) Some observations on the fire responses of two rare species in the Girraween and Bald Rock National Parks. Queensland Naturalist 33: 146–147.

Hunter, J.T. (1997a) A further record of Leucopogon cicatricatus J. Powell (Epacridaceae) for Queensland. Queensland Naturalist 34: 20–21.

Hunter, J.T. (1997b) A further record of Homoranthus lunatus for northern New South Wales. Queensland Naturalist 35: 24–25.

Hunter, J.T. (1997c) Acacia williamsiana (Fabaceae: Juliflorae) a new granitic outcrop species from northern New South Wales. Journal of the Royal Society of Western Australia 80: 235–237.

Hunter, J.T. (1998a) Two new rare species of Homoranthus (Myrtaceae: Chamelaucieae) from the Northern Tablelands of New South Wales. Telopea 8(1): 35–40

Hunter, J.T. (1998b) The botany of Howell, New South Wales: A tin granite flora (revisited). The Victorian Naturalist

Hunter, J.T. (1998c) Vegetation Survey of the Proposed Kwiambal National Park. Unpublished Report for the Glen Innes National Parks & Wildlife Service. (Armidale).

Hunter, J.T. (2003) Vegetation of Arakoola Nature Reserve, North Western Slopes, New South Wales. Cunninghamia 8: 157–284.

Hunter, J.T. (2006a) Vegetation and floristics of the Planchonella Nature Reserve. Unpublished report to the NSW Department of Environment and Climate Change

Hunter, J.T. (2006b) Vegetation and floristics of Dthinna Dthinnawan Nature Reserve. Report to NSW Parks and Wildlife Service, Tenterfield Area.

Hunter, J.T. (2009a) Vegetation and floristics of the Derra Derra section of the Bingara State Conservation Area. Unpublished report to the Department of Environment and Climate Change.

Hunter, J.T. (2009b) Vegetation and floristics of the Sepoy section of the Gwydir River National Park. Unpublished report to the NSW Department of Environment, Climate Change and Water.

Hunter, J.T. (2009c) Vegetation and mapping of Bullala National Park. Unpublished report to the NSW Department of Environment, Climate Change and Water.

Hunter, J.T. (2009d) Vegetation and mapping of Terri Hie Hie Aboriginal Area. Unpublished report to the NSW Department of Environment, Climate Change and Water.

Hunter, J.T. (2009e) Vegetation and mapping of Berygill Aboriginal Area. Unpublished report to the NSW Department of Environment, Climate Change and Water.

Hunter, J.T. (2009f) Vegetation and mapping of Courallie Aboriginal Area. Unpublished report to the NSW Department of Environment, Climate Change and Water.

Hunter, J.T. (2009g) Vegetation and mapping of the Mission Aboriginal Area. Unpublished report to the NSW Department of Environment, Climate Change and Water

Hunter, J.T. (2009h) Vegetation and mapping of Irragrippa Aboriginal Area. Unpublished report to the NSW Department of Environment, Climate Change and Water.

Hunter, J.T. (2009i) Vegetation and mapping of Campbell and Montrose Aboriginal Areas. Unpublished report to the NSW Department of Environment, Climate Change and Water.

Hunter, J.T. (2009j) Vegetation and mapping of Gunyerwarildi National Park. Unpublished report to the NSW Department of Environment, Climate Change and Water.

Hunter, J.T. (2009k) Vegetation and floristics of Horton Falls National Park. Unpublished report to the NSW Department of Environment, Climate Change and Water.

Hunter, J.T. (2009l) Vegetation and floristics of the Molroy section of the Bingara State Conservation Area. Unpublished report to the NSW Department of Environment, Climate Change and Water.

Hunter, J.T. (2010) Vegetation of the Mehi section of the Gwydir River National Park. Unpublished report to NSW Department of Environment, Climate Change and Water.

Hunter JT (2011a) A broad brush-stroke test of an assumption: does increasing Callitris cover reduce native species richness (species density)? Cunninghamia 12, 115–118

Hunter, J.T. (2011b) Vegetation and floristics of Munro South section of the Gwydir River National Park. Unpublished report to the NSW Department of Environment, Climate Change and Water.

Hunter, J.T. (2012a) Vegetation and floristics of Warialda North and South. Unpublished report to the NSW Office of Environment and Heritage.

Hunter, J.T. (2012b) Vegetation and floristics of the Salmon and Murchison sections of the Gwydir River National Park. Unpublished report to the NSW Office of Environment and Heritage.

Hunter, J.T. (2012c) Vegetation and floristics of the Mehi Section of the Gwydir River National Park Noonga Addition Supplementary Appendix. Unpublished report to the NSW Office of Environment and Heritage.

Hunter, J.T. (2013) Interactions between Callitris above-ground biomass, species density and plant form in north-eastern New South Wales. *Australian Journal of Botany* 61: 73–79.

Hunter, J.T. & Bruhl, J.J. (1996) Significant range extensions for species of vascular plants in northern New South Wales. Austrobaileya 4: 77–81.

Hunter, J.T. & Williams, J.B. (1994) A new species of Brachyloma and three new subspecies of B. daphnoides (Epacridaceae) from south-eastern Australia Telopea 6: 1–7.

Hunter, J.T., Quinn, F.C. & Bruhl, J.J. (1996) A new species of Micromyrtus (Myrtaceae) from New South Wales. Telopea 6: 77–81

Hunter, J.T., Fallavollita, E. & Hunter, V.H. (1998) Observations on the ecology of Muehlenbeckia costata m.s. (Polygonaceae), a rare fire ephemeral species occurring on the New England Batholith of north-eastern New South Wales and southern Queensland. The Victorian Naturalist 115: 9–17.

Hunter, J.T., Kingston, J. & Croft P. (1999) Vegetation and floristics of Kwiambal National Park and surrounds, Ashford, New South Wales. Cunninghamia 6: 351–378.

Huston M (1979) A general hypothesis of species diversity. *American Naturalist* 113: 81-101.

Huston M (2003) Understanding the effects of fire and other mortality-causing disturbances on species diversity. Pp. 37-70 in *Fire in Ecosystems of South-west Western Australia* (ed. by I. Abbott and N. Burrows). Backhuys Publishers, Leiden, The Netherlands.

Huston M (2004) Management strategies for plant invasions: manipulating productivity, disturbance, and competition. *Diversity and Distributions* 10: 167-178.

Johnson, RW (1964) *Ecology and control of brigalow in Queensland*. Queensland Department of Primary Industries.

Johnson GA (1999) *The role of fire in Phalaris and Paspalum control in grassy ecosystems*. Paper presented at the Australian Bushfire Conference, Albury, July 1999 http://www.csu.edu.au/special/bushfire99/papers/johnson. Accessed 5 September 2006.

Johnson, RW and Purdie, RW (1981) The role of fire in the establishment and management of agricultural systems. In AM Gill, RH Groves and IR Noble (eds) (1981), *Fire and the Australian biota*, Australian Academy of Science, Canberra, pp 497-528.

Johnston TN (1969) The effect of sheep and rabbit grazing on regeneration of white cypress pine. *Australian Forest Research* 4: 3-12.

Johnston TN & Jennings KS (1991) Management of cypress pine forests in Queensland. Pp. 182- 189 in *Forest Management in Australia* (ed. by F.H. McKinnell, E.R. Hopkins and J.E.D. Fox). Surrey Beatty, Chipping Norton, NSW.

Jurskis V and Turner J (2002) Eucalypt dieback in eastern Australia: a simple model. *Australian Forestry* 65: 81-92.

Kavanagh R & Barrott E (2001) Koala populations in the Pilliga forests. Pp. 93-103 in *Perfumed Pineries: Environmental History of Australia's Callitris Forests* (ed. by J. Dargavel, D. Hart and B. Libbis). Centre for Resource and Environmental Studies, Australian National University, Canberra.

Keane RE, Ryan KC, Veblen TT, Allen CD, Logan J & Hawkes B (2002) *Cascading effects of fire exclusion in Rocky Mountain ecosystems: a literature review*. USDA Forest Service, Rocky Mountains Research Station General Technical Report RMRS-GTR-91. Available on the internet at http://www.fs.fed.us/rm/pubs/rmrs_gtr091.html, Fort Collins, Colorado.

Keeley JE & Bond WJ (1997) Convergent seed germination in South African fynbos and Californian chaparral. *Plant Ecology* 133: 153-67.

Keith DA (1996) Fire-driven extinction of plant populations: a synthesis of theory and review of evidence from Australian vegetation. *Proceedings of the Linnean Society of NSW* 116: 37-78.

Keith DA (2004) Ocean Shores to desert dunes: the native vegetation of New South Wales and the ACT. Department of Environment and Conservation (NSW), Hurstville, NSW.

Keith DA, Akcakaya HR, Thuiller W, Midgley GF, Pearson RG, Phillips SJ, Regan HM, Araujo MB & Rebelo TG (2008) Predicting extinction risks under climate change: coupling stochastic models with dynamic bioclimatic models. *Biology Letters* 4: 560-563.

Keith DA & Bradstock RA (1994) Fire and competition in Australian heath: a conceptual model and field investigations. *Journal of Vegetation Science* 5: 347-54.

Keith DA, McCaw WL, Whelan RJ (2002a) Fire regimes in Australian Heathlands and their effects on plants and animals. In: '*Flammable Australia: the fire regimes and biodiversity of a continent*.' (Eds RA Bradstock, JE Williams and AM Gill). pp. 199-237. (Cambridge Press: Cambridge)

Keith DA, Williams JE & Woinarski JCZ (2002b) Fire management and biodiversity conservation: key approaches and principles. Pp. 401-425 in *Flammable Australia: The Fire Regimes and Biodiversity of a Continent* (ed. by R.A. Bradstock, J.E. Williams and A.M. Gill). Cambridge University Press, Cambridge.

Kelly C (2004) *The effects of fire frequency on the understorey of a subalpine snow gum forerst.* Bachelor of (Environment) Science (honours) thesis, The Australian National University.

Kenny B (2000) The influence of multiple fire-related germination cues on three Sydney *Grevillea* (Proteaceae) species. *Austral Ecology* 25: 664-669.

Kenny B, Sutherland E, Tasker E & Bradstock R (2004) *Guidelines for Ecologically Sustainable Fire Management*. NSW National Parks and Wildlife Service, Hurstville.

Kershaw AP, Clarke JS, Gill AM & D'Costa DM (2002) A history of fire in Australia. *Flammable Australia: the fire regimes and biodiversity of a continent*. (eds R.A. Bradstock, J.E. Williams, A.M. Gill), pp. 3-25. Cambridge Press: Cambridge.

Kirkpatrick, J., McDougall, K. and Hyde, M. (1995) *Australia's Most Threatened Ecosystem: the Southeastern Lowland Native Grasslands*. Surrey Beatty and Sons, Chipping Norton, NSW.

Kitchin MB (2001) *Fire Ecology and Fire Management for the Conservation of Plant Species and Vegetation Communities in a National Park in Northern NSW, Australia*. PhD thesis, University of New England.

Knox KJE & Clarke PJ (2004) Fire response syndromes of shrubs in grassy woodlands in the New England Tableland Bioregion. *Cunninghamia* 8: 348-353.

Knox KJE & Clarke PJ (2006) Response of resprouting shrubs to repeated fires in dry sclerophyll forest of Gibraltar Range National Park. *Proceedings of the Linnean Society of NSW* 127: 49-56.

Knox, KJE & Morrison DA (2005) Effects of inter-fire intervals on the reproductive output of resprouters and obligate seeders in the Proteaceae. *Austral Ecology* 30: 407-413.

Lacey CJ (1972) *Factors Influencing Occurrence of Cypress Pine Regeneration in New South Wales*. Technical Paper 21, Forestry Commission of NSW, Sydney.

Lacey CJ (1973) *Silvicultural Characteristics of White Cypress Pine*. Research Note 26, Forestry Commission of NSW, Sydney.

Ladd PG (1988) The status of Casuarinaceae in Australian forests. Pp. 63-85 in *Australia's Ever-Changing Forests: Proceedings of the First National Conference on Australia's Forest History* (ed. by K.J. Frawley and N. Semple). Department of Geography and Oceanography, Australian Defence Force Academy, Campbell, ACT.

Lamont BB, Enright NJ & Bergl SM (1989) Coexistence and competitive exclusion of *Banksia hookeriana* in the presence of congeneric seedlings along a topographic gradient. *Okios* 24: 39-42.

Lamont BB, Enright NJ, Witkowski ETF & Groeneveld J (2007) Conservation biology of banksias: insights from natural history to simulation modelling. *Australian Journal of Botany* 55: 280-92.

Lamont BB, Le Maitre DC, Cowling RM & Enright NJ (1991) Canopy seed storage in woody plants. *The Botanical Review* 57: 277-317.

Lang R (2008) Defining the original extent and floristic composition of the naturally-treeless Grasslands of the Liverpool Plains, North Western Slopes, New South Wales. *Cunninghamia* 10: 407-421.

Langkamp PJ (Ed) (1987) '*Germination of Australian native plant seed*.' Inkata, Melbourne.

Leigh JH & Holgate MD (1979) The responses of the understorey of forests and woodlands of the Southern Tablelands to grazing and burning. *Australian Journal of Ecology* 4: 25-45.

Leigh JH & Noble JC (1981) The role of fire in the management of rangelands in Australia. In: Gill, A.M., Groves, R.H. and Noble, I.R. (Eds.). *Fire in the Australian Biota.* Australian Academy of Science. Canberra, ACT.

Li J, Duggin, JA, Grant CD & Loneragan WA (2003) Germination and early survival of *Eucalyptus blakelyi* in Grasslands of the New England Tablelands, NSW, Australia. *Forest Ecology and Management* 173: 319-334.

Lindenmayer DB, Cunningham RB & Donnelly CF (1997) Decay and collapse in eastern Australian forests: impacts on arboreal marsupials. *Ecological Applications* 7: 625-641.

Lindenmayer, D, MacGregor, C, Wood, JT, Cunningham RB, Crane M, Michael D, Montague-Drake R, Brown D, Fortescue M, Dexter N, Hudson M & Gill AMl (2009), 'What factors influence rapid post-fire site re-occupancy? A case study of the endangered Eastern Bristlebird in eastern Australia', *International Journal of Wildland Fire*, vol. 18, pp. 84-95.

Lodge, G.M. & Whalley, R.D.B. (1989). *Native and natural pastures on the Northern Slopes and Tablelands of New South Wales.* Technical Bulletin No. 35 (NSW Agriculture and Fisheries)

Ludwig JA, Tongway DJ, Freudenberger DO, Noble JC & Hodgkinson KC (eds.) (1997) *Landscape ecology, function and management: principles from Australia's rangelands.* CSIRO, Melbourne.

Lunney D (2001) Causes of the extinction of native mammals of the Western Division of New South Wales: an ecological interpretation of the nineteenth century historical record. *The Rangeland Journal* 23: 44-70.

Lunt ID (1990) Impact of an autumn fire on a long-grazed *Themeda triandra* (Kangaroo Grass) Grassland: implications for management of invaded, remnant vegetations. *Victorian Naturalist* 107: 45-51.

Lunt ID (1994) Variation in flower production of nine grassland species with time since fire, and implications for grassland management and restoration. *Pacific Conservation Biology* 1: 359-366.

Lunt ID (1995) Seed longevity of six native forbs in a closed *Themeda triandra* Grassland. *Australian Journal of Botany* 43, 439-449.

Lunt ID (1996) A transient soil seed bank for the Yam-daisy *Microseris scapigera*. *Victorian Naturalist* 113, 16-19.

Lunt ID (1997a) Effects of long-term vegetation management on remnant grassy forests and anthropogenic native grasslands in south-eastern Australia. *Biological Conservation* 81: 287-297.

Lunt ID (1997b) Germinable soil seed banks of anthropogenic native grasslands and grassy forest remnants in temperate south-eastern Australia. *Plant Ecology* 130: 21-34.

Lunt ID (1998a) Two hundred years of land use and vegetation change in a remnant coastal woodland in southern Australia. *Australian Journal of Botany* 46: 629-647.

Lunt ID (1998b) *Allocasuarina* (Casuarinaceae) invasion of an unburnt coastal woodland at Ocean Grove, Victoria: structural changes, 1971-1996. *Australian Journal of Botany* 46: 649-656.

Lunt I, Barlow T & Ross J (1998) *Plains Wandering: Exploring the Grassy Plains of South-eastern Australia*. Victorian National Parks Association, Victoria.

Lunt ID, Jones N, Spooner P & Petrow M (2006) Effects of European colonization on indigenous ecosystems: post-settlement changes in tree stand structures in *Eucalyptus-Callitris* woodlands in central New South Wales, Australia. *Journal of Biogeography* 33: 1102-1115.

Lunt ID & Morgan JW (1999a) Vegetation changes after 10 years of grazing exclusion and intermittent burning in a *Themeda triandra* (Poaceae) Grassland reserve in south-eastern Australia. *Australian Journal of Botany* 47: 537-552.

Lunt ID & Morgan JW (1999b) Effect of fire frequency on plant composition at the Laverton North Grassland Reserve, Victoria. *Victorian Naturalist* 116: 84-90.

Lunt ID & Morgan JW (2000) Can competition from *Themeda triandra* inhibit invasion by the perennial exotic grass *Nassella neesiana* in native Grasslands? *Plant Protection Quarterly* 15: 92-94.

Lunt ID & Morgan JW (2002) The role of fire regimes in temperate lowland Grasslands of southeastern Australia. Pp. 177-196 in *Flammable Australia: the Fire Regimes and Biodiversity of a Continent* (ed. by R.A. Bradstock, J.E. Williams and A.M. Gill). Cambridge University Press, Cambridge.

Mackey BG (1993) Predicting the potential distribution of rainforest structural characteristics. *Journal of Vegetation Science* 4: 43-54.

Mackowski, C.M. (1984) The ontogeny of hollows in Blackbutt *(Eucalyptus pilularis)* and its relevance to the management of forests for possums, gliders and timber. Pp. 553-567 in *Possums and Gliders* (ed. By A.P. Smith and I.D. Hume). Australian Mammal Society, Sydney.

Maiden, J.H. (1906) The botany of Howell (Bora Creek): a tin granite flora. *Proceedings of the Linnean Society of New South Wales* 31: 63–72.

Maiden H & Betche E (1908) Labiatae. *Prostanthera teretifolia*, n. sp. *Proceedings of the Linnaean Society of New South Wales* 33: 313 & 314.

Major RE, Christie FJ, Gowing G, Cassis G & Reid CAM (2003) The effect of habitat configuration on arboreal insects in fragmented woodlands of south-eastern Australia. *Biological Conservation* 113: 35-48.

Marshall B & Miles J (2005) *Analysis of the Natural and Archaeological Values around Jindabyne, Adaminaby, Berridale and Dalgety*. ngh envionmental, Bega.

Martin WK (2005) Estimates of historical tree densities in the North Lachlan River Catchment, New South Wales, Australia. *Geographical Research* 43: 162-172.

Matthews JL, Mannshardt E & Gremaud P. (2013) Uncertainty quantification for climate observations. *Bull. Amer. Meteor. Soc.*, 94, 21–25.

Maynes GM (1977) Distribution and aspects of the biology of the Parma Wallaby, *Macropus parma*, in New South Wales. *Australian Journal of Wildlife Research* 4: 109-125.

McCarthy M, Gill AM & Lindenmayer DB (1999) Fire regimes in mountain ash forest: evidence from forest age structure, extinction models and Wildlife habitat. *Forest Ecology and Management* 124: 193-203.

McDonald WJF (1996) *Spatial and temporal patterns in the dry season subtropical rainforests of eastern Australia, with particular refernce to the vine thickets of central and southern Queensland*. Phd thesis, The University of New England, Armidale.

McDonald T, Wale K & Bear V (2002) Restoring blue gum high forest: lessons from Sheldon Forest. *Ecological Management and Restoration* 3: 15-27.

McFarland DC (1988) The composition, microhabitat use and response to fire of the avifauna of subtropical Heathlands in Cooloola National Park, Queensland. *Emu* 88: 249-257.

McFarland DC (1990) Flower and seed phenology of some plants in the subtropical Heathlands of Cooloola National Park, Queensland, Australia. *Australian Journal of Botany* 38: 501-509.

McGann, T.D. & Earl, J. (1999) *Floristic descriptions of grassland areas on the Moree Plains.* Report to the NSW Department of Land and Water Conservation and NSW National Parks and Wildlife Service.

McIntyre S, Heard, KM & Martin TG (2003) The relative importance of cattle grazing in subtropical grasslands: does it reduce or enhance plant biodiversity? *Journal of Applied Ecology* 40: 445-457.

Menkhorst, R., Schedvin, N. & Geering, D. (1999) *Regent Honeyeater Recovery Plan* Department of Natural Resources and Environment.

Metcalfe L, Sivertsen DP, Tindall D & Ryan KM (2003) Natural vegetation of the New South Wales wheat-belt (Cobar-Nyngan-Gilgandra, Nymagee-Narromine-Dubbo 1:250 000 vegetation sheets). *Cunninghamia* 8: 253-284.

Mitchell PB (1991) Historical perspectives on some vegetation and soil changes in semi-arid New South Wales. *Vegetatio* 91: 169-182.

Moles AT, Warton DI & Westoby M (2003) Seed size and survival in the soil in arid Australia. *Austral Ecology* 28: 575-585.

Morgan JW (1995a) Ecological studies of the endangered *Rutidosis leptorrhynchoides.* I. Seed production, soil seed bank dynamics, population density and their effects on recruitment. *Australian Journal of Botany* 43: 1-11.

Morgan JW (1995b) Ecological studies of the endangered *Rutidosis leptorrhynchoides.* 11. Patterns of seedling emergence and survival in a native Grassland. *Australian Journal of Botany* 43: 13-24.

Morgan JW (1996) Secondary juvenile period and community recovery following late-spring burning of a Kangaroo Grass *Themeda triandra* Grassland. *Victorian Naturalist* 113: 47-57.

Morgan JW (1997) The effect of Grassland gap size on establishment, growth and flowering of the endangered *Rutidosis leptorrhynchoides* (Asteraceae). *Journal of Applied Ecology* 34: 566-576.

Morgan JW (1998a) Importance of canopy gaps for recruitment of some forbs in *Themeda triandra* dominated Grasslands of south-eastern Australia. *Australian Journal of Botany* 46: 609-627.

Morgan JW (1998b) Composition and seasonal flux of the soil seed bank of species-rich *Themeda triandra* Grasslands in relation to burning history. *Journal of Vegetation Science* 9: 145-156.

Morgan JW (1998c) Comparative germination responses of 28 temperate Grassland species. *Australian Journal of Botany* 46: 209-219.

Morgan JW (1998d) Patterns of invasion of an urban remnant of a species-rich Grassland in southeastern Australia by non-native plant species. *Journal of Vegetation Science* 9: 181-190.

Morgan JW (1999) Defining Grassland fire events and the response of perennial plants to annual fire in temperate Grasslands of south-eastern Australia. *Plant Ecology* 144: 127-144.

Morgan JW (2001) Seedling recruitment patterns over 4 years in an Australian perennial Grassland community with different fire histories. *Journal of Ecology* 89: 908-919.

Morgan JW & Lunt ID (1999) Effects of time-since-fire on the tussock dynamics of a dominant grass (*Themeda triandra*) in a temperate Australian Grassland. *Biological Conservation* 88: 379-386.

Morris EC (2000) Germination response of seven east Australian *Grevillea* species (Proteaceae) to smoke, heat exposure and scarification. *Australian Journal of Botany* 48: 179-189.

Morris EC & Myerscough PJ (1983) Short note – *Banksia ericifolia* transgresses the self-thinning boundary. *Australian Journal of Ecology* 8: 199-201.

Morris EC & Myerscough PJ (1988) Survivorship, growth and self-thinning in *Banksia ericifolia. Australian Journal of Ecology* 13: 181-189.

Morrison DA (2002) Effects of fire intensity on plant species composition of sandstone communities in the Sydney region. *Austral Ecology* 27: 433-441.

Morrison, D. A., Gary, G. J., Pengelly, S. M., Ross, D. G., Mullins, B. J., Thomas, C. R. & Anderson, T. S. (1995), Effects of fire frequency on plant species composition of sandstone communities in the Sydney region: Inter-fire interval and time-since-fire. *Australian Journal of Ecology*, 20: 239–247.

Morrison DA, Buckney RT, Bewick BJ & Cary GJ (1996) Conservation conflicts over burning bush in south-eastern Australia. *Biological Conservation* 76: 167-175.

Morrison DA & Renwick JA (2000) Effects of variation in fire intensity on regeneration of co-occurring species of small trees in the Sydney region. *Australian Journal of Botany* 48: 71-79.

Morton SR, Hoegh-Guldberg O, Lindenmayer DB, Harriss Olson M, Hughes L, McCulloch MT, McIntyre S, Nix HA, Prober SM, Saunders DA, Anderson AN, Burgman MA, Lefroy EC, Lonsdale WM, Lowe I, McMichael AJ, Parslow JS, Steffen W, Williams JE & Woinarski JCZ (2009) The big ecological questions inhibiting effective environmental management in Australia. *Austral Ecology* 34: 1-9.

Mulham WE (1985) Vegetation changes after fire on two land systems in arid north-west New South Wales. *The Rangeland Journal* 7: 80-87.

Myerscough PJ (1998) Ecology of Myrtaceae with special reference to the Sydney region. *Cunninghamia* 5: 787-808.

Myerscough PJ, Clarke PJ & Skelton NJ (1995) Plant co-existence in coastal heaths: floristic patterns and species attributes. *Australian Journal of Ecology* 20: 482-493.

Nadolny C, Ranclaud H, Whalley W, McLeish T, Wheeler J, Morsley R, Croft P, McCormick L, Ede A, Hawes W, Watson C & Austin J (2003) *Grassy Vegetation in North-Western NSW and Guidelines for its Management for Conservation*. Armidale Tree Group, Armidale, NSW.

Nadolny, C., Hunter, J.T. & Hawes, W. (2009) *Native grassy vegetation in the Border Rivers-Gwydir Catchment: diversity, distribution, use and management.* Report to the Border Rivers – Gwydir Catchment Management Authority.

NPWS (National Parks and WildlifeService) (2002) *Threatened Species of the Upper North Coast of NSW*. NSW National Parks and Wildlife Service, Coffs Harbour, NSW.

NPWS (National Parks and Wildlife Service) (2003) *Targeted flora survey and mapping. NSW western regional assessments. Brigalow Belt South Bioregion (Stage 2)*. A project undertaken for the Resource and Conservation Assessment Council NSW Western Regional Assessments. Project numberWRA/ 16.

Nicholson PH (1981) Fire and the Australian Aboriginal – an enigma. In '*Fire and the Australian Biota'* (Eds. A.M. Gill, R.H. Groves and I.R. Noble). Australian Academy of Science, Canberra.

Nicolle D (2006) A classification and census of regenerative strategies in the eucalypts (*Angophora*, *Corymbia* and *Eucalyptus* – Myrtaceae), with special reference to obligate seeders. *Australian Journal of Botany* 54: 391-407.

Nieuwenhuis A (1987) The effect of fire frequency on the sclerophyll vegetation of the West Head, New South Wales. *Australian Journal of Ecology* 12: 377-385.

Noble IR & Slatyer RO (1980) The use of vital attributes to predict successional changes in plant communities subject to recurrent disturbances. *Vegetation* 43: 5-21.

Noble JC (1982) The significance of fire in the biology and evolutionary ecology of mallee *Eucalyptus* populations. Pp. 153-159 in *Evolution of the Flora and Fauna of Arid Australia* (ed. By W.R. Barker and P.J.M. Greenslade). Peacock Publications, Adelaide.

Noble JC (1984) Mallee. Pp. 223-240 in *Management of Australia's Rangelands* (ed. by G.N. Harrington, A.D. Wilson and M.D. Young). CSIRO, Canberra.

Noble, J. C. 1989. Fire studies in mallee (*Eucalyptus* spp.) communities of western New South Wales: the effects of fires applied in different seasons on herbage productivity and their implications for management. Aust. J. Ecol. **14**:169-187.

Noble, J. C. 2001. Regulating Callitris populations: tale of two pineries. Pp. 173–183 in J. Dargavel, ed. *Perfumed pineries: environmental history of Australia's Callitris forests*. Australian National University, Canberra.

Noble JC & Grice AC (2002) Fire regimes in semi-arid and tropical pastoral lands: managing biological diversity and ecosystem function. Pp. 373-400 in *Flammable Australia: the Fire Regimes and Biodiversity of a Continent* (ed. by R.A. Bradstock, J.E. Williams and A.M. Gill). Cambridge University Press, Cambridge.

Noble JC, Harrington GN & Hodgkinson KC (1986) The ecological significance of irregular fire in Australia's rangelands. Pp. 577-580 in *Rangelands: a resource under siege*. (ed. By Joss PJ, Lynch PW & Williams OB). Australian Academy of Science, Cannberra.

Norris EH, Mitchell PB & Hart DM (1991) Vegetation changes in the Pilliga forests: a preliminary evaluation of the evidence. *Vegetatio* 91: 209-218.

NSW NPWS. (2003). *Kirramingly Nature Reserve Plan of Management.* NPWS, Narrabri.

NSW Office of Water (undated). *Border Rivers Catchment.* www.water.nsw.gov.au/Water-management/Basins-and-catchments/Border-Rivers-catchment/Border-Rivers-catchment   Accessed April 2014.

NSW Scientific Committee (2004) *Fuzzy Box Woodland on alluvial soils of the South Western Slopes, Darling Riverine Plains and Brigalow Belt South Bioregions - endangered ecological communit*y. Final Determination.

O'Connor TG (1991) Local extinction in perennial Grasslands: a life-history approach. *American Naturalist* 137: 753-773.

O'Connor TG (1994) Composition and population responses of an African savanna Grassland to rainfall and grazing. *Journal of Applied Ecology* 31: 155-171.

Odgers BM (1999) Seasonal variation in buried germinable seed banks of grass species in an urban eucalypt forest reserve. *Australian Journal of Botany* 47: 623-638.

Office of Environment and Heritage. (2011) *Bluff River Nature Reserve and Bolivia Hill Nature Neserve Plan of Management*. OEH, Sydney.

Ojima DS, Schimel DS, Parton WJ & Owensby CE (1994) Long- and short-term effects of fire on nitrogen cycling in tallgrass prairie. *Biogeochemistry* 24: 67-84.

Ooi MJK, Auld TD & Whelan RJ (2006a) Dormancy and the fire-centric focus: germination of three *Leucopogon* species (Ericaceae) from south-eastern Australia*. Annals of Botany* 98: 421-30*.*

Ooi MJK, Whelan RJ & Auld TD (2006b) Persistence of obligate-seeding species at the population scale: effects of fire intensity, fire patchiness and long fire-free intervals. *International Journal of Wildland Fire* 15: 261-269.

Orr DM, Paton CJ & Playford C (2004) Dynamics of plant populations in *Heteropogon contortus* (black speargrass) pastureson a granite landscape in southern Queensland 3. Dynamics of *Aristida* spp. populations. *Tropical Grasslands* 38: 65-76.

Oxley, J. (1820) *Journals of two expeditions into the interior of New South Wales 1817–18* (J. Murray: London).

Panton WJ (1993) Changes in post World War II distribution and status of monsoon rainforest in the Darwin area. *Australian Geographer* 24: 50-9.

Parker D & Lunt ID (2000) Stand structure and recruitment patterns in *Callitris – Eucalyptus* woodlands in Terrick Terrick National Park, Victoria. *Victorian Naturalist* 117: 207-213.

Parsons, W.T. (1973) *Noxious weeds of Victoria* (Inkata Press: Melbourne).

Partridge E (2005) *'Disequilibrium ecology' - much ado about nothing*. On line at http://gadfly.igc.org/ecology/APA-ISEE.htm. Accessed 8 July 2005.

Paull D (2001) Stump count analysis of the pre-European Pilliga forests. Pp. 63-70 in *Perfumed Pineries: Environmental History of Australia's Callitris Forests* (ed. by J. Dargavel, D. Hart and B. Libbis). Centre for Resource and Environmental Studies, Australian National University, Canberra.

Pausas JG, Bradstock RA, Keith DA, Keeley JE & the GCTE (2004) Plant functional traits in relation to crown-fire ecosystems. *Ecology* 85: 1085-1100.

Peake TC (2006) The vegetation of the Central Hunter Valley, New South Wales. A report on the findings of the Hunter Remnant Vegetation Project. Hunter-Central Rivers Catchment Management Authority, Paterson.

Penman TD, Binns DL, Shiels RJ Allen RM & Kavanagh RP (2008) Changes in understorey plant species richness following logging and prescribed burning in shrubby Dry sclerophyll forests of south-eastern Australia. *Austral Ecology* 33: 197-210.

Penman, T. D., & A. York. 2010. Climate and recent fire history affect fuel loads in Eucalyptus forests: Implications for fire management in a changing climate. *Forest Ecology and Management* 260:1791-1797.

Pickett STA, Parker VT & Fiedler PL (1992) The new paradigm in ecology: implications for conservation biology above the species level. Pp. 65-88 in *Conservation Biology: the Theory and Practice of Nature Conservation Preservation and Management* (ed. by P.L. Fiedler and S.K. Jain). Chapman and Hall, New York.

Pressland, AJ (1982) Fire in the management of grazing lands in Queensland. *Tropical Grasslands* 16(3), 104-12.

Priday, S. (2006) *The native vegetation of the New South Wales South Western Slopes Bioregion (Lachlan, Murrumbidgee and Murray Catchments)*. Unpublished report to DEC Southern Office Queanbeyan.

Porteners, M.F. (1997) *Vegetation Survey of Sub-alpine Communities in Mt Kaputar National Park*. Unpublished Report to NSW NPWS, Narrabri.

Prober SM (1996) Conservation of the grassy White Box woodlands: rangewide floristic variation and implications for reserve design. *Australian Journal of Botany* 44: 57-77.

Prober SM & Brown AHD (1994) Conservation of the grassy white box woodlands: population genetics and fragmentation of *Eucalyptus albens*. *Conservation Biology* 8: 1003-1013.

Prober SM, Lunt ID & Thiele KR (2002a) Determining reference conditions for management and restoration of temperate Grassy woodlands: relationships among trees, topsoils and understorey flora in little-grazed remnants. *Australian Journal of Botany* 50: 687-697.

Prober SM, Thiele KR & Lunt ID (2002b) Identifying ecological barriers to restoration in temperate Grassy woodlands: soil changes associated with different degradation states. *Australian Journal of Botany* 50: 699-712.

Prober SM & Thiele KR (1993) The ecology and genetics of remnant grassy white box woodlands in relation to their conservation. *Victorian Naturalist* 110: 30-36.

Prober SM & Thiele KR (1995) Conservation of the grassy White Box woodlands: relative contributions of size and disturbance to floristic composition and diversity of remnants. *Australian Journal of Botany* 43: 349-366.

Prober SM & Thiele KR (2004) Floristic patterns along an east-west gradient in grassy box woodlands of Central New South Wales. *Cunninghamia* 8: 306-325.

Prober SM & Thiele KR (2005) Restoring Australia's temperate Grasslands and Grassy woodlands: integrating function and diversity. *Ecosystem Management and Restoration* 6: 16-27.

Prober SM & Thiele KR & Koen TB (2004) Spring burns control exotic annual grasses in a temperate Grassy woodland. *Ecological Management and Restoration* 5: 131-134.

Prober SM & Thiele KR & Lunt ID (2007) Fire frequency regulates tussock grass composition, structure and resilience in endangered temperate woodlands. *Austral Ecology* 32: 808-824.

Prober SM & Thiele KR & Lunt ID & Koen TB (2005) Restoring ecological function in temperate Grassy woodlands: manipulating soil nutrients, exotic annuals and native perennial grasses through carbon supplements and spring burns. *Journal of Applied Ecology* 42: 1073-1085.

Pulsford IF (1983) *Planchonella Nature Reserve proposal*. Unpub. Report. (NSW NPWS: Hurstville).

Pulsford IF, Banks JCG & Hodges L (1993) Land use history of the white cypress pine forests in the Snowy Valley, Kosciusko National Park. Pp. 85-104 in *Australia's Ever-Changing Forests II: Proceedings of the Second National Conference on Australian Forest History* (ed. by J. Dargavel and S. Feary). Centre for Resource and Environmental Studies, Australian National University, Canberra.

Purdie RW (1977a) Early stages of regeneration after burning in dry sclerophyll vegetation. I. Regeneration of the understorey by vegetative means. *Australian Journal of Botany* 25: 21-34.

Purdie RW (1977b) Early stages of regeneration after burning in dry sclerophyll vegetation. II. Regeneration by seed germination. *Australian Journal of Botany* 25: 35-46.

Purdie RW & Slatyer RO (1976) Vegetation succession after fire in sclerophyll woodland communities in south-eastern Australia. *Australian Journal of Ecology* 1: 223-236.

Read TR, Bellairs SM, Mulligan DR & Lamb D (2000) Smoke and heat effects on soil seed bank germination for the re-establishment of a native forest community in New South Wales. *Austral Ecology* 25: 48-57.

Recher H (1981) Bird communities of heath and their management and conservation requirements. In '*Heaths in New South Wales*' (ed. C. Haigh) pp.27-40. New South Wales National Parks and Wildlife Service, Sydney.

Reid JRW (1999) Threatened and declining birds in the New South Wales sheep-wheat belt: i. diagnosis, characteristics and management. Report prepared for NSW NPWS. CSIRO, Canberra.

Reid JRW (2000) Threatened and declining birds in the New South Wales Wheat Belt: landscape relationships – modelling bird atlas data against vegetation cover. Commonwealth Scientific, Industrial and Research Organisation, Canberrra, Australia.

Resource and Conservation Assessment Council of NSW (RACAC) (2004) *Joint vegetation mapping project, Brigalow Belt South Western Regional Assessment Stage 2* Resource and Conservation Division, Department of Infastructure, Planning and Natural Resources.

Richards, P.G. (1996) *Significant Plants of the Glen Innes Forest Management Area*. Unpublished Report to State Forests of New South Wales, Northern Region Planning Unit.

Richards, P.G. & Hunter, J.T. (1997) Range extensions for species of vascular plants in the Northern Tablelands of New South Wales. *Cunninghamia* 5: 275–280.

Roberts, G.W. (1983) *A Vegetation Survey of Granitic areas on part of the Northern Tablelands and upper North Western Slopes, New South Wales*. MSc. Prelim. Thesis (University of New England: Armidale).

Roberts, G. (1992) *Vegetation systems of north-east New South Wales mapped from LANDSAT TM imagery.* Unpublished report (New South Wales National Parks and Wildlife Service: Hurstville).

Robertson D (1985) *Interrelationships between Kangaroos, Fire and Vegetation Dynamics at Gellibrand Hill Park, Victoria*. PhD thesis, University of Melbourne.

Roche S, Dixon KW & Pate JS (1998) For everything a season: smoke-induced seed germination and seedling recruitment in a Western Australian *Banksia* woodland. *Australian Journal of Ecology* 23: 111-120.

Rolls E (1981) *A Million Wild Acres*. Penguin Books, Ringwood, Victoria.

Roques KG, O'Connor TG & Watkinson AR (2001) Dynamics of shrub encroachment in an African savanna: relative influences of fire, herbivory, rainfall and density dependence. *Journal of Applied Ecology* 38: 268-280.

Ross, K. A., Bedward, M., Ellis, M. V., Deane, A., Simpson, C. C. & Bradstock, R. A. (2008). Modelling the dynamics of white cypress pine Callitris glaucophylla woodlands in inland south-eastern Australia. *Ecological Modelling*, 211(1), 11-24

Rothery FM (1970) *Atlas of Bundaleer Plains and Tatala*. Australian National University Press, Canberra

Russell-Smith J, Bowman DMLS (1992) Conservation of monsoon rainforest isolates in the Northern Territory, Australia. *Biological Conservation* 59: 51-63.

Russell-Smith J & Stanton P (2002) Fire regimes and fire management of rainforest communities across northern Australia. Pp. 329-350 in *Flammable Australia: the Fire Regimes and Biodiversity of a Continent* (ed. by R.A. Bradstock, J.E. Williams and A.M. Gill). Cambridge University Press, Cambridge.

Scanlan JC (1980) Efects of spring wildfires on *Astrebla* (Mitchell grass) Grasslands in north-west Queensland under varying levels of growing season rainfall. *The Rangeland Journal* 2: 162-168.

Scanlan JC (2002) Some aspects of tree-grass dynamics in Queensland's grazing lands. *The Rangeland Journal* 24: 56-82.

Scanlan JC & Burrows WH (1990) Woody overstorey impact on herbaceous understorey in *Eucalyptus* spp. communities in central Queensland. *Australian Journal of Ecology* 15: 191-197.

Seifriz W (1953) The oecology of thicket formation. *Plant Ecology* 4: 155-164.

Semple WS (1990) Sifton bush increase: a land degradation problem on the temperate rangelands of NSW. *Australian Journal of Soil and Water Conservation* 3: 31-36.

Semple WS (1997) Native and naturalised shrubs of the Bathurst granites: past and present. *Cunninghamia* 5: 803-833.

Semple, WS & Koen, TB (1993) Some Effects of Fire on the Survival of Sifton Bush (*Cassinia arcuata* R.Br.). *The Rangeland Journal* 15(2): 320-330

Semple WS & Koen TB (2001) Growth rate and effect of sheep browsing on young eucalypts in an anthropogenic *Themeda* Grassland. *The Rangeland Journal* 23: 182-193.

Semple WS & Koen TB (2003) Effect of pasture type on regeneration of eucalypts in the woodland zone of south-eastern Australia. *Cunninghamia* 8: 76-84.

Setterfield SA, Douglas MM, Hutley LB & Welch MA (2005) Effects of canopy cover and ground disturbance on establishment of an invasive grass in an Australia savanna. *Biotropica* 37: 25-31.

Shaw NH (1957) Bunch spear grass dominance in burnt pastures in south eastern Queensland. *Australian Journal of Agricultural Research* 8: 325-234.

Shea SR, McCormick J & Portlock CC (1979) The effect of fires on regeneration of leguminous species in the northern Jarrah (*Eucalyptus marginata* Sm) forest of Western Australia. *Australian Journal of Ecology* 4: 195-205.

Shelly D (1998a) Preliminary fauna survey of belah and cypress/box woodland near West Wyalong NSW and recommendations for forest management. *Australian Forestry* 61: 147-154.

Shelly D (1998b) Survey of vertebrate fauna and habitats in a cypress pine-ironbark forest in central-west New South Wales. *Australian Zoologist* 30: 426-436.

Silcock RG & Smith FT (1990) Viable seed retention under field conditions by western Queensland Australian pasture species. *Tropical Grasslands* 24: 65-74.

Siddiqi MY, Carolin RC & Myerscough PJ (1976) Studies in the ecology of coastal heath in New South Wales. III. Regrowth of vegetation after fire. *Proceedings of the Linnean Society of New South Wales*101: 53-63.

Sivertsen, D. & Metcalfe, L. (1995) Natural vegetation of the southern wheat-belt (Forbes and Cargelligo 1:250 000 map sheets). *Cunninghamia* 4(1) 103-128

Skelly DK, Joseph LN, Possingham HP, LK Friedenburg, Farrugia TJ, Kinnison MT & Hendry AP (2007) Evolutionary responses to climate change. *Conservation Biology* 21: 1353-1355.

Smith A (1992) Sugar gliders, wattles and rural eucalypt dieback. *Australian Network for Plant Conservation Newsletter* 1: 7-10.

Somerville M (2009) *Hunter, Central & Lower North Coast vegetation classification & mapping project Volume 1: Vegetation classification technical report*. Report prepared by HCCREMS, Hunter Councils Environment Division for the Hunter Central Rivers Catchment Management Authority, Tocal, NSW

Sousa WP (1984) The role of disturbance in natural communities. *Annual Review of Ecology and Systematics* 15: 353-391.

Southwell CJ & Jarman PJ (1987) Macropod studies at Wallaby Creek 111. The effect of fire on pasture utilisation by macropodids and cattle. *Australian Wildlife Research* 14: 117-124.

Specht RL & Specht A (1989) Species richness of sclerophyll (heathy) plant communities in Australia - the influence of overstorey cover. *Australian Journal of Botany* 37: 337-350.

Spencer R & Baxter GS (2006) Effects of fire on the structure and composition of open eucalypt forests. *Austral Ecology* 31: 638-646.

Stewart B (1999) *Effects of Long-term Repeated Prescribed Burning on Soil-stored Seed Banks in a Coastal Blackbutt Forest*. Honours thesis, University of Wollongong.

Stone, C. (2005) 'Bell-miner associated dieback at the tree crown scale: a multi-trophic process', *Australian Forestry,* 68: 237-241.

Stuwe J (1994) The role of fire in ground flora ecology. *Victorian Naturalist* 111: 93-5.

Stuwe J & Parsons RF (1977) *Themeda australis* Grasslands on the Basalt Plains, Victoria: floristics and management effects. *Australian Journal of Ecology* 2, 467-476.

Tasker E (2002) The ecological impacts of cattle grazing and associated grazier burning in the eucalypt forests of northern NSW, PhD thesis, University of Sydney.

Tasker EM & Bradstock RA (2006) Influence of cattle grazing practices on forest understorey structure in north-eastern New South Wales. *Austral Ecology* 31: 490-502.

Tasker EM & Dickman CR (2004) Small mammal community composition in relation to cattle grazing and associated burning in eucalypt forests of the Northern Tablelands of New South Wales. Pp. 721-740 in *The Conservation of Australia's Forest Fauna (2nd edition)* (ed. by D. Lunney). Royal Zoological Society of New South Wales, Mosman, NSW.

Tasker EM, Whelan RJ & Baker J (2008) *The impacts of fire on Australian fauna: a synthesis of approaches in research published from 1995-2007.* Ecological Society of Australia

Thomas PB, Morris EC, Auld TD (2003) Interactive effects of heat shock and smoke on germination of nine species forming soil seed banks within the Sydney region. *Austral Ecology* 28: 674-83.

Thompson WA & Eldridge DJ (2005a) Plant cover and composition in relation to density of *Callitris glaucophylla* (white cypress pine) along a rainfall gradient in eastern Australia. *Australian Journal of Botany* 53: 545-554.

Thompson WA & Eldridge DJ (2005b) White cypress pine (*Callitris glaucophylla*): a review of its roles in landscape and ecological processes in eastern Australia. *Australian Journal of Botany* 53: 555-570.

Tierney DA (2003) Hybridization and its management implications for the rare plant *Kunzea rupestris*. Blakley. PhD thesis, The University of Sydney.

Tierney DA (2004) Towards an understanding of population change for the long-lived resprouting tree *Angophora inopina. Australian Journal of Botany* 52: 31-38.

Tierney DA (2006) The effect of fire related germination cues on the germination of a declining forest understorey species. *Australian Journal of Botany* 54: 297-303.

Tierney DA (2009) Summary of the vegetation map revision for Wyong LGA 2008. Unpublished report, Wyong Shire Council, Wyong, New South Wales.

Tierney DA (unpublished data) What drives the population viability of *Prostanthera askania*? Unpublished report. The University of Newcastle, Ourimbah, New South Wales..

Tierney DA & Wardle GM (2005) Differential seed ecology in the shrubs *Kunzea rupestris*, *Kunzea capitata* and associated hybrids (Myrtaceae): the function of thin-walled fruits in a fire-prone vegetation. *Australian Journal of Botany* 53: 313-321.

Tothill JC (1969) Soil temperatures and seed burial in relation to the performance of *Heteropogon contortus* and *Themeda australis* in burnt native woodland pastures in eastern Queensland. *Australian Journal of Botany* 17: 269-275.

Townley SJ (2000) *The Ecology of the Hastings River Mouse Pseudomys oralis (Rodentia: Muridae) in Northeastern New South Wales and Southeastern Queensland*. PhD thesis, Southern Cross University.

Tozer MG & Bradstock RA (2002) Fire-mediated effects of overstorey on plant species diversity and abundance in an eastern Australian heath. *Plant Ecology* 164: 213-223.

Tremont RM & McIntyre S (1994) Natural grassy vegetation and native forbs in temperate Australia: structure, dynamics and life histories. *Australian Journal of Botany* 42: 641-658.

Trollope SW & Trollope LA (2004) *Prescribed burning in African Grasslands and savannas for wildlife management*. Aridlands Newsletter No 55. Available at http://ag.arizona.edu/OALS/ALN/aln55/trollope.html. Accessed 9 March 2006.

Turner D, Ostendorf B & Lewis M (2008) An introduction to patterns of fire in arid and semi-arid Australia, 1998-2004*. The Rangeland Journal* 30: 95-107.

Unwin GL (1989) Structure and composition of the abrupt rainforest boundary in the Herberton Highland, northQueensland. *Australian Journal of Botany* 37: 413-428.

Uys RG, Bond WJ & Everson TM. (2004) The effect of different fire regimes on plant diversity in southern African Grasslands. *Biological Conservation* 118: 489-499.

Van der Pijl L (1982) Priciples of dispersal in higher plants. 3rd edn. Springer-Verlag. Berlin.

Vaughton G (1998) Soil seed bank dynamics in the rare obligate seeding shrub, *Grevillea barklyana* (Proteaceae). *Australian Journal of Ecology* 23: 375-384.

Verrier FJ & Kirkpatrick JB (2005) Frequent mowing is better than grazing for the conservation of lowland tussock Grassland at Pontville, Tasmania. *Austral Ecology* 30: 74-78.

Vlahos S & Bell DT (1986) Soil seed-bank components of the northern jarrah forest of Western Australia. *Australian Journal of Ecology* 11: 171-179.

Vlok JHJ & Yeaton RI (2000) Competitive interactions between overstorey proteas and sprouting understorey species in South African mountain fynbos. *Diversity and Distributions* 6: 273-281.

von Richter L, Little D & Benson D (2005) Effects of low intensity fire on the resprouting of the weed African Olive (*Olea europaea* subsp. *cuspidata*) in Cumberland Plain Woodland, western Sydney. *Ecological Management and Restoration* 6: 230-233.

Wahren C-H A, Papst WA & Williams RJ (1999) *Post-fire regeneration in Victorian alpine and subalpine vegetation*. Proceedings of the Australian bushfire conference, 1999, Albury.

Walker PJ (1976) Growth and regeneration of trees and shrubs. Pp. 8-1 to 8-26 in *Rehabilitation of Arid Lands: 10 Years of Research at Cobar, NSW, 1964-1974* (ed. by G.M. Cunningham, P.J. Walker and D.R. Green). Soil Conservation Service of NSW, Sydney.

Walker J, Condon RW, Hodgkinson KC & Harrington GN (1981) Fire in pastoral areas of poplar box (*Eucalyptus populnea*) lands. *Australian Rangeland Journal* 3: 12-23.

Walker LR, Chapin III FS. (1986a) Physiological controls over seedling growth in primary succession on an Alaskan floodplain*. Ecology* ;67:1508-1523.

Walker LR, Zasada J, Chapin III FS. (1986b) The role of life history processes in primary succession on an Alaskan floodplain*. Ecology* ;67:1243-1253.

Wardell-Johnson G & Lynch JJ (2005) *A Preliminary Review of Ecualypt Dieback Associated with Bell Miner Habitat in North-eastern New South Wales, Australia*. School of Natural and Rural Systems Management, University of Queensland, Gatton, Queensland.

Wark MC, White MD, Robertson DJ & Marriott PF (1987) Regeneration of heath and heath woodland in the north-eastern Otway Ranges following the wildfire of February 1983. *Proceedings of the Royal Society of Victoria* 99: 51-88.

Watson P (2001) *The Role and Use of Fire for Biodiversity Conservation in Southeast Queensland: Fire Management Guidelines Derived from Ecological Research*. SEQ Fire and Biodiversity Consortium, Brisbane.

Watson PJ (2005) *Fire Frequencies for Western Sydney's Woodlands: Indications from Vegetation Dynamics*. PhD thesis, University of Western Sydney.

Watson P (2006a) *Fire Frequency Guidelines and the Vegetation of the Northern Rivers Region. Draft 2.* Hotspots Fire Project, Nature Conservation Council of NSW, Sydney.

Watson P (2006b) *Fire and the Vegetation of the Southern Rivers Region. Draft for Comment.* Hotspots Fire Project, Nature Conservation Council of NSW, Sydney.

Watson P (2007) *Fire and the Vegetation of the Central West Region.* Hotspots Fire Project, Nature Conservation Council of NSW, Sydney.

Watson P., and Tierney, D. (2009a) *Fire and the Vegetation of the Hunter-Central Rivers Region.* Hotspots Fire Project, Nature Conservation Council of NSW, Sydney.

Watson P., and Tierney, D. (2009b) *Fire and the Vegetation of the Namoi Region.* Hotspots Fire Project, Nature Conservation Council of NSW, Sydney.

Watson PJ, Bradstock RA & Morris EC (2009). Fire frequency influences composition and structure of the shrub layer in an Australian subcoastal temperate grassy woodland. *Austral Ecology* **34**, 218–232.

Watson, P., and G. Wardell-Johnson. (2004). "Fire frequency and time-since-fire effects on the open-forest and woodland flora of Girraween National Park, south-east Queensland, Australia." *Austral Ecology* 29 (2): 225-236.

Wellington AB & Noble IR (1985) Post-fire recruitment and mortality in a population of the mallee *Eucalyptus incrassata* in semi-arid, south-eastern Australia. *Journal of Ecology* 73: 645-656.

Westoby M, Walker B & Noy-Meir I (1989) Opportunistic management for rangelands not at equilibrium. *Journal of Range Management* 42: 266-274.

Westoby M & Burgman M (2006) Climate change as a threatening process. *Austral Ecology* 31: 549-550.

Whelan RJ (1995) '*The ecology of fire.*' (Cambridge University Press, Cambridge)

Whelan RJ, De Jong NH & von der Burg S (1998) Variation in bradyspory and seedling recruitment without fire among populations of *Banksia serrata* (Proteaceae). *Australian Journal of Ecology* 23:121-128.

White ME. (1994) *After the greening: the browning of Australia*. Kangaroo Press, Kenthurst.

Williams JE & Gill AM (1995) *The Impact of Fire Regimes on Native Forests in Eastern New South Wales*. NSW National Parks and Wildlife Service, Sydney.

Williams JE, Whelan RJ & Gill AM (1994) Fire and environmental heterogeneity in southern temperate forest ecosystems: implications for management. *Australian Journal of Botany* 42: 125-137.

Williams P. R., Congdon R.A., Grice A. C. & Clarke P. J. (2003) Effect of fire regime on plant abundance in a tropical eucalypt savanna of north-eastern Australia. *Aust. Ecol. 28, 327–38*

Williams PR & Clarke PJ (2006) Fire history and soil gradients generate floristic patterns in montane sedgelands and wet heaths of Gibraltar Range National Park. *Proceedings of the Linnean Society of New South Wales*. 127: 27-38.

Williams, J.B. (1969) *Plants recorded from Howell, south of Inverell, on granite hills*. Unpublished checklist. (University of New England: Armidale).

Williams, J.B. (1972) *The plants of Bald Rock State Park: A preliminary checklist*. Unpublished checklist. (University of New England: Armidale).

Williams, J.B. (1991b) *List of plants seen at Strathbogie Aboriginal Centre*. Unpublished checklist. (University of New England: Armidale).

Williams, J.B. (1992*) Notes on the vegetation and flora of Mount Yarrowyck west of Armidale.* Unpublished checklist. (University of New England: Armidale).

Williams, J.B. (1994) *Bolivia.* Unpublished checklist. (University of New England: Armidale)

Williams, J.B. (1998*). The Vegetation and Flora of Planchonella Hill, near Yallaroi, north- west slopes of NSW*. Unpublished report, Botany Department, University of New England, Armidale

Williams, J.B. & Wissmann, H.J. (1991) Torrington Region, New South Wales: Checklist of Vascular Plants occurring on the Mole Granite & Associated Rocks. Appendix H in G. Stewart, (ed.), *Torrington Region Draft Land Assessment* (N.S.W. Dept. of Lands: Armidale)

Williams RJ, Cook GD, Gill AM & Moore PHR (1999) Fire regime, fire intensity and tree survival in a tropical savanna in northern Australia. *Australian Journal of Ecology* 24: 50-59.

Willis AJ & Groves RH (1991) Temperature and light effects on the germination of seven native forbs. *Australian Journal of Botany* 39: 219-228.

Windsor DM (1999) A review of factors affecting regeneration of box woodlands in the Central Tablelands of New South Wales. Pp. 271-285 in *Temperate Eucalypt Woodlands in Australia: Biology, Conservation, Management and Restoration* (ed. by R.J. Hobbs and C.J. Yates). Surrey Beatty and Sons, Chipping Norton, NSW.

Withers J & Ashton DH (1977) Studies on the status of unburnt *Eucalyptus* woodland at Ocean Grove, Victoria. I. The structure and regeneration. *Australian Journal of Botany* 25: 623-637.

Woinarski, J. C. Z. 1999. Fire and Australian birds: a review. Pages 57-112 in A. M. Gill, J. C. Z. Woinarski, and A. York, editors. *Australia's biodiversity - responses to Fire: plants, birds and invertebrates*. Biodiversity Technical Paper No.1. Woinarski and A. York). Department of the Environment and Heritage, Canberra. Environment Australia, Canberra. http://www.deh.gov.au/biodiversity/publications/technical/fire

Wong N, O'Brien E, Morgan J & Price J (2006) Conservation management options for native Grasslands of northern Victoria. *Australasian Plant Conservation* 14: 12-13.

Wood PD (2001) *What are the Germination Signals of the Cumberland Plain Woodland Soil Seed Bank?* Honours thesis, University of Western Sydney.

Wu J & Loucks OL (1995) From balance of nature to hierarchical patch dynamics: a paradigm shift in ecology. *Quarterly Review of Biology* 70: 439-466.

Yates CJ & Hobbs RJ (2000) Temperate eucalypt woodlands in Australia - an overview. Pp. 1-5 in *Temperate Eucalypt Woodlands in Australia: Biology, Conservation, Management and Restoration* (ed. by R.J. Hobbs and C.J. Yates). Surrey Beatty and Sons, Chipping Norton, NSW.

Yates CJ, Hopper SD, Brown A & van Leeuwen S (2003) Impact of two wildfires on endemic granite outcrop vegetation in Western Australia. *Journal of Vegetation Science* 14: 185-194.

Yates CJ, Ladd PC, Coates DJ & McArthur S (2007) Hierarchies of cause: understanding rarity in an endemic shrub *Verticordia staminosa* (Myrtaceae) with a highly restricted distribution. *Australian Journal of Botany* 55: 194-205.

Yibarbuk D, Whitehead PJ, Russell-Smith J, Jackson D, Godjuwa C, Fisher A, Cooke P, Choquenot D & Bowman DMJS (2001) Fire ecology and Aboriginal land management in central Arnhem Land, northern Australia: a tradition of ecosystem management. *Journal of Biogeography* 28: 325-343.

York A (1999) Long-term effects of repeated prescribed burning on forest invertebrates: management implications for the conservation of biodiversity. Pp. 181-266 in *Australia's Biodiversity - Responses to Fire. Plants, Birds and Invertebrates* (ed. by A.M. Gill, J.C.Z. Woinarski and A. York). Department of the Environment and Heritage, Canberra.

York A (2000) Long-term effects of frequent low-intensity burning on ant communities in coastal blackbutt forests of southeastern Australia. *Austral Ecology* 25: 83-98.

Zammit C & Westoby M (1987) Population structure and reproductive status of two *Banksia* shrubs at various times after fire. *Vegetatio* 70: 11-20.