Fire Severity and Plant Community Dynamics in the Australian Alps, Southeastern New South Wales

by

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Candidate's Declaration

This thesis contains no material which has been accepted for the award of any other degree or diploma in any university. To the best of the author's knowledge, it contains no material previously published or written by another person, except where due reference is made in the text.

Michael Damien Doherty MDoherty. Date: 4TH MARCH 2021

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Abstract

The effects of fire on plant communities are a subset of a broader range of spatial and temporal influences on plant community dynamics. Widespread unplanned fires in the Australian Alps in 2003 provided a unique opportunity to investigate the response of vascular plant species to high severity fire and to test whether there was a differential response to variations in fire severity that may affect long-term vegetation dynamics within and between subalpine, montane and tablelands plant communities. A series of pre-existing sites established in the late 1990s in the northern parts of the Australian Alps in Brindabella National Park, Bimberi Nature Reserve and Burrinjuck Nature Reserve were resurveyed and monitored from 2003-2012. Two research questions were posed: 1) what is the response to high severity fire of vascular plant species found in subalpine, montane and tablelands plant communities? 2) does high severity fire result in significant or permanent change to vascular plant species richness and plant community composition as compared to low severity fire in subalpine, montane and tablelands plant communities and does the recovery rate vary with severity?

The fire responses of over 400 vascular plant species in 21 plant communities were documented, with 24% of species being fire-killed and responding via canopy and soil seed banks and 76% via resprouting. There was no loss of plant species from the study areas and all vascular plant species recovered rapidly post-fire, but some fire-killed species exhibited slow recovery rates. Although vegetation structure is affected by fire severity, it is fire severity interacting with fire frequency that has the potential to change vegetation structure in the long term, particularly via the loss of structurally dominant obligate seeder species.

The overall trend in vascular plant species richness was an increase in species richness post-fire irrespective of fire severity, with a time since fire decline. Significantly for forest ecosystems, this peak and decline response of vascular plant species richness post-fire is modified and varies over time in relation to rainfall, decreasing in drier years. This fluctuation in vascular plant species richness is driven by both native and introduced species. Habitat complexity, vegetation type, understorey cover and season of sampling are also major influences on overall vascular plant species richness with variation over space due to environmental gradients and variation over time both within and between vegetation types in relation to time since fire and rainfall. The results have implications not only for pre- and post-fire sampling but also for vegetation sampling in general, as any initial sample in a sequence of samples over time will not reflect the total vascular plant species richness of a site. The full expression of species richness at a point can only be measured as the result of resampling over multiple years, where variations in climatic and disturbance variables cause a wider array of species to become apparent on site.

While there was an initial difference in floristic compositional dissimilarity between low and high severity sites in the Brindabella sites and greater dissimilarity in the short to medium term, this dissimilarity diminished over time and there was no evidence of permanent compositional change in high severity sites. In high severity sites, there was an increase in the density of many dominant shrub species leading to differences in the short-term post-fire. As shrub layers started to senesce, both the changes in composition and the changes in abundance reduced and by approximately 10 years post-fire any compositional difference due to severity was not detectable. The initial significant response immediately post-fire was partially due to differences in species composition pre- and post-fire but was also due to changes in the abundances of species pre- and post-fire. While there were some initial differences post-fire between low and high severity sites, these differences diminished over time. The Burrinjuck study area did not exhibit these differences in relation to fire severity but did exhibit differences in richness and composition based on season of sampling. Because of the interaction of time since fire and rainfall, species richness and community composition are not constant over time. However, the results of this research show that sites maintain a strong compositional self-similarity over time after both low and high severity fire and that the plant communities investigated are buffered against long-term shifts in composition, even after a large infrequent disturbance event such as the 2003 fires. Although fire influences plant species richness and plant community composition, it sits within a matrix of other major environmental influences such as rainfall, and post-fire effects are transient rather than permanent. Infrequent high severity fire events in the Australian Alps should be regarded as natural disturbances, and not as disruptive perturbations, as they do not lead to permanent changes in the plant communities investigated. However, if high severity fire events become more frequent under climate change with multiple fires per decade, then resultant changes in structure, composition and richness are likely to occur.

This research has vastly improved our understanding of how subalpine, montane and tablelands plant communities respond to high severity fire and provides reserve managers with detailed species and community level responses to better inform fire management. The research also provides a baseline with which to compare the effects of future changes to fire regimes in these communities under climate change. More broadly, the research has shown how long-term data sets and ongoing monitoring play a crucial role in understanding how species richness, community composition and vegetation structure vary over time in relation to the combined influences of both climate and fire.

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List of Acronyms and Abbreviations

ACT	Australian Capital Territory
CSIRO	Commonwealth Scientific and Industrial Research Organisation
GLMM	Generalised Linear Mixed Model
HRV	Historic Range of Variability
NP	National Park
NPWS	National Parks and Wildlife Service NSW
NR	Nature Reserve
NSW	New South Wales
OEH	Office of Environment and Heritage NSW

Glossary and Terms

Basal Sprouting / Resprouting: recovery of plant stems from buds at or below ground level at the base of a main stem, usually after fire. Lower parts of the trunk usually have either thicker bark or structures such as lignotubers which protect living tissue from the effects of heat

Epicormic Sprouting / Resprouting: recovery of plant stems along main trunk and branches either from protected buds, or in the case of eucalypts, bud strands – a ‘bud bank’ - which are triggered to produce buds after stem damage from fire. Frost, hail, drought and insect herbivory also produce an epicormic response

Fire Event: a particular fire, or else series of fires occurring spatially and temporally in close proximity e.g. the 2003 Australian Alps fires

Fire Frequency: typically, the interval between fires, but the term can also cover a variety of other measures involving time such as the number of fires over time at a point in the landscape and time since fire

Fire Intensity: the heat output from a fire. While fire intensity is a direct measure of energy, it is rarely measured during a fire, so the effect of fire intensity – fire severity – is often measured post-fire

Fire-Killed Species: species which are killed by 100% leaf scorch

Fire Regime: the particular combination of fire season, fire frequency and fire intensity occurring at a point in the landscape over time after numerous fire events. Fire type can also be referred to as a component of the fire regime, namely whether fires are peat fires, surface fires or crown fires. While the fire regime will vary from point to point, there will be tend to be a ‘typical’ regime across broad vegetation types, but this will still vary locally with slope, aspect and geology.

Fire Season: whether fires typically occur in summer, autumn, winter or spring

Fire Severity: the effect of a fire, usually as measured by scorch on vegetation

Leaf Scorch: the degree or amount of scorching evident on leaves as measured post-fire – commonly simply estimated as ‘green’, ‘brown’ or ‘black’

Leaf Consumption: where leaves are fully consumed during a fire

Planned Fire: this term covers all management fires, including ‘prescribed burns’, ‘control burn’ and ‘hazard reduction burns’

Seed Bank: a store of seed produced by mature plants. This may be held above the ground in the canopy of mature plants (canopy seed banks) or else accumulate in the soil (soil seed banks)

Seeder / Reseeder Species: plant species in which mature plants are killed by 100% leaf scorch. These species rely on canopy or soil seed banks from which germination occurs after fire

Sprouting / Resprouting Species: plant species in which mature plants survive after 100% leaf scorch and regrow from protected buds or bud strands. Note: the ability to resprout can vary within a species based on interactions between age of plant, stem diameter, pre- and post-fire climate and fire intensity

Topkill: where the main upper trunk of a tree is killed by fire but the tree recovers from basal sprouting

Unplanned Fire: this term covers ‘wildfires’, ‘bushfires’ and ‘grassfires’. Note that unplanned is not the same as un-natural; and that some planned fires attempt to mimic natural regimes

Note on Taxonomy

When using scientific names for plants throughout the thesis, plant taxonomic authorities will not be cited in the text, but can be found against the relevant taxa in Appendix 1. Common names of plant species are also included in Appendix 1.

Chapter 1: Introduction

1.1 Fire in the Australian Alps

The Australian Alps region extends from the highlands of Victoria, through New South Wales (NSW) and into the Australian Capital Territory (ACT) and includes Australia’s highest peak, Mt. Kosciuszko, at 2228 m. The region is approximately 1.2 million hectares in extent and is primarily in the Australian Alps Bioregion, with some extension into the South Eastern Highlands Bioregion (Thackway and Cresswell, 1995). Twelve conservation reserves form the Australian Alps National Parks system which protect extensive areas of grasslands, heathlands, shrublands, woodlands and forests across a range alpine (>1850 m), subalpine (1400-1850 m), montane (1100-1400 m) and tablelands (<1100 m) ecosystems (Carnahan, 1986; Costin, 1954; Doherty *et al.*, 2015; Gellie, 2005; Good, 1992; Ingwersen, 1983; Keith, 2004; Walsh *et al.*, 1986; NSW Department of Planning Industry and Environment, 2020) (Figure 1.1).

Figure 1.1: The Australian Alps National Parks system. Source: Australian Alps National Parks (2020).

While there are occasional small scale fire events in the Australian Alps in the order of hundreds to thousands of hectares, it is only in years of exceptional drought that large scale fire events in the order of tens of thousands to millions of hectares occur, and when these fires do occur, they burn extensive areas with a wide range of fire intensities (Banks, 1989; Vines, 1974; Zylstra, 2006). In January 2003, a number of fires were ignited in the Australian Alps by a series of lightning strikes and subsequently burned through more than 1.5 million hectares of land in the ACT, NSW and Victoria, predominantly in subalpine and montane ecosystems. Many areas burnt in 2003 had not experienced high fire intensities, or in some cases any fires, since 1938-1939 (Zylstra, 2006). Although other large unplanned fires have occurred in the Australian Alps in the recent past, as for example in 1926 and in 1938-1939, such events are infrequent and evidence suggest that in the Australian Alps, similar large scale fire events have only occurred on five other occasions in the last 4000 years (Hope *et al.*, 2006). The area burnt in the NSW section of The Australian Alps in 2003 was similar in extent to the area that burnt in 1938-1939, but the 2003 fires burnt less of the Victorian section of the Australian Alps than was burnt in 1938-1939 and in the case of the ACT, a larger areas was affected by fire in 2003 as compared to 1938-1939 (Figures 1.2 and 1.3).

Rather than being uniform in their effects, the unplanned summer fires of 2003 provided a mosaic of fire severities across a wide range of landscape elements and vegetation types. Unlike previous fire events in southeastern Australia, extensive fire severity mapping was undertaken after the 2003 Australian Alps fires by the NSW Office of Environment and Heritage and forms an important adjunct to field based site measurements of fire severity and vegetation response (Barrett, 2006).

Because of the extent and initial physical visual impact of the fires, land and fire management practices in the Australian Alps have been the subject of much subsequent discussion and ongoing debate, particularly regarding fuel management (Cary *et al.*, 2003a; Kanowski *et al.*, 2005; Whittaker and Mercer, 2004). Unfortunately, the debate around the impact of the 2003 unplanned fires and fire management in the Australian Alps in general has not been informed by a detailed understanding of how ecosystems in the Australian Alps actually respond to extensive high intensity fires (Williams and Bradstock, 2008). Although major fire events are infrequent in the Australian Alps (Banks, 1989; Gill, 1981d; Luke and McArthur, 1978; Zylstra, 2006), when they do occur, hundreds of thousands of hectares may be burnt at high intensity (Williams and Bradstock, 2008; Williams *et al.*, 2008). These types of landscape fire events have been termed large infrequent wildfires (Meyn *et al.*, 2007) or large landscape fires (Williams and Bradstock, 2008).

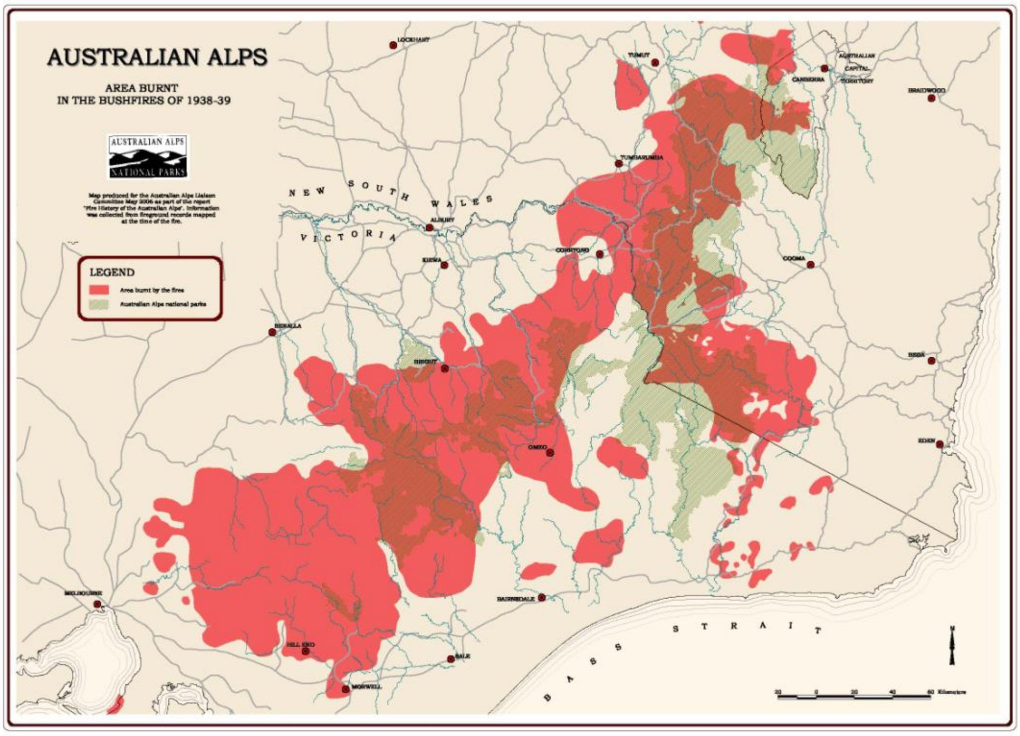


Figure 1.2: Extent of fires in the Australian Alps in the 1938-1939 summer fire season. Australian Alps National Parks shaded green. Source: Zylstra (2006).

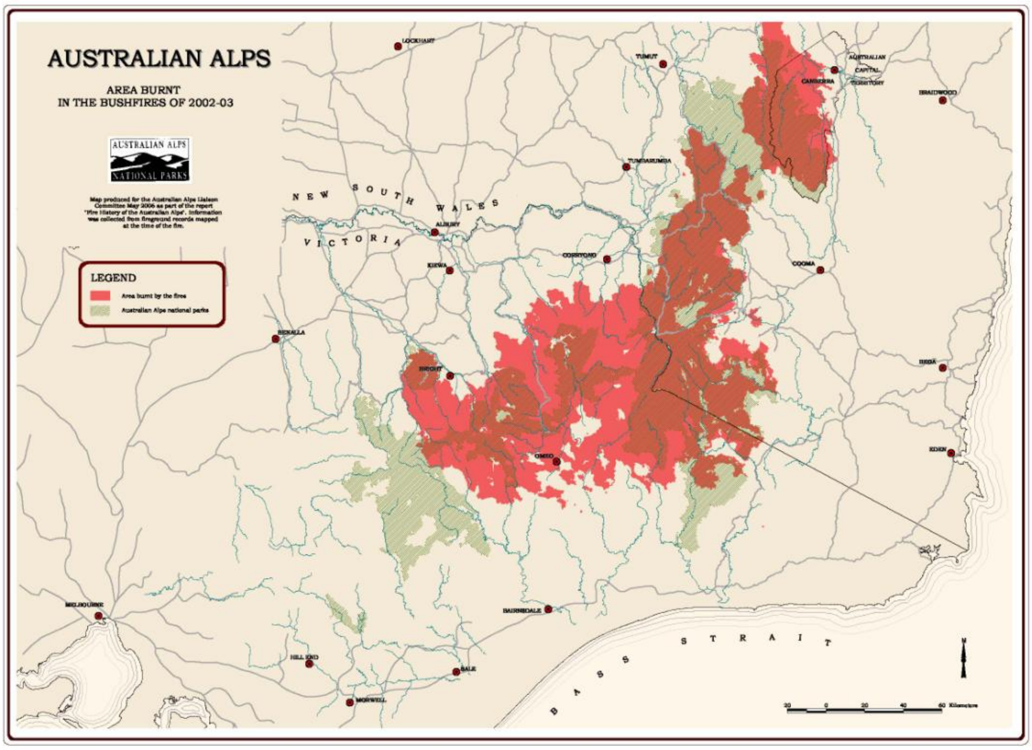


Figure 1.3: Extent of fires in the Australian Alps in the 2002-2003 summer fire season. Australian Alps National Parks shaded green. Source: Zylstra (2006).

While the short-term economic and social impacts of such large infrequent wildfires in southeastern Australia are known and acknowledged (Bradstock, 2003; Cary *et al.*, 2003a; Cary *et al.*, 2003b; Gill *et al.*, 2013; Henri, 2003; Kanowski *et al.*, 2005; Leonard, 2003), the effects of such fires on vascular plant species richness, plant community composition and vegetation structure are little studied in non-commercial montane ecosystems. The few post-fire vegetation studies undertaken after the 1938-1939 fire event lacked detailed pre-fire site data to enable before and after comparisons of plant community dynamics (Ashton, 2000).

Fires that occur infrequently, yet burn over extensive areas, occur in many other ecosystems across the world and parallels to the 2003 fires in the Australian Alps can be drawn with the Yellowstone fires in 1988 both in terms of the extent and intensity of the fires and also the public reaction to them (Anderson *et al.*, 1999; Christensen, 1991). Research in temperate forest ecosystems, particularly in the western USA, has shown that large, infrequent high intensity fires can play a key role in shaping long-term landscape patterns and ecosystem processes (Halpern, 1988). Evidence of the influence of these events can also be found in chaparral shrublands (Borchert and Odion, 1995; Keeley *et al.*, 2008; Keeley and Zedler, 2009) and in temperate coniferous forests (Agee, 1993; Agee, 1998; Christensen, 1991; Despain and Romme, 1991; Keane *et al.*, 2008). Australian temperate shrublands have also been studied in relation to high intensity fire (Keeley *et al.*, 2012; Noble and Bradstock, 1989; Specht, 1981c), but there are fewer studies investigating the response of Australian temperate forest ecosystems to such events (Williams and Bradstock, 2008). Research on high intensity fire regimes in Australian montane forests and associated communities has primarily been undertaken in ecosystems containing commercially important timber species such as *Eucalyptus regnans* in the central highlands of Victoria (Ashton, 1976; Ashton, 1981b; Ashton and Martin, 1996; Ashton, 2000) and *Eucalyptus delegatensis* subsp. *delegatensis* and *Eucalyptus delegatensis* subsp. *tasmaniensis* in NSW, Victoria and Tasmania (Ashton, 1976; Ashton and Martin, 1996; Bowman and Kirkpatrick, 1986a; Florentine *et al.*, 2008; O'Dowd and Gill, 1984).

Extensive research on the recovery of alpine and subalpine ecosystems from the effects of both grazing and burning has been undertaken in both NSW (Barker, 1988; Leigh *et al.*, 1987; Leigh *et al.*, 1991; Newman, 1954; Wimbush and Costin, 1979a; Wimbush and Costin, 1979b; Wimbush and Forrester, 1988) and in Victoria (Carr and Turner, 1959a; Carr and Turner, 1959b; Wahren *et al.*, 2001; Williams, 1990b). However, there is a paucity of research on the fire ecology of the subalpine, montane and tablelands forest communities that occur at lower elevations in the Australian Alps, partially because of the low frequency of fire and partially because of the dominance of these communities by species of non-commercial significance. The 2003 fire event created research opportunities for studying fire response in wide range of plant communities whose response to fire was poorly understood, particularly as regards high severity fire (Gill, 2006; Vivian *et al.*, 2008). The range in fire severity across the Alps enabled comparisons to be made between vegetation response at both low and high severity within and between plant communities.

1.2 Fire Severity Research in Southeastern Australia

Of the three main components of the fire regime (Gill, 1975): frequency, season and intensity, fire intensity in particular is an especially important variable to consider in interpreting post-fire plant responses as it is a measure of the potential damage to plant tissue arising from direct and indirect exposure to heat and is therefore a critical factor affecting plant survival and subsequent community dynamics (Bond and van Wilgen, 1996). As fire intensity increases and effects on vegetation become greater, differences in water, light and nutrient availability can drive vegetation dynamics via changes in both interspecific and intraspecific competitive relationships (Chapin and Van Cleve, 1981; Hanes, 1971; Keith and Bradstock, 1994; Raison, 1980; Shafi and Yarranton, 1973). However, fire intensity is rarely measured directly and for this reason, the effect of fire intensity - fire severity - is used as a surrogate measure, as it can be directly related to fireline intensity after accounting for other factors such as vegetation height (Heward *et al.*, 2013; Knox and Clarke, 2016; Rothermel and Deeming, 1980; van Wagner, 1973).

This variation in fire intensity and resultant fire severity that occurs with large unplanned fire, and particularly large areas burnt at high severity, is of particular ecological significance. Fire intensities achieved by prescribed fire are in the low to moderate range (Moreno and Oechel, 1989; Perez and Moreno, 1998) and hence the impact of high intensity fire on ecosystem processes can only be studied after large unplanned fire events. Not only are large fires likely to be associated with higher fire intensities, they will also impact on vegetation types that are generally less fire prone and because of this, mesic forest communities and communities occurring in water courses and are much more likely to burn (Gill and Allan, 2008).

Despite the importance of understanding how differences in fire severity may affect plant species and plant community dynamics, fire research in southeastern Australian ecosystems has generally not focussed on intensity and severity as variables of interest. Instead, a significant body of research has been undertaken on the effects of changes in fire frequency on plant species persistence, vegetation composition and vegetation structure, with research undertaken in many ecosystems over many decades (Bradstock *et al.*, 1997; Fox and Fox, 1986; Lunt and Morgan, 1999; Morrison *et al.*, 1995; Nieuwenhuis, 1987; Wakefield, 1970; Watson and Wardell-Johnson, 2004; Watson *et al.*, 2009). There has also been some research on differences between the season within which fires occur, but this research is limited in southeastern Australia, particularly as compared to northern Australia with its distinct wet and dry seasons, where fire frequency is high and the relationship between season and fire intensity and fire severity is strong (Edwards and Russell-Smith, 2009; Werner and Franklin, 2010; Williams *et al.*, 2002). Overall, the effects of fire intensity on vegetation dynamics have received less research attention in the southeastern Australia (Campbell and Clarke, 2006; Morrison, 1995; Morrison and Renwick, 2000; Morrison, 2002; Russell and Roberts, 1996) and in particular, vegetation response in relation to differences in fire severity remains little studied in upper montane ecosystems (Morgan and Nield, 2011; Vivian *et al.*, 2008). Whelan and Muston (1991) highlighted the need for a range of fire intensities for the maintenance of biodiversity, but stressed that little is known about the ecological effects of large wildfires. The long-term monitoring of vegetation dynamics after major fire events is therefore an essential step in creating a greater understanding of the ecological effects of large fires and their influence of long-term vegetation dynamics.

1.3 Plant Community Dynamics

Two factors are of primary significance in explaining the spatial patterning and temporal dynamics of plant species and plant communities: a) the differential response of plant species to environmental gradients and b) the role of disturbance in mediating plant species coexistence (Austin, 1985; Austin, 1987; Austin, 1990; Canham and Marks, 1985; Pickett, 1980; Pickett *et al.*, 1987; Pickett and Cadenasso, 2005; Silvertown, 2004; Walker and Willig, 1999; Willig and Walker, 1999).

Environmental gradients can occur at local, regional and continental scales and consist of *indirect gradients* (e.g. elevation and latitude, acting through local and regional changes in rainfall and temperature); *direct gradients* (e.g. pH and soil depth which have a direct physiological impact on plant growth, but are not consumed) and *essential resource gradients* (e.g. light, water, nutrients which are consumed by plants in order to grow) (Austin, 1980b; Austin and Smith, 1989; Guisan and Zimmermann, 2000). Changes along indirect and direct gradients at regional and continental levels is reflected in changes to the patterns of occurrence of species and the composition of plant communities ("beta" diversity of Whittaker, 1965) and it is these environmental gradients that form the dominant influence on the broad geographical patterns of species that we discern across landscapes.

However, there is an interplay of regional and local processes that affect species diversity (Ricklefs, 1987). Changes in species richness and composition over time at particular locations within communities ("alpha" diversity of Whittaker, 1965) are more specifically determined by species' competitive interactions for essential resources at a local level (Austin, 1990; Tilman, 1985; van Andel, 2005), particularly in relation to prevailing disturbance regimes. At the local level, disturbance (e.g. drought, fire) acts on plant communities to create gradients in time (Gleason, 1917; Tilman, 1985), where resource availabilities fluctuate and opportunities are created for species to germinate, grow and reproduce in an environment where competition from other species is reduced or eliminated, based on the life history and response traits of the species available at a particular site.

Fire acting on plant communities is a specific disturbance type among this broader range of spatial and temporal influences on plant community dynamics. Although disturbances such as fire events may change resource availability and hence affect competitive relationships in the short-term creating a transient dynamic, it is unclear whether increasing the intensity/severity of such disturbances may result in permanent change to species richness or to species composition, particularly where high intensity/severity events are rare (Frelich and Reich, 1999; Malanson, 1984; Smith *et al.*, 2016).

There has been significant debate as to the role or influence of species richness in relation to community dynamics and ecosystem function, including whether increased species richness and ecosystem complexity results in greater ecosystem stability (Holling, 1973; Waide *et al.*, 1999). This debate is ongoing and unresolved (Fonseca and Ganade, 2001) as, depending on whether studies have looked at dual or multiple species interactions or small scales or landscapes, the evidence suggests that greater complexity emerging from multiple species interactions may result in either increased or decreased ‘stability’ (Doherty *et al.*, 1998). Rather, changes in species richness at a variety of scales can be utilised as a means to examine fluxes in the overall pool of species that are available to respond to disturbance events. Disturbance events may result in transient fluxes in species richness, or else result in permanent increases or decreases in species richness and it is important to ascertain the patterns and drivers of any resulting dynamics. This, combined with a complementary assessment of changes in compositional dynamics can inform our ability to predict whether species and communities are likely to persist or change under different levels and types of disturbance.

1.4 Research Approach, Research Question and Hypotheses

The 2003 unplanned fires burnt through areas in which long-term monitoring sites had been established by the author between 1996 and 1998 (Doherty, 1997; Doherty, 1998b) in the northern part of the Australian Alps. This created a ‘natural experiment’ (Diamond, 1983) and extensive vegetation floristic and structural information collected by the author before the 2003 unplanned fires in permanent vegetation monitoring sites was available for two study areas: a) Brindabella (Brindabella National Park and the northern parts of Bimberi Nature Reserve) where the original pre-fire data was collected in 1996 and 1997) and b) Burrinjuck (Burrinjuck Nature Reserve) where the original pre-fire data was collected in 1998. These same sites were relocated and resampled after the 2003 fire event and monitored until 2013 and provide the underlying data set used for subsequent analyses. Because of the large spatial extent of the fires, only 2 out of 163 pre-fire sites remained unburnt. Post-fire research using these sites has therefore focussed on comparing between levels of fire severity and has not compared between burnt versus unburnt sites.

This long-term data set spanning a wide range of plant species and plant communities and consistently collected by the same observer provides a unique opportunity to greatly expand our knowledge of fire ecology in the Australian Alps, and to test existing assumptions and theory about fire, plants and vegetation dynamics.

Differences in fire severity within and between communities after the 2003 fires in the Australian Alps provide an opportunity to study whether there is a differential response to variations in fire severity that affects long-term vegetation dynamics. For a given large fire event, a greater range of plant communities are likely to be affected and within each community, a range of fire intensities may be expected. This enables comparisons to be made both between and within communities of the effects of fire, controlled for fire season, but with variation in fire intensity. Past fire history is also an important factor and such information, combined with the use of long-term sites with pre-fire data, can assist in examining any variation between communities and also between sites within communities. Although there are expected to be major differences in responses to high intensity fire between fire-killed dominated vegetation types versus resprouter dominated vegetation types, overall trends in vascular plant species richness, plant community composition and vegetation structure after fire, and especially high intensity fire, are poorly understood in the vegetation types found in the region.

Bond and van Wilgen (1996) distinguish between interval-dependent dynamics (frequency) and event-dependent dynamics (season, intensity) as alternative explanations of how fire drives plant population change. Keeley (2005) suggests that as examples of fire-interval effects, risk of extinction from immaturity (fire-interval < primary juvenile period) or from senescence (fire-interval > seed production /seed bank viability) can be regarded as forms of the intermediate disturbance hypothesis. In reality, multiple explanations may be relevant at different stages either within a system or between different systems over long periods of time. In chaparral, intervals above a threshold of a decade have little impact on recovery, but in drier sageshrub communities, interval and intensity remain as critical factors in recovery over extended periods. Keeley *et al*. (2005) suggest that there is also an environmental filter hypothesis, whereby post-fire changes are regulated by environmental differences varying across landscapes and sorting along environmental gradients.

As the temporal and spatial extent of the 2003 fires resulted in large areas burnt with varying intensity, these differences provide an important opportunity to focus particularly on event-dependent dynamics. The 2003 event provides a rare opportunity to study the effects of infrequent but high intensity fire on plant communities and to gain insight into species responses to disturbance and also to evaluate theory dealing with post-fire secondary succession.

It is likely that in relation to the response of plant communities to high severity fire, that as fire severity increases, there are greater changes in composition, richness and structure, compared with the pre-fire communities. The open question is whether these changes are transient or permanent. Hence, two alternative pathways can be envisaged after high severity fire: a) although potentially significantly changed in the short term, the plant community ultimately trends back toward a similar pre-disturbance composition, richness and structure compared with the pre-fire community, or b) a threshold of fire severity is reached, beyond which the community changes and a new compositional and or structural trajectory is initiated, which is different to the pre-fire community. While patterns of species richness with time-since-fire in many Australian ecosystems show strong increases and then gradual decreases with long periods post–fire (Whelan, 1995), there is a lack of studies investigating whether richness patterns post-fire vary with differences in fire severity. Additionally, species richness in and of itself does not enable a comparison of changes in plant community dynamics, where species richness may remain similar but species composition may change. The local and regional species pool which is available will determine the composition of plant communities post-disturbance and affect the course of secondary succession, but there may be differences in the availability of propagules or a differential in growth responses in relation to fire severity, which may be reflected in changes in plant species richness within plant communities. There is also a need to understand the basic fire responses of plants to be able to interpret the subsequent dynamics of any changes in plant species richness or plant community composition.

Therefore, using pre- and post-fire data from long established sites affected by the 2003 Australian Alps unplanned fires, the aims of my thesis are:

a) to document the post-fire responses of vascular plant species in subalpine, montane and tablelands plant communities, particularly in relation to high severity fire;

b) to document and understand post-fire vegetation dynamics of subalpine, montane and tablelands plant communities in relation to variation in fire severity, focussing on vascular plant species richness and plant community compositional dynamics and

c) to evaluate the results in light of existing models of post-fire species richness responses and post-fire secondary succession.

A stratified random sampling approach was used to determine site selection in the initial pre-fire survey work, enabling sampling across variation in environmental gradients – rainfall, temperature, lithology, aspect – which influence plant distribution and abundance. Analysis of the site data collected using this approach determined pre-fire communities and the use of permanent sites with photographs assisted in accurate resampling after the fire event.

Changes in the cover-abundance of native and exotic canopy and understorey vascular plant species collected pre- and post-fire are used in the analyses to assess patterns in vascular plant species richness and vegetation community composition before and after fire, using fire severity as the variable of interest. To inform the analysis of these patterns, vascular plant species responses were catalogued immediately post-fire to assess mechanisms of persistence under disturbance.

Based on the current state of knowledge in relation to secondary succession and fire severity generally and knowledge gaps in the fire ecology of subalpine, montane and tablelands ecosystems in southeastern NSW specifically, two research questions were posed:

1) What is the response to high severity fire of vascular plant species found in subalpine, montane and tablelands plant communities?

The answers to this question informed the interpretation of the second research question:

2) Does high severity fire result in significant or permanent change to vascular plant species richness and plant community composition as compared to low severity fire in subalpine, montane and tablelands plant communities and does the recovery rate vary with severity?

In relation to the second question two associated hypotheses were posed to quantitatively test the relationship between fire severity and patterns in vascular plant species richness and plant community composition:

Hypothesis 1*: High severity fire is associated with a greater increase in plant species richness within plant communities post-fire as compared to low severity fire and the duration of this effect is positively associated with fire severity*.

Hypothesis 2*: High severity fire is associated with a greater change in plant community floristic composition as compared with low severity fire and such change occurs for a longer period of time. At the upper end of fire severity, threshold effects induce permanent compositional change in plant communities*.

The remainder of the thesis is structured as follows:

Chapter 2 reviews research on plant succession, disturbance regimes and fire regimes to provide context and background to my research approach, and explores plant responses to high severity fire which underpin changes in species richness and community composition over time post-fire.

Chapter 3 introduces the study areas and their biophysical characteristics and the nature and extent of the 2003 unplanned fires and outlines the vegetation types, pre-fire sampling methods and post-fire sampling methods used in the research.

Chapter 4 summarise s the immediate post-fire responses of plants in the study areas to the 2003 fires, focussing on plant response traits and whether species are fire-killed or resprout after 100% leaf scorch and assessing in particular the status of potentially vulnerable fire-killed species.

Chapter 5 is a case study evaluating persistence traits in a fire-killed species of major structural importance throughout the Australian Alps – *Eucalyptus delegatensis* subsp. *delegatensis* – a species whose loss regionally from repeated short interval fires would have major structural and environmental impacts.

Chapter 6 investigates the on-site and external variables that affect patterns of vascular plant species richness over time post-fire, particularly fire severity, and evaluates the results from the Brindabella and Burrinjuck study areas in light of existing theory on vascular plant species richness dynamics.

Chapter 7 investigates the post-fire dynamics of the 21 plant communities found across the study areas in relation to the effects of fire severity, focussing on whether the floristic composition of sites within communities is persistent after disturbance and how these results can be interpreted in light of disturbance theory and successional trajectories.

Chapter 8 synthesises the results from Chapters 4-7 and discusses the implications of the research for long-term fire management and suggests areas for further research.

Chapter 2: Plant Succession, Disturbance and Fire Regimes

2.1 Plant Succession

2.1.1 Concepts Underpinning Plant Succession

Early 20th century concepts of vegetation change and definitions of plant communities were based on the concept of plant succession, where species replace one another in an orderly and predictable sequence over time. Successional pathways were seen as unidirectional; idealised 'climatic climax' plant communities were defined for regions; directional species replacement and relay floristics were key, and any deviations or changes from the climax community via agents such as fire, were seen as 'disclimax' communities (e.g. Clements, 1916; Clements, 1936). Such formulations of succession theory were not without critics at the time (Tansley, 1935) and subsequent critiques have stressed that because early succession theory was based on rigid developmental laws and typological thinking, it hindered a more thorough exploration of vegetation change for many decades (Drury and Nisbet, 1973; Peters, 1991). Because of this history, Burrows (1990) suggested abandoning the term succession in favour of the more neutral term 'vegetation change', but because the term succession is well entrenched in the literature, subsequent critiques have focussed more on redefining and expanding, rather than abandoning, the term.

Two key concepts in plant ecology arising in the mid 20th century transformed early successional ideas into a more dynamic concept of vegetation communities and vegetation change, where underlying environmental conditions and disturbance were recognised as being critical factors (McIntosh, 1987). Firstly, Whittaker (1953), proposed the concept of a vegetation continuum, with vegetation varying continuously along environmental gradients and succession being a process involving individual species responding to their local environment to form communities (sensu Gleason, 1926), with no absolute climax vegetation community for an area. Subsequent theory development based on the continuum concept has also used niche theory to study the nature of the relationship between environmental resource gradients and species abundance (see Austin, 1985; Austin, 2005), both with competition - the realised niche sensu Hutchinson (1957; 1959) - and without competition - the fundamental niche sensu Elton (Elton, 1927). The second key concept was the role of disturbance in vegetation dynamics (Denslow, 1980; Glenn-Lewin and van der Maarel, 1992; Levin and Paine, 1974; Pickett *et al.*, 2008; White, 1979). Rather than being a minor factor or one that deflected communities from their 'true path', disturbance is seen to be fundamental to an understanding of plant communities and vegetation change. A variety of disturbance agents acting over time and space mean that long-term site stability and the existence of an endpoint in succession are not certainties (Drury and Nisbet, 1973; Glenn-Lewin *et al.*, 1992; Pickett and White, 1985).

The combination of these two key concepts provides a model of vegetation dynamics where a pool of candidate species available in a region is selected for at a particular location both by the immediate physical environment over space and by disturbance events over space and time (Austin, 2005; Pickett and Cadenasso, 2005). While vegetation types and plant communities can still be defined using this approach, such plant assemblages should be seen as ‘snapshots’ in time, reflecting processes up to the point of observation, rather than necessarily being rigidly defined communities. Whether such assemblages can be considered stable over time will then depend on the timeframe under consideration (Chapin *et al.*, 1996). Subtle differences in initial conditions may result in very different plant communities and even in a similar environment, there may be multiple successional pathways (Fastie, 1995). Pickett *et al*. (2008) in synthesising nearly a century of succession theory, suggest the use of a broad concept of succession defined as either a change in species composition or a change in vegetation structure, or both, and where directionality or termination of succession is not assumed. This broader definition of succession is more readily accommodated within conceptualisations of vegetation change, under the general term of vegetation dynamics. Three major differentials as drivers of vegetation dynamics are suggested by Pickett *et al*. (2008):

1) differential site availability via differences in basic resource levels supported by a site or as a result of coarse scale disturbance;
2) differential species availability via differences in propagule or seed rain or the persistence of propagules in the soil or substrate; and
3) differential species performance via tolerance, facilitation or inhibition (Connell and Slatyer, 1977) from species interactions.

These have been put forward in the form of a ‘law of vegetation dynamics’ (Pickett and McDonnell, 1989) that states:

1) if a site becomes available, or
2) if species become differentially available at that site, or
3) if species perform differentially at that site, then
4) composition and/or the structure of vegetation will change through time.

Importantly, this conceptualisation puts emphasis on changes in both composition and structure as components of vegetation dynamics, as vegetation structure not only reflects resource availability on a site but is also an important factor mediating both species availability and species performance on a site (Peet, 1992). However, an important question is whether such change is permanent or only temporary. A ‘change through time’ could be regarded as referring to a permanent change in composition or structure, or it could also reflect more transient responses after disturbance events.

2.1.2 Primary and Secondary Succession

A dichotomy between primary succession (colonisation and growth on bare or newly exposed surfaces) and secondary succession (post-disturbance regrowth in areas of pre-existing vegetation) arose from early conceptualisations of vegetation change and these terms are still in general use in vegetation studies (Horn, 1974). Peet (1992) proposed new definitions of primary and secondary succession in which primary succession is dominated by mechanisms involving long-term environmental change and secondary succession is dominated by mechanisms involving population processes. Peet's definitions provide some flexibility for longer term feedbacks between primary and secondary succession over long periods of time at one location, depending on the time frame of observation. That is, a site may be gradually changing in composition over millennia as the environment changes (effectively an ongoing primary succession), but will also undergo secondary succession at shorter intervals from various forms of disturbance in the short term (Pickett *et al.*, 2008). These shorter return time disturbances will then affect the species pool which is available for longer term successional processes.

While there are specific examples of 'classic' primary succession on areas of glacial retreat (Matthews, 1999) and on newly formed volcanic surfaces (del Moral and Grishin, 1999), secondary succession is by far the more common process (Wilson, 1999) and as Pickett and Cadenasso (2005: p. 191) state: '...few successions begin with a clean slate'. Areas undergoing secondary succession have an existing suite of plant species and communities growing on them and using Peet's definitions, observations of change at a point will generally involve population processes rather than environmental processes due to the shorter time frames of observation. Because secondary succession is initiated in areas that already have a stock of adults, juveniles and propagules, there are many biological legacies (Dale *et al.*, 2005a; Franklin *et al.*, 2000) that affect post-disturbance vegetation dynamics and Dale *et al*. (2005b) define secondary succession as the process that follows disturbances which leave substantial legacies of earlier ecological systems.

All of these varying definitions of secondary succession involve the response of in situ biota to a disturbance event which by definition, removes biomass (Wilson, 1999). Clearly, because one of the fundamental requirements of fire is fuel, fire as a disturbance agent consumes existing vegetation and hence post-fire vegetation dynamics is logically a process of secondary succession (Hanes, 1971; Horn, 1974). While Noble and Slatyer (1977; 1980) provided a framework to examine post-fire secondary successional pathways using 'vital attributes' including the nature of seedbanks and whether species are killed or sprout after fire and partly incorporating the potential net effects of facilitation, tolerance and inhibition (Connell and Slatyer, 1977), there are large gaps in our knowledge of basic plant responses to fire to inform such modelling approaches.

The nature of succession, particularly secondary succession, necessitates the concept of disturbance agents and disturbance regimes acting upon an existing biota. Section 2.2 evaluates our current understanding of disturbance and disturbance regimes and in particular the effects of infrequent disturbance events such as large landscape fires, as occurred in 2003.

2.2 Disturbance

2.2.1 Disturbance Regimes

Disturbance is recognised as a major factor shaping ecosystem composition, structure and function (Connell and Slatyer, 1977; Connell, 1978; Connell, 1979; Denslow, 1980; Huston, 1979; Noble and Slatyer, 1977; Noble and Slatyer, 1980; Petraitis *et al.*, 1989; Rykiel, 1985; Sousa, 1984; Walker and Willig, 1999; Walker, 2011; White, 1979). Of the many possible types of disturbance, fire in particular is acknowledged as playing a critical role in ecological processes in many of the world's ecosystems (Bond and van Wilgen, 1996; Bond *et al.*, 2005; Bond and Keeley, 2005; Bradstock *et al.*, 2002; DeBano *et al.*, 1998; Mutch, 1970). Although disturbance is generally observed during specific events in time and space, two key factors in the ecological study of disturbance over long time periods are the frequency or return time of a disturbance and the intensity or severity with which events occur. The development of the concept of disturbance regimes (Denslow, 1980; Malanson, 1984), enabled the quantification and comparison of disturbances between sites. In particular, the concept of the fire regime, consisting of fire frequency, fire seasonality, fire intensity and fire type (Gill, 1975) has been widely used in ecological research both in Australia (Bradstock *et al.*, 1996; Clark and McLoughlin, 1986; Gill and Williams, 1996; Whelan and Muston, 1991; Williams and Gill, 1995) and internationally (Beaty and Taylor, 2001; Johnson *et al.*, 2001; Keeley and Fotheringham, 2001a; Taylor and Beaty, 2005) as a means of quantifying attributes of the sequence of fires at particular locations. The concept can be applied across larger landscapes if vegetation and environmental conditions are similar, but the concept is primarily applied at a point within a landscape (Gill and Allan, 2008).

There are varying definitions of what constitutes ecological disturbance (Clark, 1989; Pickett *et al.*, 1989; Rykiel, 1985; Sousa, 1984; Sprugel, 1991) and White and Jentsch (2001) suggest that these can be simplified into 'relative' definitions and 'absolute' definitions. Those definitions stressing deviation from the 'normal' dynamics of a community can be termed relative definitions; those definitions stressing the use of measurable changes in physical variables or in the disposal of resources, particularly the loss of biomass, to examine disturbance impacts, can be termed absolute. White and Jentsch argue that relative definitions of disturbance suffer from an inability to clearly define what the normal dynamics of a community are, including whether as system is at equilibrium or not and whether any equilibrium is determined over space (e.g. patch dynamics) or time (e.g. site dynamics) or both. By contrast they argue that absolute definitions allow the concept of a disturbance regime to be more readily quantified and incorporated into a research approach, with variation in response both within and between plant communities being measured against differences in frequency, intensity and season of a disturbance.

2.2.2 Large Infrequent Disturbances

The concept of ‘large infrequent disturbances’ (LIDs) as a major influence on ecosystem composition, structure and function is a more recent research focus of disturbance ecology (Dale *et al.*, 1998; Foster *et al.*, 1998; Romme *et al.*, 1998; Turner and Dale, 1998; Turner *et al.*, 1998). Examples of such disturbance events include volcanic eruptions (Dale *et al.*, 2005b; del Moral and Grishin, 1999; Ernst, 1908; Thornton, 1996), earthquakes (Veblen and Ashton, 1978), cyclones, hurricanes and severe windstorms (Boose *et al.*, 1994; Webb, 1958; Webb, 1999; Webb and Scanga, 2001), floods (Bornette and Amoros, 1996) and large fire events (Meyn *et al.*, 2007; Moritz, 1997). Such events provide a complex of disturbance agents acting directly and indirectly on organisms and their environment.

However, because LIDs are infrequent and their occurrence in space and time is unpredictable, they are problematic phenomena to study. A formal manipulative experimental research design is constrained by the lack of predictability of these major disturbance events and by the operational constraints of implementing a landscape design. LIDs by their very nature can only be studied as ‘natural experiments’ (Diamond, 1983; Recher *et al.*, 1975). Further, there may be no undisturbed control sites for a fully balanced experimental ‘Before-After-Control-Impact’ (BACI) design (Faith *et al.*, 1991; Underwood, 1991; Underwood, 1992).

Ideally, pre-disturbance data can be utilised where it is available, supplemented by post-disturbance monitoring (Greene, 1993; Hinds, 1984). Because these events often occur over extensive areas they not only provide an opportunity to compare ecological responses between different communities from a single event, they can also result in a range of disturbance severities, providing an opportunity to study differing ecological responses within communities to one event at different levels of disturbance. The research focus is one of assessing degrees of impact through variation in the same disturbance event, rather than differences between impact and control sites (Hargrove and Pickering, 1992; Oksanen, 2001; van Mantgem *et al.*, 2001).

Formal planned experimental approaches using manipulative field experiments with fires of low to moderate intensity are also not without potential limitations. Replicate study sites may occur within areas experiencing the same disturbance event and hence pseudo-replication in sampling (Hurlbert, 1984; Turner and Dale, 1998; van Mantgem *et al.*, 2001) and spatial autocorrelation in relation to closely placed subsamples (Bataineh *et al.*, 2006; van Mantgem and Schwilk, 2009) are potential issues if a traditional statistical approach is used.

2.2.3 Large Fire Events as Large Infrequent Disturbances

Fire events such as those that occurred in 1938-1939 and 2003 in the Australian Alps are examples of large infrequent disturbances (Williams *et al.*, 2008). Such events will only occur if climate and weather are conducive for landscape spread (Bradstock *et al.*, 1998; Bradstock *et al.*, 2009; Bradstock *et al.*, 2010; Gill and Moore, 1996; Hammill and Bradstock, 2009; Hennessy *et al.*, 2005). Weather and antecedent climatic trends are critical factors affecting both the timing and size of fires (Cary *et al.*, 2006; Cary *et al.*, 2009; Flannigan and Wotton, 2001). Maximum fire intensity in Australian forests is attained where fires burn through dry heavy fuel loads located on steep slopes under the influence of strong winds, especially during drought conditions (Gill, 1997; Lucas *et al.*, 2007; Pitman *et al.*, 2007; Sharples *et al.*, 2012). This was the case with montane communities burnt in the 2003 fires in the Australian Alps, particularly on the 18th of January 2003. On this one day under extreme fire weather conditions, many thousands of hectares burnt with high intensity and resultant severity, compared to areas which were burnt with low to moderate severity with little to no canopy scorch in the preceding and following weeks (Barrett, 2006). Although climate, weather, terrain and fuel type are major determinants of fire severity, the effect of large high intensity fires also varies considerably between ecosystems. In some ecosystems, large high intensity fires are considered a part of the natural disturbance regime whereas in others, large high intensity fires can have long lasting deleterious effects resulting in potentially permanent compositional and structural change (Schoennagel *et al.*, 2008).

Williams and Bradstock (2008) suggest three strong themes arising from a global survey of large fires from temperate ecosystems including Australia (Bradstock, 2008; Williams *et al.*, 2008), the USA (Burton *et al.*, 2008; Keane *et al.*, 2008) and the Mediterranean Basin (Pausas *et al.*, 2008):

a) large fires are usually a part of the historical range of variability (HRV), particularly in temperate biomes;
b) large fires are inherently heterogeneous and leave persistent patterns of spatial and temporal diversity and
c) large fires do not necessarily have ‘disastrous’ long-term consequences.

Rather than being seen as ‘ecological disasters’, infrequent large landscape fires can be viewed as ‘templates for diversity’ (Bradstock, 2008). They provide a wide range of post-fire conditions within which species can differentially respond and persist, either via sprouting or via seed banks. However, despite a natural background occurrence of large high severity fires, there is emerging evidence that under the influence of changes in the drivers of these types of fires, the frequency of recurrence is showing an increase over recent decades (Barbero *et al.*, 2015; Cai *et al.*, 2014; Sharples *et al.*, 2016). This puts particular pressure on interval-dependent species which require a minimum recovery time to recruit, flower and fruit between fire events. Given this, the documentation of the effects of high severity fires is critical to managing floristic and structural change that may result from any future changes to fire regimes.

Because fire acts as a disturbance agent and disturbance needs to be assessed in relation to patterns over both time and space, as disturbance regimes, Section 2.3 provides a background on fire regimes as a particular form of disturbance regime and examines the major components that influence plant and community response and recovery after fire. It also surveys how plant species respond to fires of different intensity and resultant severity as a basis for understanding and interpreting subsequent post-fire plant community dynamics. A detailed discussion of patterns of post-fire species richness and post-fire plant community dynamics can be found in the introductory sections of Chapters 6 and 7 respectively.

2.3 Fire Regimes, Plant Community Dynamics and Plant Responses

2.3.1 The Fire Regime Concept

Fire regimes consist of the long-term interaction of fire type, fire season, fire frequency and fire intensity at a point in the landscape (Gill, 1975) and have been the basis for structuring fire research in Australia and internationally for many years (Falk *et al.*, 2007; Gell *et al.*, 1993; Gill and Allan, 2008; Keeley, 1981; Olson, 1981; Whelan and Muston, 1991; Whelan, 1995; Williams *et al.*, 2009). Type, season, frequency and intensity are common metrics used to describe both the characteristics of fire at a point over time – the true regime – as well as to describe the characteristics of particular fire events.

A significant proportion of fire research in southeastern Australia has focussed on fire frequency, without comprehensively addressing fire intensity, or its measured effect, fire severity (Gill *et al.*, 1994; Williams *et al.*, 1994; Williams and Gill, 1995). This may be due to the relatively infrequent occurrence of large-area high severity fires or because other aspects of the fire regime are perceived as playing a lesser role in southeastern Australia e.g. season, as compared to northern Australia, where fire frequency is high and there is a distinct relationship between season and fire severity (Edwards and Russell-Smith, 2009; Werner and Franklin, 2010; Williams *et al.*, 2002). While there has been a somewhat disproportionate focus on vegetation response and fire frequency in the Australian fire ecology literature dating back many years (Bradstock *et al.*, 1997; Fox and Fox, 1986; Lunt and Morgan, 1999; Morrison *et al.*, 1995; Nieuwenhuis, 1987; Russell-Smith *et al.*, 2002; Wakefield, 1970; Watson and Wardell-Johnson, 2004; Watson *et al.*, 2009), vegetation response and fire intensity has been a focus for a much shorter period of time (Campbell and Clarke, 2006; Morrison, 1995; Morrison and Renwick, 2000; Morrison, 2002; Russell and Roberts, 1996) and vegetation response and fire severity per se is not often studied (Vivian *et al.*, 2008).

Although type, season and frequency are all readily assessable for a given area after a fire event and over time, fire intensity is rarely if ever assessed because it is an instantaneous variable that can only be estimated during a fire (Malanson, 1984; Simard, 1991). However, the effect of fire intensity on vegetation can be measured immediately after a fire event as fire severity, which can be used as a surrogate for fire intensity (Keeley *et al.*, 2008; Perez and Moreno, 1998). The linking of such fire characteristics to the responses of plant species provides some predictive ability in relation to the impact of a particular fire event as well as the longer term cumulative impact of a particular fire regime. Linking biological responses to disturbance events and regimes leads to an obvious question: what are the bounds within which a species can persist (i.e. survive and reproduce) under various fire disturbance scenarios?

2.3.2 Fire Regimes and Plant Community Dynamics

There has been a considerable amount of research undertaken on large fires in Mediterranean ecosystems globally, particularly in shrublands, including chaparral in the USA, matorral in Spain and Chile, garrigue and maquis in France, fynbos in South Africa and heathlands in southern Australia. These shrubland ecosystems have been termed crown-fire ecosystems (Keeley and Fotheringham, 2001b; Pausas *et al.*, 2004) because when they burn, they inevitably do so with full crown consumption and at high intensity. Such systems have been studied post-fire (Hanes, 1971; Trabaud and Lepart, 1980) and evidence from chaparral ecosystems in the USA suggests that they are highly resilient to high intensity burning and that high intensity fire has a negative effect on alien species cover and diversity (Keeley *et al.*, 2008). However, investigation into differential responses within a community to variation fire intensity has not been undertaken, particularly during a single fire event.

Crown fire shrubland ecosystems (Specht, 1981c) may be contrasted with multi-layered forest and woodland ecosystems (Specht and Morgan, 1981), where low intensity understorey fires can occur without affecting the crown, but within which high intensity crown fires will occur from time to time (Ashton, 1981b; Ashton, 1981c; Christensen *et al.*, 1981; Gill and Catling, 2002). Temperate forest and woodland ecosystems, including some forest and woodland types within the Mediterranean ecosystem biome, fall into this latter category.

While studies of temperate montane forest ecosystems in the western USA have demonstrated differential successional effects arising from differences in fire severity (Collins *et al.*, 2011; Miller and Urban, 1999; Stephenson *et al.*, 1991; Swetnam *et al.*, 2009), there is a lack of parallel studies in Australian temperate montane forest ecosystems investigating fire severity and successional pathways (Gill *et al.*, 2004; Gill, 2006). Direct comparison is problematic, as there are major differences between montane forests in the USA and Australia: temperate montane forest ecosystems in the western USA are dominated by conifers which are generally killed by intense crown fire (with notable sprouting exceptions e.g. *Sequoia sempervirens* and *Sequoiadendron giganteum*; see also Hanson and

North (2009)) whereas Australian temperate montane forest ecosystems are dominated by resprouting eucalypts which can withstand even intense crown scorch and leaf consumption (with notable fire-killed exceptions e.g. *Eucalyptus delegatensis* and *Eucalyptus regnans*). Additionally, the open nature of eucalypt-dominated communities results in the maintenance of high species richness (Westoby, 1985), with no evidence that shade tolerance or shade intolerance is an important factor in canopy vegetation dynamics, other than in wet eucalypt and rainforest ecosystems (Specht and Morgan, 1981; Specht, 1981a; Specht and Specht, 1989a; Specht and Specht, 1989b).

Bond and van Wilgen (1996) suggested that there are three competing, but not necessarily mutually exclusive, hypotheses to explain population changes in fire prone environments: an interval-dependent hypothesis; a self-regulatory (density-dependent) hypothesis and an event-dependent hypothesis. The first hypothesis emphasises interval-dependent processes operating in relation to differences in fire frequency. The second hypothesis emphasises density-dependent feedbacks which regulate population size even under constant conditions, but essentially also operating in relation to fire frequency. The third hypothesis emphasises the differences arising due to the nature of particular fire events and primarily in relation to differences in fire season and fire intensity, which can vary independently from fire frequency. All three hypotheses involve processes affecting both the recruitment and survival of plants and can therefore be seen to be consistent with Peet’s (1992) definition of secondary succession involving population processes. Bond and van Wilgen suggest that fire frequency is the most important factor where fire occurs at very short and very long intervals and that fire intensity and fire season are more important at intermediate intervals. They further suggest that stochastic effects are larger where there is greater variability in fire intensity, with patch size and structure and associated differences in intensity playing a crucial role in survival and reproduction of species. Although not stated explicitly, there appears to be an implicit assumption that a significant proportion of the species involved in such processes are variable in their response with differences in survival associated with different fire intensities or season of fire. In many ecosystems this is an open question and detailed data on plant species responses across a range fire intensities is lacking.

Particular components of the fire regime are more important for certain species, as for example in the case of fire-killed herbs and shrubs, where fire frequency (expressed as inter-fire interval) is a critical factor affecting persistence (Gill and Allan, 2008). A second fire occurring in a period less than the time to maturation for a fire-killed individual increases the risk of local extinction, particularly if the species has a canopy rather than a soil seed bank (Benson, 1985). In the case of fire-killed trees, the effect of fire intensity is also important - although there may be survival of adults from low intensity fires, there will be no survival from high intensity fires (Ashton, 1981b). Nonetheless, there is still a potential variation in response to fire for what are generally seen as ‘fire sensitive’ species and a differential response based on variable fire intensity has important implications for post-fire landscape recovery and structural dynamics.

2.3.3 Fire Intensity, Fire Severity and Plant Responses

While experimental planned ignitions can control for fire season and fire frequency, they cannot easily replicate high intensity fire. Experimental studies undertaken on fire behaviour have only been undertaken at low to moderate intensities, as high intensity prescribed fire is well beyond acceptable prescriptions for planned fires (Gould *et al.*, 2007). This means that the effects of high fire intensity can only be studied after unplanned fires and the challenge in focussing on fire intensity as a variable of interest is to be able to control as far as possible for fire season and fire frequency so that only fire intensity varies as part of any comparison being undertaken. Comparison of sites that have experienced the same fire event but with a range of fire intensities provides the only real means of gauging the impact of fire intensity across a full spectrum from low to high conditions, within and between vegetation types.

Fire intensity is most commonly measured as fireline intensity (Byram, 1959) and represents the energy released as heat during various phases of a fire. It is influenced by factors such as available fuel, moisture and temperature, chemical factors, wind and topography (Whelan, 1995). Ideally, establishing a direct relationship between fire intensity and effect on vegetation is desirable. Rothermel and Deeming (1980) suggest that there is a positive linear relationship between fireline intensity and post-fire scorch height. However, DeBano *et al.* (1998) maintain that fire intensity is rarely if ever measured and models are still lacking to adequately quantify post-fire vegetation responses directly with fire intensity.

Because heat output is a function of fireline intensity and damage caused to plant tissue increases with heat output, the resultant damage caused to plants can be related back to fireline intensity. Scorch height is commonly used as a measure of fire severity and there is a relationship between height of crown scorch and fire intensity (Keeley, 2009; Rothermel and Deeming, 1980; van Wagner, 1973) although this relationship varies to some degree with fuel type and fuel moisture (Alexander and Cruz, 2012). Despite being a variable that is directly relevant for understanding and managing post-fire recovery, fire severity has only recently started to be spatially incorporated into fire event mapping in Australia (Barrett, 2006; Chafer *et al.*, 2004; Hammill and Bradstock, 2006; Morgan and Nield, 2011; Vivian *et al.*, 2008). This is due both to increases in the availability of finer resolution remote sensing technology and also greater access to post-fire resourcing after large scale fire events.

Given this, the effect of fire intensity - fire severity - is usually measured as the variable of interest (Simard, 1991). Fire severity refers to measures that have been used to describe how fire intensity affects ecosystems (Keeley, 2009) and can be measured in different ways. It can be measured directly as the physical loss or decomposition of organic matter aboveground or belowground (Keeley, 2009). It can also be measured indirectly as the height and degree of char and scorch on vegetation vertically within a stand (Gill and Moore, 1994), or as remotely sensed patterns of before and after productivity using Normalised Difference Vegetation Index (NDVI) (Chafer *et al.*, 2004; Hammill and Bradstock, 2006). Keeley (2009) notes that the term fire severity was created to describe the effect of fire intensity on ecosystems in the absence of any direct measure of fire intensity during wildfires, stressing the difference between fire severity as an effect, and subsequent ecosystem responses after a fire event. Fire severity can be thought of as the effect of intensity on vegetation and as such, measures of vegetation loss or of scorch height on vegetation can be considered surrogate measures of fire intensity. Other measures which involve processes of recovery such as recolonisation or regeneration are examples of ecosystem responses, rather than measures of fire severity.

The superficial effects of fire can sometimes mask different levels of fire intensity and Brown *et al.* (1994) point out that a stand replacement fire (i.e. a stand-killing fire), can still vary in intensity and hence severity (i.e. there can be tree or canopy death from either a ‘lethal surface fire’ or a ‘crown fire’): the former produces leaf scorch, whereas the latter involves leaf consumption. Overall, greater impact is expected in coniferous systems from true crown fires, particularly as there are many coniferous species which can recover from leaf scorch (Hanson and North, 2009). In the Australian context, Gill (1978) terms these ‘fire-scorched’ and ‘fire-defoliated’ effects respectively. Although fire-killed eucalypts such as *Eucalyptus regnans* and *Eucalyptus delegatensis* respond in a similar manner, the large majority of eucalypts which are resprouters are more likely to resprout via epicormic sprouts or coppice rather than being killed by greater fire intensity (Gill, 1997). Post-fire surveys to date in montane eucalypt ecosystems have found mortality rates of between 3% and 15% after high intensity fire for resprouting species (Gill, 1997; Noble, 1984; Strasser *et al.*, 1996; Vivian *et al.*, 2008). Gill (1978) found a similar amount of branch death in both fire-scorched and fire-defoliated specimens of *Eucalyptus dives*, a resprouter, but it may be the case that with greater fire intensity, individuals may respond via basal coppicing rather than by epicormic resprouting, as upper stem death increases (i.e. although the individual plant remains alive, increasing intensity may result in greater above ground impact in these cases).

Although our knowledge of the recovery time to pre-fire composition or abundance of plants and animals from fires of different intensity is still incomplete, we can nonetheless construct an approach that will accommodate both frequency and intensity data (as measured by severity effects) as it becomes available. The ecological response of wet sclerophyll forests to fire is, arguably, better understood than many other drier vegetation types if only because of their greater economic value as a timber resource and their importance in water catchment protection. For example, the variation with which *Eucalyptus delegatensis* responds to fire is well documented in relation to fire frequency and fire intensity. The species is not killed by fires of low intensity but is killed by higher intensity fires resulting in full crown scorch. Although it has a lifespan of some hundreds of years; if a mature stand is killed, it can take 20 – 30 years until adequate flowering and fruit production has taken place to restore the canopy seed bank (Bowman and Kirkpatrick, 1986a; Bowman and Kirkpatrick, 1986b; Bowman and Kirkpatrick, 1986c; Battaglia and Reid, 1993; Ellis *et al.*, 1980; Florentine *et al.*, 2008).

However, some precocious and apparently suppressed individuals can flower and produce viable seed much earlier at 5-6 years after high intensity fire (Cremer *et al.*, 1984; Fielding, 1956). Additionally, there is also information on nutrient cycling in these forests which suggests a return time of approximately 12 years after a low intensity fire before nitrogen levels return to pre-fire levels, although other nutrients may take longer and the effects of high intensity fire are not known (Raison *et al.*, 1983; Raison *et al.*, 1985; Raison *et al.*, 1986; Woods and Raison, 1983). There is no documentation of the long-term understorey vegetation dynamics in this community, and there is the possibility that the floristic and structural dynamics of canopy and understorey will play out differently over space and time depending on the fire regime.

Although the Australian Alps have a large proportion of montane wet sclerophyll forest ecosystems, these forests are not necessarily easily compared with montane ecosystems internationally in relation to ecosystem responses. For example, a considerable body of research has been undertaken on fire intensity and fire severity in montane forest ecosystems in the western USA, particularly in the Cascade Range, the Sierra Nevada and the Rocky Mountains, where conifers are the dominant canopy trees. With few exceptions (e.g. giant redwood, *Sequoiadendron giganteum*) these conifers are killed by 100% leaf scorch and hence variation in fire intensity can have significant effects on tree survival and post-fire vegetation dynamics. The long absence of fire in these ecosystems is known to alter compositional dynamics, particularly in relation to the establishment of shade tolerant species (e.g. increased dominance of white fir, *Abies concolor*, in the absence of fire). However, climatically similar temperate ecosystems in Australia are dominated by eucalypts, the majority of which sprout either basally or via epicormic buds after 100% leaf scorch and within which no clear successional changes in dominant species are apparent in the long absence of fire, notwithstanding the wet sclerophyll – rainforest fire-mediated dynamics found in Victoria and Tasmania (Gilbert, 1959; Hill and Read, 1984; Jackson, 1968).

Borchert and Odion (1995) surveyed vegetation recovery after fire in chaparral and found that shrub survival was a function of both fire intensity and inherent species characteristics. Keeley (2006) found that for most large chaparral shrubs there was no difference in resprouting success with age, but that for four out of five subshrub species, younger plants had greater success at resprouting than older plants, possibly because adventitious buds may become buried in woody tissues. There is evidence from semi-arid shrublands in western NSW that mortality in some sprouters is greater at higher fire intensities, but this response is complex and is related to the interaction of shrub age, stem thickness and shrub height (Hodgkinson, 1998).

Research since 2003 research on fire and vegetation response in the alpine and subalpine zones of the Australian Alps has concluded that alpine flora and fauna are highly resilient to infrequent, large extent high severity fires (Bear and Pickering, 2006; Green and Sanecki, 2006; McDougall, 2007; Walsh and McDougall, 2004; Williams *et al.*, 2006; Williams *et al.*, 2008). In one of the few studies investigating the effects of variation in fire severity after the 2003 fires, Camac *et al*. (2013) concluded that fire severity had little impact on post-fire regeneration or recovery in closed and open alpine heathlands on the Bogong High Plains. However, apart from the work of Vivian *et al*. (2008) cited previously, there have been few published studies on post-fire responses in montane wet sclerophyll forests at lower elevations (<1600 m). Previous research in montane wet sclerophyll forests the Brindabella Ranges focussed on the response of the dominant eucalypt species to high severity fire (Noble, 1984) and used these response data to model successional dynamics using modified forest gap models (Shugart and Noble, 1981; Strasser *et al.*, 1996). Although these models can realistically recreate what could be seen as the dynamics of a primary succession on an initially vacant patch, overall, the post-fire successional processes occurring in these montane forests remain largely unknown, particularly in relation to high intensity fire.

Of the range of potential effects of fire intensity on ecosystems, the effect on vegetation structure is the most obvious post-fire and forms a fundamental component for measures of fire severity on vegetation. Turner *et al*. (1999) provided evidence that in coniferous forests dominated by fire-killed species, stand structure rather than tree species composition, is most affected in the long term by fire severity. Additionally, areas of high fire severity had greater forb cover and less graminoid and shrub cover as compared to low intensity or unburned areas. A similar result is reported for sprouting species in the Mediterranean Basin (Baeza *et al.*, 2006).

The canopy architecture of conifers and eucalypts is subtly different. Conifers have dense tapering pyramidal canopies often half to two thirds of the tree height, within which much dead branch and leaf material is retained (Ex *et al.*, 2015). Such architecture adds to the ability of fire to move from ground to canopy and to promote crown fires via a ‘ladder fuel’ arrangement (Schwilk, 2003). Eucalypts have sparse globular canopies often only one third of the tree height, with smaller branches which naturally abscise and they do not retain large amounts of fine twig or leaf material in the canopy. In smooth-barked eucalypts (e.g. *Eucalyptus mannifera*, *Eucalyptus rossii*) this means that there is a fuel discontinuity between the ground and the canopy. However, this lack of continuity is reduced in other groups of eucalypts where bark is retained either on the trunk (stringybarks, e.g. *Eucalyptus macrorhyncha*; peppermints, e.g. *Eucalyptus robertsonii*) or as annually decorticating ribbons accumulating on branches and lower trunks (ribbon gums, e.g. *Eucalyptus viminalis* and *Eucalyptus dalrympleana* subsp. *dalrympleana*) can lead to greater fuel continuity from ground layer to canopy (Jacobs, 1955; Kirkpatrick, 1997).

Bark thickness is an important factor in relation to the structural effects of fire, but the type of bark in eucalypts is also important. However, the bark types in eucalypts that are more likely to transmit and maintain heat into the cambium are somewhat counter-intuitive. Although stringybark and ironbark eucalypt species would appear to have greater insulation against the effects of heat penetration because of their thick outer bark, they are in fact more prone to heat damage than smooth-barked species (Gill and Ashton, 1968; Wesolowski *et al.*, 2014) . This is due to the interaction of bark type and thickness with overall bark moisture; smooth-barked eucalypt species being able to maintain non-lethal cambial tissue temperatures for longer compared to rough barked eucalypts. However, smooth-barked species in the ash group of eucalypts have thin bark on the trunk and are also only weak epicormic resprouters and are more susceptible to injury even at low fire intensities. This applies particularly within the ash group of eucalypts, including *Eucalyptus regnans* and *Eucalyptus delegatensis* (Ashton and Martin, 1996; Bowman and Kirkpatrick, 1986a). *Pinus* species are more likely to develop thick sapling bark early in their development where fires are frequent but of low intensity, as compared to species occurring in areas with infrequent fire (Jackson *et al.*, 1999). Because of their resprouting ability, eucalypt-dominated ecosystems may be more resistant to fire-induced successional change and therefore there may be little difference between the effects of low and high intensity fires in terms of canopy composition. In eucalypts, epicormic tissue strands keep pace with outward growth of the trunk (Burrows, 2002; Burrows, 2008) enabling persistent resprouting after fire even as trees age.

The majority of eucalypt species have a juvenile leaf form and an adult leaf form, with juvenile leaves generally larger, often covered in wax coatings and in a horizontal plane as compared to the grey-green vertically oriented adult leaves (Penfold and Willis, 1961; Jacobs, 1955). After crown scorch or full crown consumption, the original canopy structure is lost and epicormic resprouts emerge along trunk and branches and over time transition into an adult leaf form, eventually reforming the pre-fire crown structure (Burrows, 2013; Gill, 1975). Gill (1978) found that after 3 years post-fire in *Eucalyptus dives*, the transition from juvenile epicormic leaves to adult leaves was nearly complete and the branching patterns were restored. Leaf weight was restored in 8 months. However, visual observations of crown recovery in *Eucalyptus dalrympleana* subsp. *dalrympleana* suggested that it took at least 6 years for crown recovery to occur. Gill (1980) found that bark thickness in *Eucalyptus dalrympleana* subsp. *dalrympleana* was not fully restored even 7 years after a high intensity (full crown scorch) fire. This suggests a level of vulnerability (i.e. possible stem death or at least reduced vigour) to a second high intensity fire within the 7 year or more recovery interval, based on overall recovery of both bark and crown. Interestingly, the dead bark was retained on the tree and not shed until approximately 2 years after the fire. Noble (1984) found no difference between the rates of survival of lignotuberous seedlings of *Eucalyptus pauciflora*, *Eucalyptus dalrympleana* subsp. *dalrympleana* and *Eucalyptus dives* after a high intensity fire, with overall resprouting rates (pre- to post-fire) of approximately 60% for all species. This does imply an overall mortality of 40% of lignotuberous individuals due to the fire.

Dynamic forest gap modelling has been undertaken on the ACT side of the Brindabella Range using the BRIND model (Shugart and Noble, 1981; Strasser *et al.*, 1996). The BRIND model predicts the establishment, growth and death of individual tree species on southeast facing slopes using stochastic functions. Based on model results, the long absence of fire was postulated to favour single species stands such as *Eucalyptus delegatensis* subsp. *delegatensis* over *Eucalyptus dalrympleana* subsp. *dalrympleana* and *Eucalyptus pauciflora*, which only persisted with episodic fire. However, the overall dynamics of eucalypt regeneration and recruitment in subalpine woodlands and montane wet sclerophyll forest areas remain unclear and neither interspecific seedling competition (Burdon and Pryor, 1975) nor insect herbivory (Williams, 1990a) have been shown to be important factors that give advantage to one species over another. Even after major disturbance by fire, Vivian (2008) found no change in the boundary between *Eucalyptus delegatensis* subsp. *delegatensis* dominated and *Eucalyptus fastigata* dominated stands in the Brindabella Range after the 2003 fires, with self-replacement of the stands being the observed pattern. In this instance, there was potential for change given one species is fire-killed at 100% leaf scorch (*Eucalyptus delegatensis* subsp. *delegatensis*) and the other species is an epicormic sprouter (*Eucalyptus fastigata*), but neither out-competed the other under the observed conditions.

2.4 Conclusion

Early concepts of vegetation change envisaged predictable successional states, with species replacing each other over time in an orderly sequence, particularly in the case of primary succession. Although primary succession does occur, it is less predictable and orderly than originally conceived and varies markedly with the local and regional species pool. By far the more common form of plant succession is secondary succession where species that are already in place respond to environmental changes. Vegetation floristic composition and structure can change over local and regional environmental gradients, as well as over long periods of time in relation to major climatic changes. Plant species and the vegetation types that they form are also subject to the influence of disturbance events over shorter time periods. Whether a change through time is directional and permanent, or whether it is transient and temporary, can depend not only on which time frame is applied, but also the frequency and intensity of disturbance events, such as fire. Fire acting as a disturbance agent is a global phenomenon, but the effect of fire on ecosystems varies according to variation in components of the fire regime – fire frequency, fire season and fire intensity. Of particular research interest in relation to the issue of permanent vegetation change is the effect of fire at the upper end of fire intensity, where plant species are subject to full crown scorch or full crown consumption, that is, high severity fire. The effects from, and community dynamics after, high severity fire in crown fire ecosystems have been well researched and documented, particularly the Mediterranean-type shrublands found in Europe, the USA and Australia and there is also a large literature on the fire ecology of coniferous forests in the USA, including montane forest communities, in relation to high severity fire. Locally, a large amount of research on plant species and plant community responses to fire has also been undertaken over many decades in the alpine zone of the Australian Alps across a range of fire severities. However, there is a significant gap in our knowledge of the effects of fire on plant response and plant community dynamics in plant communities in the subalpine, montane and tableland zones of the Australian Alps, especially in relation to high severity fire events. More broadly, there is a lack of long-term studies - particularly involving both pre- and post-fire data – investigating plant responses, plant species richness and plant community composition in southeastern Australia (Doherty *et al.*, 2014) despite many shorter term post-fire studies. While there is a large amount of literature investigating plant succession, disturbance and fire regimes, both globally and in Australia, the exhortations of Whelan (1995) and those of Bond and van Wilgen (1996) for a focus on long-term ecological studies of fire are still relevant.

Chapter 3 outlines the location, environmental characteristics and plant communities of the study area and details the pre- and post-fire sampling approaches used to gather data for investigating the relationship between fire severity, plant response and plant community dynamics after the 2003 fires.

Chapter 3: Environment, Plant Communities and Sampling

3.1 Environmental Background

3.1.1 Location of the Study Areas

Pre- and post-fire research by the author was undertaken in two study areas located to the west and northwest of the ACT, in the most northerly parts of the Australian Alps. The Brindabella study area, comprising Brindabella National Park and the northern parts of Bimberi Nature Reserve, is located immediately to the west of the ACT border and runs from the Goodradigbee River in the north to Snowgum Hill in the south. The Burrinjuck study area is comprised of Burrinjuck Nature Reserve which is located to the northwest of the ACT and is a link between the Australian Alps, South Eastern Highlands and NSW South Western Slopes Bioregions (Thackway and Cresswell, 1995). The location of the two study areas within the broader region is shown in Figure 3.1 and a more detailed map of each separate study area is shown in Figure 3.2 and Figure 3.3.

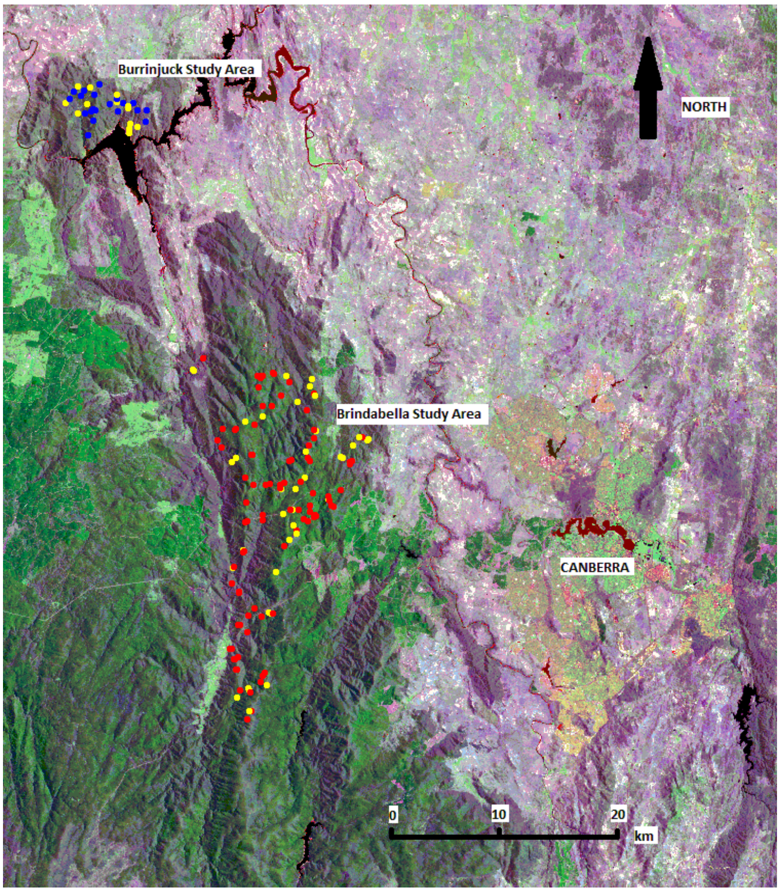


Figure 3.1: Burrinjuck and Brindabella study areas in relation to Canberra. Yellow dots are sites sampled pre- and post-fire but not after 2004; blue dots in Burrinjuck and red dots in Brindabella are sites sampled pre- and post-fire and from 2004-2012. Base map is Landsat TM false colour image.

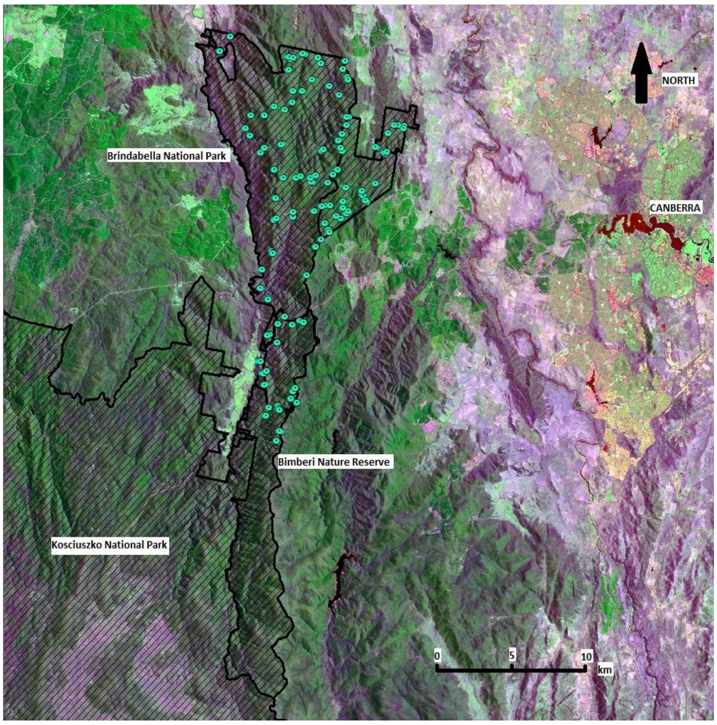


Figure 3.2: Northern extent of the Australian Alps showing Brindabella National Park, Bimberi Nature Reserve and Kosciuszko National Park. Location of the 130 Brindabella sites indicated as green circles. Base map is Landsat TM false colour image.

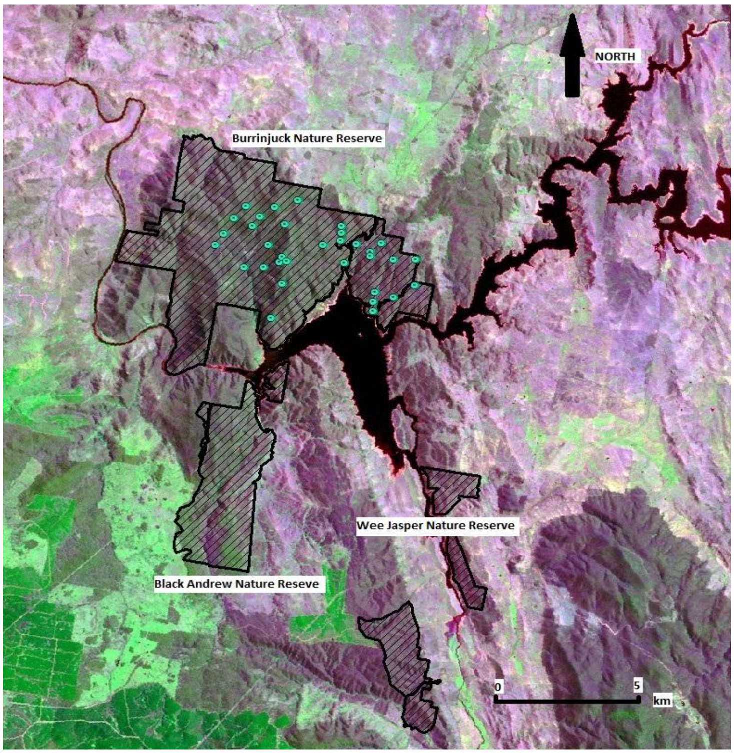


Figure 3.3: Burrinjuck Nature Reserve, Black Andrew Nature Reserve and Wee Jasper Nature Reserve. Location of the 33 Burrinjuck sites indicated as green circles. Base map is Landsat TM false colour image

The sites in each study area were originally established as part of vegetation surveys for the NSW National Parks and Wildlife Service (NPWS) (Doherty, 1997; Doherty, 1998b). The original site data was collected by the author in these two study areas from 1996-1998, comprising 130 sites in Brindabella and 33 sites in Burrinjuck. Resampling of all of the same sites occurred in 2003 after the fire event, with a smaller stratified subset of the total sites sampled from 2004-2012. The sampling approach and data collection methods for the original pre-2003 sampling are detailed in Section 3.2 and those for the post-2003 resampling in Section 3.3.

3.1.2 Elevation, Climate, Geology and Landforms

There is an altitudinal range of over 1100 m in the Brindabella study area ranging from 380 m on the Goodradigbee River in the north to 1485 m at Snowgum Hill in the south. This altitudinal range provides a range of rainfall and temperature combinations, with mean annual rainfall of 800 mm in the northeast to 1300 mm in the south. Most of the study area is within the 900 to 1200 mm range. Mean annual temperature shows a similar but opposite pattern, with temperatures of 14^0C to the north and 6^0C at Snow Gum Hill (Adomeit *et al.*, 1977; Bureau of Meteorology, 2020a). The Goodradigbee Graben forms a large undulating area bounded by the Long Plain Fault Zone near the Goodradigbee River to the west and the Dingo Dell Fault to the east. A smaller area comprises the higher Cotter Horst, bounded to the west by the Goodradigbee Fault (south) and the Dingo Dell Fault (north) and to the east by the Cotter Fault (south) and the Pig Hill Fault (north). The study area forms the upper catchment for streams flowing west into the Goodradigbee River and then into Burrinjuck Reservoir (and eventually the Murrumbidgee River) and streams flowing north and east directly into Murrumbidgee River. Two major lithologies occur in the area: large areas to the west and north composed of early Devonian extrusive acid volcanics (the Mountain Creek Volcanics - rhyolite, andesite, dacite, agglomerate, tuff) and a narrower band to the east composed of late Ordovician sediments (the Nungar Beds - greywacke, sandstone, slate, chert, limestone, quartzite). These lithologies correspond to the Goodradigbee Graben and Cotter Horst respectively (Strusz, 1971).

The altitudinal range in the Burrinjuck Nature Reserve ranges from less than 400 m around the foreshores of Lake Burrinjuck to 965 m on the top of Barren Jack Mountain. The climate found in the reserve is transitional between a cooler moister montane regime and a drier warmer tablelands regime and annual mean rainfall is approximately 700-950 mm, with annual mean temperature of approximately 13-14^0C (Adomeit *et al.*, 1977; Bureau of Meteorology, 2020a). Burrinjuck Nature Reserve is situated on the Black Range Synclinorium consisting of early Devonian volcanics and sediment with open folding and some faulting. The western portion of area lies on rhyolite, dacite, andesite, tuff and agglomerate. To the east, the Reserve is composed of tuff, shale, tuffaceous siltstone, rhyolite and agglomerate. Between these two major areas and bounding the eastern section of the reserve there is a small strip composed of shale, siltstone, tuffaceous sandstone and tuff (Cramsie *et al.*, 1975).

3.1.3 The Extent and Severity of the 2003 Fires

Lightning strikes initiated a series of fires in the Australian Alps on the 7th and 8th of January 2003. Over 160 fires were initiated in the higher parts of Victoria, NSW and the ACT (Ellis, 2004; McLeod, 2003). Drought conditions throughout 2002 prior to the lightning strikes had been severe, with most of eastern Australia experiencing dry conditions, known as the Millennium Drought, from 1996-2010 (Bureau of Meteorology, 2020b; Zhao *et al.*, 2017). In the Brindabella and Burrinjuck study areas, fires that had been initiated on the 8th of January had been spreading for many days burning at low to moderate severities but on the 18th of January, conditions worsened and the fires spread rapidly at high intensity with resultant high severity effects. Conditions moderated after this day but fires were still burning in the study areas up until the end of January 2003. Although fire intensity data is typically not measured during fire events, fire severity mapping was undertaken immediately after the 2003 fires for the NSW section of the Australian Alps using remote sensing (Barrett, 2006). The one month span within which the fires burned resulted in a mosaic of fire severities, particularly in the Brindabella study area (Figure 3.4). This fire severity data layer enabled comparison of on-ground assessments of fire severity with mapped severity for the Brindabella study area. The mapping did not extend to the Burrinjuck study area.

Post-fire sampling in the Brindabella and Burrinjuck study areas incorporated an on-ground assessment of upper canopy scorch as a percentage of original upper canopy height (Section 3.3). For the Brindabella study area, a comparison between on-site data collected in late 2003, and fire severity classes for those sites based on the fire severity mapping, showed very good correspondence between on-site and remotely sensed classes. Low-Moderate remote sensing classes essentially corresponded to a range of 0-70% on-site scorch and High-Very High remote sensing classes corresponded to a range of 70-100% on-site scorch. Figure 3.5 shows remotely sensed severity classes sequenced from left to right in order of percent canopy scorch of original canopy height from the site data.

Because the structure of vegetation is such that species of high abundance form definable strata within plant communities (Specht, 1970), different strata can burn at different severities on the same site during a fire event. In the case of the effects of the 2003 fires on the vegetation types in the Brindabella and Burrinjuck study areas, the on-site severity assessments relate to scorch on the upper canopy and are therefore comparable with remotely sensed data. On High-Very High severity sites, both understorey and upper canopies burnt at high severity. However, on Low-Moderate severity sites, even though the upper canopies burnt at low-moderate severity, the understorey components as a layer burnt at high to very high severity, but did not affect the upper canopy cover.

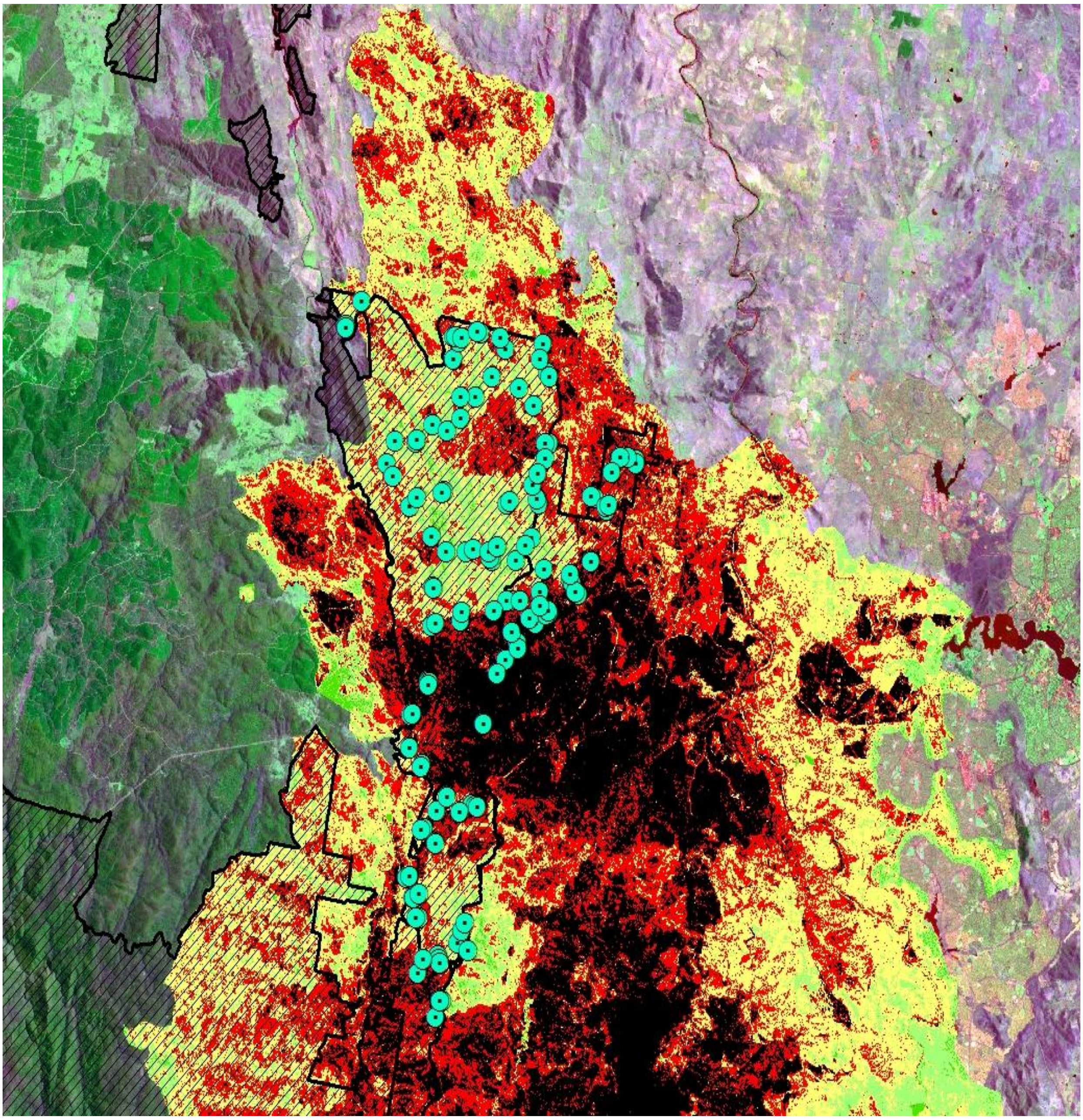


Figure 3.4: Fire extent and fire severity from the 2003 fires in the Brindabella study area. Hatched areas are conservation reserves. Pre-fire vegetation survey sites overlain as blue dots. Within the mapped area, green is unburnt/low severity; yellow is moderate severity, red is high severity and black is very high severity Source: Base map is Landsat TM false colour image Barrett (2006).

Of the 130 sites in the Brindabella study area, only two sites did not burn during the 2003 fire event and of those that did burn, 61 burnt at Low-Moderate severity and 67 burnt at High-Very High severity. Of the 33 sites in Burrinjuck, 22 burnt at Low-Moderate severity and 11 burnt at High-Very High severity in the 2003 fires. Because of proportional sampling, vegetation types which were more widespread had more sample sites and had an array of sites within them that were burnt at Low-Moderate and at High-Very High severity (Figure 3.6). Essentially, the conditions during the fires, particularly on the 18th of January, were such that all vegetation types burnt and there were no topographic constraints on fire spread when fire weather conditions were extreme. There is also evidence that topographic 'fire channelling', where local topography channels fire spread, occurred in the Brindabella area, exacerbating the intensity and rate of spread of the fires (Sharples *et al.*, 2012) and their penetration into mesic sites. Protected slopes and gully environments that would normally only experience low intensity fire burnt at high intensity on the 18th of January 2003. Details of the immediate effects of the 2003 fires on vascular plant species and vegetation types are detailed in Chapter 4.

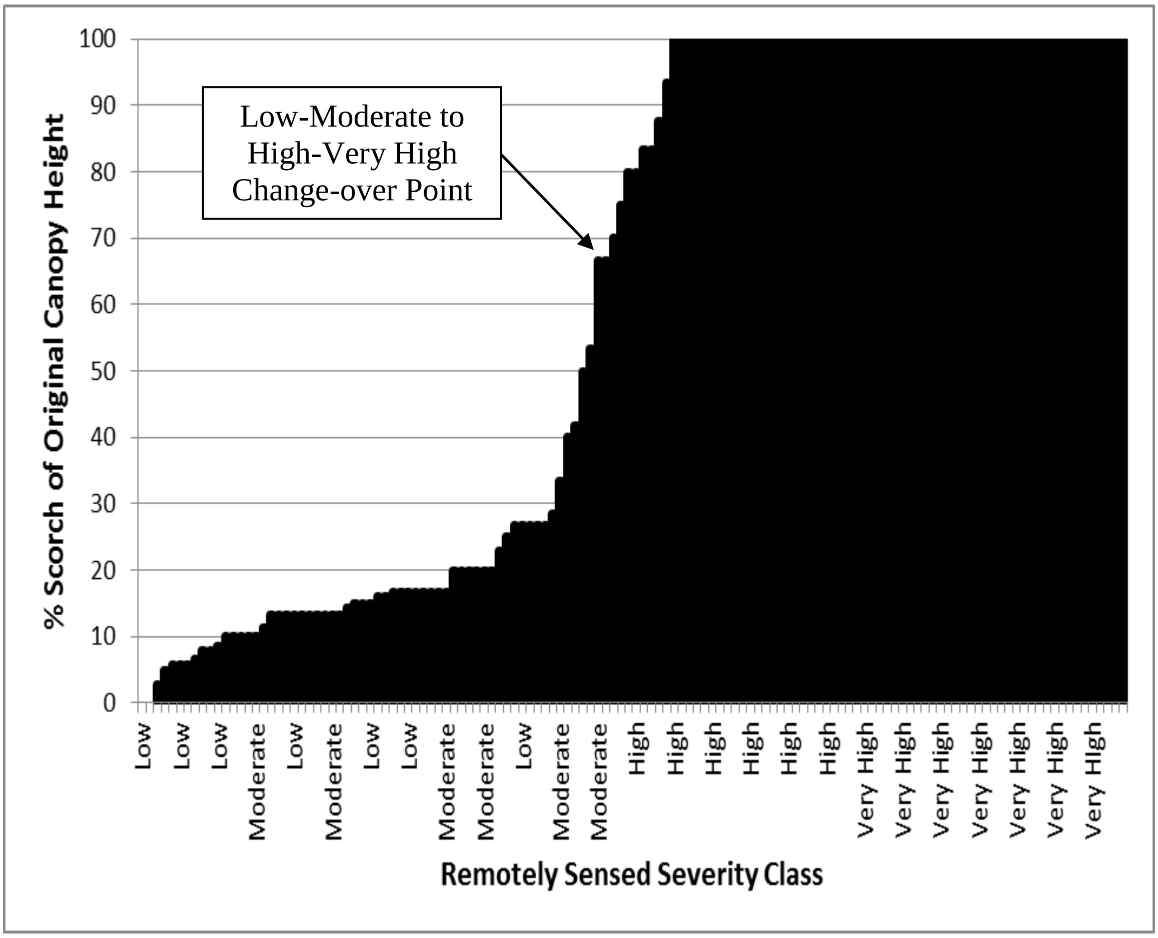


Figure 3.5: Remotely sensed fire severity classes for the Brindabella study area. Classes are ordered by % scorch of original canopy height.

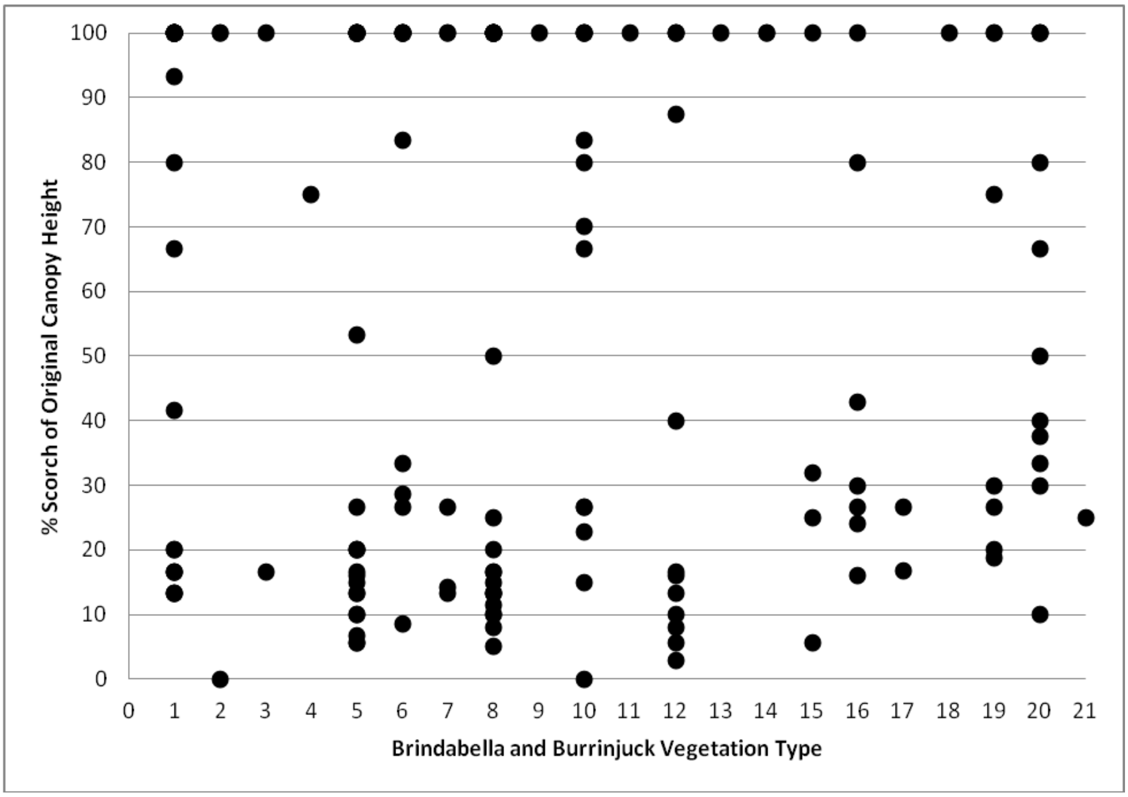


Figure 3.6 Range of percentage scorch of original canopy height on sites, by vegetation type, for the Brindabella and Burrinjuck study areas. Vegetation types 1-14 occur in the Brindabella study area and vegetation types 15-21 occur in the Burrinjuck study area. Summary descriptions of the vegetation types can be found in Table 3.1.

3.2 Pre-Fire Sampling and Plant Communities

3.2.1 Pre-fire Sampling in 1996/1997 and 1998

3.2.1.1 Landscape Sampling Design

The survey design used in the original vegetation survey design in 1996/1997 for Brindabella and 1998 for Burrinjuck (Doherty, 1997; Doherty, 1998b) was that of stratified random sampling involving sampling within unique environmental cells that were environmentally representative, geographically randomised and replicated (Austin and Heyligers, 1989). Given that plant species respond to environmental gradients, an environmental stratification ensures that all potential variation can be sampled (Austin, 2005; Belbin, 1991a). The survey design involved stratification using toposequences within these environmental cells. One sample in each environmental cell consisted of a number of sites which reflected the topographic variation found within that cell. The survey aim was to sample all cells at least once and to complete at least two samples in those environmental cells of larger extent. At least three sites per altitudinal unit were sampled and were chosen from six potential topographic positions: crest, upper crest, mid-slope, lower slope, flat and open depression. Cells to be sampled were located well within cell boundaries and site placement was random, but within definite topographic units.

The Brindabella area was stratified by lithology, rainfall and temperature to determine the environmental cells where sites were to be located, consisting of two lithology classes: Volcanics and Sediments; 5 rainfall classes: < 800 mm; 800-900 mm; 900-1000 mm; 1000-1100 mm; 1100-1200 mm; > 1200 mm and 5 temperature classes: < 10^0 C; > 10^0C and < 11^0C; > 11^0C and < 12^0C; > 12^0C and < 13^0C; > 13^0C.

As Burrinjuck Nature Reserve is small in size and only has a small altitudinal range a stratification was used based on lithology and altitude and consisted of 3 lithology types: Volcanics, Shale and Tuff and four altitude levels: <400 m; 400-600 m; 600-800 m and >800 m.

3.2.1.2 Site Sampling

Site samples were 20 m x 20 m unless they occurred on drainage lines in which case they were elongated to 40 m in length but only 10 m in width to avoid sampling non-riparian vegetation found on slopes adjacent to the drainage lines. This gave a sampling area size for all sites of 0.04 ha. Sites were placed such that the sides were parallel to the direction of maximum slope; in drainage lines the long axis of the site faced downstream. All sites were permanently marked with a metal fencing dropper driven into the centre of the site and all droppers had a metal tag attached to their top with a site number from 1 to 130 in the case of Brindabella and 1 to 33 in the case of Burrinjuck. Slope and aspect measurements were taken at the centre of each site and colour photographs were taken from the centre-rear of each site looking down slope toward the centre. Sites sampled were georeferenced on marked-up hard copy 1:25000 topographic maps in the case of Brindabella and on 1:50000 topographic maps in the case of Burrinjuck.

All vascular plant species on each site were recorded and the cover of each species on the site was visually estimated using a 6-point modified Braun-Blanquet cover scale (Poore, 1955) as follows: 1 = 1 or a few individuals but < 5% cover; 2 = Many individuals but < 5% cover; 3 = 5 - <25% cover; 4 = 25 - <50% cover; 5 = 50 - <75% cover; 6 = 75 -100% cover.

In the pre-fire survey, 373 vascular plant species were sampled in the 130 sites at Brindabella (407 vascular plant species were recorded overall in Brindabella including species found outside sites in the original survey) and 158 vascular plant species were sampled in the 33 sites at Burrinjuck (190 vascular plant species were recorded overall in Burrinjuck including species found outside sites in the original survey). Fungi, liverworts and mosses were not sampled but specimens recorded for the study area based on herbarium records are provided in Appendix 1 as a component of the overall plant species lists for the study area. The identity of plant specimens was verified at the CSIRO Herbarium, Black Mountain, Canberra. Botanical nomenclature for species lists follows the taxonomy used in NSW PlantNET (http://plantnet.rbgsyd.nsw.gov.au/search/simple.htm). Throughout the thesis, plant taxonomic authorities will not be cited in text when scientific names are used, but can be found with the relevant taxa in Appendix 1.

Structural data was also collected at each site: the maximum height and estimated percentage cover of the dominant species in each layer was recorded for upper, middle and lower layers with the possibility for multiple middle and lower layers, and any emergent layer if present. In addition, a fauna Habitat Complexity Score was derived at each site (Newsome *et al.*, 1975; Newsome and Catling, 1979). This aggregate measure has been used as a means of predicting habitat suitability for small mammals based on the structural attributes of a site – the higher and more complex the score, the greater the likelihood of the presence of small mammals (Catling and Burt, 1995; Coops and Catling, 1997b; Coops and Catling, 1997a). It complements the structural measures and is in effect an aggregate measure of the overall structural complexity of a site.

All structural and botanical field measurements and laboratory plant identifications were undertaken by the author, with field technicians assisting in site layout and data recording.

3.2.2 Pre-2003 Fire Plant Communities

Doherty (1998a, 1998b) described fourteen vegetation types - plant communities - for the Brindabella study area and seven vegetation types for the Burrinjuck study area. The vegetation type nomenclature used consists of a combination of an overstorey dominant canopy species descriptor with a vegetation structural formation descriptor sensu Specht (Specht, 1970). These vegetation types are briefly summarised in Table 3.1 with their equivalent NSW Vegetation Class descriptor (Keith, 2004) and greater detail on the distribution and characteristic species found in the vegetation types is summarised in Appendix 2. Although there is overlap of vascular plant species between these areas (Appendix 1), the vegetation types are distinct between the two study areas and form a continuum of changing types from high elevation subalpine (1100-1400 m) types to mid elevation montane (700–1100 m) types to low elevation tablelands (1100 m) types.

Vegetation types 1 (Brindabella) and 20 (Burrinjuck) have an overstorey of *Eucalyptus dives*, *E. macrorhyncha* and *E. mannifera* in common but differ in their understorey floristics in the mid and lower strata. Similarly, vegetation types 4 (Brindabella) and 17 (Burrinjuck) have an overstorey of *E. bridgesiana* and *E. melliodora* in common but also differ in their understorey floristics in the mid and lower strata.

The vegetation types from this point on in the thesis are sequentially numbered from 1 to 21 incorporating the types across both of the study areas from Brindabella to Burrinjuck for the purposes of the post-2003 analyses. References to vegetation type numbers throughout the thesis from this point are based on these communities and the numbering in Table 3.1.

Table 3.1: Vegetation types in the Brindabella and Burrinjuck study areas. Detailed descriptions of the vegetation types are in Appendix 2.

Vegetation Type Number	**Study Area**	**Dominant Canopy Species**	**Canopy Height Range**	**Vegetation Structure (Specht, 1981b)**	**NSW Vegetation Class (Keith, 2004)**
1	Brindabella	*Eucalyptus dives - Eucalyptus macrorhyncha - Eucalyptus mannifera (+/- Eucalyptus rubida)*	8-25 m	Open Forest	Southern Tableland Dry Sclerophyll Forests
2	Brindabella	*Eucalyptus macrorhyncha - Eucalyptus nortonii*	8-10 m	Low Open Forest	Southern Tableland Dry Sclerophyll Forests
3	Brindabella	*Eucalyptus dives - Eucalyptus mannifera - Eucalyptus macrorhyncha*	12-15 m	Open Forest	Southern Tableland Dry Sclerophyll Forests
4	Brindabella	*Eucalyptus bridgesiana - Eucalyptus melliodora*	15-20 m	Open Forest	Southern Tableland Grassy Woodlands
5	Brindabella	*Eucalyptus dalrympleana - Eucalyptus dives (+/- Eucalyptus pauciflora)*	10-35 m	Open Forest	Southern Tableland Dry Sclerophyll Forests
6	Brindabella	*Eucalyptus dalrympleana - Eucalyptus robertsonii*	15-35 m	Tall Open Forest	Southern Tableland Wet Sclerophyll Forests
7	Brindabella	*Eucalyptus dalrympleana - Eucalyptus delegatensis*	30-35 m	Tall Open Forest	Montane Wet Sclerophyll Forests
8	Brindabella	*Eucalyptus dalrympleana - Eucalyptus pauciflora*	15-35 m	Woodland / Open Forest	Subalpine Woodlands
9	Brindabella	*Eucalyptus camphora subsp. humeana*	20-25 m	Open Forest	Southern Tableland Wet Sclerophyll Forests
10	Brindabella	*Eucalyptus viminalis - Acacia melanoxylon*	15-35 m	Open Forest	Southern Tableland Wet Sclerophyll Forests
11	Brindabella	*Casuarina cunninghamiana*	25-30 m	Tall Open Forest	Eastern Riverine Forests
12	Brindabella	*Eucalyptus dalrympleana - Eucalyptus fastigata*	20-40 m	Tall Open Forest	Southern Tableland Wet Sclerophyll Forests
13	Brindabella	*Leptospermum micromyrtus*	1-2 m	Closed Heath	Southern Montane Heaths
14	Brindabella	*Leptospermum myrtifolium – Baeckea utilis*	2-5 m	Shrubland	Montane Bogs and Fens
15	Burrinjuck	*Eucalyptus bridgesiana*	18-35 m	Open Forest	Southern Tableland Grassy Woodlands
16	Burrinjuck	*Eucalyptus bicostata – E. robertsonii – E. dalrympleana*	18-35 m	Tall Open Forest	Southern Tableland Wet Sclerophyll Forests
17	Burrinjuck	*Eucalyptus bridgesiana – E. melliodora*	15-18 m	Woodland / Open Forest	Southern Tableland Grassy Woodlands
18	Burrinjuck	*Calytrix tetragona – Westringia eremicola*	1-2 m	Open Heath	Southern Montane Heaths
19	Burrinjuck	*Eucalyptus goniocalyx – E. macrorhyncha*	10-20 m	Open Forest	Southern Tableland Dry Sclerophyll Forests
20	Burrinjuck	*Eucalyptus dives – E. macrorhyncha – E. mannifera*	12-30 m	Open Forest	Southern Tableland Dry Sclerophyll Forests
21	Burrinjuck	*Eucalyptus rossii*	10-12 m	Open Forest	Southern Tableland Dry Sclerophyll Forests

The original pre-fire vegetation classification produced in 1998 was based on a multivariate analysis of quantitative full floristic site data using modified Braun-Blanquet cover scores as ordinal values. This pre-fire data set consisted of 130 sites by 371 species for Brindabella and 33 sites x 158 species for Burrinjuck and these matrices were analysed using the PATN multivariate software package (Belbin, 1995; Belbin, 1991a). Compositional dissimilarity between sites was calculated using the quantitative version of the Bray-Curtis coefficient (Bray and Curtis, 1957; Gauch, 1973) (Equation 1). Dissimilarity measures quantify the compositional dissimilarity between different sites, with values between sites varying between 0 (sites have all species in common i.e. totally similar) and 1 (sites have no species in common i.e. totally dissimilar).

$$BC_d = \frac{\sum \left| x_i - x_j \right|}{\sum \left(x_i + x_j \right)}$$

x_{ik} is the abundance of species *i* in sample *k*; x_{jk} is the abundance of species *j* in sample *k*

Equation 1: Quantitative version of the Bray-Curtis formula for calculating site dissimilarity

A flexible unweighted pair group arithmetic averaging (UPGMA) clustering strategy (Austin and Belbin, 1982; Faith *et al.*, 1987; Sneath and Sokal, 1973) was used to produce a hierarchical clustering of the sites, with a beta dilation value of -0.1. A beta of -0.1 enables slight dilation of the resulting groups so that groups are not overly clustered together (Belbin *et al.*, 1992). UPGMA provides the best fit between the input association measure and the distances implied in the resulting dendrogram and has been tested against its ability to recover known data structures (Belbin, 1991a). The resulting dendrogram – ‘tree diagram’ - produced after clustering shows site to site dissimilarity and the hierarchical relationships between clusters, moving up and down the dendrogram. Site relationships were interpreted from the resulting dendrogram based on floristic relationships and environmental and structural data from the field to produce the fourteen Brindabella and seven Burrinjuck vegetation types (Doherty, 1997; Doherty, 1998b). This multivariate approach was also used in the multivariate analyses undertaken in Chapter 7, using classification of sites as per the original pre-fire sampling described above, with the addition of analyses using ordination, the methods of which are discussed in Section 7.2. The approach used in post-fire resampling from 2003 onwards is detailed in section 3.3.

3.3 Post-fire Sampling

As the fires occurred in the study area in January 2003, the resampling campaign was designed to be implemented in the first spring after the fires in November 2003. Although some reconnaissance work was undertaken in the field immediately after the fires, it was decided that sampling too early post-fire would provide too many false negative species records (i.e. there would have been too little time for species to initiate recovery), particularly entering the cooler months post-fire. Beginning in November 2003, relocating and sampling was undertaken of all 130 sites in Brindabella and all 33 sites in Burrinjuck. All except seven sites in Brindabella were successfully relocated and resampled, giving a total first post-fire resample of 156 sites: 123 in Brindabella and 33 in Burrinjuck.

Sites were resampled based on the same pre-fire methods used by the author in 1996-1997 and 1998 outlined previously in Section 3.2.1. All vascular plant species on each site were recorded using the same modified Braun-Blanquet 1-6 cover score as per pre-fire sampling and similarly, vegetation structure categorised by main structural layers was also recorded as per pre-fire sampling. Additional information was collected on each site, with the addition of accurate GPS recordings of site positions, replicate site photos looking downslope from the upslope rear of each site and also additional photos looking upslope and down slope from the centre of each site. In addition, all plant species were assigned a fire response category based on observations of that species across the range of sites on which it occurred and reproductive state was also recorded (flowering, fruiting, seeding). The fire response categories used were those of Gill and Bradstock (1992).

In the first spring after the 2003 fire, all but seven of the potential 163 sites in the three reserves were resampled. This first resampling of sites took 3 months to complete with the field sampling campaign starting in November 2003. Because of time limitations, an annual field sampling campaign of 3 months over many years was not attempted, and a subset of sites was selected for ongoing sampling in the long term. The subset was stratified by vegetation type and fire severity and sites were selected for ongoing monitoring based on sampling at least two low and two high intensity sites within each vegetation type or, if there were 4 or less sites in a vegetation type, then one low and one high intensity site were sampled and if only one site occurred in vegetation type, then that site was sampled. Low intensity sites were defined as those with <35% scorch of original canopy height; moderate as 35-70% scorch of original canopy height and high as >70% scorch of original canopy height. This resulted in 52 sites being selected for continuing sampling from the 2004 sampling season onwards: 39 sites in Brindabella and 13 sites in Burrinjuck. Including the full initial post-fire survey and subsequent re-surveys of the subset of sites, sites were resampled in spring-summer: 2003, 2004, 2005, 2006, 2007, 2010 and 2012 (Figure 3.7 and Figure 3.8).

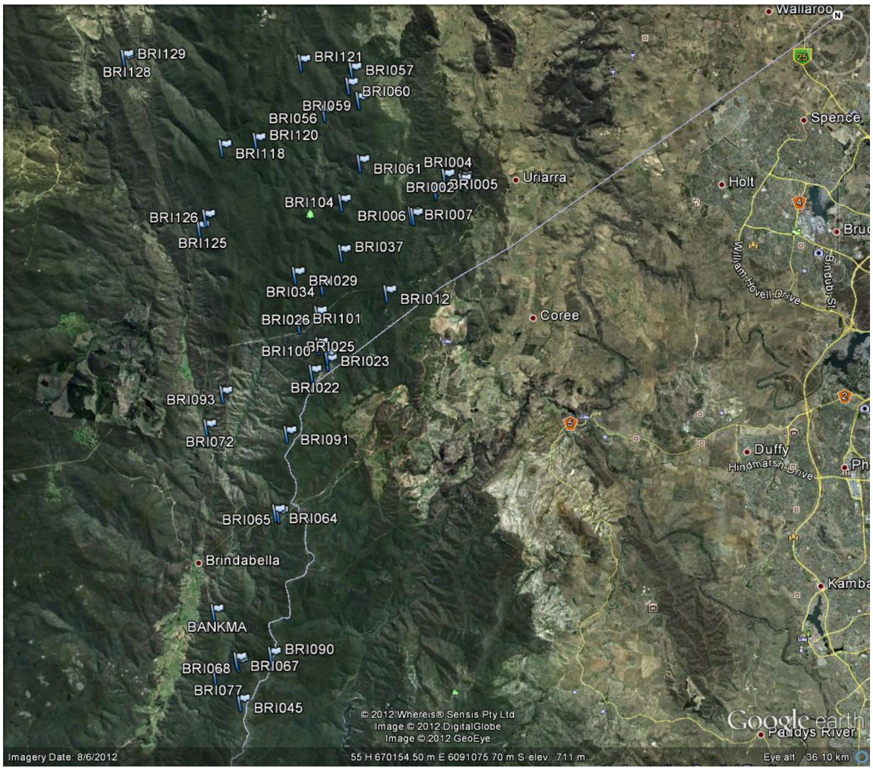


Figure 3.7: Brindabella study area showing the location of 39 ongoing sampling sites

Figure 3.8: Burrinjuck study area showing the location of 13 ongoing sampling sites

In subsequent analyses in following chapters, results will be differentiated in relation to whether they are pre-fire versus first post-fire sample comparisons; or pre-fire versus ongoing sample comparisons (i.e. second to seventh post-fire samples). Table 3.2 outlines the terminology used in chapters 4 to 8 in relation to the sampling undertaken in the Brindabella and Burrinjuck study areas from 1996 to 2012; the sampling identifier will be used when discussing comparisons between years.

Table 3.2: Sampling terminology and number of sites sampled in the Brindabella and Burrinjuck study areas from 1996 to 2012

Sampling Identifier	Year	Sample Period	Number of Sites Sampled in Brindabella	Number of Sites Sampled in Burrinjuck	General Sampling Terminology	
Pre-fire sample	1996/1997 (BRIND) or 1998 (BURR)	1	130	33	Pre-fire Sampling	
First post-fire sample	2003	2	123	33		Post-fire Sampling
Second post-fire sample	2004	3	39	13	Ongoing Sampling	
Third post-fire sample	2005	4	39	13		
Fourth post-fire sample	2006	5	39	13		
Fifth post-fire sample	2007	6	39	13		
Sixth post-fire sample	2010	7	39	13		
Seventh post-fire sample	2012	8	39	13		

Overall, in relation to the data used in the analyses in the following chapters, there was a total of 480 site x time samples across 571 vascular plant species (495 native; 76 exotic: 13% exotic) from 1997–2012 for the Brindabella study area, and a total of 147 site x time samples across 395 vascular plant species (316 native; 79 exotic: 20% exotic) from 1998–2012 for the Burrinjuck study area. It should be noted that the total number of species for the study areas detailed in Appendix 2 is greater than the total number of species used in the analyses, as there were species recorded from the study areas that were not located on sample sites.

Analytical methods used for the later analyses of vascular plant species richness and plant community composition are detailed in Section 6.2 and Section 7.2 respectively. Chapter 4 surveys and summarises the initial responses of the vascular plant species and plant communities in the study areas to high severity fire as an essential component to the later interpretation of the quantitative analyses undertaken investigating the relationships between fire severity, vascular plant species richness and community composition in Chapters 6 and 7.

Chapter 4: Post-Fire Responses of Vascular Plant Species

Note: Components of this chapter have been presented at Bushfire 2004 (2004, Adelaide); Bushfire 2006 (2006, Brisbane) and the Nature Conservation Council of NSW Bushfire Conference (2011, Sydney).

4.1 Introduction

Fire acts in a wide range of ecosystems mediating plant species coexistence and driving plant community dynamics (Bond and van Wilgen, 1996; Bond and Keeley, 2005; Bond *et al.*, 2005). Fires of differing season, frequency and intensity lead to the persistence, loss, or gain of plant species, as determined by the interactions between species traits, plant competition and fire regimes (Clark *et al.*, 2002; Gill, 1981c; Gill and Bradstock, 2003; Morgan *et al.*, 2001; Pausas and Lloret, 2007; Whelan *et al.*, 2002). Post-fire studies of plant response can focus on different levels of response, from the individual, to the species to the community (Lawes and Clarke, 2011; Whelan *et al.*, 2002). At the individual and species level, functional traits such as whether a plant is killed by fire or whether it resprouts and whether it has a canopy or a soil seed bank (Enright *et al.*, 1998a; Enright *et al.*, 1998b; Pausas *et al.*, 2004) form a basis for predicting species responses and population dynamics to fire events and to changes in fire regimes (Bradstock and Myerscough, 1981; Bradstock and O'Connell, 1988). The aggregation of such responses across many interacting species over changing environmental gradients and under varying fire regimes forms the basis for research on plant community dynamics (Bond and van Wilgen, 1996; Capitanio and Carcaillet, 2008; Johnson, 1995). Dale *et al.* (2005b) suggest that species may be able to cope with and recover from rare disturbances, such as infrequent fire, because of adaptations to more common disturbances such as drought and it is arguable as to whether plants can be said to be adapted to fire per se (Bradshaw *et al.*, 2011; Keeley, 1991; Pausas *et al.*, 2004). Keeley (2006) suggests that resprouting should be thought of as an adaptation to disturbance, fire being only one possible agent of disturbance which may include drought or severe wind storms (Allen *et al.*, 2010; Benwell, 2007; Davidson and Reid, 1989; Everham and Brokaw, 1996; Pook *et al.*, 1966; Webb, 1999; Webb and Scanga, 2001).

The classification of plant responses in Australia is based on whether a plant is killed or whether it sprouts after 100% leaf scorch from a fire event (Gill, 1981a; Gill, 1981b; Gill, 1981c; Gill and Bradstock, 1992). The majority of fire-killed species rely on canopy or soil seed banks for persistence, with a minority arriving from ex situ wind born propagules (Lamont *et al.*, 1991; Lamont, 1991; Whelan, 1986). Species persisting via sprouting may regrow from protected or buried buds or a from a variety of underground storage organs (Bond and van Wilgen, 1996; Bond and Midgley, 2001). Vesk and Westoby (2004) found that this seeder/sprouter dichotomy was strongest at high disturbance intensities, but that the dichotomy was less marked as the intensity of a disturbance decreased.

In wet sclerophyll forests in northern NSW, Campbell and Clarke (2006) found that greater than 80% of woody understorey species were resprouters irrespective of fire intensity. However, within species that sprout after 100% leaf scorch, there can still be variation in sprouting response based on high fire intensities and their effects on buried buds. This response varies from crown and epicormic sprouting, to basal sprouting, to plant death, as fire intensity increases (Bellingham and Sparrow, 2000; Bond and Midgley, 2001; Moreira *et al.*, 2008). For this reason, Rowe (1983) proposed a broader biological classification of plant response to disturbance, including fire, within which it is possible for a species to have a different response in space and/or time, based on their life stage. The categorisation of the response of an individual plant to fire (and by sufficient observations, the typical response of a species), is therefore made complicated by a number of factors:

a) plant response can vary with fire intensity (Moreira *et al.*, 2008)
b) plant response can vary with life stage (age and/or height) such that younger or smaller individuals are more likely to be killed by moderate or high intensity fire (Hodgkinson, 1998; Rowe, 1983);
c) plant response can vary with interactions of frequency and season such that mortality is greater in two successive fires in certain seasons as compared to others (Noble, 1989);
d) plant response may vary along environmental gradients or in different geographic locations (Pausas *et al.*, 2004; Whelan *et al.*, 2002).

Characterising species as obligate seeders (which only recover from seedlings post-fire); obligate resprouters (which resprout post-fire, but recruit seedlings between fires) or facultative seeders (which resprout and also recruit some seedlings post-fire) (Keeley *et al.*, 2012; Pate *et al.*, 1990) can fail to capture the full potential range of a species' response to fire. A species' response across a wide range of fire intensities and environments is needed to fully assess its potential for and mode of recovery after a particular fire event (Vivian, 2010). This in part explains why there is considerable variation and inconsistency across plant fire response databases (Vivian *et al*. 2010). The 2003 fires provided an opportunity to document the full range plant species responses to a spectrum of fire severity across a broad range of vegetation types over a long period of time post-fire. Although species with a strong resprouting ability are likely to persist, there is a potential vulnerability for local extinction for fire-killed tree and shrub species if the time between successive fires is too short to enable recruitment, flowering and the establishment of a seed bank (Benson, 1985).

Variation if fire severity also influences vegetation structure. The structure of a plant community can be regarded as the spatial arrangement of vegetation layers defined by height and by the abundance of contributing species (Brown, 1991). Those species forming definable strata within plant communities (Specht, 1970) can be termed structural dominants, both in relation to their relative abundance and also to their influence on ecosystem processes (Avolio *et al.*, 2019). For the purposes of defining the structural response of communities in the study areas to high severity fires, the response of the dominant upper canopy species across the survey sites was recorded.

This chapter summarises the response to fire of vascular plant species and vegetation structure within the Brindabella and Burrinjuck study areas. The impacts of the 2003 fires on regionally significant plant species including those that are rare or threatened and species at their distributional limits, identified in Doherty (1998a; 1998b) are also assessed. These data are used to answer the first research question posed in Section 1.4: **What is the response to high severity fire of vascular plant species found in subalpine, montane and tablelands plant communities?**

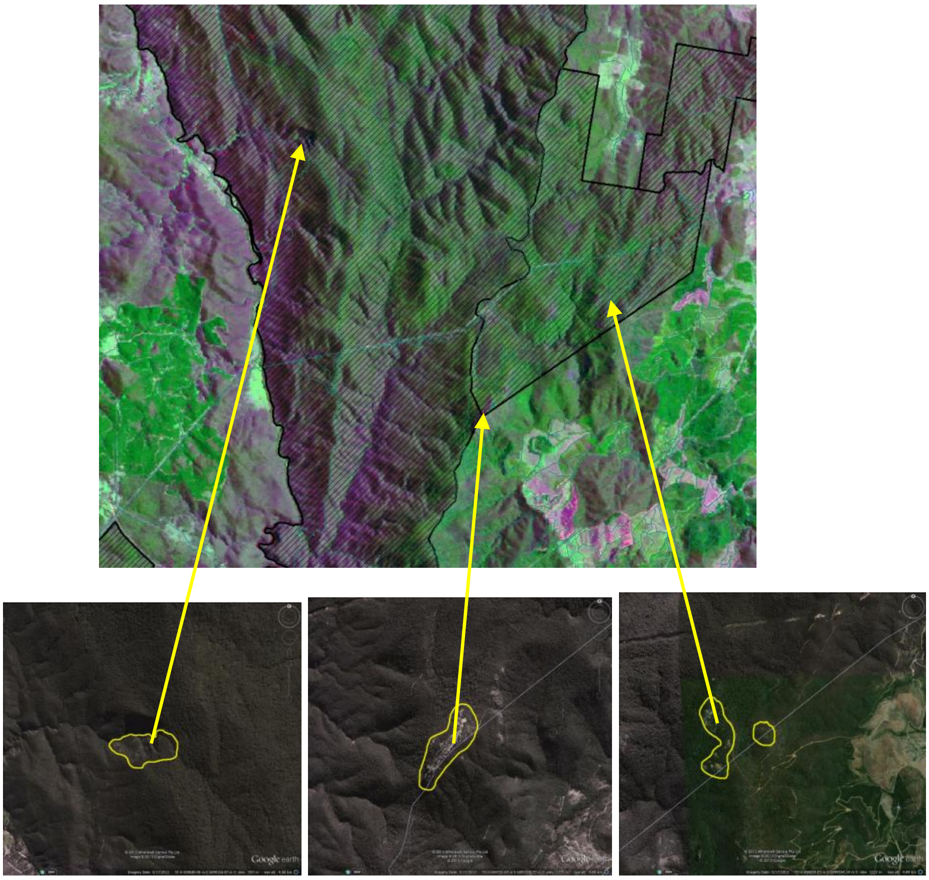
4.2 Methods

In the first spring after the 2003 fire, all but seven out of a potential 163 sites in the two study areas were resampled using the methods outlined in Section 3.3. For sampling from 2004 onwards, a subset of 52 sites were sampled until 2012. The responses of all vascular plant species in Brindabella and Burrinjuck based on the resampling data for the first two post-fire samples in 2003 and 2004 was recorded as part of the resampling after the fires. Additionally, targeted surveys were also undertaken after the 2003 fires outside of the monitoring sites in specific rocky outcrop habitats in the Brindabella study area – Waterfall Creek, Mt. Coree, Genges Trig – known to contain pre-fire populations of fire-killed plant species which are rare or threatened, uncommon, or of conservation significance (Figure 4.1). Observations on species responses were consolidated after two seasons post-fire (i.e. after the 2004 sample), to allow for any staggered germination or delayed resprouting responses that may have occurred.

Populations of *Leptospermum micromyrtus* on the west side of Mt. Coree (Figure 4.1) were surveyed in spring 2013, 10 years post-fire. The original pre-fire populations of *Leptospermum micromyrtus* were dominated by a small number of very large individuals which were all killed in the 2003 fires. Ring counts on selected cut-through stems indicated that the stand established in approximately 1950.

The fire response of the vascular plant species across the study areas was summarised using the Gill and Bradstock (1992) seven category system to categorise the response of reproductively mature plants to 100% leaf scorch (see Table 4.1). The use of this proportion standardises the response to the extreme end of fire intensity where plants will either die or resprout via a variety of mechanisms. Assessing mature plants ensures that any variation in susceptibility when plants are immature and either cannot resprout or do not have an in situ seed bank, is avoided. For fire-killed species in the study areas, this immaturity risk (Keeley and Fotheringham, 2000), was evaluated, with a particular focus on rare plant species and also for those species which are important structural dominants whose loss would lead to major structural and ecosystem changes (Avolio *et al.*, 2019; Fisher *et al.*, 2009; Grime and Hillier, 2000; Lunt *et al.*, 2012).

Because high severity fire effects the upper canopy structure, which has implications for subsequent understorey dynamics (Specht and Morgan, 1981), the fire responses of dominant canopy species were evaluated across the range of vegetation types to define broad structural response types.



Waterfall Creek **Mt. Coree** **Genges Trig**

Figure 4.1: Locations of reconnaissance surveys for fire-killed species on rocky outcrops in the Brindabella study area

4.3 Results

4.3.1 Recovery Mechanisms and Rates

There was a rapid initial recovery after the fires, regardless of fire severity. There was no loss of plant species from the study area irrespective of type of response and all plant species recorded pre-fire re-established post-fire, including species of conservation significance (i.e. rare species and species at their geographical northern distributional limits).

Responses were detailed for 441 vascular plant species across both study areas, comprising 362 native species and 79 exotic species (Appendix 3). Although additional species were recorded in sites over other years in subsequent samples, fire response data were restricted to the first two post-fire sampling observations where evidence was unequivocal in regard to a particular species' response to the 2003 fire event and response data were based on species responses on sites with at least 100% leaf scorch. Table 4.1 provides a summary of the plant species post-fire responses recorded in 2003 and 2004. Species can occur in more than one category due to multiple recovery mechanisms and therefore the response type tally is greater than the 441 species observations, with percentages calculated across 476 response type observations. Of the vascular plant species where a recovery mechanism could be ascertained, 23.7% of vascular plant species recovered via seed from seed banks, either stored in the canopy or in the soil (Categories 1 and 2) and 76.3% of vascular plant species recovered vegetatively after full canopy scorch via basal or epicormic resprouting or resprouting from underground storage organs (Categories 4, 5, 6 and 7).

Table 4.1: Vascular plant species fire responses to the 2003 fires, showing the number of species occurring in each of 7 categories

	Cat 1 **Fire-killed: Canopy Stored Seed**	**Cat 2** **Fire-killed: Soil Stored Seed**	**Cat 3** **Fire-killed: No Propagules**	**Cat 4** **Resprout: Root or Sucker**	**Cat 5** **Resprout: Basal Stem Buds or Lignotuber**	**Cat 6** **Resprout: Epicormic Shoots**	**Cat 7** **Regrowth: Aerial Buds**
Native	6	48	0	259	53	25	6
Exotic	1	60	0	18	0	0	0
TOTAL	**7**	**108**	**0**	**277**	**53**	**25**	**6**

In relation to the types of species occurring in each category in the study areas and the percentages of total response that they represent (Table 4.1), Category 1 plants represented 1.5% of the total and consisted of medium to large native shrubs, the fire-killed canopy dominant *Eucalyptus delegatensis* subsp. *delegatensis* and the exotic serotinous conifer *Pinus radiata.* Category 2 plants represented 22.7% of the total and were a mixture of exotic Mediterranean annual grasses and forbs, including the common weed species, *Conyza bonariensis* and *Cirsium vulgare,* and some small native annual species including *Ranunculus pumilio* subsp. *pumilio* and *Ranunculus sessiliflorus.* Category 3 had no species present either before or after the 2003 fires. Category 4 plants represented 58.2% of the total and were predominantly native grasses, lilies, orchids and many forbs, many of which were geophytes (sensu Raunkiaer (1934)). Category 5 plants represented 11.1% of the total and included 18 of the 19 eucalypts found in the study area as well as medium to large shrub species including the common dominant understorey shrub, *Daviesia mimosoides* subsp. *mimosoides.* Category 6 plants represented 5.2% of the total number of species and were primarily eucalypts, with some larger shrub species able to resprout via stems buds including two species only found in moist gullies, *Olearia argophylla* and *Pomaderris aspera*. Category 7 represented 1.3% of the total number of species and was entirely composed of ferns and the grasstree, *Xanthorrhoea glauca* subsp. *angustifolia*. Of the 19 species of eucalypts found across both study areas, all species readily resprouted by basal sprouting (Category 5) and also by epicormic resprouting (Category 6), except for *E. delegatensis* subsp. *delegatensis*, (Category 1) which has a canopy stored seed bank. All of the eucalypts also had some degree of seed germination in addition to epicormic and basal sprouting. In some instances, all three recovery mechanisms occurred in the same species on the same site, as for example in the Burrinjuck study area, where *Eucalyptus bicostata* recovered by all three mechanisms post-fire in 2003 on Site 4 (Figure 4.2).

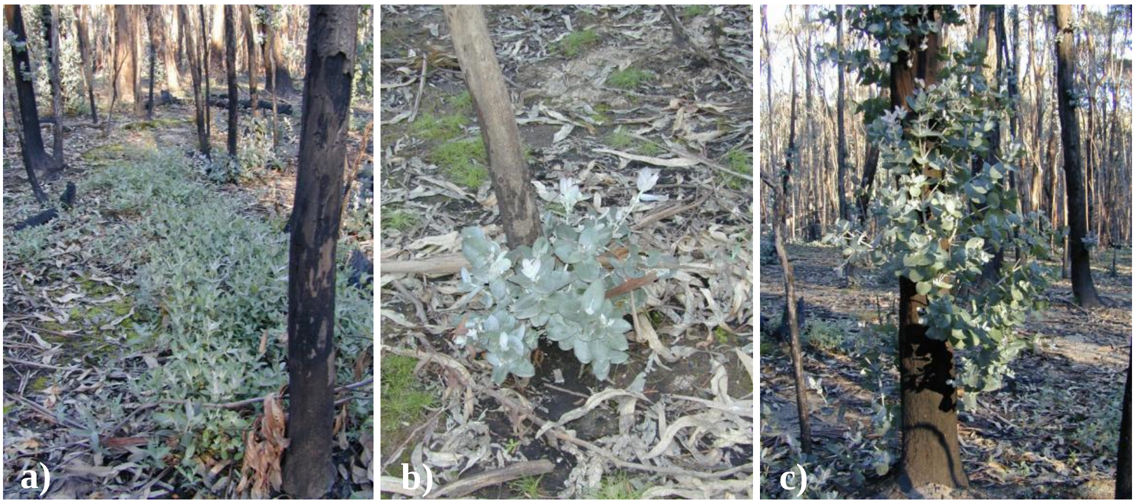


Figure 4.2: Multiple recovery mechanisms in *Eucalyptus bicostata* regenerating on Burrinjuck Site 4 in 2003: a) from seed, b) from basal sprouts and c) from epicormic sprouts.

Variation in fire response types over environmental gradients, between regions and particularly between different observers is common (Vivian *et al.*, 2010). However, in the two study areas, the type of response whether fire-killed or resprouting, was consistent within species over the sites sampled, even though some species exhibited multiple forms of resprouting (Figure 4.2). Nonetheless, there are known regional differences in the fire response of certain species. For example, although *Banksia marginata* in the Brindabella study area is fire-killed at 100% leaf scorch, populations further south in Namadgi National Park resprouted after 100% leaf scorch after the 2003 fires (Doherty, 2008). It is important to reiterate that the assessments in Table 4.1 were made for sites where species experienced 100% leaf scorch.

The growth response of plant species after the 2003 fires was extremely rapid in certain instances. In an initial reconnaissance of the northern Brindabella study area undertaken 1 month after the 2003 fires, species including *Eucalyptus fastigata* and the two ferns *Dicksonia antarctica* and *Blechnum nudum* were responding rapidly by resprouting from epicormic growth in the case of the former species, and from aerial buds in the case of the latter two species, in moist gullies where 100% leaf scorch had occurred (Figure 4.3).

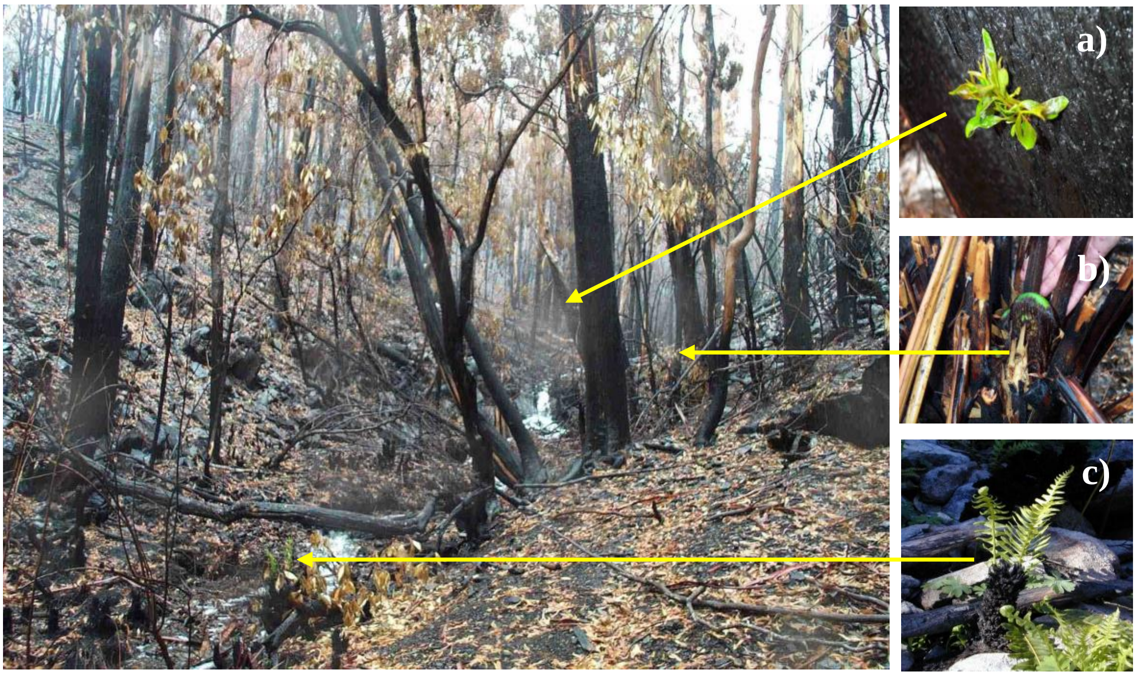


Figure 4.3 Moist gully dominated by *Eucalyptus fastigata* in the northern part of the Brindabella study area, one month post-fire in February 2003. Regenerating species are: a) *Eucalyptus fastigata,* b) *Dicksonia antarctica* and c) *Blechnum nudum*.

A second reconnaissance in June 2003 to the northern Brindabella study area six months post-fire showed a significant recovery already underway in the canopy eucalypts from epicormic resprouting (*Eucalyptus dalrympleana*, *Eucalyptus robertsonii*, *Eucalyptus fastigata*, *Eucalyptus dives*, *Eucalyptus pauciflora*) and from basal resprouting (*Eucalyptus pauciflora*, *Eucalyptus dives*, *Eucalyptus stellulata*). In the case of *Eucalyptus pauciflora* in particular, both forms of resprouting could be found within the same stand. The recovery of *Eucalyptus pauciflora* over the 10 years within which monitoring was undertaken saw very rapid recovery of stems, particularly from basal sprouts. On Brindabella site 91, a site which burnt at high severity on the 18th of January 2003, the pre-fire height of *Eucalyptus pauciflora* was 15 m. By November 2003, basal sprouting had already occurred and was approximately 40 cm high; by November 2012 three of these basal sprouts had become dominant and reached a height of 12 m, 80 % of the pre-fire canopy height (Figure 4.4).

Figure 4.4: An example of rapid recovery from basal sprouting in *Eucalyptus pauciflora* on Brindabella Site 91, north of Mt. Coree. Sprouts at: a) 1 year post-fire November 2003 and b) 9 years post-fire in November 2012

Although these examples represent the upper end of recovery rates, the growth rates of eucalypts were rapid overall across the sites sampled (Appendix 4). Additionally, because of the ability of eucalypts to recovery vigorously from epicormic or basal growth, tree mortality across the study area – notwithstanding the fire-killed stands of *Eucalyptus delegatensis* subsp. *delegatensis* – was uniformly low across the sites, with most sites having no mortality of eucalypts.

The 2003 fires provided an opportunity to establish a baseline response as to whether adequate recruitment was occurring to replace fire-killed adult populations. Of the obligate seeder component, there were some native post-fire increaser species (e.g. *Gynatrix pulchella*, *Ranunculus pumilio* subsp. *pumilio* and *Ranunculus sessiliflorus*) and many exotic, but essentially ephemeral, annual species (e.g. *Conyza bonariensis*, *Cirsium vulgare*). These species responded with high cover abundance values (CA 4-5) on the sites where they occurred in the first sample post-fire, but had short adult lifespans and, by the second sampling season post-fire, the cover abundance of these species had reduced considerably (CA 1 - 2). Fire-killed shrub and tree species which are of conservation significance or structural importance are discussed in greater detail in Section 4.3.4.

Overall and across all of the sites resampled, there was only one instance of what could be termed a recruitment failure of a dominant species. On Brindabella Site 68, the pre-fire understorey consisted of a layer of *Daviesia mimosoides* subsp. *mimosoides* to 2.5 in height with a cover of 75%. The site was burnt at low severity, with a scorch height of 4 m under a 30 m canopy of *Eucalyptus delegatensis* subsp. *delegatensis*. By the last sample in 2012, the *Daviesia* layer had barely begun to recover, with only 18 seedlings present. This compares with abundant regeneration from both basal sprouting and from seed germination across the rest of the sites where *Daviesia mimosoides* subsp. *mimosoides* was found, where cover abundance values had typically reached values of 25-50% in the second sample post-fire. Although the likely cause of this lack of regeneration was a fire intensity that was high enough to kill stems but not high enough to germinate the seed bank (Auld and O'Connell, 1991; Auld, 1996; Santana *et al.*, 2010), the site remains anomalous as all other sites in the study areas where a *Daviesia* layer was present, and which experienced a low severity fire, had adequate regeneration from resprouting and from the soil seed bank.

4.3.2 Structural Response Types

Structural response types based on the response of the dominant canopy species to high severity fire across the range of vegetation types were developed based on the response of mature individuals after the 2003 fires. The responses of smaller but mature sub-canopy layers of the same species are not considered in this structural summary as the responses of immature individuals such as saplings may differ from those of adults as, for a given fire intensity, their canopy will be closer to the ground and the stems smaller. Hence, basal or lignotuberous resprouting may occur, or plants may be killed, in small individuals of species that would otherwise be epicormic resprouters as adults (Noble, 1984).

Given these considerations, three major structural response types based on the post-fire response of mature canopy structural dominants to high severity fire were recognized in the study areas:

1) Epicormic Resprouting Dominants (Figure 4.5). This canopy response type occurred in wet and dry sclerophyll forests and grassy woodlands (vegetation types 2, 4, 5, 6, 8, 12, 15, 16, 17, 19). The structural dominants in this response type are: *Eucalyptus dalrympleana* subsp. *dalrympleana*, *Eucalyptus robertsonii*, *Eucalyptus fastigata*, *Eucalyptus bicostata*, *Eucalyptus bridgesiana* subsp. *bridgesiana*, *Eucalyptus goniocalyx*, *Eucalyptus nortonii*, *Eucalyptus melliodora*, *Eucalyptus polyanthemos* and *Eucalyptus macrorhyncha*.

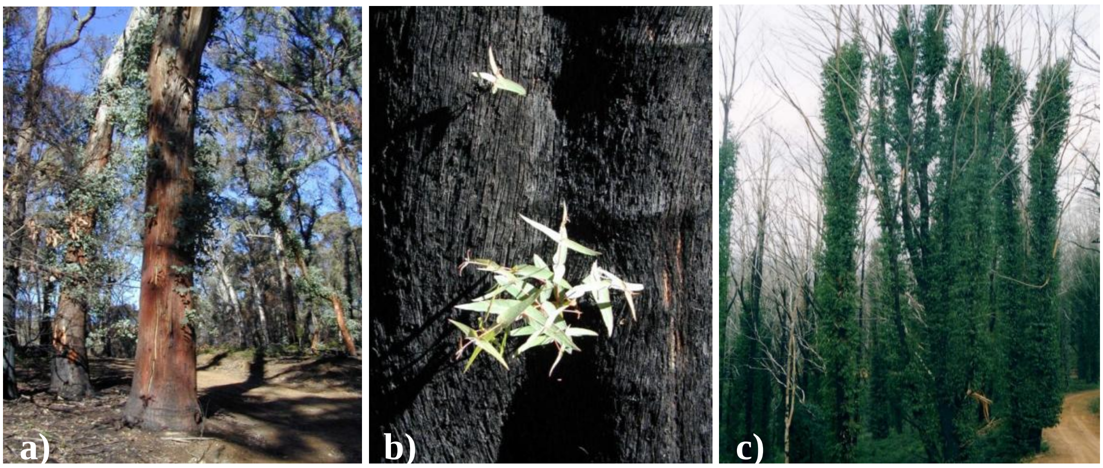


Figure 4.5: Examples of Epicormic resprouting canopy types. a) *Eucalyptus dalrympleana* subsp. *dalrympleana,* b) *Eucalyptus robertsonii* and c) *Eucalyptus fastigata.* Brindabella study area, near Mt. Coree

2) Basal Resprouting Dominants (Figure 4.6). This canopy response type occurred in wet and dry sclerophyll forests and subalpine woodlands (vegetation types 1, 2, 3, 9, 10, 11, 20, 21), and was a function of very high fire severity. That is, canopy species in this group resprouted via epicormics at sites experiencing moderate to high levels of severity including 100% leaf scorch, but reverted to basal resprouting at sites experiencing very high severity fire. Species in this group were susceptible to 'topkill' (Hoffmann *et al.*, 2009) where the upper parts of the trunk were killed but the basal tissue survives to resprout, often forming multi-stemmed clumps. The structural dominants in this response type were: *Eucalyptus pauciflora*, *Eucalyptus stellulata*, *Eucalyptus viminalis*, *Eucalyptus rubida* subsp. *rubida*, *Eucalyptus dives*, *Eucalyptus camphora* subsp. *humeana*, *Eucalyptus mannifera* subsp. *mannifera*, *Eucalyptus rossii*.

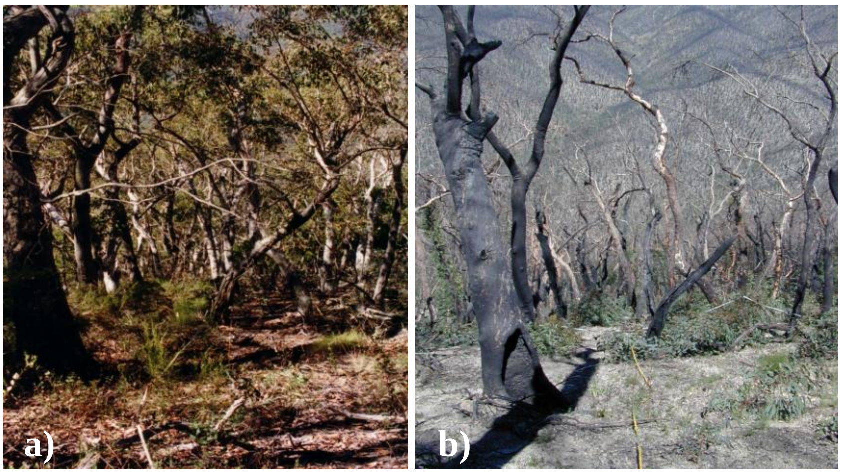


Figure 4.6: Examples of Basal resprouting canopy types. a) *Eucalyptus mannifera* subsp*. mannifera and Eucalyptus dives* pre-fire in 1997 and b) post-fire in 2003*.* Brindabella Site 94, above Flea Creek to the west of Mt. Coree

The differences between these two types of resprouting response were associated with a productivity and canopy height interaction. That is, the epicormic canopy response type was found in those species occurring in forest types on moderate to high productivity sites, with a canopy height of greater than 20 m. The basal sprouting response type was associated with either lower productivity sites or with subalpine sites, with canopy heights of less than 20 m. For a given fire intensity, severity was less in sites with a more elevated canopy as the canopy is further aware from the flame and scorch zone. Growth and recovery rates are less optimal in colder sites or sites with a lower productivity, with high severity fire in turn affecting site productivity (Raison, 1981).

3) Fire-killed Dominants (Figure 4.7). These canopy species were fire-killed and regeneration is by soil or canopy stored seed banks, and while the onset of structural recovery post-fire is rapid if adequate germination ensues, the time with which the canopy dominants take to reach reproductive maturity creates a vulnerability to a second fire which may lead to structural change. This canopy response type occurred in stands of montane heath dominated by *Leptospermum micromyrtus* (vegetation type 13, Brindabella) and by *Westringia eremicola – Calytrix tetragona* (vegetation type 18, Burrinjuck) and also in stands of tall wet sclerophyll forest dominated by *Eucalyptus delegatensis* subsp. *delegatensis* (vegetation type 7, Brindabella). *Eucalyptus delegatensis* subsp. *delegatensis* shows a clear demarcation in relation to the effects of intensity as it translates into fire severity. In adjacent sites burnt at the same time and at the same intensity in 2003 to the east of Mt. Coree, *Eucalyptus delegatensis* subsp. *delegatensis* was killed, but in adjacent stands of *Eucalyptus dalrympleana* subsp. *dalrympleana, Eucalyptus dives* and *Eucalyptus robertsonii* with the same slope, aspect and lithology, all individuals resprouted from epicormics. Additionally, individuals of *Eucalyptus dalrympleana* subsp. *dalrympleana* in the fire-killed stand of *Eucalyptus delegatensis* subsp. *delegatensis* recovered via epicormic regrowth.

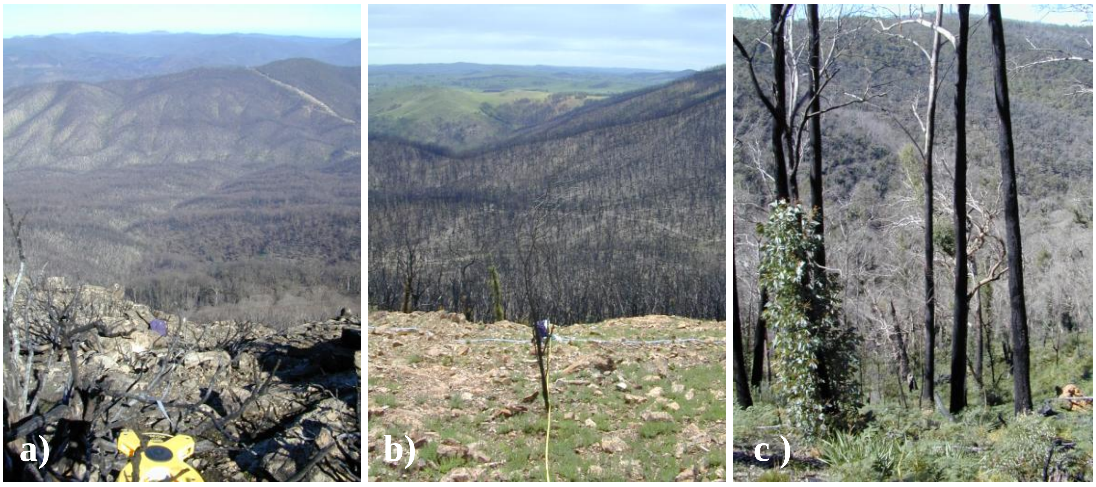


Figure 4.7: Examples of fire-killed seeder canopy types. a) *Leptospermum micromyrtus,* Brindabella Site 22, b) *Westringia eremicola – Calytrix tetragona* Burrinjuck Site 27 and c) *Eucalyptus delegatensis* subsp. *delegatensis* Brindabella Site 65. All photos are post-fire samples in 2003-2004

These three canopy dominant responses are only broadly mappable to vegetation types, as a 1:1 translation is complicated by the preponderance of eucalypts to form mixed stands (Austin *et al.*, 1983b; Duff *et al.*, 1983) where canopy response varies by species. As indicated above, stands of fire-killed *Eucalyptus delegatensis* subsp. *delegatensis* are rarely in pure stands and individuals of the epicormic resprouter *Eucalyptus dalrympleana* subsp. *dalrympleana* are often present (Figure 4.7), as well as occasional *Eucalyptus fastigata.* Similarly, *Eucalyptus dalrympleana* subsp. *dalrympleana* occurs as scattered individuals in subalpine woodlands dominated by basal sprouting *Eucalyptus pauciflora*. Because the dominant species in the third canopy structural response type are potentially vulnerable to short-interval fires, they form the focus of later research on reproductive maturity discussed in Section 4.3.4 and in Chapter 5.

4.3.3 Response of Exotic Species

The overall percentage of exotic species in the Brindabella area flora ranged between 7.8% pre-fire to 15.6% immediately post-fire, but by the 2012 survey had reduced to 8.4%. The percentage of exotic species in the Burrinjuck area flora ranged between 10.8% pre- fire to 21.3% immediately post-fire but by the 2012 survey was at 17.4%. Immediately post-fire in 2003, the proportion of exotic species observed approximately doubled in each study area as compared with pre-fire (Table 4.2), with significant increases in the proportion of exotics in Brindabella (Z Ratio= 3.3; P = 0.001) and in Burrinjuck (Z Ratio = 2.818; P = 0.0048). These increases were maintained into 2004 in Brindabella (Z Ratio= 2.155; P = 0.0312) and in Burrinjuck (Z Ratio = 2.468; P = 0.0136) but ultimately by the 2012 sample period, there was no statistically significant difference in the proportion of exotic species in Brindabella (Z Ratio = 0.298; P = 0.7657) or Burrinjuck (Z Ratio = 1.782; P = 0.0747) compared to pre-fire. However, between the two study areas, Burrinjuck maintained a significantly higher mean percentage of exotic species as compared with Brindabella (mean difference of 7.35 +/- 2.64; P<0.001) across the sample sites post-fire.

Table 4.2: Native and exotic vascular plant species by sample year and study area

Sample Year	Total Number of Vascular Plant Species		Number of Native Species		Number of Exotic Species		% Exotic	
	Brindabella	Burrinjuck	Brindabella	Burrinjuck	Brindabella	Burrinjuck	Brindabella	Burrinjuck
1996-1998	346	158	319	141	27	17	7.80	10.76
2003	423	290	357	228	66	62	15.60	21.38
2004	327	260	285	208	42	52	12.84	20.00
2005	317	248	292	205	25	43	7.89	17.34
2006	285	179	260	150	25	29	8.77	16.20
2007	311	231	272	183	39	48	12.54	20.78
2010	305	229	276	187	29	42	9.51	18.34
2012	308	207	282	171	26	36	8.44	17.39

The transient nature of the exotic species response in the study area can be seen using Burrinjuck Site 5 as an example. Post-fire in November 2003 the site was dominated by a large cover of exotic annual and biannual species and in particular, a high cover (75%) of *Carduus pycnocephalus* up to 1.5 m in height. By the second post-fire sample in November 2004, this large biomass of *Carduus* had collapsed and, although there was still a high cover of exotic annuals, the majority of the cover was turning over to native species such that by 2005, the majority cover on the site was native (Figure 4.8).

Figure 4.8: Changes in overall exotic cover on Burrinjuck Site 5 over the first three post-2003 samples. a) November 2003, b) November 2004 and c) November 2005.

4.3.4 Response of Fire-Killed Species

Native species which were recorded as being fire-killed in the study areas were assessed for short- to medium-term threats focussing on whether these species had germinated post-fire in known populations and whether they were flowering and setting seed during each subsequent post-fire resampling survey. Native annual herbs and forbs which had a rapid onset of flowering and fruiting immediately post-fire are not included in this threat assessment, although all species were recorded on each site when sampled as part of the ongoing site recording of species presence and abundance. Fire-killed species not occurring at sites were assessed at known pre-fire populations (Figure 4.1). In total, the post-fire response of 23 fire-killed species was assessed in relation to whether the species had a canopy seed bank or a soil seed bank, whether it was a structural dominant and whether it was rare in the study area i.e. restricted to 1 or 2 localised populations (Table 4.3).

Table 4.3: Characteristics of populations of fire-killed native tree and shrub species in the Brindabella and Burrinjuck study areas

Fire-Killed Native Species in Brindabella and Burrinjuck	**Vegetation Type**	**Recovery from Canopy Seed Bank**	**Recovery from Soil Seed Bank**	**Restricted Occurrence (i.e. Rare)**	**Species is a Structural Canopy Dominant**
Banksia marginata	5, 8	Yes			
Calytrix tetragona	13, 18		Yes		Yes
Dillwynia phylicoides	20, 21		Yes		
Dillwynia sericea	20		Yes		
Casuarina cunninghamiana	11	Yes			Yes
Daviesia mimosoides subsp. *acris*	13		Yes		
Dodonaea boroniifolia	18		Yes		
Eucalyptus delegatensis subsp. *delegatensis*	7	Yes			Yes
Grevillea oxyantha subsp. *oxyantha*	5, 13		Yes		
Grevillea victoriae subsp. *brindabella*	5, 13		Yes	Yes	
Gynatrix pulchella	10, 11		Yes		
Hovea asperifolia subsp *asperifolia*	6, 10		Yes		
Leionema lamprophyllum subsp. *obovatum*	13		Yes		
Leptospermum grandifolium	10, 11	Yes			
Leptospermum micromyrtus	13	Yes			Yes
Micrantheum hexandrum	13		Yes		
Ozothamnus stirlingii	7, 8		Yes		
Phebalium squamulosum subsp. *ozothamnoides*	13		Yes		
Pimelea treyvaudii	20		Yes		
Prostanthera lasianthos	10, 11		Yes		
Prostanthera rotundifolia	10, 11, 13		Yes		
Prostanthera sp. (Undescribed)	13		Yes	Yes	
Westringia eremicola	18		Yes		Yes

Of the fire-killed native shrub species, 52% are found on rocky outcrops and were associated with heath communities and some of these species also form the major structural dominants in these communities (e.g. *Leptospermum micromyrtus* on Mt. Coree in Brindabella National Park and *Calytrix tetragona* and *Westringia eremicola* in Burrinjuck Nature Reserve).

All non-rare and non-structurally dominant fire-killed species in the study area had recruited successfully in the 2003 and 2004 seasons post-fire and were all flowering and setting fruit by the spring of 2006. The legumes, *Dillwynia phylicoides, Dillwynia sericea*, *Daviesia mimosoides* subsp.

acris and *Hovea asperifolia* subsp. *asperifolia* in particular germinated *en masse* from soil stored seed banks. Although the serotinous seeders, *Banksia marginata* and *Leptospermum grandifolium*, both recruited successfully post-fire, they had yet to flower by the cessation of sampling in 2013.

The two species indicated as rare in the study areas (Table 4.2) also successfully germinated rapidly post-fire. *Grevillea victoriae* subsp. *brindabella* is a restricted species (Stajsic, 2010) and the northern limit of the species is on rocky outcrops at Genges Trig, northeast of Mt. Coree, with other populations at Waterfall Creek and further south along the Baldy Range (Figure 4.1). All of the populations in known stands in the northern part of the Brindabella study area were killed in the 2003 fires. Although prolific regeneration occurred in all fire-killed stands, the species was only recorded as flowering in 2013. Populations in the Genges Trig area in the northern Brindabella Range numbering in the hundreds were fully mature and flowering prolifically in that year (Figure 4.9). However, a detailed inspection of live plants and the soil surface in the stand produced only two fruits on plants evident from previous years flowering indicating that an adequate soil seed bank is unlikely to have been established at this point.

Figure 4.9: *Grevillea victoriae* subsp. *brindabella* a) habit and b) inflorescence in 2013, 10 years post-fire. Genges Trig

Prostanthera sp. (Undescribed) primarily occurs on rocky outrcops surrounding the upper portion of Waterfall Creek and also occurs on scatterred rocky outcrops to the south along Webb's Range as well as west of Genges Trig (Figure 4.1). Although no germination had occurred within one year of the 2003 fires, significant germination occurred in all populations by the second season in 2004. By 2010, all known populations had flowered prolifically and fruited and when surveyed in 2013, were also flowering and fruiting (Figure 4.10). Seed viability and soil seed bank studies have not been undertaken at this point, but the successful production of large amount of fruit suggests that the soil seed bank of this species was rapidly replenished post-fire.

Figure 4.10: *Prostanthera* sp. (Undescribed). Waterfall Creek

Of particular interest in relation to post-fire recovery were those species which were fire-killed but took some years to form a soil or canopy seed bank and which also form the dominant vegetation structure where they occur. Fire-killed species are vulnerable to what has been termed immaturity risk (Keeley and Fotheringham, 2000) where too frequent fire may lead to local extinction because sufficient time has not elapsed for the formation of adequate seed banks. Five species fall into this category in the study area: *Casuarina cunninghamiana*, *Eucalyptus delegatensis* subsp. *delegatensis* and *Leptospermum micromyrtus* in the Brindabella study area; and *Westringia eremicola* and *Calytrix tetragona* in the Burrinjuck study area.

Casuarina cunninghamiana occurs along the Goodradigbee River and lower parts of associated tributaries on the western edge of the Brindabella study area as well as in the Burrinjuck study area. Although a few individuals in the study areas were observed to weakly sprout epicormically after 100% leaf scorch, greater than 95% of those individuals experiencing 100% leaf scorch died both immediately after the 2003 fires or as a result of failed resprouting. This species had very sparse recruitment on Brindabella Site 72 where it was present pre-fire, with only two seedlings recorded post-fire. Although in theory a serotinous species, and although it has a winged seed enabling wide dispersal, it is reliant on recruitment after flood events rather than fire events (Benson and McDougall, 1995), so when killed by fire, there is not necessarily significant recruitment back onto site until seeds have been deposited in suitable sites among flood debris on river banks. Because *Casuarina cunninghamiana* is regionally common and many riverine populations in the region remained unburnt or were burnt at low intensity in 2003, populations of the species in the study areas were not regarded as being at risk through a second fire.

In relation to the other structural canopy dominants, regeneration was highly variable. In the Burrinjuck study area, *Westringia eremicola* germinated rapidly and by the end of 2004 had grown up to 25 cm in height, half the height of mature pre-fire individuals. In some instances where *Westringia eremicola* was growing in a more protected environment under *Eucalyptus dives*, the species had also flowered, indicating in this instance a primary juvenile period of less than 2 years. However, *Calytrix tetragona* was slow to re-establish and individuals did not flower until 2007 on the same site (Burrinjuck Site 27). In the case of these co-occurring structural dominants, it is probable that while the soil seed bank of *Westringia eremicola* was accruing rapidly, that of *Calytrix tetragona* was accumulating at a much slower rate. Because of this, these fire-killed heathlands were regarded as being at risk if a second fire occurs in the short term. In the Brindabella study area, the northern limit of *Leptospermum micromyrtus* occurs on rocky outcrops to the north-east of Devils Peak and to the west of Blue Range Road and on the western flanks of Mt. Coree. Ring counts taken from dead specimens after the 2003 fires on Mt. Coree indicate that the species is long lived and the stand on Mt. Coree may have arisen after stand replacing fires in 1952. All individuals in known stands in the study area were killed in the 2003 fires. Although sub-populations on outcrops to the northeast of Mt. Coree regenerated well and had many individuals with fruit, subpopulations on Mt. Coree were slow to recover with very few individuals having fruit (Figure 4.11).

Figure 4.11: Regeneration of *Leptospermum micromyrtus* on Mt. Coree in 2013

As the stems of these fire-killed individuals were still distinctive and readily counted, a ‘pre-fire’ count of the population on Mt. Coree was undertaken. Six random transects 50 m long and 2 m wide were completed parallel to the slope on the west side of Mt. Coree. Post-2003 germinants were counted within this same transect and whether germinants were flowering or fruiting was recorded (Table 4.4).

Table 4.4: Flowering and fruiting status of populations of *Leptospermum micromyrtus* on Mt. Coree, Brindabella study area

Transect No.	Live Individuals Without Fruit	Live Individuals With Fruit	Total Live Individuals
1	20	2 (2 with < 5 fruits/plant)	22
2	12	1 (1 with < 5 fruits/plant)	13
3	6	3 (2 with < 5 fruits/plant; 1 with > 60 fruits/plant)	9
4	6	0	6
5	34	2 (2 with < 5 fruits/plant)	36
6	55	4 (4 with < 5 fruits/plant)	59
TOTALS	**133**	**12**	**145**
Mean and S.D.	22 +/-19.2	2 +/-1.4	
%	92	8	

Although flowering and fruiting was noted in this population in 2006, by 2013, only 8% of *Leptospermum micromyrtus* recruits surveyed had any fruit and of these, only one individual had a large number of fruits. Although the mean number of recruits per transect was 22, there is a great deal of variation from transect to transect. Using a ratio approach and taking 180 pre-fire individuals (or 30 pre-fire individuals per transect) as a standard pre-fire number of individuals with which to compare, the replacement after the 2003 fires is less than 1:1, with 0.7 new individuals for every 1 pre-fire individual. The low number of recruits and the low numbers of individuals fruiting means that populations of *Leptospermum micromyrtus*, particularly on Mt. Coree, remained vulnerable to another fire in the short term. This species requires much longer fire free periods to persist as compared to the other fire-killed species in the study area.

Although not a rare species, *Eucalyptus delegatensis* subsp. *delegatensis* is fire-killed after full canopy scorch and is potentially vulnerable to either lack of recruitment post-fire or else the loss of post-fire germinated saplings if a second fire occurs before individuals reach sexual maturity, which has may take decades. Of the stands in the study area, including Namadgi National Park, approximately 30% of all stands suffered crown scorch and death from the 2003 fires. However, there was prolific overall recruitment of *Eucalyptus delegatensis* subsp. *delegatensis* in fire-killed stands regionally. In the fire-killed stands investigated, there was germination and recruitment by the end of 2003 and by early 2004 the most advanced saplings were over 1.5 m high and, despite continuing drought conditions, growth was rapid. Despite rapid regeneration and growth, populations of the species are potentially vulnerable to a second fire before flowering, fruiting and seed set. However, during the course of survey work post-fire, it was noted that in some stands, some 'precocious' individuals from the post-fire germination from 2003 were flowering and fruiting within 6 years of the fires. Fruit samples from such individuals were taken and the seed was tested for germinability (Doherty *et al.*, 2017a). The results of this study are detailed in Chapter 5.

4.4 Discussion

Because fire, and especially high severity fire, has been episodic and infrequent in much of the study area, rates of recovery post-fire for lower subalpine, montane and tablelands plant communities and for similar communities further south in the Australian Alps, have been until now poorly documented. The overall rapid and full recovery of the vascular plant species found in the Brindabella and Burrinjuck study areas is consistent with the results of other assessments undertaken in the Australian Alp in areas of treeless subalpine vegetation in Kosciuszko National Park after the 2003 fires which are recovering their pre-fire flora (McDougall *et al.*, 2015; Walsh and McDougall, 2004). Nonetheless, the fire-killed component of the flora which is reliant on the establishment of adequate canopy and soil seed banks is potentially more sensitive to fire intensity and fire frequency effects in the short term. Additionally, there has been some slow recovery in heathland dominated by fire-killed species and also in swamps in the study areas, but this matches recovery patterns for other heath and swamp communities in the Australian Alps which may take 10 to 15 years to achieve a pre-fire structure and floristic diversity (Wahren *et al.*, 2001). In the study areas, there were no instances where the overall initial impact of the fires has impeded plant recovery over the 10 years of monitoring undertaken. In contrast, alpine and sub-alpine bogs in particular are much more fire sensitive than any communities monitored in the study areas, and further south in Namadgi National Park and Kosciuszko National Park, intensive post-fire conservation management intervention has been necessary because of significant impacts from the 2003 fires on the *Sphagnum* cover (Clarke and Martin, 1999; Good, 2004).

The documentation of plant responses to fire is critical to managing biodiversity for both planned and unplanned fire events. Systems such as Gill and Nicholls' (1989) doubling of primary juvenile periods as a conservative buffer to manage fire frequency have been used to determine inter-fire intervals for management, but such guidelines can be regarded as signalling minimal times between consecutive fire events, rather than necessarily optimal times between fire events. This approach essentially underpins the use of maximum and minimum inter-fire intervals in fire and biodiversity management, particularly in NSW (Kenny *et al.*, 2003), the ACT (Kitchin, 2008) and Victoria (Chambers, 1977; Wouters *et al.*, 2002). The approach is based primarily on plant species responses to fire and utilises data on the primary juvenile periods of fire-killed species within particular vegetation types and builds upon the work of Benson (1985) and Gill and Nicholls (1989). Although these operationalised instances are based on the concept of the fire regime, they have to date primarily focussed on fire frequency, and have not addressed another critical aspect of the fire regime, fire intensity, or its measured effect, fire severity. Fire intensity is known to be a key factor in germination for leguminous shrubs with a soil seed bank and also for large trees with a canopy seed bank (Auld and O'Connell, 1991; Auld *et al.*, 2000; O'Dowd and Gill, 1984). Much of the area burnt in the 2003 fires burnt at moderate to very high severity ensuring that leguminous seed banks were stimulated to germinate across the study areas and significant recruitment occurred across a range of species in the Fabaceae and Mimosaceae.

Approaches used in post-fire vegetation monitoring primarily use time since fire as a surrogate for time since disturbance, but this effectively assumes that all fires have the same impact or effect, with fire frequency or fire return time assumed to be of primary importance. There can be markedly different structural responses to fires of different intensity, which are known to affect post-fire succession (Catling and Newsome, 1981; Fox, 1978; Fox and McKay, 1981; Fox and Fox, 1986; Newsome *et al.*, 1975). Habitat quality and quantity differ with severity within a fire event (Smucker *et al.*, 2005) and available resources for fauna will also differ significantly over time between lower intensity prescribed fire - planned fire - and higher intensity unplanned wildfire (Haslem *et al.*, 2011). Such differences in structure not only potentially affect plant community response, but also the post-fire response of fauna for which vegetation provides habitat. Monamy and Fox (2000) found that vegetation density determined small mammal succession, and not the time elapsed since disturbance, pointing to the potential differences arising in vegetation structural response depending whether a fire is low or high severity. The interplay between planned and unplanned fire requires adequate monitoring not only of time since fire but also fire severity, in relation to both individual plant responses and also to the responses of dominant canopy species. The knowledge of whether dominant species in the canopy are fire-killed or resprout in combination with fire severity mapping can better inform post-fire monitoring and give conservation managers more site level information as to recovery of an area with increasing time since fire. Knowledge of the variability of response to fire severity is critical, as although species such as *Eucalyptus delegatensis* subsp. *delegatensis* are fire-killed at 100% leaf scorch, they will survive a low intensity understorey fire, as was the case in most of Namadgi National Park south of the present study areas. It is important to consider that assessing plant species and community recovery in relation to time since fire, in the absence of knowledge of fire intensity or fire severity, can lead to misunderstandings of what is meant by a species being categorized as ‘fire-killed’.This is particularly the case in forest ecosystems where the state of ‘fire-killed’ canopy species, such as *Eucalyptus delegatensis* subsp. *delegatensis* is not adequately indicated by time since fire alone given its ability to withstand fires of low intensity and resultant severity. The response of resprouting species to high severity fire in the study areas was both rapid and vigorous, and where mortality has occurred in such species, it has been less than 10% of pre-fire standing eucalypts even in areas of intense crown fire (pers. obs.). However, such species are not immune to attrition if fire return times are too frequent. In other areas of the Australian Alps in Victoria, other large fires have occurred post-2003, particularly in 2006 and 2009, burning some areas that were still recovering from the previous recent fire events. This resulted in some areas being burnt twice or three times at moderate to high severity within 10 years (Bassett *et al.*, 2015). Under such conditions, re-sprouter species such as *Eucalyptus pauciflora*, while showing resilience to high severity disturbance, are not immune from the effects of frequent recurrent fire (Fairman *et al.*, 2016; Fairman *et al.*, 2018).

Major disturbances, particularly high severity fires where biomass is consumed and bare soil is exposed, have the potential to foster an increase in the richness and cover of exotic plant species-weeds – in the immediate post-fire environment (Fornwalt *et al.*, 2010; Johnston and Johnston, 2003). Although there is a paucity of post-fire monitoring of exotic species post-fire in Australia, research in the USA has focussed on whether there are opportunities for weeds to not only increase or become established but the degree to which they may hinder native plant species reestablishment post-fire (Fornwalt *et al.*, 2010). There appeared to be a very transient weed response in the study areas. Although there was an increase in weeds post-fire in both study areas, it was still proportional to the increase in native species post-fire and the numbers of species trend in a similar manner over time post-fire. In relation to differences in the response of exotic species between the two study areas overall, the pool of exotic species was greater in Burrinjuck than in Brindabella and is likely to be a reflection of the grazing and disturbance history of the Burrinjuck study area, which was grazed and logged up until the 1960s (NSW National Parks and Wildlife Service, 2009; NSW National Parks and Wildlife Service, 2010). The Burrinjuck study area also has many areas of flat and moderately fertile terrain proximal to grazing and agricultural activities, as compared to the Brindabella study area. Hence there is both a historical legacy and ongoing source of weed seed from surrounding areas to maintain a diversity of exotic species.

The high proportion of fire-killed species on rocky outcrops has parallels with many other such areas across Australia. Clarke (2002) found that rocky outcrops in the Sydney Basin, New England Tablelands, southwest Western Australia and Kakadu National Park all share similar characteristics, being ‘islands’ of fire-killed species in a ‘sea’ of resprouting species. Clarke proposed that these rocky outcrops are not fire refugia as such (Krawchuk *et al.*, 2020), but are areas that experience fire less frequently than surrounding landscapes. The rocky outcrops found in the Brindabella and Burrinjuck areas show similar characteristics with a high proportion of fire-killed plant species on rocky outcrops as compared to the surrounding forest matrix. The topographic position of these rocky outcrops, namely on high points in the landscape, means that when large landscape fires do occur and fire runs head upslope under extreme condition, these habitats tend to burn at high severity (Hammill and Bradstock, 2009). This was the case for both the Brindabella and Burrinjuck study areas.

In most vegetation types, a small number of species - the structural dominants - provide the bulk of cover or biomass at a given point in time. However, when fires of moderate to high intensity occur in vegetation, these structural dominants may take many years to recover their biomass and this provides an opportunity for other plant species to respond during that time and to germinate or resprout, flower and produce propagules (Purdie, 1977b). At the individual plant level, the ability of plant species to recover from fires of even high intensity gives the vegetation types that they constitute a level of resilience to disturbance (Holling, 1973; Walker, 1995). More broadly at the ecosystem level, the aggregated responses of what may have previously been uncommon plant species can provide ecosystem resilience to erosion and exotic species invasion. This ability of ecosystems to provide suites of species that respond in similar functional ways to disturbance has been termed 'ecological redundancy' (Walker, 1992) in that the more species and the wider the variety of plant responses present in an ecosystem, the more likely it is that the ecosystem can recover from disturbance and avoid degradation by functional substitution of one species for another, particularly in the short term. In the Brindabella study area, *Stellaria pungens* responded rapidly after the fires by resprouting and from soil seed banks and was flowering and fruiting within one year of the fires. This species ultimately provided extensive areas of high cover in the northern Brindabella Range and potentially reduced the amount of rain splash and erosion that would have occurred in the absence of a rapid recovery of plant cover. *Wurmbea biglandulosa* also responded in a similar way in the Burrinjuck study area and although more patchy in distribution than *Stellaria pungens*, nevertheless provided significant cover rapidly after the fires. Further rapid recovery of a variety of herbs and grass species in both study areas led to a high degree of soil stability in the immediate post-fire environment. Although rain events immediately after the 2003 fires led to significant erosion in steep areas of sedimentary and granitic geology to the west and south of the study areas, the rapid recovery of plants on the volcanic geology of the central and northern Brindabellas and in Burrinjuck maintained soil stability post-fire.

4.5 Conclusion

This Chapter set out to answer the question: **what is the response to high severity fire of vascular plant species found in subalpine, montane and tablelands plant communities?**

The overall conclusion is that all of the vascular plant species found across the subalpine, montane and tablelands plant communities in both study areas recovered rapidly via resprouting or from soil or canopy seed banks, irrespective of fire severity. The response to high severity fire of 441 vascular plant species was documented and the recovery of plant species of conservation significance was tracked over the 10 years that sites were monitored post-2003. This study has enabled a greater understanding of the underlying plant fire response traits that enable persistence of plant species and ultimately plant communities and fills in large knowledge gaps in relation to the fire ecology of non-alpine plants and plant communities in the Australian Alps.

There is nonetheless a need to understand not only the direct response of individuals and populations to fire, but also the aggregate dynamics and complex interplay between patterns of vascular plant species richness and plant community composition after high severity fire. The characterisation of plant response to high severity fire underpins the further interpretation of post-fire dynamics. In light of the information analysed on plant responses detailed in this chapter, the dynamics of species richness and community composition are explored in detail in Chapters 6 and 7 respectively.

This chapter highlighted how the majority of vascular plant species in the study areas are resprouters even after high severity fire and that of those species which are fire-killed, all germinate and most reach reproductive maturity rapidly, after even high severity fire. Because monitoring was undertaken for up to 9 years post-fire, plant fire responses were documented over a period of time that enabled confidence that any variation in resprouting or germination success would be observed. There still remains a question as to the long-term viability of population of fire-killed structurally important species in the study areas. However, due to the 9 year time frame of post-fire sampling, variation in the primary juvenile period of *Eucalyptus delegatensis* was observed which has important implications for its management. Chapter 5 explores an adaptation to fire that may enable the persistence of this extensively occurring, but fire-killed structural dominant, under short return intervals of high severity fires.

Chapter 5: Persistence of Fire-Killed Species: Seed Germinability in Early Maturing *Eucalyptus delegatensis* subsp. *delegatensis*

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5.1 Introduction

Eucalyptus delegatensis subsp. *delegatensis* is an important component of montane forest ecosystems in southeastern Australia, with high forestry and conservation values (Good, 1992; Hamilton and Cowley, 1991; Sebire and Fagg, 2009). Recent extensive landscape-scale fires in the Australian Alps in 1996, 2003, 2006–07, 2009 and 2013 affected many stands of *Eucalyptus delegatensis* subsp. *delegatensis* in ACT, NSW and Victoria, with 61% of the extent in NSW and 87% of the extent in Victoria being burnt (Bowman *et al.*, 2014; Wright and Robertson, 2014; Zylstra, 2006). Although fire severity was variable in these events, with some stands experiencing low intensity fire with little or no canopy scorch, large areas of mature trees experienced high intensity fire resulting in full canopy scorch or in some cases full leaf consumption, particularly in 2003 (Barrett, 2006).

Unlike the majority of eucalypts, which are strong epicormic or basal resprouters (Burrows, 2013; Gill, 1975), *Eucalyptus delegatensis* subsp. *delegatensis* and other related ‘ash group’ eucalypts including *Eucalyptus regnans* are killed after 100% canopy scorch (Ashton, 1976; Bowman and Kirkpatrick, 1986a). However, provided that the fire-killed trees are mature and contain an adequate canopy stored seed bank, a large quantity of seed is shed, resulting in ‘wheat-field’ regeneration in the first spring, with saplings growing rapidly in the post-fire environment (Bowman and Kirkpatrick, 1986b).

Understanding the range of variation in time to viable seed production for such fire-killed eucalypts – the primary juvenile period – is an essential requirement for managing species persistence and maintaining forest structure under altered fire regimes that are likely with global warming (Cary, 2002; Stephens *et al.*, 2013; Williams *et al.*, 2011). Fire-killed individuals and populations dependent on soil or canopy seed banks for regeneration are subject to ‘immaturity risk’ (Keeley and Fotheringham, 2000) or ‘interval-squeeze’ (Enright *et al.*, 2015), which occurs when, among other contributing aspects, fire frequency increases such that individuals cannot produce viable seed between recurring fire events to enable post-fire recruitment. Unmanaged interval-squeeze at landscape scales for *Eucalyptus delegatensis* subsp. *delegatensis* could result in loss of the species and forest structure over large areas with consequent loss or shift in the provision of ecosystem services including timber production, carbon storage and wildlife habitat (Bowman *et al.*, 2014; Colloff *et al.*, 2016; Lavorel *et al.*, 2015). Although the primary juvenile period of *Eucalyptus delegatensis* subsp. *delegatensis* is stated as being in the order of 20 years (Cremer *et al.*, 1984; Ellis and Lockett, 1991; Fagg *et al.*, 2013; Sebire and Fagg, 2009), this is essentially a management 'rule-of-thumb' rather than a definitive measure of actual time to maturity. It has been reported previously that individuals in some stands can flower and fruit at less than 10 years after a stand-replacing fire (Cremer *et al.*, 1984; Fielding, 1956; House, 1997) although the actual viability of these individuals are not documented in the literature. The reasons for early maturation are not clear, but it has been noted that early flowering occurs in suppressed individuals in even aged stands of *Eucalyptus regnans*, just before death (A.R. Griffin pers. comm., as quoted in House 1997, p. 45) and this may be the case with *Eucalyptus delegatensis* subsp. *delegatensis*. Takeno (2012) in a global survey of nonphotoperiodic flowering responses concluded that many cases of early flowering can be labelled 'stress-induced', but can result from a variety of causes. In the case of *Eucalyptus delegatensis* subsp. *delegatensis* if early flowering is indeed stress induced, it may be attributable to individuals being crowded by more vigorously growing neighbours with resultant low light intensity leading to stunted growth and early flowering.

Currently, there is a poorly documented gap in our knowledge as to the actual viability and hence potential adaptive importance of this early flowering and fruiting trait in *E. delegatensis* subsp. *delegatensis* and in the ash group eucalypts generally. In Victoria, local recruitment failure has occurred in some stands of *Eucalyptus delegatensis* subsp. *delegatensis* which have been burnt multiple times – up to three fires in 10 years – in the past decade (Bowman *et al.*, 2014) and in these cases, aerial seeding has been used to successfully re-establish stands (Bassett *et al.*, 2015). However, if large scale landscape fire events burn through considerable areas of immature *Eucalyptus delegatensis* subsp. *delegatensis* regeneration, broad scale reseeding may prove problematic due to a lack of seed supplies (Doherty *et al.*, 2017b; Ferguson, 2011). Hence, it is important to document and investigate populations with characteristics that may provide some buffering against increases in the frequency of high severity fire in *Eucalyptus delegatensis* subsp. *delegatensis* ecosystems, at least in some parts of the species range.

In 2010, early or 'precocious' (Bolotin, 1975) flowering and fruiting was observed in stands of post-2003 sapling *Eucalyptus delegatensis* subsp. *delegatensis* near Bull's Head in Namadgi National Park in the ACT, and also in similar aged stands near Smokers Trail in Namadgi National Park in 2013 (Figure 5.1). The presence of mature fruit observed in 2010 confirmed that at least some individual *Eucalyptus delegatensis* subsp. *delegatensis* saplings had flowered and fruited only 6 years after the stand replacing fires of 2003. Given bud initiation and formation in *Eucalyptus delegatensis* subsp. *delegatensis* (and *Eucalyptus regnans*) takes up to 2 years (Cunningham, 1957; Fielding, 1956), bud initiation in these individuals would have begun less than 4 years after germination from the 2003 fires. In 2013, opportunistic sampling of seeds was undertaken from the stand of *Eucalyptus delegatensis* subsp. *delegatensis* in the Smokers Trail area in order to investigate if seed in early-maturing individuals was viable, as a first step in a broader investigation of the distribution and abundance of the phenomenon.

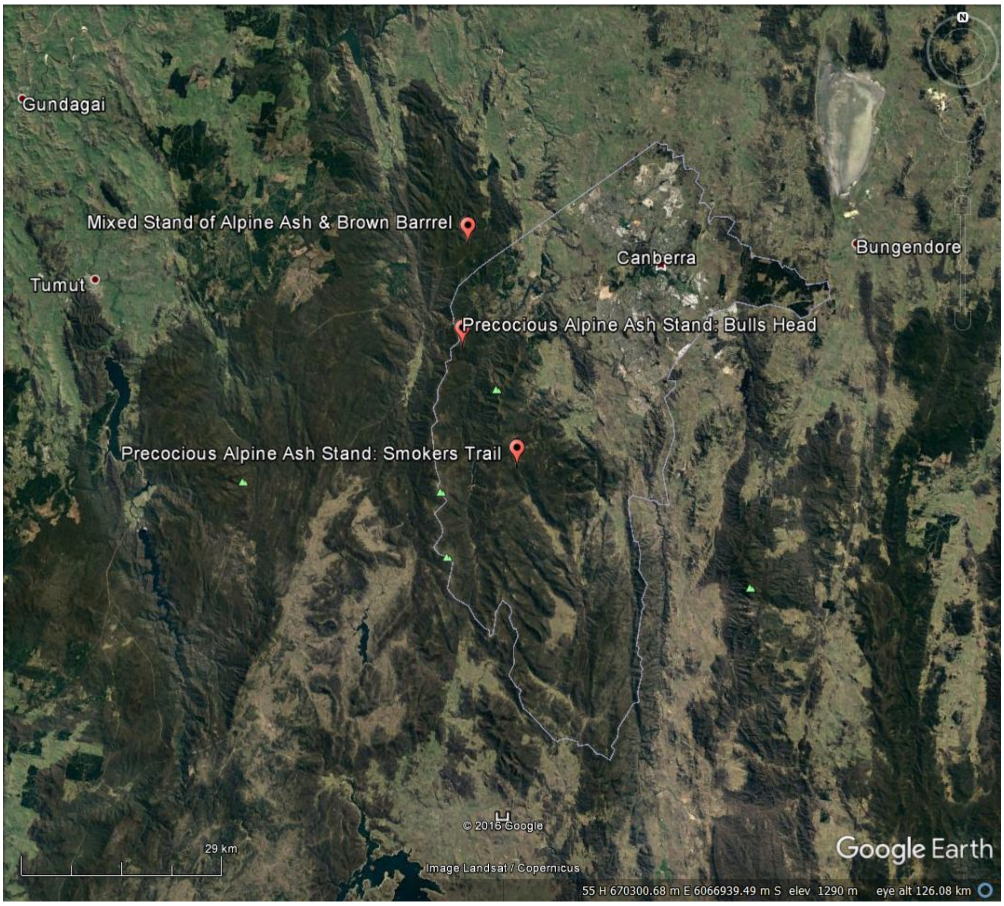

Figure 5.1: Locations of *Eucalyptus delegatensis* subsp. *delegatensis* stands mentioned in Chapter 5. Germinability of seeds from the Smokers Trail stand are those reported in the chapter

5.2 Methods

5.2.1 Field Collection

Branches bearing capsules from early maturing *Eucalyptus delegatensis* subsp. *delegatensis* individuals were sampled in March 2013 from a stand on the west side of Smokers Trail, south of the junction with Square Rock Track, in Namadgi National Park, ACT (site grid reference GDA94 Zone 55 Easting 672197; Northing 6064772). The stand is located in undulating terrain with a westerly aspect on granitic geology at an elevation of 1288 m, at the north-eastern edge of the species' distribution. Average annual rainfall in this area is ~900–1000 mm, which is considerably drier than areas further south in NSW and Victoria within the range of *Eucalyptus delegatensis* subsp. *delegatensis,* which can receive up to 1800 mm (Adomeit *et al.*, 1977). The closest Bureau of Meteorology weather station to the site, Honeysuckle Creek (elevation 1116 m), has a mean maximum temperature of 15.6^{o}C, a mean minimum temperature of 4.8^{o}C and a mean annual rainfall of 1009 mm. The area was burnt at high intensity in January 2003, resulting in tree death and subsequent ‘wheat- field’ regeneration from canopy-stored seed. Walking transects were undertaken to sample capsules and to note the characteristics of saplings that were exhibiting precocious budding, flowering and fruiting.

5.2.2 Seed Extraction

Given the difficulty in separating chaff from viable seed in eucalypts, combined chaff and seed mixes (hereafter called the ‘CS mix’) were used during the germination experiment. This approach and the other methods used in the current experiment are consistent with those used by Boland *et al*. (1980). The fruiting branches were taken back to the laboratory, a total of 139 capsules were cut from the branches, and all pedicels were removed from the capsules. The capsules were air-dried indoors at room temperature for 8 weeks on an open tray, with most chaff and seed falling from the capsules without intervention, and with remaining material shaken out by hand. Of the 139 capsules, 22 had recalcitrant valves and these were manually dissected to remove chaff and seed. One capsule was discarded due to insect damage but, overall, only six seeds from all of the seeds derived from the other 138 capsules showed any sign of insect damage. Impurities such as dirt and powdery frass were removed from the CS mix before further analysis. A total of 2.42 g of CS mix was extracted from the 138 capsules. The combined mix from all of the capsules was homogenised and then stratified for 8 weeks at 4^{0}C in a refrigerated cabinet, as previous work has demonstrated that cold stratification is required for germination in *Eucalyptus delegatensis* (Boland *et al.*, 1980; Grose, 1963).

5.2.3 Germination Testing

Germination testing was undertaken according to the method described by Boland *et al*. (1980) to facilitate comparison with known germination results for *Eucalyptus delegatensis* subsp. *delegatensis*. Therefore, like Boland *et al*., our test was based on 0.5 g samples of CS mix for germinability testing, multiplied up to a germinability estimate based on a 10 g CS mix. Four pseudo-replicates of 0.5 g from the pooled CS mix were placed on wet filter paper over moist vermiculite in covered Petri dishes in a Thermoline L+M TLMRIL396-1-SD growth cabinet. The cabinet was set at 20^{0}C during day and night, with cycles of 12 h of daylight and 12 h of darkness. The growth cabinet test ran for 7 weeks from September 2013. Petri dishes were checked for any germination weekly, with the first germination occurring at 3 weeks. Germinants were counted and then removed from the Petri dishes as the experiment progressed. Dishes were checked weekly until no further germination was noted after 6 weeks since commencement of the trial. The experiment ran for one further week but no further germination occurred.

5.3 Results

5.3.1 Field Observations

Given that eucalypt capsules take some months to mature and only persist unopened on the tree for 1–2 years (Sebire and Fagg, 2009), the sampled capsules had formed from a flowering event in late 2011 or 2012, some 8–9 years after the 2003 stand- replacing fire. Although the height of post-2003 saplings in the stand ranged from 2 to 10 m, saplings exhibiting early budding and fruiting were 5–6 m in height. That is, the largest and smallest individuals from the germination event after the 2003 fires showed no evidence of precocious flowering and fruiting; only the intermediate size saplings exhibited the phenomenon.

5.3.2 Seed Germination

The germination pattern was one of rapid emergence at 3 weeks with a total of 41 germinants counted at this time, followed by a steady decline to no new germinants over the succeeding 4 weeks. The cumulative number of germinants across all replicates reached 91 by week 6. Figure 5.2 shows values for mean germinants per week and mean cumulative number of germinants over time, using combined data from the four pseudo replicates. Overall, after 7 weeks in the growth cabinet, the mean number of germinants across the four pseudo replicates was 22.75 (s.d. = 6.95) per 0.5 g CS mix, equivalent to 455 (s.d. = 139) per 10 g of CS mix. The actual range in germinants across the four pseudo replicates was 15–30 per 0.5 g CS mix, equivalent to 300–600 germinants per 10 g CS mix.

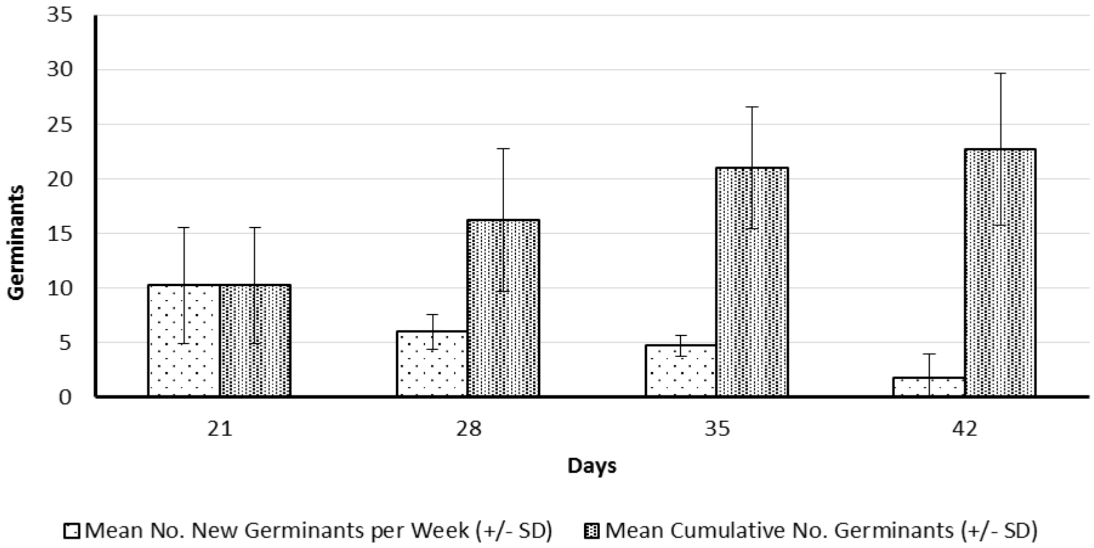

Figure 5.2: Mean numbers of *Eucalyptus delegatensis* subsp. *delegatensis* germinants over 7 weeks.

Adjusting for the amount of CS mix used in the current experiment to calculate a seed to capsule ratio (i.e. 2.42 g CS extracted from 138 capsules, but only 2.00 g used in the experiment), precocious *Eucalyptus delegatensis* subsp. *delegatensis* exhibits a germinability of 91 seeds per 114 capsules, or ~0.80 viable seeds per capsule, once again at the lower end of the germinability spectrum.

5.4 Discussion

The viability of seed from *Eucalyptus delegatensis* subsp. *delegatensis* found in this study was lower than that found by Boland *et al.* (1980), who quote a mean figure for *Eucalyptus delegatensis* of 1070 (s.d. = 550) per 10 g of CS mix. Although the germinability of the seed in our experiment is much lower than that quoted by Boland *et al*. (1980), precocious *Eucalyptus delegatensis* subsp. *delegatensis* do nonetheless produce some viable seed. It has proved difficult to compare the results with studies other than Boland *et al*. as other studies have used pure viable seed to test variation in response to environmental factors, rather than looking at absolute viability per se (Battaglia, 1993; Forestry Tasmania, 2010). However, using another metric, Forestry Tasmania (2010), quoting Grose (1957), state that the average number of viable seeds per capsule in *Eucalyptus delegatensis* subsp. *delegatensis* is 3.7, ranging from zero to seven.

5.4.1 Implications for Climate Change Adaptation and Fire Management

The findings have important implications for fire management, particularly under a changing climate. The potential catastrophic loss of large tracts of *Eucalyptus delegatensis* subsp. *delegatensis* in the Australian Alps from abrupt fire regime change is a significant issue in southeastern Australia and recent research has highlighted that although there needs to be a renewed focus on pre- and post-fire management, there are still gaps in our knowledge of the state and transition dynamics of *Eucalyptus delegatensis* subsp. *delegatensis* stands (Bowman *et al.*, 2013; Bowman *et al.*, 2014; Bowman *et al.*, 2015; Doherty *et al.*, 2017b). Current post-fire management assumes that *Eucalyptus delegatensis* subsp. *delegatensis* stands across their geographic range are vulnerable to local extinction after two high intensity fires in rapid succession, e.g. two or more high intensity fires in 20 years, due to either no, or an inadequate amount of viable canopy stored seed. It is now clear that at least at the northern extent of the range of *Eucalyptus delegatensis* subsp. *delegatensis* in the ACT and NSW, this 20 year primary juvenile period is not a binary ‘viable/ non-viable’ threshold. Viability is occurring at much younger ages than the notional primary juvenile period currently used to define fire management thresholds. There is therefore a potential for this response trait to provide a buffering mechanism against the loss of some *Eucalyptus delegatensis* subsp. *delegatensis* populations to multiple short-interval fire events in some parts of the species range. Although the amount of regeneration under these circumstances may not be adequate from a silvicultural point of view (Fagg *et al.*, 2013), it may be just adequate to allow the persistence of the species in less dense stands, satisfying a stocking standard suitable in a National Park context (Bassett *et al.*, 2015). Fire management of *Eucalyptus delegatensis* subsp. *delegatensis* stands does not need to be relaxed in light of early viability; rather, early viability may both explain the persistence of *Eucalyptus delegatensis* subsp. *delegatensis* in multiply burnt stands and also enable some management flexibility as regarding where and when to reseed stands.

Although this research has focussed on the phenomenon in the ACT, 36 000 ha (45%) of *Eucalyptus delegatensis* subsp. *delegatensis* stands in Kosciuszko National Park were fire-killed in 2003 (Wright and Robertson, 2014) and early flowering and fruiting has also been observed in several regenerating stands, with somewhat less than 5% of individuals flowering and fruiting up until 2013 (G Robertson, pers. comm.). Disturbance regimes and post-disturbance succession are an important influence on species evolution (Banks *et al.*, 2013; Pickett, 1976), and in particular, changing fire regimes are a strong influence on the evolution of plant fire adaptive traits (Pausas *et al.*, 2004; Pausas and Keeley, 2014). Nonetheless, persistence through a short period of multiple fire events may not translate to persistence in the face of major long-term changes in fire frequency. That is, although the early flowering trait may have assisted *Eucalyptus delegatensis* subsp. *delegatensis* in persisting in the past during random stochastic multiple high intensity fire events, a permanent change in fire frequency may be beyond the coping bounds of the trait (Fairman *et al.*, 2016).

Importantly, germinability as tested in the laboratory does not necessarily mean that successful establishment will occur in the field and it is at this point not known to what degree germination, if it occurs, leads to successful establishment, persistence, flowering, fruiting and seed set in the next generation. There are many factors that operate against germination and seedling establishment. For example, O'Dowd and Gill (1984) found that constant harvesting by ants removes a large proportion of *Eucalyptus delegatensis* subsp. *delegatensis* seeds that fall in unburnt stands, but that after fire, large amounts of seed falling from mature trees achieves predator satiation such that significant germination occurs. Much younger stands with much smaller seed crops may not reach this threshold and hence adequate germination may not occur from early maturing *Eucalyptus delegatensis* subsp. *delegatensis*, particularly as the seeds need some months of cold temperatures in situ before they germinate, exposing them to potential ant predation. Other major factors that may affect demographic dynamics include competitive effects from understorey vegetation (Cunningham and Cremer, 1965; Florentine and Westbrooke, 2004) and post-fire climatic conditions including drought (Fagg *et al.*, 2013; Kitajima and Fenner, 2000). However, successful sapling regeneration has been observed in small tree fall gaps within mature stands of alpine ash in the vicinity of Bulls Head in the absence of high intensity fire or mass seed release at the stand level (M. Doherty pers. obs.).

Independent of the early flowering trait to persist, the potential for hybridisation between *Eucalyptus delegatensis* subsp. *delegatensis* and other eucalypt species, such as *Eucalyptus fastigata*, has the potential to introduce other fire adaptive traits such that mixed hybrid stands of *Eucalyptus* *delegatensis* subsp. *delegatensis* may better persist under increased fire frequency. Hybridisation between *Eucalyptus regnans* and *Eucalyptus obliqua* (Ashton, 1981a) and also between *Eucalyptus regnans* and *Eucalyptus macrorhyncha* (Ashton and Sandiford, 1988) has been observed on ecotonal edges of that species range in Victoria. Similarly, hybridisation has been noted between *Eucalyptus delegatensis* subsp. *delegatensis* and *Eucalyptus pauciflora* in the northern extent of its range (Pryor, 1953). Pryor identifies areas in the upper Cotter River catchment of the ACT where post-disturbance hybrid swarms of eucalypts occur, particularly in ecotonal areas. Although introgressive hybridisation is often seen as a threat to, for example, endangered taxa, it can also be seen as a positive adaptive response to changing environmental pressures (Arnold, 2008). Hybridisation between taxa with different fire adaptive traits, as between fire-killed and resprouting eucalypts, raises possibilities for eucalypt adaptation under changing climatic and fire regimes, especially if the resulting *Eucalyptus delegatensis* subsp. *delegatensis* hybrids gained, for example, thicker bark (Pryor *et al.*, 1956). *Eucalyptus regnans*, despite being fire-killed at high intensity, can resprout weakly in the upper canopy if temperatures are sufficient only to kill leaves without penetrating the bark (Cremer, 1962), and it also has an epicormic structure, albeit generally non-responsive after fire due to a thin bark layer that provides inadequate protection from heat during all but the lowest intensity fires (Waters *et al.*, 2010). Weak resprouting of mature *Eucalyptus delegatensis* subsp. *delegatensis* has been noted in some stands in Bimberi Nature Reserve in 2017 after low intensity fire (M Gale, pers. comm.) and recent research observations in Tasmania has demonstrated that populations of *Eucalyptus delegatensis* subsp. *tasmaniensis* also weakly resprout in the upper canopy (Lunn *et al.*, 2018). Although the buds of *Eucalyptus delegatensis* subsp. *delegatensis* occur in single umbellasters as per most eucalypts, one specimen with occasional paired umbellasters was noted in the stand investigated. This could indicate possible hybridisation with brown barrel, which has paired umbellasters and which grows in moist forests in close proximity to stands of *Eucalyptus delegatensis* subsp. *delegatensis*, both species being in the ash group of eucalypts (Brooker and Kleinig, 2006; Pryor and Johnson, 1971). *Eucalyptus delegatensis* subsp. *delegatensis* and *Eucalyptus fastigata* barrel co-occur in mixed stands in the northern parts of the Brindabella range, near the northern limit of *Eucalyptus delegatensis* subsp. *delegatensis* (Doherty, 1997) and on the eastern edge of the species' distribution in Namadgi National Park (Vivian *et al.*, 2008), although no obvious hybrids have been noted in these stands. More commonly, where *Eucalyptus delegatensis* subsp. *delegatensis* grows in mixed stands, *Eucalyptus dalrympleana* subsp. *dalrympleana* is the co-dominant eucalypt species. Whether the structure and physiology of *Eucalyptus delegatensis* subsp. *delegatensis* and other related ash group eucalypts is similar to *Eucalyptus regnans* in relation to hybridisation is an open research question.

5.4.2 Further Research

There is a window of opportunity for future targeted sampling in NSW, the ACT and Victoria within stands burnt in 2003 and in subsequent fire events to document the spatial extent of the phenomenon of early maturation in *Eucalyptus delegatensis* subsp. *delegatensis*. In particular, *Eucalyptus delegatensis* subsp. *delegatensis* stands burnt by high intensity fires in the Victorian Alps over the past 20 years provide a mosaic of recent fire histories within which to further examine the phenomenon. To date, several twice and thrice burnt *Eucalyptus delegatensis* subsp. *delegatensis* stands in Victoria have been artificially reseeded in order to enable regeneration (Bassett *et al.*, 2015; Fagg *et al.*, 2013). A comparison of the fate of any similar but untreated areas will be useful in order to gauge if any recruitment is occurring from early flowering individuals in non-reseeded stands. Further detailed examination of the phenomenon would require a stratified sampling approach based on samples across the geographic range of the species to cover major environmental gradients plus further stratification based on lithology, slope, aspect, time since fire and fire severity. If early maturing in *Eucalyptus delegatensis* subsp. *delegatensis* is more widespread, it may provide buffering for at least some stands potentially affected by increased fire frequency; if the phenomena is more localised, seed from stands exhibiting early maturation could be used to trial seeding into more vulnerable stands to augment their resilience to future, frequent fire as a ‘planned adaptation’ action (Doherty *et al.*, 2017b).

The present study raises several questions requiring further research before such management interventions might be undertaken:

- *Are there any specific stand structural characteristics associated with early maturing trees?* Although it was observed that intermediate sized individuals exhibited early flowering in the stand investigated, a more detailed analysis of stem density, stem size, stem height and stand age in relation to early flowering in the Alps is an essential next step in understanding relationships between time since fire and maturation.
- *How widespread is the phenomenon of ‘early-maturing’ in Eucalyptus delegatensis* subsp. *delegatensis?* There is no published research indicating any systematic survey of *Eucalyptus delegatensis* subsp. *delegatensis* stands after the 2003 fires to document the phenomenon, and there are no reports or observations of the phenomenon from Victoria to date. Particularly given the greater number of recent high intensity fires in the Victorian Alps, there are numerous stands that will have a variety of recent high intensity fires within which to observe if precocious flowering and fruiting is occurring.
- *Is there a relationship between environment and/or fire history and the phenomenon of ‘early-maturing’ in Eucalyptus delegatensis* subsp. *delegatensis?* That is, is the phenomenon more common at the distributional limits of the species compared with elsewhere?

- *To what degree is the response is environmentally and/or genetically determined?* Competition from faster growing individuals and consequent suppression of slower growing individuals could lead to early flowering and fruiting simply due to chance local variations in environment where all individuals have the capacity to flower and fruit early if suppressed. It could also be the case that there is a proportion of the population that is inherently slow growing and prone to early flowering and fruiting and will always form a separate subpopulation within a stand.

5.5 Conclusion

In the absence of detailed data or observations, current models of the dynamics of *Eucalyptus delegatensis* subsp. *delegatensis* forests still assume a 20 year primary juvenile period (Bowman *et al.*, 2016). A more detailed understanding of how the species' post-fire ecology varies over its geographic range, especially in NSW and the ACT, may enable greater flexibility when managing the species under frequent high intensity fire scenarios and predicting range extensions or contractions. Local variations in primary juvenile period may interact with, and be the product of, variations along broad climatic gradients as well as fine scale local environmental and topographic influences, which can dramatically alter predicted persistence under climate change (Austin and van Niel, 2011). A detailed understanding of how potentially vulnerable species' ecology and response traits vary geographically will enable sounder predictions of changes to species distributions.

Chapter 6 investigates patterns of plant species richness after fire in relation to fire severity.

Chapter 6: Fire Severity and Vascular Plant Species Richness

6.1 Introduction

As summarised in Chapter 4, the majority of vascular plant species in the study area are able to resprout after high intensity fire, and for those species that are fire-killed, germination from canopy and soil stored seed banks has resulted in continued persistence post-fire. Species richness of vascular plants after fire is determined by such in situ resprouting and germination from seed banks and there is also the potential for supplementation by colonisation from dispersal of propagules from nearby unburnt areas (Whelan, 1986). In addition, plant species not previously recorded prior to a fire may emerge from the soil seed bank and, in the absence of any loss of pre-fire plant species, germination or colonisation of previously unrecorded species has the potential to increase species richness, for a varying period post-fire (Doyle *et al.*, 1998; Whelan, 1995). Increases in species richness post-fire can be maintained in the short term, particularly after high severity fire, where canopy dominance of high cover species is temporarily reduced and there is greater access to nutrients, light and water (Myerscough *et al.*, 1996; Specht and Specht, 1989a; Specht and Specht, 1989b; Williams and Clarke, 1997). The effects of heat and smoke can also stimulate germination of recalcitrant species which may not otherwise germinate (Clarke and French, 2005; Thomas *et al.*, 2007; Wills and Read, 2002). Fire effectively creates ‘safe sites’ (Harper, 1977), where species can germinate and establish in zones where conditions would otherwise be unsuitable. The availability of such safe sites varies over both space and time for particular species [‘spatial dispersers’ and ‘temporal dispersers’ (Keeley and van Mantgem, 2008)] and fire events provide an opportunity for species in both functional types to establish. Barnes *et al*. (2006) modelled self-thinning and patchiness as they affect species persistence and suggest that disturbance and senescence provide the opportunity for regeneration of inferior competitors.

At the broadest level, changes in species richness can be used to represent changes in underlying environmental gradients or successional states, and the intermediate disturbance hypothesis (IDH) postulates that diversity will be greatest at intermediate levels of disturbance (i.e. intermediate disturbance frequency; intermediate time after a disturbance or intermediate size of a disturbance) (Connell and Slatyer, 1977; Connell, 1978; Connell, 1979). However, testing the hypothesis using a variety of field data and modelling approaches has resulted in conflicting interpretations, and the definition of what constitutes an intermediate disturbance remains problematic. Although tested in theoretical contexts (Johst and Huth, 2005; Roxburgh *et al.*, 2004; Shea *et al.*, 2004), it has not been adequately tested empirically. The IDH has been less frequently used in the context of intermediate intensities or severities of disturbance (Li *et al.*, 2004; Woods, 2004) but in relation to fire in particular, it may not be readily applicable, due to complexities involving the effects of scale (Schwilk *et al.*, 1997). The IDH does not explicitly deal with scale, and it is known that scale affects immigration and recolonisation after disturbances such as fire, with differences between the dynamics of small patch disturbance as compared with large landscape disturbance (Barnes *et al.*, 2006; Pimm, 1984; Roxburgh *et al.*, 2004). There are also interactions between area and frequency of disturbance, which can lead to different patterns of diversity across scales (Svensson *et al.*, 2009).

Interactions between disturbance size and frequency are considered important factors from a patch dynamics and recolonisation perspective for those species which cannot persist on site after fire (Gill and Bradstock, 2003; Miller, 1982). However, when species can persist on site either via resprouting (Bellingham and Sparrow, 2000) or via a canopy or soil seed bank (Thompson, 2000), then intensity may be the critical factor affecting patterns of species persistence and species richness, in which case Egler's (1954) initial floristics composition model may apply (Malanson, 1984). Egler suggested that secondary succession did not involve facilitation of one species by another, but was in fact simply a result of differential growth of species already present on a site. Work undertaken in dry sclerophyll forest types in the Canberra region (Purdie and Slatyer, 1976; Purdie, 1977b; Purdie, 1977a) has shown an initial floristic composition type response (Egler, 1954; Wilson *et al.*, 1992) for fires of low severity in drier woodland and open forest communities in the Canberra region, but responses to fires of higher severity are lacking. Capitanio and Carcaillet (2008) assessed the applicability of the initial floristic composition model to post-fire vegetation dynamics in Mediterranean plant communities and found that all species present in later stages were present at the beginning of the succession and it was the slow growth rates of the dominant species that allowed dynamics to occur in the short term. Although they assert that according to the IDH species richness should not be highest immediately after disturbance, as is the case with an initial floristic model, this is not necessarily the case as richness is maintained by the former model by intermediate frequency and potentially intensity of disturbance, not necessarily at intermediate times after a particular disturbance event. Whether richness peaks one, two, three or more seasons after a disturbance may not be as critical as the fact that there is an increase in species richness at all after a disturbance.

The number of species that are present at a site at a point in time is a subset of the total species pool that may be available, as the conditions for emergence from dormant storage organs or from seed banks, as well as the persistence of species at a site, varies across species and over time (Murdoch and Ellis, 2000; Merritt *et al.*, 2007; Ooi, 2010). Gill (1999) has termed the pool of plants visible above ground at a point in time at a site 'on-site apparent richness' (Figure 6.1). This visible species pool changes over time with previously unrecorded species emerging from the soil seed bank post-fire ('on-site in-soil only richness') or arriving and colonising from off-site post-fire ('off-site dispersal-reach richness). Differences over time in on-site apparent richness will influence the species present that contribute to fuel dynamics, with some species producing more or less, or different types, of fuel and therefore influencing fire regimes over time which in turn can affect species richness and composition.

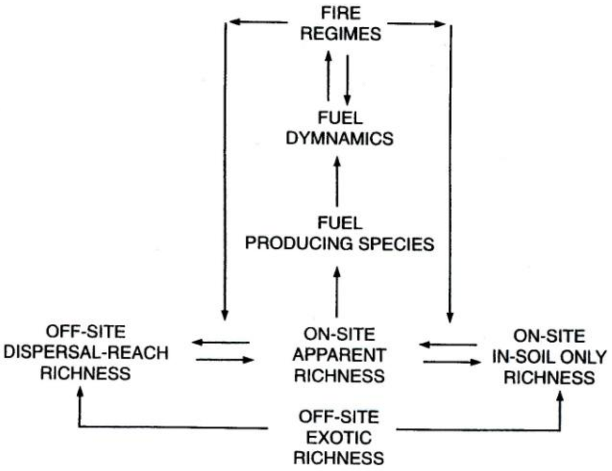

Figure 6.1: Terminology for and relationships between various types of plant species richness in relation to fire (Diagram from Gill 1999)

Vascular plant richness on a site is composed of both visible (‘apparent’) richness and also soil seed bank (‘unapparent’) richness (Gill 1999). When fire acts as a disturbance agent, there are post-fire opportunities for changes in species richness and patterns over time. A decrease in species richness would be possible under some circumstances where a lack of regeneration has led to a loss of species. However, given the known ability of plant species to recover post-fire via resprouting or from seedbanks, the most likely pattern of response is either recovery to a similar level of pre-fire species richness or an increase in species richness. If there is a post-fire change in species richness, the question arises as to how long the effect is maintained. Gill (1999) suggests five possible post-fire patterns in species richness (Figure 6.2a-6.2e). All of these curves start at a notional state of 0 for on-site apparent species richness immediately post-fire, which can be taken as a state of 100% leaf scorch as a result of high intensity fire conditions where no green material remains (Gill, 1975; Gill and Bradstock, 1992). However, as outlined in Chapter 4, rapid post-fire recovery means that post-fire on-site apparent species richness (i.e. visible photosynthetic plants) is only ‘absent’ (i.e. unobservable) for a very short time.

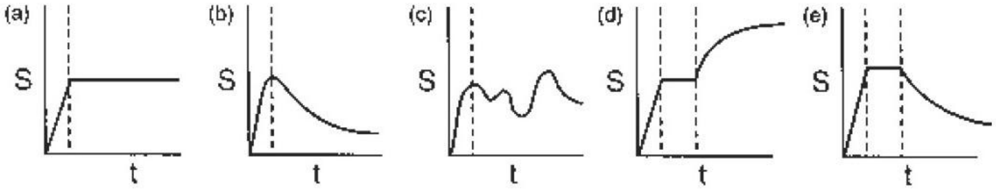

Figure 6.2: Potential response curves of species richness over time post-fire. S is species richness and t is time. (Diagram from Gill 1999).

Although idealised, the curves represent plausible patterns of change over time that can be related to a variety of scenarios involving plant competition, especially interaction between plant forms and plant lifespans (Specht and Specht, 1989a; Specht and Specht, 1989b). All models assume a rapid return of on-site apparent species richness but differ in how this metric subsequently changes over time. The pattern represented in Figure 6.2a represents a null model with recovery to pre-fire numbers and no change over time; Figure 6.2b represents a gradual decrease to a low stable number after initial recovery; Figure 6.2c represents fluctuations over time from the appearance and disappearance of species; Figure 6.2d represents a period of stability followed by a gradual increase due to immigration and colonisation that reaches a high stable number; and Figure 6.2e is similar to Figure 6.2ba but has a period of stability followed by a gradual decline to a low stable number. Long term pre- and post-fire data enables an evaluation of these conceptual models.

Research undertaken in coniferous forests in the USA has confirmed both the inhibitory effect of canopy on understorey as well as the increase in species richness that follows fire events, in the context of an initial floristic composition framework. Doyle *et al*. (1998) found that species richness was greater in severely burned stands and also that species important during the first 17 years of succession emerged in the first year post-fire and were present pre-fire. Laughlin and Grace (2006) found that old growth *Pinus ponderosa* in montane ecosystems in SW USA had an inhibitory effect on the abundance of understorey plants leading to lower understorey richness, which increased after fire. For Australian heathlands, shrublands and some coastal forests there are known patterns of species richness over time in relation to the temporary reduction in canopy dominance after fire, with an initial increase in vascular plant species richness immediately post-high severity fire, followed by a gradual reduction in species richness with increasing time since fire (Cohn *et al.*, 2002; Freestone *et al.*, 2015; Hobbs and Atkins, 1990; Keith and Bradstock, 1994; Penman *et al.*, 2008; Russell and Parsons, 1978; Specht *et al.,* 1958; Specht, 1981c). However, the post-fire species richness response of subalpine, montane and tablelands forest ecosystems in Australia is poorly documented and hence the 2003 fires in the Australian Alps provide a rare opportunity to study species richness responses post-fire in ecosystems that only burn infrequently at high severity. The data set accumulated from pre-fire sampling and subsequent post-fire resampling of the Brindabella and Burrinjuck sites provides measures of vascular plant species flux over time. This includes both pre- and post-fire richness data in relation to structural and cover values which were measured concurrently, as well as in relation to variation in fire severity on sites within and between vegetation types.

In this chapter, I test the first hypothesis, as outlined in Chapter 1.4, in relation to my second research question: **Does high severity fire result in significant or permanent change to *vascular plant species richness* and plant community composition as compared to low severity fire in subalpine, montane and tablelands plant communities?**

Hypothesis 1*: High severity fire is associated with a greater increase in plant species richness within plant communities post-fire as compared to low severity fire and the duration of this effect is positively associated with fire severity.*

6.2 Methods

6.2.1 Pre- and Post-fire Sampling

The number of vascular plant species was recorded at each survey site in 1996/1997 (Brindabella) and 1998 (Burrinjuck) before fire, and in 2003, 2004, 2005, 2006, 2007, 2010 and 2012 after the 2003 fire that burned the same sites. Species numbers were tabulated for 130 Brindabella and 33 Burrinjuck full floristic sites, but because seven of the Brindabella sites were not resampled post-fire in 2003, these were omitted from analyses resulting in data for 123 sites. All native and exotic species were recorded on each site. A full listing of vascular plant species for the two study areas can be found in Appendix 1. To investigate pre- and first year post-fire changes, the 1996/1997 (Brindabella) and 1998 (Burrinjuck) pre-fire data and first post-fire sample in 2003 data were used. For further investigation of changes over time beyond the 2003 sample, the pre-fire and first year post-fire data were used with the addition of the long-term sub-set site data from the 39 sites in Brindabella and 13 in Burrinjuck, which were sampled from 2004 and 2012, beyond their initial post-fire sampling in 2003 giving a total of 52 sites.

6.2.2 Data Exploration

To investigate whether species richness was significantly different pre- and post-fire and whether there was a relationship between richness and fire severity, paired t-tests were undertaken in Genstat (Version 17) based on the pooled data across sites and vegetation types for the 1996/1997/1998 pre-fire samples and the first post-fire samples in 2003. The tests were between the mean pre-fire richness across all sites combined, by study area, and the mean post-fire richness across all sites for the first post-fire sample in 2003, by study area. Two-sample t-tests (paired) were used to test the following null hypotheses (sub-hypotheses of Hypothesis 1):

$H0^{1a}$: mean first post-fire sample species richness = mean pre-fire sample species richness

$H0^{1b}$: mean difference between first post-fire sample and pre-fire sample species richness Low Severity = mean difference between first post-fire sample and pre-fire sample species richness High Severity

Subsequently, potential relationships between species richness and environmental variables were explored in the form of box plots and correlation matrices to investigate general patterns of species richness across vegetation types and over time and to explore relationships among a range of structural and environmental variables influential in determining apparent species richness. This assisted in identifying variables and types of relationships to be further explored with statistical modelling.

6.2.3 Statistical Modelling

The nature of relationships among species richness and fire severity, and variables measuring vegetation structure and vegetation cover associated with the dataset, as well as rainfall, was explored using Generalised Linear Mixed Models (GLMMs) in Genstat (Version 17). GLMMs are an extension of Generalised Linear Models allowing for both fixed (e.g. rainfall) and random effects (e.g. site) in the model and are particularly useful for analysing non-normal repeated measures data, such as the data sets for pre- and post-fire sampling at Brindabella and Burrinjuck (Harrison *et al.*, 2018).

The vascular plant species richness data used in the analyses consists of count data and the GLMMs were structured with a Poisson distribution and a logarithm link function, and with Site used as the random model (Lee and Nelder, 1996; Lee and Nelder, 2001). The fixed models were then constructed incorporating the candidate variables with the Wald Statistic used for testing significance and also in the selection of model terms (Bolker *et al.*, 2008). In the model runs, all candidate variables were initially tested and those terms where the Wald Statistic was not significant were dropped from the model, and the model was rerun with the remaining terms. This process was iterated until a final model was produced where all of the remaining terms were significant at a P level of at least 0.05. Although for significance testing of final models the dispersion parameter should ideally approximate 1, where there is over- or under-dispersion, the Poisson GLMM module in Genstat takes account of this when the option for estimating the dispersion parameter is chosen, and an automatic adjustment of the deviance ratio and F-test and t-test is undertaken: for dispersion parameter > 1 probabilities increase (i.e. less significant); for dispersion parameter < 1 probabilities decrease (i.e. become more significant). Candidate terms for the analysis were derived from data collected at each site and are shown in Table 6.1. Those variables with a wide spread of values, both percentage variables and high value continuous variables, were normalised for further analysis by transformation as follows:

Overstorey Cover: log_10+1
Rainfall: log_10
Time Since Fire: log_10
Total Cover: log_10+1
Understorey Cover: log_10+1.

Given the objective of the analyses was to ascertain fluxes in vascular plant species richness, and influences on these fluxes, over time rather than over space, the majority of variables used in the analyses are those collected on sites over time, both pre- and post- the 2003 fire event. Hence changes in structure and cover and their potential influence on richness are key variables of interest.

Table 6.1: Candidate variables for use in the GLMMs

Term: Variate (V) or Factor (F)	Range of Values
Habitat Complexity (V)	0 - 12
Number of Strata (V)	1 - 6
Overstorey Cover (V)	0 - 90 %
Overstorey Height (V)	0 - 40 m
Rainfall (V)	358-1568 mm (total actual rainfall for each sample year)
Scorch Proportion (V)	0 - 1 (varying between 0 for unscorched up to 1for fully scorched)
Season (F)	1, 2 or 3 (Spring, Summer, Autumn)
Time Since Fire (V)	1 - 64 years
Total Cover (V)	0 - 195% (additive and overlapping across strata)
Understorey Cover (V)	0 - 175% (additive and overlapping across strata)
Vegetation Type (F)	1 – 21 (1-14 Brindabella; 15-21 Burrinjuck)

Season of sampling (late spring; summer; early autumn) was included to account for any variation due to resampling a particular site at different seasons across sampling years. Additionally, as an external variable that changes over time, rainfall was included as it is a potential major influence on vascular plant species richness. For the rainfall data, the Fairlight climate station was used for the Brindabella study area and the Burrinjuck Dam climate station was used for the Burrinjuck study area. Mean annual rainfall for Fairlight is 810 mm and mean annual rainfall for Burrinjuck Dam is 924 mm (Bureau of Meteorology, 2020a). In relation to the Fairlight climate station, there are other climate stations close by in the study area, namely Mount Ginini, Bendora Dam and Brindabella (Koolami), but these have incomplete yearly rainfall records for the study period from 1997 to 2012. Data from the Fairlight climate station data were used as it provided the closest complete rainfall record for the entire study period.

Time since fire data for the sample sites, both pre- and post-2003, was obtained from digital layers supplied by the NSW NPWS. Including Vegetation Type in the analyses enabled variation in the available vascular plant species pool, as manifested by differences between vegetation types, to be factored into the analyses.

The iterative variable elimination process based on dropping non-significant terms using the Wald Statistic was used to develop two models of the relationship between vascular plant species richness and environmental variables: a model for Brindabella and a model for Burrinjuck.

6.3 Results

6.3.1 Data Exploration

6.3.1.1 Vascular Plant Species Richness and Vegetation Type

There is a wide range of variation in vascular plant species richness between vegetation types and also within vegetation types over time. In Brindabella, the full range of vascular plant species richness spans from just over 10 species in vegetation type 13 (long unburnt montane heath) to over 90 species in vegetation type 10 (creekline Tall Open Forest), the means ranging from 18 to 70 species. In Burrinjuck, there was a similar span of values with just above 10 species in vegetation type 19 (dry Open Forest) to over 100 species in vegetation type 15 (creekline Open Forest), with means ranging from 21 to 62 species. The overall pattern across the vegetation types is for greater species richness in more mesic sites. Some of this variation is due to variation in the number of sites per vegetation type, however in the original pre-fire sampling, stratification of sites based on environmental variation ensured that environments, and hence their resultant vegetation types, were sampled proportionally to their occurrence in the landscape.

6.3.1.2 Vascular Plant Species Richness Pre- and Post-Fire

Overall, for the Brindabella study area, 90 sites had greater species richness post-fire compared to pre-fire, nine sites had no difference in species richness and 24 sites had less species richness post-fire compared to pre-fire. In Burrinjuck, all 33 sites had greater species richness post-fire as compared to pre-fire. There is a clear pattern for most sites to be above the 1:1 line of no difference, showing a tendency for greater species richness post-fire as compared to pre-fire conditions for most sites (Figure 6.3). Results from a two-sample t-test (paired) showed that aboveground plant species richness was greater ($P<0.001$) in the first sample after the 2003 unplanned fire, compared with pre-fire species richness, for each of the two study areas. However, there was no significant difference in species richness pre- and immediately post-fire between low and high severity sites for either the Brindabella ($P=0.76$) or Burrinjuck ($P=0.33$) study areas. Although an increase in species richness was associated with fire, it was not related to fire severity and species richness increased significantly at both low and high severity sites. In relation to the null hypotheses proposed in Section 6.2.2:

$H0^{1a}$: mean first post-fire sample species richness = mean pre-fire sample species richness is rejected as there is a significant difference between pre- and post-fire species richness, with species richness being greater post-fire than pre-fire in both study areas;

$H0^{1b}$: mean difference between first post-fire sample and pre-fire sample species richness Low Severity = mean difference between first post-fire sample and pre-fire sample species richness High Severity is accepted as there is no significant difference between low and high severity sites in either of the two study areas.

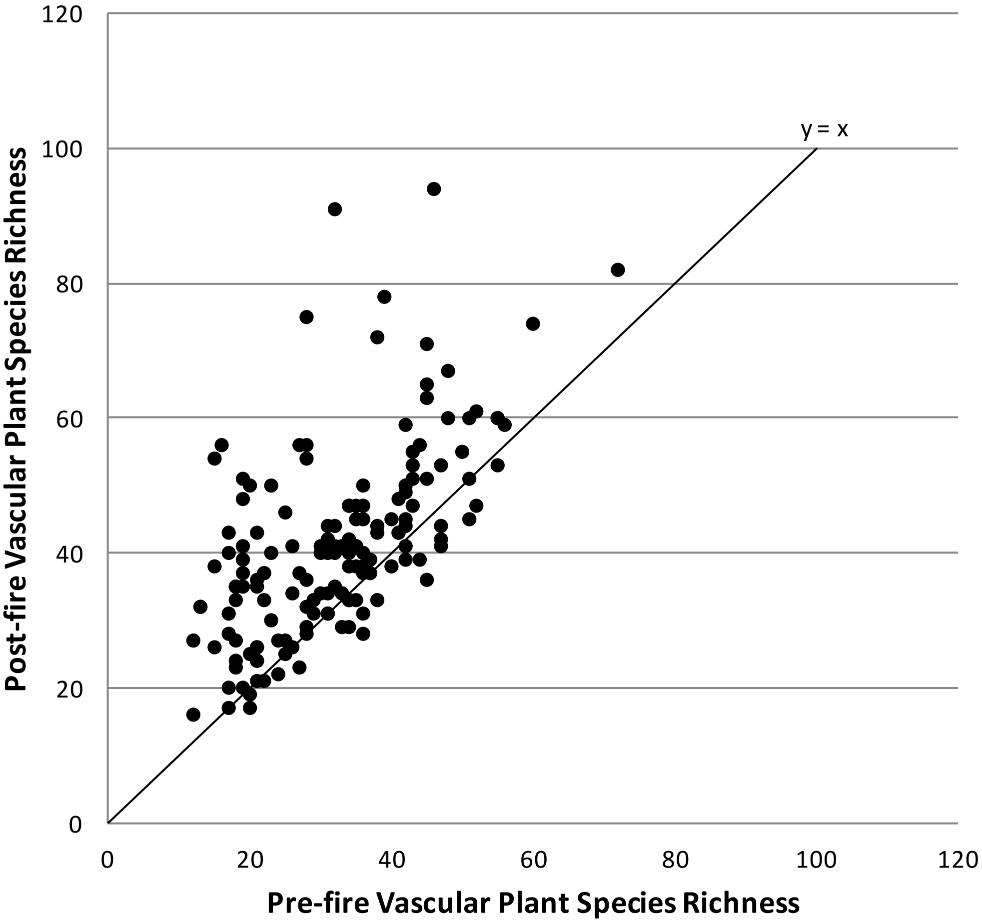

Figure 6.3: Pre-fire vascular plant species richness (/40m^2) versus first post-fire sample vascular plant species richness (/40m^2) for combined Brindabella and Burrinjuck study areas data. 1:1 line of y=x plotted for reference.

In relation to changes in vascular plant species richness after fire, Figure 6.4 shows mean vascular plant species richness over time from 1996 to 2012 for the Brindabella and Burrinjuck study areas (Appendix 5 shows the changes in species richness over time across sites before and after fire in relation to fire severity in greater detail).

6.3.1.3 Relationships Among Environmental Variables and Species Richness

There was a moderate to large drop in species richness across the sites (17 out of 21 – see Appendix 5) in sample period 5, which was November 2006 (Figure 6.4). Investigation of rainfall records showed that this year was very much below average for rainfall and further investigation of the rainfall records for the study areas also revealed that the rainfall in the 1997 pre-fire sampling period was also well below average (Figure 6.5). This relationship between plant species richness and rainfall, particularly a marked decline in species richness in drier years, means that testing for the effects of fire - time since fire; fire severity - without also testing for the effects of rainfall, may confound an interpretation of how species richness changes over time post-fire.

In relation to cover, there was a general trend of increasing species richness with understorey cover, but a reduction in species richness in relation to overstorey cover (Figure 6.6). The former pattern may be interpreted as a relationship between greater numbers of species in a site leading to an overall great understorey cover, whereas the latter trend may reflect increased overall competition and particularly light availability in sites with high canopy cover, especially for example heath and shrub communities with high cover of dominant species.

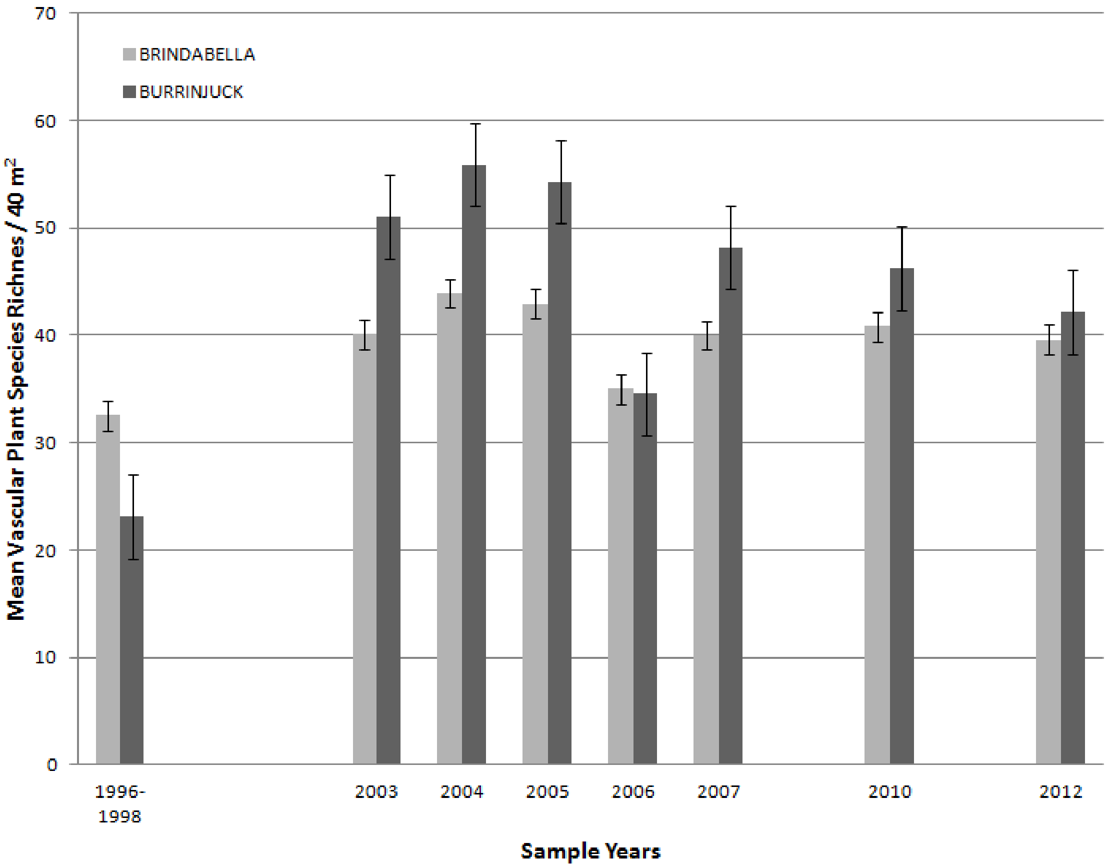

Figure 6.4: Mean vascular plant species richness for sampling years between 1996 and 2012 for the Brindabella and Burrinjuck study areas

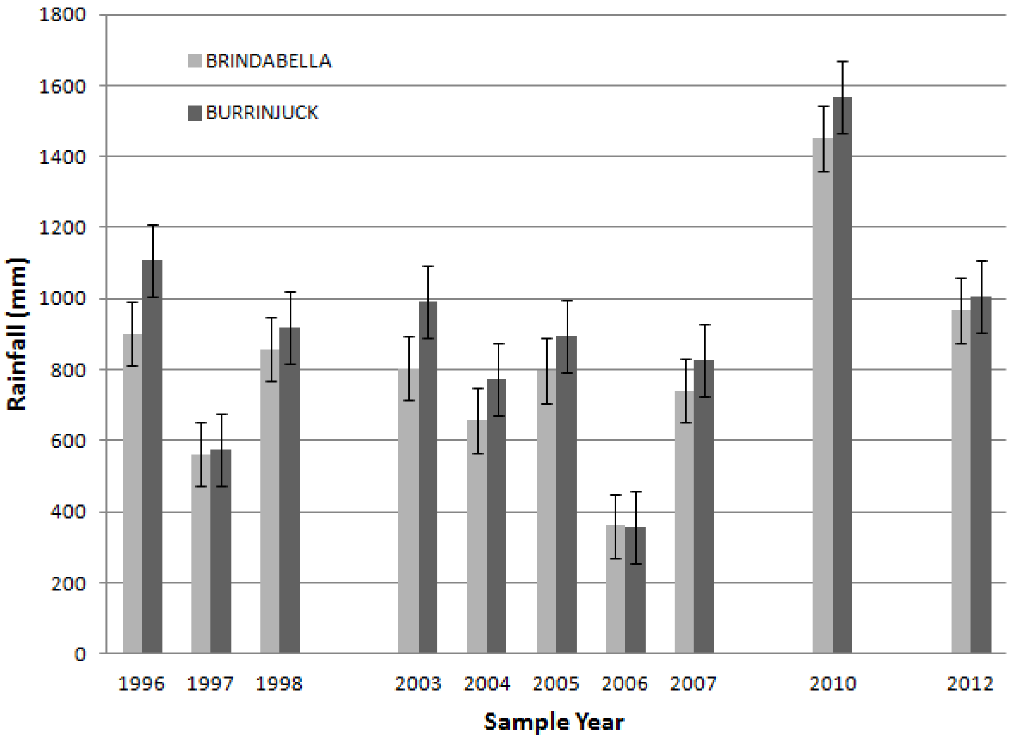

Figure 6.5: Total annual regional rainfall for sampling years between 1996 and 2012 for the Brindabella and Burrinjuck study areas

There was also a trend of increasing species richness with increasing overstorey height which may reflect greater species richness in mesic sites, particularly creeklines (Figure 6.6). These patterns conform to the general relationships previously outlined in section 6.1 in relation to cover, plant competition and species richness. In terms of structure, there is an increase in species richness with increasing habitat complexity and with number of strata (Figure 6.7). The relationship with total cover is similar to that for understorey cover, which reflects the fact that on many sites, most of the total cover on a site is provided by understorey species. In relation to rainfall and time since fire, there is an increase in species richness with increasing rainfall but a decrease in species richness with increasing time since fire (Figure 6.8). Species richness in the Brindabella sites appears unrelated to scorch, but there seems to be a positive relationship between scorch and species richness for the Burrinjuck sites (Figure 6.8). The correlation between eight of the nine variables and plant species richness was highly significant ($P<0.001$), with only the correlation with scorch being non-significant ($P=0.080$). Two-sided tests of correlations different from zero, on 624 observations, are shown in Table 6.2.

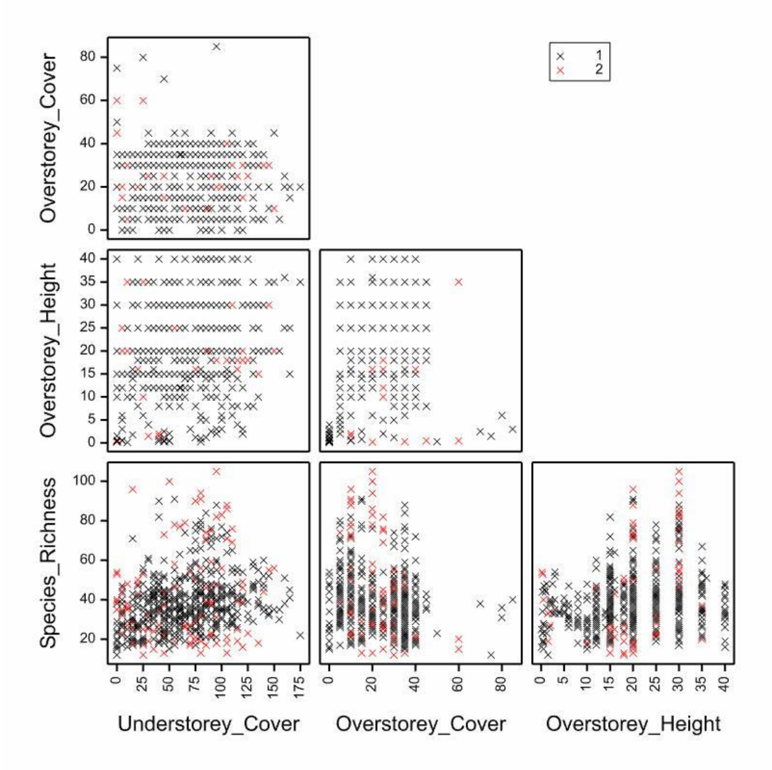

Figure 6.6: Correlation matrix among explanatory and response variables: richness versus understorey cover, overstorey cover and overstorey height. Black crosses are Brindabella sites; red crosses are Burrinjuck sites.

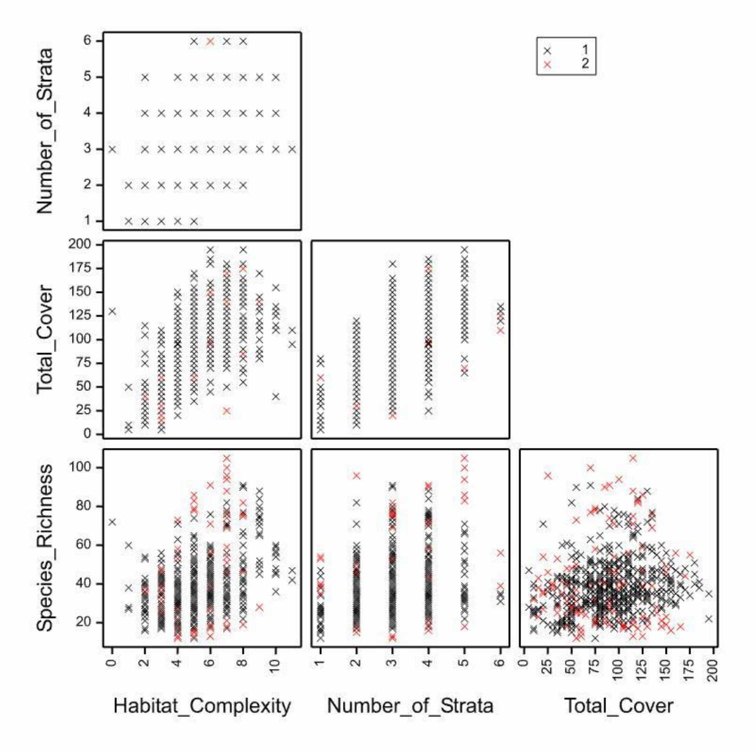

Figure 6.7: Correlation matrix among explanatory and response variables: richness versus habitat complexity, number of strata and total cover. Black crosses are Brindabella sites; red crosses are Burrinjuck sites.

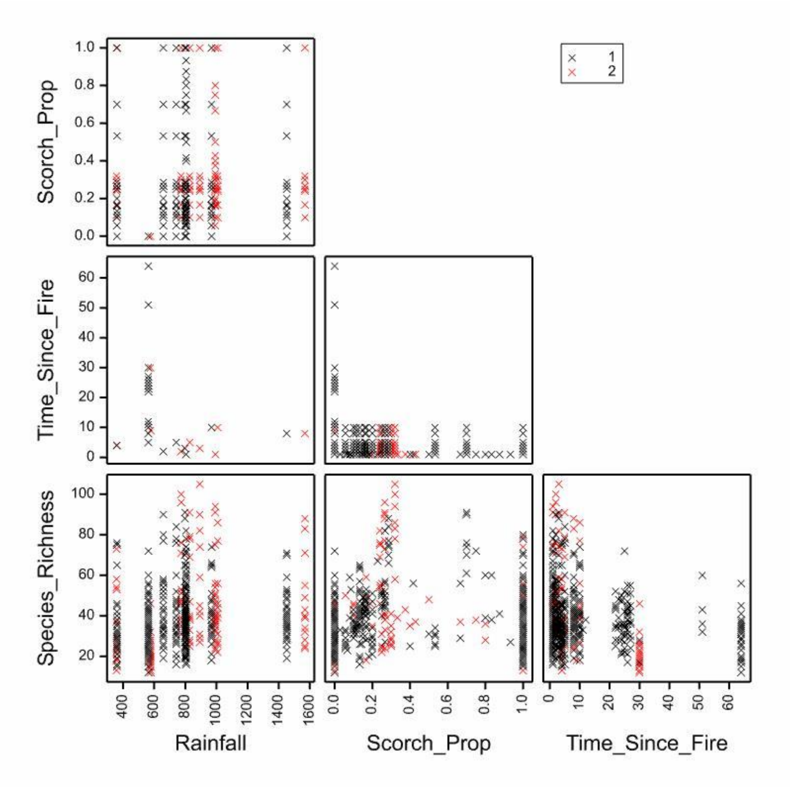

Figure 6.8: Correlation matrix among explanatory and response variables: richness versus rainfall, scorch and time since fire. Black crosses are Brindabella sites; red crosses are Burrinjuck sites.

Table 6.2 Results of two-sided tests of correlations different from zero for explanatory and response variables for combined data for Brindabella and Burrinjuck study areas

Species Richness vs Cover and Height

Species_Richness	1	-			
Understorey_Cover	2	<0.001	-		
Overstorey_Cover	3	<0.001	0.0027	-	
Overstorey_Height	4	<0.001	<0.001	0.0023	-
		1	2	3	4

Species Richness vs Habitat, Strata and Total Cover

Species_Richness	1	-			
Habitat_Complexity	2	<0.001	-		
Number_of_Strata	3	<0.001	<0.001	-	
Total_Cover	4	<0.001	<0.001	<0.001	-
		1	2	3	4

Species Richness vs Scorch; Rainfall; Time Since Fire

Species_Richness	1	-			
Rainfall	2	<0.001	-		
Scorch_Prop	3	0.0443	<0.001	-	
Time_Since_Fire	4	<0.001	<0.001	<0.001	-
		1	2	3	4

6.3.2 Statistical Modelling

Because the Total Cover variable was derived as an additive term consisting of Understorey Cover + Overstorey Cover, it could not be fitted at the same time as each separate term due to the strong correlation between the constituent variables and the aggregate variable, hence in the model test runs either Understorey Cover and Overstorey Cover (first model run, Appendix 6 a and Appendix 7 a) were fitted or else Total Cover was fitted (second model run, Appendix 6 b and Appendix 7 b), in separate fits of the model. The third model run (Appendix 6 c and Appendix 7 c) used only significant terms from the first and second model runs to develop the final model for each study area. All of the GLMM model fitting runs to produce final models for the Brindabella and Burrinjuck study areas are detailed in Appendices 6 and 7 respectively.

For the GLMM of apparent vascular plant species richness in the Brindabella study area, vegetation type, time since fire, rainfall and habitat complexity were all highly significant (P<0.001), with significant influence of understorey cover (P=0.001) (Table 6.3). Because vascular species richness varied across the 14 vegetation types in the Brindabella study area, there is a significant influence of vegetation type on predicting vascular plant species richness. Broadly across all range of vegetation types, there was an overall decline in vascular plant species richness with time since fire but this was offset in some years by rainfall, with an increase in species richness in years of higher rainfall. Habitat complexity and understorey cover also had a significant positive influence of species richness, but were less influential than vegetation type, time since fire and rainfall.

Table 6.3: Brindabella study area final species richness GLMM model

Fixed Term	**Estimate with SE**	**Wald statistic**	**d.f.**	**P**
Vegetation_Type	Varies +/- by Type	107.59	13	<0.001
Time_Since_Fire_log10	-0.1138 (0.01264)	83.67	1	<0.001
Rainfall_log10	0.2705 (0.04317)	43.53	1	<0.001
Habitat_Complexity	0.01611 (0.005949)	15.04	1	<0.001
Understorey_Cover_log10_1	0.09192 (0.028502)	10.40	1	0.001

Term	**Model(order)**	**Parameter**	**Estimate**	**s.e.**
Dispersn	Identity	Sigma2	0.630	0.0475

For the GLMM of apparent vascular plant species richness in the Burrinjuck study area, sample season, habitat complexity, time since fire and rainfall were all highly significant (P<0.001) (Table 6.4). Although there was variability in vascular plant species richness across the seven vegetation types found in the Burrinjuck study area, they were not a significant predictor of the overall variation in vascular plant species richness found there. The difference in pre-fire and post-fire sampling seasons in Burrinjuck as compared with Brindabella has resulted in season of sampling being a significant predictor of vascular plant species richness in Burrinjuck, where richness is less in autumn samples as compared to spring samples. A decrease in species richness occurs with time since fire, with the rate of decrease being more rapid than that in Brindabella. There is an increase in species richness with rainfall, and this effect is also greater than that in Brindabella. Habitat complexity also positively influences affects species richness in Burrinjuck and tis effect is also greater than in Brindabella.

Table 6.4: Burrinjuck study area final species richness GLMM model

Fixed Term	Estimate with SE	Wald statistic	d.f.	P
Sample_Season	-0.4623 (0.07394)	288.52	1	<0.001
Habitat_Complexity	0.06314 (0.013736)	23.46	1	<0.001
Time_Since_Fire_log10	-0.2386 (0.04494)	22.51	1	<0.001
Rainfall_log10	0.4125 (0.09481))	18.93	1	<0.001

Term	Model(order)	Parameter	Estimate	s.e.
Dispersn	Identity	Sigma2	1.029	0.140

Importantly, fire severity was not a significant predictor of vascular plant species richness in either study area. The common pattern between the two study areas was a decrease in vascular plant species richness with time since fire and an episodic temporary increase in vascular plant species richness in higher rainfall years.

To determine the form of both the time since fire effect and rainfall effect on apparent vascular plant species richness, the combined models were used to predict species richness across a potential range of values from 0 to 100 years post-fire and from 200 to 1650 mm rainfall for each study area based on the predictions from the GLMMs (Figures 6.9 – 6.12). There was an initial rapid decline in species richness in both study areas with time since fire but this rate slowed over time. The rainfall curve acted to reinforce decreases in species richness at points in time with greater predicted drops in species richness at lower rainfall, but less impact on species richness at higher rainfall.

Using a mean annual rainfall of 900 mm for both study areas, the GLMM was used to predict likely increases and decreases in apparent vascular plant species richness (Table 6.5). This increase effect will fluctuate over time and richness will increase and decrease relative to annual rainfall. Burrinjuck is more responsive to rainfall than Brindabella.

Table 6.5: Predicted values of vascular plant species richness at set annual rainfalls against mean annual rainfall using predictions from the GLMMs. Comparison taken as 900 mm annual rainfall which is approximately the mean annual rainfall in each study area. Figures in brackets indicate percentage increase or decrease in species richness compared to this approximate mean annual rainfall.

Decreasing Rainfall				Approximate Mean Annual Rainfall	Increasing Rainfall	
	400 mm	**600 mm**	**800 mm**	**900**	**1000 mm**	**1200 mm**
Brindabella Mean Predicted Species Richness	35 (-7.9%)	36 (-5.3%)	38 (0%)	38	39 (+2.6%)	39 (+2.6%)
Burrinjuck Mean Predicted Species Richness	30 (-14.3%)	33 (-5.7%)	34 (-2.8%)	35	36 (+2.8%)	37 (+5.7%)

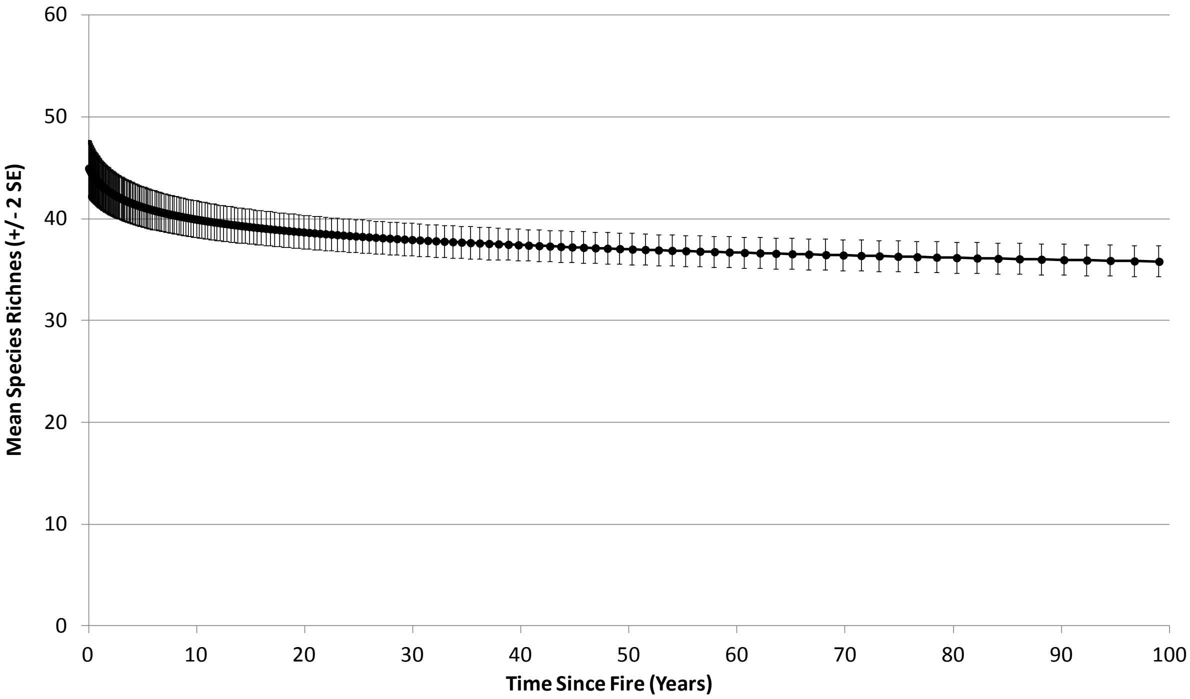

Figure 6.9: Predicted apparent vascular plant species richness vs time since fire: Brindabella model

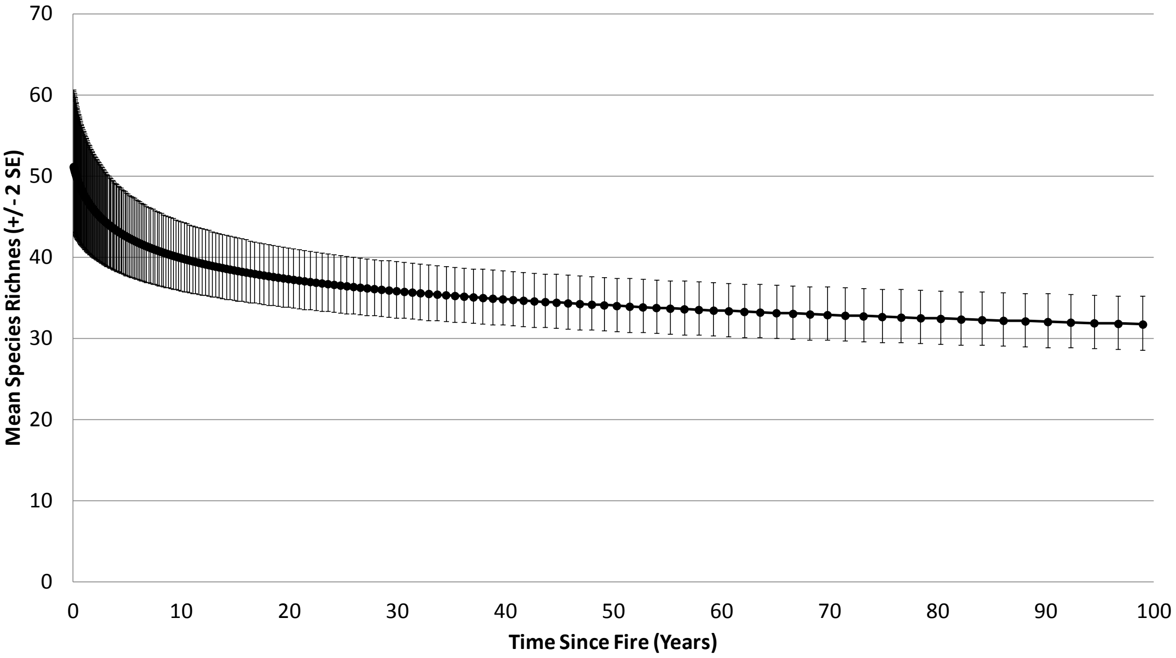

Figure 6.10: Predicted apparent vascular plant species richness vs time since fire: Burrinjuck model

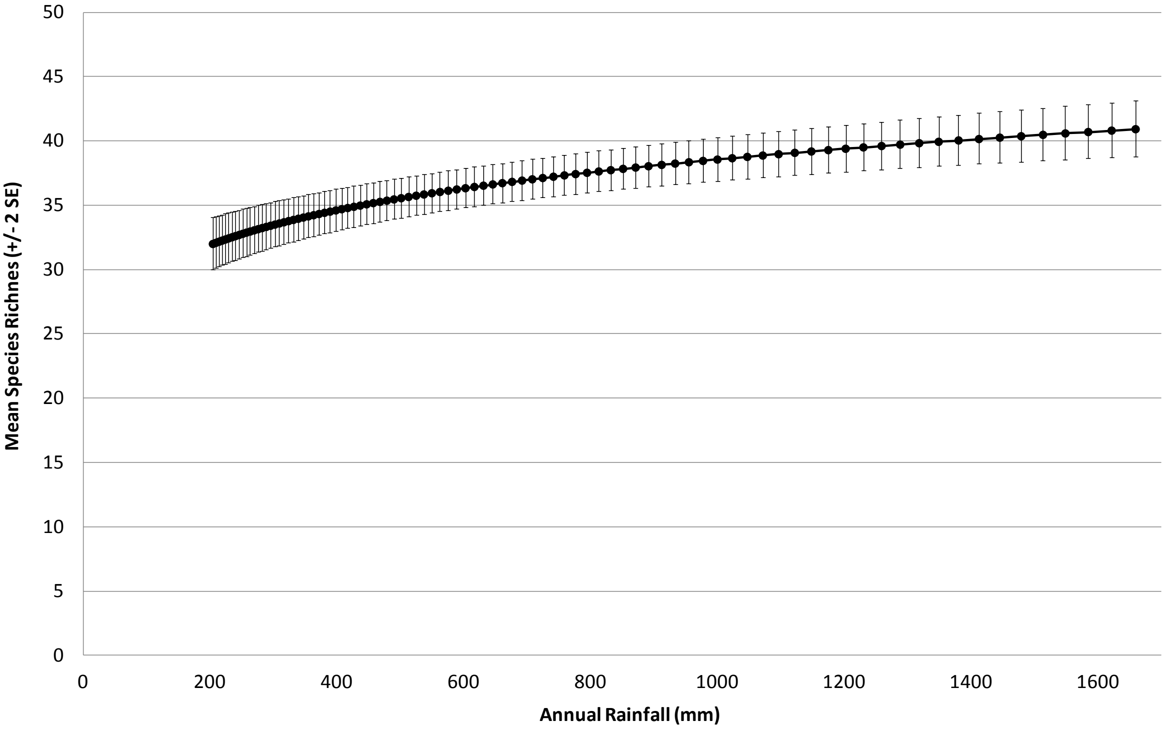

Figure 6.11: Predicted apparent vascular plant species richness vs annual rainfall: Brindabella model

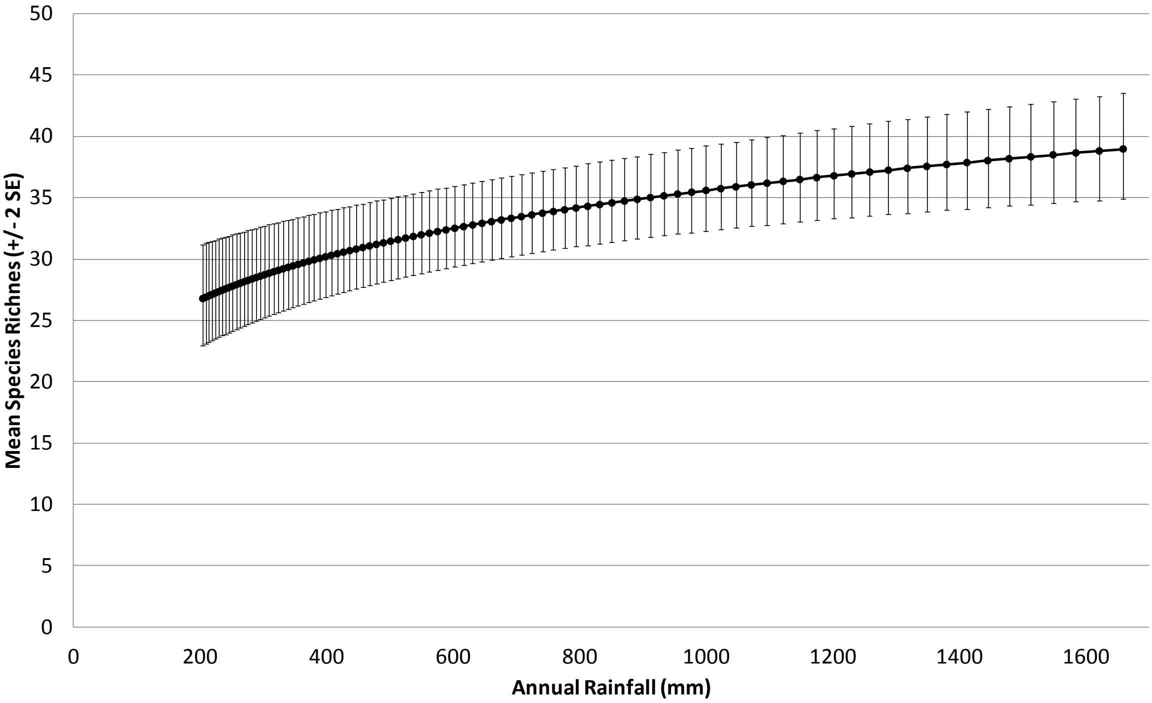

Figure 6.12 Predicted apparent vascular plant species richness vs annual rainfall: Burrinjuck model

Using time since fire points of 1, 5 10 and 20 years, the GLMM was used to predict percentage decreases in apparent vascular plant species richness over time taken as compared to 1 year post-fire (Table 6.6). This decline continued over time irrespective of rainfall. The Burrinjuck study area has a more rapid decline in species richness and greater difference in numbers pre- to post-fire and at 20 years post-fire compared to Brindabella.

Table 6.6: Predicted mean species richness values at set times since fire using predictions from the GLMMs. Comparison point taken as 1 year post-fire. Figures in brackets indicate percentage increase or decrease in species richness compared to 1 year post-fire.

	1 year	**5 years**	**10 years**	**20 years**
Brindabella	44	41 (-6.8%)	40 (-9.1%)	39 (-11.4%)
Burrinjuck	49	43 (-12.3%)	40 (-18.4%)	37 (-24.5%)

There was an underlying decrease in species richness over time post-fire, irrespective of fire severity, with year to year variation due to rainfall variability. The sampling season also affected ‘baseline’ measures of apparent vascular plant species richness as fewer species were apparent in autumn compared to spring. This was particularly the case for geophytes and some forbs, but the dominant shrub and tree species were present in a given site sample at any season. There is therefore variation around a baseline level of vascular plant species richness, forming the core composition of the sites over time. In dry years, species which are responsive to rainfall such as geophytes remain dormant and the apparent richness of sites is less than wetter years. Species groups which are less apparent in dry years include Liliaceae, Orchidaceae and annual species of *Ranunculus* and *Geranium.* A list of genera that contribute to this flux in on-site apparent vascular plant species richness are shown in Box 6.1.

Pre-fire values will vary over time in relation to a ‘sampling starting point’ and many sites in Brindabella and Burrinjuck were sampled at a point in time that was greater than 20 years post-fire. Additionally, the Burrinjuck sites were originally surveyed in autumn in the pre-fire survey, whereas post-fire samples were taken in spring, and a significant seasonal effect was detected. However, when this seasonal effect is taken into account, there are still significant influences of time since fire and rainfall. In both study areas, there was a significant overall increase in vascular plant species richness post-fire, but there was no significant influence of fire severity on this species richness as compared to the other variables from the 2003 fire event and particularly the influence of subsequent rainfall patterns.

Box 6.1: Native genera contributing to changes in apparent species richness in relation to rainfall, based on Brindabella and Burrinjuck site sampling from 1997 to 2012

Species within the following genera that are more likely to be apparent in average to above-average rainfall years, and non-apparent in drier years

Anthericaceae: *Arthropodium, Caesia, Euchiton, Thysanotus*

Asteraceae: *Craspedia, Podolepis, Senecio*

Droseraceae: *Drosera*

Geraniaceae: *Geranium*

Hypericaceae: *Hypericum*

Juncaceae: *Juncus*

Orchidaceae: *Acianthus, Caladenia, Chiloglottis, Dipodium, Microtis, Prasophyllum, Pterostylis, Thelymitra*

Plantaginaceae: *Plantago, Veronica*

Poaceae: *Deyeuxia, Dichelachne, Panicum*

Ranunculaceae: *Ranunculus*, including 2 seasonal annuals: *R. pumilio* var. *pumilio* and *R. sessiliflorus* var. *sessiliflorus.*

Rubiaceae: *Galium*

6.4 Discussion

Vascular plant species richness in the study area is determined by both underlying spatial, and by episodic temporal, influences. There is a signal from underlying regional environmental variation across gradients of rainfall, temperature, aspect, slope and lithology leading to variation in vascular plant species richness *between* vegetation types (Austin and Cunningham, 1981; Austin *et al.*, 1983a; Austin, 1987). There is also a signal from fire and rainfall events that provides variation in vascular plant species richness over time *within* vegetation types.

In relation to Gill’s proposed models of vascular plant species richness with time since fire (Section 6.1, Figure 6.2), findings from this study suggest that the general form of the relationship in subalpine, montane and tableland ecosystems in southeastern NSW is equivalent to response ‘b’, with an increase in richness post-fire followed by a gradual decline with time, but only if time since fire is considered separately. However, when fluctuations in rainfall are taken into account, the general form of the relationship is more like response ‘c’, with a decline in vascular plant species richness with time since fire moderated by year to year fluctuations in rainfall and hence differences in apparent vascular plant species richness.

Post-fire and year to year variation in vascular plant species richness in both study areas was not due to local extinction and colonisation processes in relation to fire but was due to variation in the response of existing individuals found on site and how detectible they were. Because the majority of plants either resprout or recover from canopy and soil seed banks, vascular plant species recover rapidly post-fire in situ and so Egler’s Initial Floristic Composition model (Egler, 1954) appears appropriate in the ecosystems studied. This is also consistent with studies in lower altitude ecosystems in the broader southern tablelands region (Purdie and Slatyer, 1976) and also in Mediterranean and forested ecosystems in the northern hemisphere (Anderson and Romme, 1991; Capitanio and Carcaillet, 2008). Despite evidence from coniferous communities in the northern hemisphere that post-fire overstorey dynamics and competition have a significant long-term effect on understorey plant species richness (Laughlin and Grace, 2006) there was no clear effect on vascular plant species richness of overstorey cover in the Brindabella and Burrinjuck study areas. While there is an influence of overstorey cover on understorey richness applicable across geographic space and between vegetation types over environmental gradients (Specht, 1983; Specht and Specht, 1989b), the effect of overstorey on understorey specie richness dynamics in the communities studied appears to be a transient effect in relation to disturbances such as fire, and there was no evidence that this competitive dynamic has long-term effects on vascular plant species richness in the ecosystems examined. The significant relationship between understorey cover and vascular plant species richness in Brindabella may reflect the fact that more mesic sites with higher understorey cover also have a higher mean species richness than drier vegetation sites. Whether there is a relationship between richness and cover per se, or whether this relationship reflects underlying productivity gradients, is not clear.

Species richness post-fire showed a sharp increase, peaking in the second season post-fire, then a gradual decease over time in relation to a time since fire effect (i.e. a general increase in richness post-fire with a subsequent gradual decrease in apparent richness over time) but was also affected by rainfall such that in very dry years richness decreased, but increased again when rainfall increased, in the intermediate years post-fire. In later years, even with very high rainfall, species richness continued to decline. Similarly, in the immediate post-fire years, richness was maintained despite fluctuations in rainfall. The initial pre- versus post-fire increase in richness appears to be attributable to both differences in rainfall between the samples, and also a post-fire effect in relation to an easing of competitive restraints, as discussed in Section 6.1. Hence, both the pre-fire 1997/1998 sample year and the 2006 sample year are low rainfall years and both show a reduction in apparent vascular plant species richness. However, the system responds when rainfall returns to near average conditions. In other biomes including rangelands, heathlands, woodlands and grasslands, rainfall has been shown to be a significant factor affecting species richness at a point in time (Cowling *et al.*, 2018; Hall *et al.*, 1964; McIntyre and Lavorel, 1994). In the two study areas, changes in species richness are related to year to year rainfall patterns acting upon patterns set in train by a fire event, rather than to fire severity in particular, in contrast to conifer dominated systems in the USA where fire severity has been shown to negatively impact species richness (Richter *et al.*, 2019). The 1997/1998 season of original sampling was one of driest years in recent records up to that time and hence, raises issues when comparing species richness over time. What could be envisaged as a ‘benchmark’ measure is essentially only a starting measure for the comparison of changes over time. The strong relationship between changes in vascular plant species richness and rainfall data means that any given measure of richness can only be compared to years of similar rainfall.

Even though there was higher than average rainfall in the 2010 sample year, there was no concomitant species richness response apparent across the sites. This pattern suggests that in the temperate ecosystems found in the study area, apparent vascular plant species richness is reduced in dry periods, but that there is no pool of plant species that responds to higher than average rainfall. In temperate ecosystems, rainfall is not a primary limiting factor for growth and reproduction, with few ephemerals in the flora, although germination and establishment can be affected by unusually wet or unusually dry years and drought events can cause mortality in established plants (Benwell, 2007; Davidson and Reid, 1989; Pook *et al.*, 1966).

This is in contrast to semi-arid and arid ecosystems where rainfall is limiting on plant growth and establishment and there are large annual and ephemeral components in the flora, with significant episodic germination events after heavy rain periods from soil seed banks (Adler and Levine, 2007; Archibold, 1995; Hall *et al.*, 1964; Maconochie, 1982). In these ecosystems, rainfall is recognized as major factor influencing plant community dynamics both in relation to apparent vascular plant species richness and also in relation to overall compositional and structural recovery after fire (Fensham *et al.*, 2017; Nano *et al.*, 2017). Fox (1990) found that in semi-arid mallee ecosystems, seasonality in relation to annual rainfall was a major factor affecting species richness over time. In dry years apparent vascular plant species richness was greatly reduced, ranging from only 25-42% of cumulative richness across all years. This large variation was due to the flux in a large number of ephemeral species which were not apparent in years of low rainfall but persisted in the soil seed bank and germinated when rainfall was adequate. Kenny *et al*. (2018) found that although time since fire was the major factor determining structural recovery in semi-arid mallee shrublands, rainfall variability also affected recovery. Griffin (1992) found in arid spinifex grasslands that cumulative rainfall post-fire was highly influential on cover and dominance.

Species richness trends after fire in the Brindabella and Burrinjuck study areas are consistent with other research on post-fire changes in species richness to the south and north of the region. Research on the south coast of NSW shows that maximum richness in coastal heath occurs at 4 years post-fire, with species richness greater post-fire compared to pre-fire (Posamentier *et al.*, 1981) and that species richness in dry and wet sclerophyll forests also declines with time since fire (Penman *et al.*, 2009; Penman *et al.*, 2011). Hammill *et al*. (2016) in the Sydney Basin found a strong influence of rainfall on species richness, with rainfall being more influential than temperature or fire frequency when these were controlled for other environmental variation.

Although plant species in the study area show a strong ability to recover and recruit after fire, exhibiting fire-stimulated recruitment in the broad sense, ‘fire-dependent recruitment’ (Keeley and Fotheringham, 2000) is not a strong element found in the flora found there. *Eucalyptus delegatensis* subsp. *delegatensis* is a major exception although it too can recruit to a small degree between fires (Chapter 5). Because many areas in the Brindabella study area burnt in 2003 had not been burnt since 1939, soil seed banks for some species may have declined in the long absence of fire such that local extinction of populations could occur. However, evidence from heathy woodlands in Victoria suggests that at least in some forest ecosystems, there is no decline or major difference in the soil seed banks between recently burnt areas and areas remaining unburnt for 75 years (Chick *et al.*, 2016) and studies in Jarrah forest in Western Australia show no difference in seed banks between recently burnt areas that have been fire free for up to 22 years (Koch *et al.*, 2009). More broadly, while fire undoubtedly stimulates seed banks to germinate, often *en masse*, seed banks can be very long-lived (Murdoch and Ellis, 2000) and there is still occasional germination due to random events such as tree, shrub and grass death and also soil disturbance from animal activity between fire events (Mallen-Cooper *et al.*, 2019). However, some seed banks remain dormant in the absence of fire, with germination triggers necessary from heat and smoke in some plant species, particularly in coastal shrubland and dry forest ecosystems (Merritt *et al.*, 2007; Probert, 2000; Thomas *et al.*, 2007;

Warcup, 1980; Wills and Read, 2002). Because light and nutrient availability are also greater in the post-fire environment, without detailed smoke and heat studies on montane species, it is difficult to separate the importance of the effects of these factors from the direct effects of a fire in seed germination after disturbance (Fenner and Thompson, 2005)

Grubb (1977) stresses the importance of what is termed the ‘regeneration niche’ in the maintenance of species richness in plant communities, with differential requirements for germination across species interacting with environment and disturbance events to enable the persistence of multiple species at a site over time. While fire events create disturbance and regeneration opportunities, the results to date from long-term monitoring of sites in the Brindabella and Burrinjuck study areas suggests that fire severity as such is not a factor in this post-fire dynamic.

6.5 Conclusion

Hypothesis 1, that: *high severity fire is associated with a greater increase in plant species richness within plant communities post-fire as compared to low severity fire and that the duration of this effect is positively associated with fire severity* is not supported by the analyses undertaken. Fire severity was not a significant influence on vascular plant species richness response in either the Brindabella study area or the Burrinjuck study area when separate models were used. Time since fire, rainfall, vegetation type and season of sampling are far more significant as predictors of vascular plant species richness overall. Although fire severity did not have a significant effect on vascular plant species richness, there was an overall increase in species richness immediately post-fire across both low and high severity sites. This decreased with time since fire but interacted with annual rainfall, such that lower than average rainfall years were associated with a reduction in apparent on site vascular plant species richness.

By controlling for other environmental factors, a sample at a point in time will reflect both time since fire and rainfall influences, and hence samples taken in drier than normal years will be sampling lower species richness values than would otherwise be the case for a given time since fire. While many studies have focused on the effects of rainfall on plant establishment and persistence, the findings of the present study show that there is a need to incorporate rainfall influences when measuring species richness after fire.

This chapter has focussed on overall numbers of vascular plant species present at varying points in time, and Chapter 7 examines the relationship between fire severity and plant community composition in greater detail to track plant community responses based in detailed floristic composition data.

Chapter 7: Fire Severity and Plant Community Composition

Note: Components of this chapter have been presented at the Association for Fire Ecology 5th Annual Fire Ecology and Management Congress (2012, Portland, Oregon, USA)

7.1 Introduction

The degree to which disturbance events change ecosystem properties and the duration of any such change in those properties is a much studied and much debated theme in ecology (Elmqvist *et al.*, 2003; Fonseca and Ganade, 2001; Groffman *et al.*, 2006; Grubb and Hopkins, 1986; Harrison, 1979; Holling, 1973; May, 1973; O'Neill *et al.*, 1986; Peet, 1992; van Hulst, 1980; Westman, 1986; Westman, 1978). Research approaches relating to disturbance and change have ultimately focussed on the concept of what has been termed ecosystem ‘stability’, as it relates to ecosystem properties, using a variety of measures including species richness (Chapter 6) and community compositional stability. Pimm (1984) consolidated the concepts proposed by Holling (1973) and Harrison (1979) on stability and complexity in ecosystems, which were later refined by Whitford *et al*. (1999). Both Pimm’s and Whitford *et al.*’s’ syntheses can be summarised into four measures relating to ecosystem stability that are both measurable and ecologically relevant:

Stability: the return to an initial state following a disturbance

Resistance: the degree to which a variable has moved from a pre-disturbance state

Resilience: the time taken for a variable to return to a pre-disturbance state

Persistence: the time a variable lasts before it is changed to a new value

To measure ecosystem changes and ecosystem stability based on these definitions, there is clearly an assumption that there is at least one, or potentially multiple, variables for which there is pre-disturbance data and that there are conceptual models of the states and behaviour of those variables. In other words, the expected bounds of natural variability need to be understood before post-disturbance dynamics can be interpreted. White and Pickett (1985) distinguish between natural disturbances and perturbations, where perturbations are events that force ecosystems to depart from the bounds of their natural variability (i.e. the ecosystem by definition becomes non-stable). Despite the term stability logically implying the existence of an unstable state, the literature varies in how stability is defined, and rather than implying a system that is ready to ‘collapse’, an unstable system can be more broadly defined as one that is quantitatively and permanently different from its natural state and may be trending in a successional direction toward a different state, as measured for example by changes in structure or composition (Strand *et al.*, 2019). To define such bounds and to detect change from a pre-existing state, and in order to distinguish between long-term directional change and short-term transient dynamics, the concept of the 'Historical Range of Variability' (HRV) has been used to address these issues. Broadly, HRV approaches seek to combine data from multiple sources using a variety of complementary approaches (e.g. written historical records, photography, dendrochronology, palynology, ecology) to specify the ecological bounds within which a system has persisted in the past (Morgan *et al.*, 1994; Wiens *et al.*, 2012), particularly in regard to the effects of fire on forest ecosystems (Keane *et al.*, 2009). In the absence of defined models of variation in an ecosystem subject to disturbance, comparisons need to be made of variables that are available ideally both before and after disturbance, or alternatively, only after a disturbance. Disturbance events then become starting points that can, if monitored in the long term, enable the construction of conceptual models of ecosystem dynamics (Gosper *et al.*, 2018; Lauvaux *et al.*, 2016; Vankat, 2013). That is, in many areas, historical data is lacking to readily estimate the HRV and so recent pre-disturbance data may be the only benchmark with which to compare post-disturbance data (Falk *et al.*, 2019). This is the case in relation to the Brindabella and Burrinjuck study areas.

Because post-fire succession analysis occurs across environmental gradients in space and also involves changes in situ over time involving potentially hundreds of species, multivariate approaches provide a tractable way of analysing large data sets in relation to these characteristics. Ordination as a technique has been used to study vegetation patterns in relation to environmental gradients in space (Bray and Curtis, 1957; Gauch *et al.*, 1974; Ter Braak, 1986; Whittaker, 1956; Whittaker, 1960; Whittaker, 1967; Whittaker, 1975). However, fewer studies have utilised ordination techniques to study vegetation patterns in time (Austin, 1977; Austin and Belbin, 1981; Austin, 1980a; van der Maarel, 1969). Such analysis of sites over time can be termed 'trajectory analysis' and examples include the pioneering work of van der Maarel (1969); Austin (1977), Swaine and Greig-Smith (1980) and Halpern (1988). Although the use of trajectory analysis is still not widespread, the approach has been utilised in some fire ecology research to investigate post-fire succession, notably in Californian coastal sage scrub (Westman and O'Leary, 1986) and in Mediterranean garrigue (Malanson and Trabaud, 1987). The latter study found no transition or change in Garrigue communities in ordination space and found strong inertia in floristic composition and hence a high resilience to fire and a resistance to change in the plant communities studied.

With or without a measure of a pre-disturbance variable, a multivariate conceptualisation of vegetation dynamics after a disturbance enables a comparison of sites over time. Measuring the initial displacement between pre-fire and first post-fire samples as measured by dissimilarity can be used to measure resistance, and both the accumulated and final difference between pre-fire and post-fire samples as measured by dissimilarity can be used to measure resilience. Both Laughlin *et al.* (2004) and Benscoter and Vitt (2008) used comparisons of repeated measures of plant community composition to investigate stability after disturbance and calculated Bray-Curtis dissimilarities between pre-fire and post-fire states and used this to derive a univariate statistic (equivalent to the difference between 2 values in a paired *t*-test) to investigate resilience and resistance of forest communities to fire.

Matthews *et al*. (2013) framed four pertinent questions around long-term studies of the compositional stability of communities after disturbance, particularly in relation to a trajectory analysis approach:

- *What is the relative importance of 'event-driven' vs gradual change?*
- *Do communities typically change states after some threshold is reached or after catastrophic disturbance?*
- *Do communities follow long-term directional trajectories that can be related to species-specific life history or physiological traits?*
- *What are the effects of repeated disturbances over short periods of time?*

These questions are summarised diagrammatically in Figure 1 from their paper, reproduced below as Figure 7.1

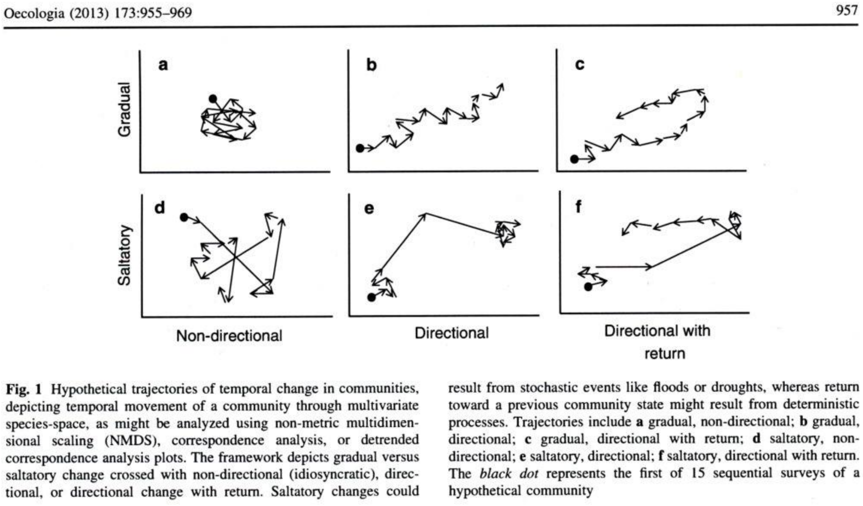

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Fig. 1 Hypothetical trajectories of temporal change in communities, depicting temporal movement of a community through multivariate species-space, as might be analyzed using non-metric multidimensional scaling (NMDS), correspondence analysis, or detrended correspondence analysis plots. The framework depicts gradual versus saltatory change crossed with non-directional (idiosyncratic), directional, or directional change with return. Saltatory changes could result from stochastic events like floods or droughts, whereas return toward a previous community state might result from deterministic processes. Trajectories include **a** gradual, non-directional; **b** gradual, directional; **c** gradual, directional with return; **d** saltatory, non-directional; **e** saltatory, directional; **f** saltatory, directional with return. The *black dot* represents the first of 15 sequential surveys of a hypothetical community

Figure 7.1: Hypothetical trajectories of temporal change in communities. Figure 1 from Matthews *et al*. (2013).

Mathews *et al*.'s conceptual diagrams are a useful summary of potential community behaviour after disturbance, incorporating both the time frame of potential change and also directionality. Non-directional trajectories, whether gradual or saltatory (Figure 7.1a and d), are likely to indicate transient dynamics arising from secondary succession processes without permanent compositional change. Directional trajectories (Figure 7.1b and e) may indicate permanent compositional change, particularly if sampling has been undertaken for many years after a disturbance event. Directional trajectories with return (Figure 7.1c and f) imply some measurable impact from a disturbance event, but with slow compositional recovery over time trending back to pre-disturbance composition.

Although rapid recovery can be seen as being more desirable than slow directional return, the rapidity of recovery may in reality be less important than the fact that a system returns to its pre-fire state, as variation in reproductive strategies and growth rates will affect the pace of recovery (Falk *et al.*, 2019). To pose and answer such questions for a particular system requires a considerable amount of long-term data, but the questions are amenable to studies investigating the short to medium term impacts of disturbance, if only so as to set up a baseline for further long-term monitoring of ecosystem change (Hinds, 1984). There are a range of studies indicating that disturbance can have either transient or long-term effects and there is a need to discern whether composition is more or less stable under disturbance (Morgan *et al.*, 2014; Strand *et al.*, 2019). Studies on compositional stability under disturbance have been conducted across a wide variety of taxonomic groups and ecosystems including soil bacteria (Weedon *et al.*, 2017); stream communities (Marchant and Dean, 2014) and vascular plants (Belote *et al.*, 2012; Frelich and Reich, 1999; MacDonald *et al.*, 2015; Riginos *et al.*, 2018) – the latter studies involving disturbances including logging, fire and grazing. Studies on the effects of logging and grazing show that there are short-term significant differences over 10 years and that stability in these cases will not necessarily be apparent until many years have passed. However, fire events per se will not necessarily be a major perturbation as the forest communities studied have been found to be both resistant and resilient to fire as natural disturbance, with human modifications to fire regimes more likely to lead to directional and potentially permanent change (Keeley and Pausas, 2019).

The fire response traits of species summarised in Chapter 4 are critical in understanding why a system may either be highly compositionally resistant or resilient or else be prone to compositional change. Species persistence, the persistence niche (Bond and Midgley, 2001), is fostered for example by the ability to resprout, which may maintain composition even where some structural change is occurring, and may maintain ecosystem function and ecosystem services under disturbance regimes, especially climate induced changes to fire regimes (Doherty *et al.*, 2017b; Lawes and Clarke, 2011). Additionally, the relationship between vascular plant species richness and plant community composition - floristics - is complex as in some cases although species richness has not changed. A relevant example is Brindabella Site 91 with 45 species before the fire and 45 species recorded immediately after the fire, but the composition had changed in the short term with only 26 of 45 species recorded before the fire re-recorded after the fire in the first post-fire sample. In other cases, there was an increase in species richness, as was the case with Burrinjuck Site 27, with 15 species recorded before the fire and 55 species recorded immediately after the fire in the first post-fire sample, but with virtually all (14 of 15) species recorded before the fire re-recorded after the fire. Tracking changes in composition through time enables a greater ability to assess detailed changes post-fire compared to using vascular plant species richness alone.

In this investigation of the effects of the 2003 fires on plant community composition in the Australian Alps, the research focus was on potential differences in composition associated with differences in fire severity rather than fire per se. The 2003 fires provided variation in the severity of the disturbance and enable testing whether a more intense disturbance results in greater compositional change or indeed permanent compositional change in the plant communities found in the Australian Alps.

In this chapter, I test the second hypothesis, as outlined in Chapter 1.4, in relation to my second research question: **Does high severity fire result in significant or permanent change to vascular plant species richness and *plant community composition* as compared to low severity fire in subalpine, montane and tablelands plant communities?**

Hypothesis 2: *High severity fire is associated with a greater change in plant community floristic composition as compared with low severity fire and such change occurs for a longer period of time. At the upper end of fire severity, threshold effects induce permanent compositional change in plant communities.*

7.2 Methods

7.2.1 Data

Data collection methods for vascular plant species recorded on sites were as outlined previously in Sections 3.2 and 3.3, and the approach used to investigate vascular plant species richness was as described in Chapter 6, two components of pre- versus post-fire data were used in the multivariate analyses:

a) 1996/1997 (Brindabella) and 1998 (Burrinjuck) pre-fire data and data from the first post-fire sample in 2003 were used to investigate initial pre- and first year post-fire changes and

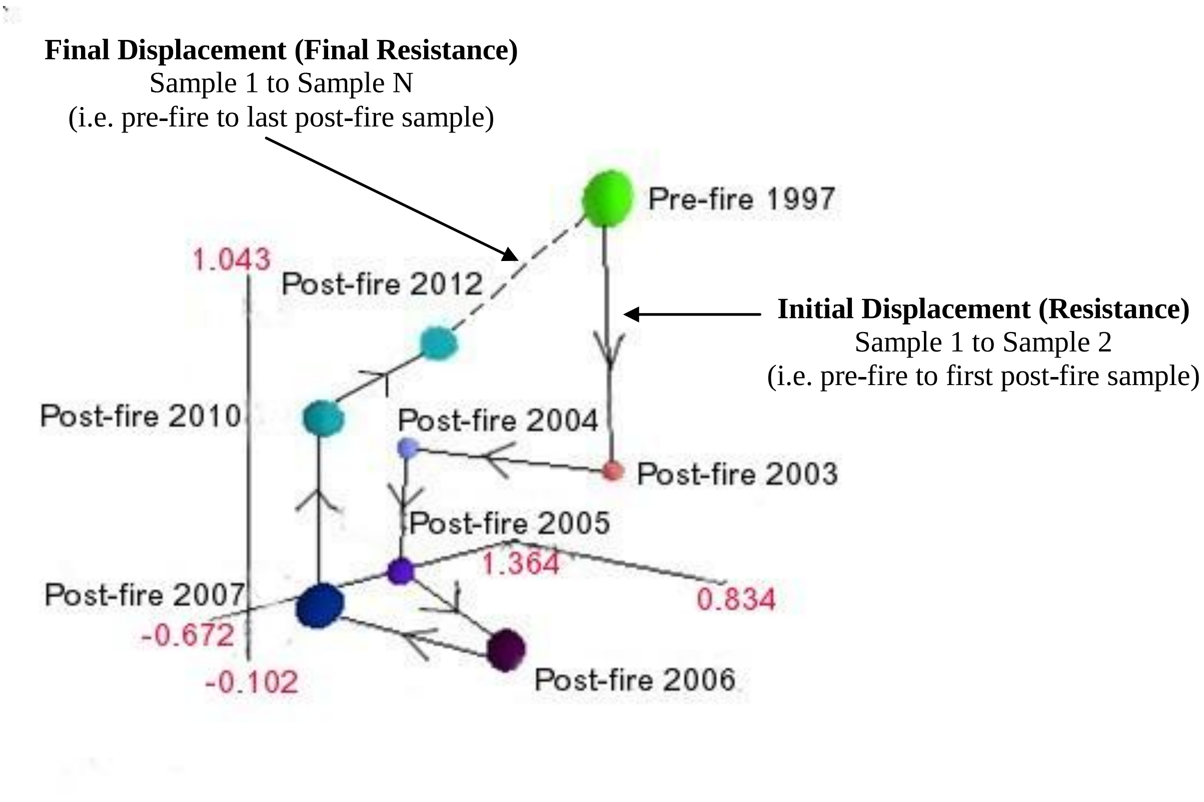
b) for further investigation of changes over time beyond the first post-fire 2003 sample, data from the ongoing subset of 39 sites in Brindabella and 13 sites in Burrinjuck, which were sampled from 2004 and 2012, including the first post-fire sampling in 2003, were used.

7.2.2 Classification and Ordination Analyses

Fire events have the potential to affect the vascular plant composition of sites and, given the large number of species-site-time combinations in long-term data sets, a multivariate approach was used to analyse the data set. Such an approach is amenable to utilising a ‘trajectory’ analysis measuring the degree of dissimilarity within and between sites and assessing whether there are directional, non-directional or long-term changes in compositional trajectories over time (Austin, 1977; Matthews *et al.*, 2013).

For the multivariate analyses undertaken, species cover abundance data were used in their existing 6 point ordinal form (1 = 1 or a few individuals but < 5% cover; 2 = Many individuals but < 5% cover; 3 = 5 - <25% cover; 4 = 25 - <50% cover; 5 = 50 - <75% cover; 6 = 75 - 100% cover) and were not transformed. Dale (1989) suggests that transformation of cover-abundance ordered categorical scales is unnecessary and that such measures provide a robust global ordering of data for multivariate analysis. In particular, the Bray-Curtis measure of dissimilarity used in the analyses (Equation 1) is well suited to untransformed data of this type and, in relation to the multi-dimensional scaling ordination method used (see 7.2.2.2 below) is not sensitive to ordinality. Ricotta and Feoli (2013) also strongly support the direct use of ordinal scales in classification and ordination of vegetation data. The data analyses used the quantitative form of the Bray-Curtis measure and this is the form shown in Equation 1.

To quantify how pre- and post-disturbance change can be measured within and between sites over many samples using an ordination trajectory approach, three metrics were trialled based on the Bray-Curtis dissimilarity values (Figure 7.2). Firstly, the dissimilarity distance between the pre-fire sample and the first post-fire sample can be regarded as an ‘initial displacement’, akin to measuring the *initial resistance* of a site to disturbance. Secondly, the dissimilarity distances of a site as they change over time can be regarded as the ‘travel distance’ of that site through dissimilarity space, akin to measuring *resilience* of the site to disturbance. This method was trialled using two variants: a) ‘radial travel distance’ whereby the distances from the pre-fire sample to each separate post-fire sample are summed and b) ‘total travel distance’ whereby the dissimilarity between each consecutive sample are summed. The time taken for samples of a site to reduce in their dissimilarity, if indeed they have become dissimilar, can be gauged by assessing each site-time sample. Finally, the dissimilarity distance between the pre-fire sample and the last post-fire sample can be regarded as a ‘final displacement’, akin to measuring the *final resistance* of a site to disturbance, which is alternatively a measure of overall s*tability.* Using these concepts, site-time changes between sites burnt at low severity and sites burnt at high severity using mean values of Bray-Curtis distances for each measure by Low-Moderate and High-Very High severity classes, were tested using the method of Laughlin *et al.* (2004) and Benscoter and Vitt (2008).



Final Displacement (Final Resistance)
Sample 1 to Sample N
(i.e. pre-fire to last post-fire sample)

Initial Displacement (Resistance)
Sample 1 to Sample 2
(i.e. pre-fire to first post-fire sample)

Radial Travel Distance = (1997 to 2003) + (1997 to 2004) + (1997 to 2005) + …..(1997 to 2012)

Total Travel Distance = (1997 to 2003) + (2003 to 2004) + (2004 to 2005) +.....(2010 to 2012)

Figure 7.2: An example of an ordination site-time trajectory from 1997 to 2012, showing terminology used in the trajectory analyses. The scale on the ordination axes is derived from standardized association measure distances. The ranges appear as thin black lines and the range values are displayed in red at each end.

While the approach taken by Matthews *et al.* (2013) (Figure 7.1) is a succinct way of summarising trajectory analyses and what they may look like in relation to compositional change over time, in practice, the portrayal of multiple sites over many samples in three dimensions is problematic. These diagrams are very useful in an initial interpretation of site-time groupings, enabling a visualisation of site trajectories through dissimilarity space over time and show that each site occupies a separate discrete part of dissimilarity space. However, they cannot be used to formally test hypotheses in relation to the impact of fire severity on compositional dissimilarity and so a univariate statistical approach using the Bray-Curtis distances was used to test for changes in composition associated with fire severity. These tests are detailed in Sections 7.2.2.1 and Section 7.2.2.2.

The overall steps in undertaking these analyses were a) initially compare the pre-fire and first post-fire samples to investigate if there was change in the allocation of sites to vegetation types (Table 3.1; Appendix 2) using a classification approach according to the original pre-fire vegetation classification; b) test whether differences in this initial pre- and post-fire state were associated with fire severity; c) use ordination of the pre-fire and ongoing post-fire samples to track changes in trajectories over time and d) test whether differences in pre- and subsequent post-fire trajectories were associated with fire severity. This approach is detailed in Section 7.2.2.1 and Section 7.2.2.2 below.

7.2.2.1 Pre-fire Versus First Post-fire Samples

Classification

To compare whether there was any initial change in the post-fire groupings of sites into vegetation types as compared to their pre-fire classification, the 123 site by 572 species matrix for the Brindabella study area and the 33 site x 395 species matrix for the Burrinjuck study area were re-classified using the Windows PATN software package (Belbin, 2000). Dissimilarities were derived for all site-time combinations to explore differences in dissimilarity pre- and immediately post-fire, irrespective of fire severity. Compositional dissimilarity between sites was calculated using the quantitative version of the Bray-Curtis coefficient (Bray and Curtis, 1957; Gauch, 1973) (Equation 1) and an agglomerative hierarchical fusion using flexible unweighted pair group arithmetic averaging (UPGMA) clustering strategy (Austin and Belbin, 1982; Faith *et al.*, 1987) with beta = -0.1 was then used to produce a hierarchical classification. An assessment of changes in the groupings of sites into vegetation types was undertaken by comparing the pre-fire and first post-fire samples for each site within the classification dendrograms. To make the pre- and post-fire classifications comparable in relation to vegetation types (i.e. site groups from the classification), a 14 group cut-off and a 7 group cut-off were used in the Brindabella and Burrinjuck classifications respectively.

Testing for Fire Severity Effects

Comparing mean dissimilarity distances of the pre- versus first post-fire samples was undertaken to enable a quantification of changes in dissimilarity over time, as compared to the qualitative initial dendrogram assessment. A two-sample t-test (paired) using a categorisation of sites into severity classes was used to test if fire severity influenced the composition of pre- and first post-fire samples. These two-sample t-tests (paired) were undertaken using Genstat (Version 17) utilising Bray-Curtis dissimilarity distances between site-time samples to test the following null hypothesis (a sub-hypothesis of Hypothesis 2):

$H0^{2a}$: mean pre-fire to first post-fire dissimilarity distances Low Severity = mean pre-fire to first post–fire dissimilarity distances High Severity (Initial Displacement)

Comparisons were made using fire severity as a factor with two values; a site being either burnt in 2003 at Low-Moderate Severity or else burnt in 2003 at High-Very High Severity. Pre-fire composition was then compared to post-fire composition with the 2003 fires acting as the disturbance event and changes evaluated post-fire in relation to Bray-Curtis dissimilarity values and their change within a site-time group from 2003-2012.

7.2.2.2 Pre-fire Versus Ongoing Samples

Ordination

The analysis of ongoing samples utilised the long-term subset of 52 sites monitored pre-fire and from 2003-2012 post-fire, comprising the 39 site by 485 species matrix for the Brindabella study area and the 13 site x 365 species matrix for the Burrinjuck study area. While the classification and associated dendrogram were appropriate to compare the vegetation classification, and ordination approach is more suited to exploring changes in site-time trajectories post-fire. For ordination of the site data, the same Bray-Curtis metric and clustering strategies were used as per the classification of the site data, and then Semi-strong Hybrid Multidimensional Scaling (SSH) was used to produce an ordination of the data. SSH is a robust method for transforming multivariate ordinal data into interval data, while at the same time reducing dimensionality (Belbin, 1991b). Ordination diagrams showing the position of sites over time in dissimilarity space were produced for each vegetation type (Appendix 8). The Windows PATN package was used to display the output from the SSH MDS ordination as rotating visual 3-D images. Although difficult to portray in two dimensions, it enabled the ordinations to be rotated on-screen so that cluster proximity could be visually evaluated, such that clusters which may appear close together on two axes may in reality be well displaced in dissimilarity spaces.

Testing for Fire Severity Effects

As in the case of the comparison of the pre- and first post-fire samples, mean dissimilarity distances of the pre- and post-fire ongoing samples were compared to quantify changes in dissimilarity over time across all of the post-fire samples. A two-sample t-test (paired) using a categorisation of sites into severity classes was used to test if fire severity influenced the site trajectories. Two-sample t-tests (paired) were undertaken using Genstat (Version 17) utilising Bray-Curtis dissimilarity distances between site-time samples to test a series of four null hypotheses (sub-hypotheses of Hypothesis 2):

$H0^{2b}$: mean pre-fire to first post-fire dissimilarity distances Low Severity = mean pre-fire to first post-fire dissimilarity distances High Severity (Initial Displacement)

$H0^{2c}$: mean Low Severity radial travel distance = mean High Severity radial travel distance

$H0^{2d}$: mean Low Severity total travel distance = mean High Severity total travel distance

$H0^{2e}$: mean pre-fire to seventh post-fire dissimilarity distances Low Severity = mean pre-fire to seventh post-fire dissimilarity distances High Severity (Final Displacement)

Comparisons were made using fire severity as a factor with two values; a site being either burnt in 2003 at Low-Moderate Severity or else burnt in 2003 at High-Very High Severity. Pre-fire composition was then compared to post-fire composition with the 2003 fires acting as the disturbance event and changes evaluated post-fire in relation to Bray-Curtis dissimilarity values and their change within a site-time group from 2003-2012.

7.3 Results

7.3.1 Pre-fire Versus First Post-fire Samples

7.3.1.1 Classification of Pre-fire Versus First Post-fire Samples

There was only a small change between the pre- and post-fire groupings of sites based on the classification of all pre- and post-fire site samples for the Brindabella study area. Only four site pairs (i.e. pre- and first post-fire sample pairs for a given site) out of 123 site pairs (3.2%) occurred in different vegetation groupings post-fire as compared to pre-fire. The floristic composition of pre-fire samples was strongly maintained post-fire, with a strong ‘site identity’ and no clear change in site pair affinities. Of the four site pairs where pre- and post-fire samples were in different groupings, three of the sites (Brindabella Sites 3, 89, 101) occur in Vegetation Type 1: *Eucalyptus dives - Eucalyptus macrorhyncha - Eucalyptus mannifera* (+/- *Eucalyptus rubida*) Open Forest (Species Rich), representing 12.5% of sites in that type. Although these three sites burnt at high severity in 2003 which may have influenced their change in groups, 11 other sites in this vegetation type were also burnt at high severity and showed no pre- to post-fire change in their group affinities. The other site (Brindabella Site 24) occurred in Vegetation Type 8: *Eucalyptus dalrympleana – Eucalyptus pauciflora* Open Forest, representing only 10% of sites in that type that changed groupings. This site was burnt at high severity, but 9 other sites in this vegetation type were also burnt at high severity and did not change groupings pre- to post-fire.

The classification of the Brindabella study area sites shows that for a given site, pre-fire and first post-fire samples were clustered together, mostly as very closely related pairs in the dendrogram and that these site clusters also have a high fidelity to their pre-fire vegetation type. The implication from this comparison of pre-fire to first post-fire samples is that while there was some compositional movement of some sites within related vegetation types, there was no overall compositional movement toward unrelated vegetation types in the Brindabella study area. Any slight compositional differences immediately post-fire between vegetation types were still bounded within broader vegetation groups such as for example wet sclerophyll forest types and dry sclerophyll forest types. The tight clustering of site-time samples pre- and post-fire in the classification suggests that there has not been a major change to composition within sites due to fire.

However, the classification of the Burrinjuck study area samples consisting of the pre-fire samples and the first post-fire samples, showed that there were marked changes between pre-fire samples and first post-fire samples for a given site and also mixing between vegetation types. In the Burrinjuck study area, 16 out of 33 pre- to first post-fire site pairs (48.5%) were in different groups post-fire compared to pre-fire. The differences were such that many of the pre-fire samples across multiple sites clustered in their own distinct group in the dendrogram. Instead of pre- and first post-fire site samples clustering together, many clustered in separate pre- and post-fire groups within vegetation types. The post-fire composition of the Burrinjuck sites is very different to the pre-fire composition across a range of sites and vegetation types, compared with the Brindabella study area. Many sites within a sampling year in different vegetation types are closer to one another compositionally than a particular site is to its post-fire sample. There was no clear pattern in relation to fire severity in this pre- to post-fire variation in Burrinjuck as the sites affected showed a range of fire severities from low to high within vegetation types (Table 7.1).

Given the significant influence of rainfall and season of sampling on apparent vascular plant species richness demonstrated in Chapter 6, the question arises as to whether the differences in the Burrinjuck study area are due to the effects of fire severity, or whether this post-fire difference is actually due to seasonal and rainfall differences affecting composition. Similarly, although there appears to be little compositional change in the Brindabella study area, the potential effect of fire severity may be masked. Bray-Curtis distances from the multivariate analyses were used to calculate mean dissimilarities between low-moderate and high-very high severity sites, pre- and post-fire in order to test the degree to which fire severity influenced the post-fire composition of sites in both study areas.

Table 7.1: Sites where pre- and post-fire groupings were different in the Burrinjuck study area, by vegetation type and scorch category

Site	Vegetation Type	% Scorch Original Canopy Height
Bur_1	15 *Eucalyptus bridgesiana* Open Forest	6
Bur_2	15 *Eucalyptus bridgesiana* Open Forest	25
Bur_9	15 *Eucalyptus bridgesiana* Open Forest	32
Bur_13	16 *Eucalyptus bicostata - Eucalyptus robertsonii - Eucalyptus dalrympleana* Open Forest	24
Bur_15	16 *Eucalyptus bicostata - Eucalyptus robertsonii - Eucalyptus dalrympleana* Open Forest	27
Bur_16	16 *Eucalyptus bicostata - Eucalyptus robertsonii - Eucalyptus dalrympleana* Open Forest	30
Bur_24	16 *Eucalyptus bicostata - Eucalyptus robertsonii - Eucalyptus dalrympleana* Open Forest	100
Bur_31	17 *Eucalyptus bridgesiana - Eucalyptus melliodora* Open Forest/Woodland	26
Bur_29	19 *Eucalyptus goniocalyx - Eucalyptus macrorhyncha* Open Forest	18
Bur_21	19 *Eucalyptus goniocalyx - Eucalyptus macrorhyncha* Open Forest	27
Bur_5	19 *Eucalyptus goniocalyx - Eucalyptus macrorhyncha* Open Forest	100
Bur_23	20 *Eucalyptus dives - Eucalyptus macrorhyncha - Eucalyptus mannifera* Open Forest	33
Bur_33	20 *Eucalyptus dives - Eucalyptus macrorhyncha - Eucalyptus mannifera* Open Forest	37
Bur_28	20 *Eucalyptus dives - Eucalyptus macrorhyncha - Eucalyptus mannifera* Open Forest	50
Bur_26	20 *Eucalyptus dives - Eucalyptus macrorhyncha - Eucalyptus mannifera* Open Forest	66
Bur_30	20 *Eucalyptus dives - Eucalyptus macrorhyncha - Eucalyptus mannifera* Open Forest	100

7.3.1.2 Fire Severity Effects in Pre-fire Versus First Post-fire Samples

Mean dissimilarity distances using all of the pre- versus first post-fire sample data using a two-sample t-test (paired) provided greater insight into the preliminary results from the site classification. The statistical analysis revealed that there was a highly significant difference between the mean dissimilarities between low-moderate and high-very high sites such that low-moderate severity sites had less initial displacement than high-very high severity sites (P<0.001) in the Brindabella study area. In contrast to this important result in the Brindabella data, there was no significant difference between the mean dissimilarities between Low-Moderate and High-Very High sites for Burrinjuck (P = 0.829) (Table 7.2). On the one hand, there was a subtle compositional effect due to fire severity for the Brindabella sites compared to the classification, but the more obvious compositional changes shown for the Burrinjuck sites are not attributable to fire severity. The results from the pre-fire to first post-fire sample comparisons revealed differences in dissimilarity both within sites and between sites, with an influence of fire severity in the Brindabella study area but not in the Burrinjuck study area, despite major floristic changes pre- and post-fire in the Burrinjuck study area.

Table 7.2: Results of two-sample t-tests of effects of fire severity on pre- versus first post-fire samples for all sites

Brindabella Pre-fire to First Post-fire

Sample	Size	Mean	Variance	Standard deviation	Standard error of mean
Low	60	0.3342	0.005919	0.07693	0.00993
High	63	0.4062	0.009949	0.09975	0.01257

Difference of means: -0.0720
Standard error of difference: 0.0160

95% confidence interval for difference in means: (-0.1037, -0.04026)
Test statistic t = -4.49 on approximately 116.06 d.f.
Probability < 0.001

Burrinjuck Pre-fire to First Post-fire

Sample	Size	Mean	Variance	Standard deviation	Standard error of mean
Low	22	0.5160	0.01375	0.1173	0.02500
High	11	0.5063	0.01531	0.1237	0.03731

Difference of means: 0.0096
Standard error of difference: 0.0441

95% confidence interval for difference in means: (-0.08030, 0.09953)
Test statistic t = 0.22 on 31 d.f.
Probability = 0.829

The results of these statistical analyses can be used to interpret the hypotheses as follows:

$H0^{2a}$: mean pre-fire to first post-fire dissimilarity distances Low Severity = mean pre-fire to first post-fire dissimilarity distances High Severity

Brindabella			
Pre-First Post-fire	P = < 0.001	Reject Null	High Severity > Low Severity

Burrinjuck			
Pre-First Post-fire	P = 0.829	Accept Null	High Severity = Low Severity

7.3.2 Pre-fire Versus Ongoing Samples

7.3.2.1 Ordination of Pre-fire Versus Ongoing Samples

The site-time ordinations of the ongoing samples reflected similar patterns to those for the pre- versus first post-fire classifications, but because each site has multiple samples over time, the site-time clusters and patterns of site-time self-similarity are strongly apparent in relation to the dissimilarity trajectories after fire. Vegetation Type 5 and Vegetation Type 8 are examples of site-time clusters for Brindabella, and ‘snapshots’ from the rotated ordinations with site-time clusters identified by site and sample number, show discrete groupings of site-time groups over time. Although there was movement through dissimilarity space over time, that movement was such that site-time clusters remained discrete from one another and did not overlap in dissimilarity space, constrained within separate bounds of dissimilarity space for each site-time group.

Within Vegetation Type 5, sites Brindabella 12, 34, 67, 59, 120 (all Low-Moderate Severity) and 5, 23, 64, 100 (all High-Very High Severity) remain discrete from one another and there was no separate clustering of Low versus High Severity sites (Figure 7.3). Within Vegetation Type 8, sites Brin 029, 061, 090, 104 (all Low-Moderate Severity) and 025, 045, 091 (all High-Very High Severity) also remained discrete from one another and there was no separate clustering of Low versus High Severity sites (Figure 7.4). This pattern of strong site-time similarity is reflected in the ordination diagrams from the other Brindabella ongoing site samples (Appendix 8).

The ongoing site-time trajectories for the Burrinjuck study area also showed similar patterns, but do show a much greater initial displacement of pre- to first post-fire samples as compared to the Brindabella sites. Within Vegetation Type 16, the site-time samples for Burrinjuck site 13 (Low-Moderate Severity) and Burrinjuck site 24 (High-Very High Severity) remained discrete from one another (Figure 7.5) and there is a greater initial displacement in this instance in the case of the High

Severity site as compared to the Low Severity site. However, in Vegetation Type 19, sites Burrinjuck 6 (High-Very High Severity) and Burrinjuck 8 and 21 (all Low-Moderate Severity) showed similar discrete trajectories, but the low severity sites showed a greater initial displacement as compared to the high severity site (Figure 7.6). Overall there is a general pattern of greater initial displacement in the Burrinjuck site-time ordinations (Appendix 8), despite the site-time groupings remaining discrete from one another in a similar pattern to the Brindabella study area site-time trajectories.

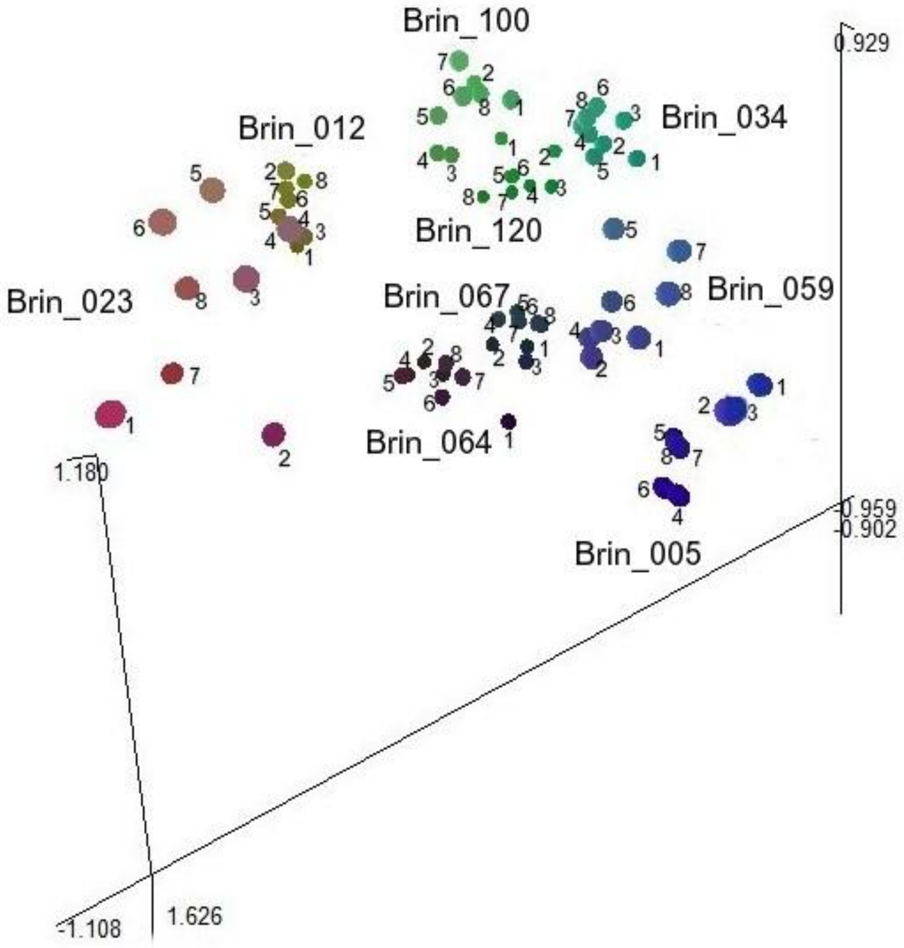


Figure 7.3: SSH ordination showing site-time clusters for Vegetation Type 5, 1996/1997-2012. Site clusters labelled as ‘Brin_XXX’. Time samples numbered from 1 to 8 and in similar colour tones. The scale on the ordination axes is derived from standardized association measure distances. The ranges appear as thin black lines and the range values are displayed at each end.

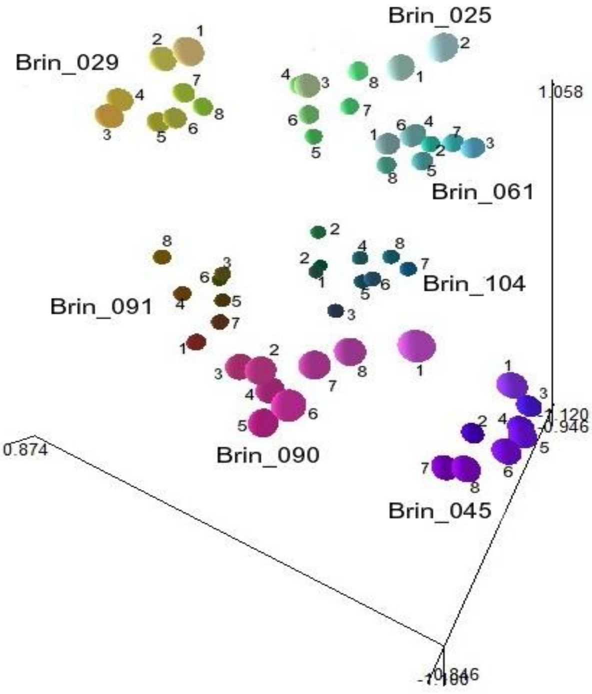


Figure 7.4: SSH ordination showing site-time clusters for Vegetation Type 8, 1996/1997-2012. Site clusters labelled as 'Brin_XXX'. Time samples numbered from 1 to 8 and in similar colour tones. The scale on the ordination axes is derived from standardized association measure distances. The ranges appear as thin black lines and the range values are displayed at each end.

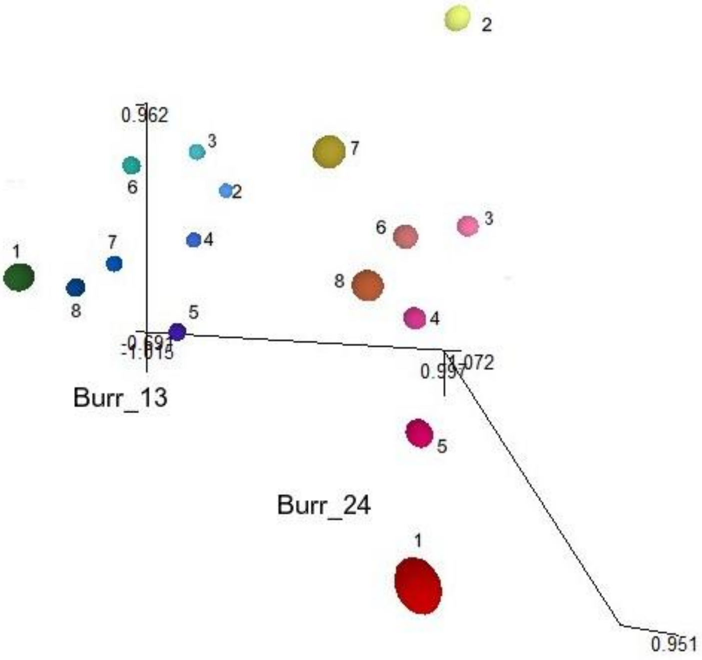


Figure 7.5: SSH ordination showing site-time clusters for Vegetation Type 16, 1998-2012. Site clusters labelled as 'Burr_XX'. Time samples numbered from 1 to 8 and in similar colour tones. The scale on the ordination axes is derived from standardized association measure distances. The ranges appear as thin black lines and the range values are displayed at each end.

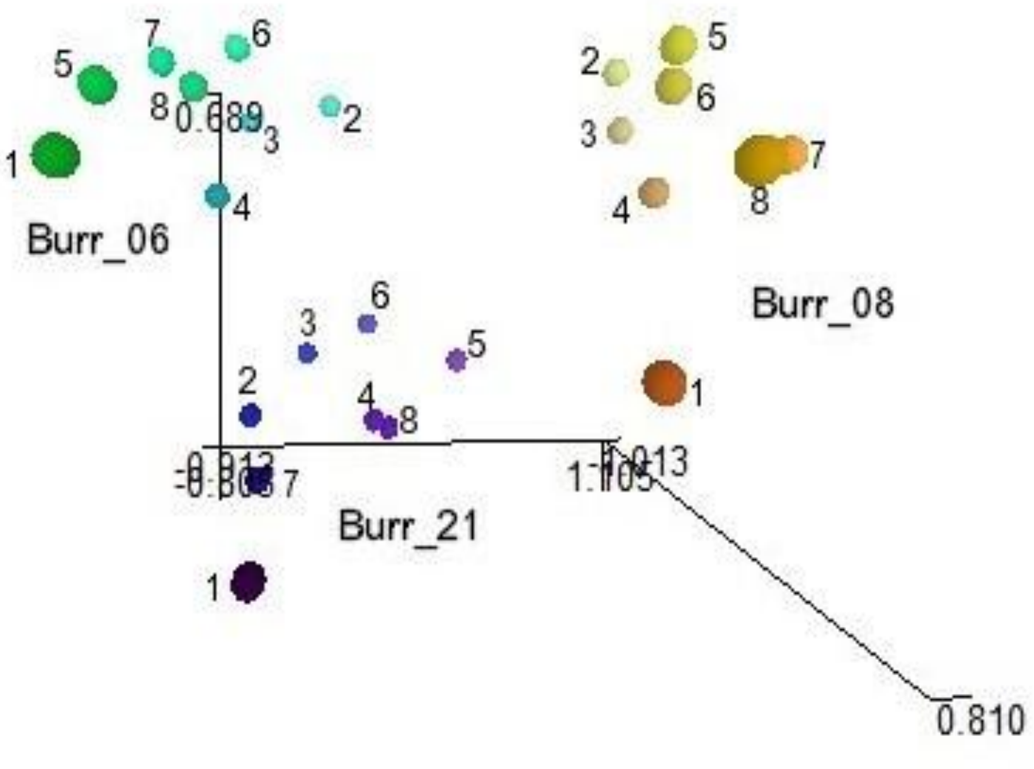


Figure 7.6: SSH ordination showing site-time clusters for Vegetation Type 19, 1998-2012. Site clusters labelled as 'Burr_XX'. Time samples numbered from 1 to 8 and in similar colour tones. The scale on the ordination axes is derived from standardized association measure distances. The ranges appear as thin black lines and the range values are displayed at each end.

7.3.2.2 Fire Severity Effects in Pre-fire Versus Ongoing Samples

In relation to the Brindabella study area, there was a significant difference between the means of Low-Moderate and High-Very High Severity sites in relation to pre-fire to first post-fire distance (P = 0.005), radial Travel l (P = 0.010) and the alternative travel measure total travel (P = 0.016). According to these three measures, dissimilarity was smaller in Low-Moderate severity sites compared to High–Very High Severity sites indicating greater initial compositional change by high severity fire.

However, by the seventh post-fire sample, this difference was non-significant (P = 0.161) indicating that although fire severity caused an initially greater displacement in relation to composition, this compositional difference reduced with time since fire and in less than 10 years post-fire, there was no significant compositional difference in the dissimilarity means between low-moderate and high-very high severity sites. The results of these statistical analyses can be used to interpret the hypotheses for the Brindabella study area as follows:

$H0^{2b}$: mean pre- to first post-fire distance Low Severity = mean pre- to first post-fire distance High Severity

Pre-First Post-fire P = 0.005 Reject Null High Severity > Low Severity

$H0^{2c}$: mean Low Severity radial travel distance = mean High Severity radial travel distance

Radial Travel P = 0.010 Reject Null High severity > Low Severity

$H0^{2d}$: mean Low Severity total travel distance = mean High Severity total travel distance

Total Travel P = 0.016 Reject Null High severity > Low Severity

$H0^{2e}$: mean pre- to seventh post-fire distance Low Severity = mean pre- to seventh post-fire distance High Severity

Pre-Seventh Post-fire P = 0.161 Accept Null High severity = Low Severity

In relation to the Burrinjuck study area, there was no significant difference between the means of Low-Moderate and High-Very High severity sites in relation to pre-fire to first post-fire distance (P = 0.369), radial travel (P = 0.408), the alternative travel measure total travel (P = 0.347) or pre-fire to seventh post-fire sample (P = 0.879). Despite initial large differences in floristic composition between pre- and post-fire samples in the Burrinjuck study area these differences cannot be attributed to the influence of fire severity. The results of these statistical analyses can be used to interpret the hypotheses for the Burrinjuck study area as follows:

$H0^{2b}$: mean pre-fire to first post-fire distance Low Severity = mean pre-fire to first post-fire distance High Severity

Pre-First Post-fire P = 0.369 Accept Null High Severity = Low Severity

$H0^{2c}$: mean Low Severity radial travel distance = mean High Severity radial travel distance

Radial Travel P = 0.408 Accept Null High Severity = Low Severity

$H0^{2d}$: mean Low Severity total travel distance = mean High Severity total travel distance

Total Travel P = 0.347 Accept Null High Severity = Low Severity

$H0^{2e}$: mean pre-fire to seventh post-fire distance Low Severity = mean pre-fire to seventh post-fire distance High Severity

Pre-Seventh Post-fire P = 0.879 Accept Null High Severity = Low Severity

Despite the initial significant impact of fire severity on plant community composition for the Brindabella study area, sites remained most similar to themselves over time and clustered around a bounded area of dissimilarity space such that there was no overlap between sites in terms of compositional change over time. Sites remained nested within vegetation types in a similar fashion. For the Brindabella study area pre- and first post-fire samples, differences due to fire severity were unclear from the classification and ordination diagrams, but formal testing showed that fire severity influences floristic composition between pre- and post-fire samples, but with a diminished effect over time.

Although there were large compositional differences between pre- and first post-fire samples in the Burrinjuck study area, tracking a subset of pre- and post-fire sites over a longer time period and formally testing for severity effects showed that the initial difference was due to differences in the season of sampling, and not due to the effects of fire severity. Fire severity had no significant effect on initial displacement, radial travel, total travel or final displacement for the sites samples in the Burrinjuck study area. Dissimilarity differences over time in the Burrinjuck study area were not due to the effects of fire severity but were influenced by the overall effects of fire, season in which sites were sampled and rainfall. As outlined in Chapter 6, there was a very strong effect of sampling season on vascular plant species richness in the Burrinjuck study area and also a strong response of exotic annuals to rainfall. These factors have a far greater influence than fire severity on the composition of plant communities in the Burrinjuck study area post-fire.

7.4 Discussion

Overall, fire severity did have an effect on floristic composition in the short term in the Brindabella study area, but not in the Burrinjuck study area. The results in the Burrinjuck study area reflect the very strong influence of season of sampling on floristic composition at a point in time. Studies of the seasonal differences in floristic composition for temperate forest ecosystems have not been a focus of research but in semi-arid areas of western NSW, Schultz *et al.* (2014) found variation in season and year to year rainfall had a significant influence on the composition of native grazed pastures. Despite this seasonal and year to year influence, multiple samples at sites over time grouped together. Although semi-arid ecosystems are found in a more variable environment that the current study area, there are similarities in the strong site identity of sites sampled over multiple years across seasonal and rainfall variation. The question arises as to whether fire events necessarily have major ecosystem impacts if they occur within historical disturbance parameters.

Keeley and Pausas (2019) suggest that perturbations to parameters of the fire regime (i.e. changes to fire frequency and fire intensity outside the HRV), as well as changes to fuel types and landscape burning patterns, are the types of perturbations that will result in loss of stability and permanent ecological changes in fire-prone ecosystems. In the case of the Brindabella and Burrinjuck study areas, the historical range of variability remains poorly defined for the plant communities located there due to a lack of detailed data on ecosystem states over the past few hundred years. There are no early explorer records for these areas and, up until detailed vegetation surveys were undertaken in the late 1990s (Doherty, 1997; Doherty, 1998b), there was no description of the structure or composition of the plant communities found there. While fire history records exist, these are only accurate for planned and unplanned fires from the 1990s onwards and previous fire boundaries, especially those from 1938-1939, are largely conjectural (Zylstra, 2006). Of particular relevance is that there is no information regarding fire severity for these historic fires. Some fire scar dating and chronologies have been developed immediately to the south of the Brindabella study area Banks (1989) found that fire frequency in this part of the Australian Alps increased with European settlement and mining activities in the Brindabellas in the mid-1800s, then became less frequent after the 1950s. This historical burning is likely to have affected vegetation structure, at least in the short term, but the effect on floristics, especially the effect on fire-killed plant species, is not known. Whether such species which are rare in the study area now were more common before European colonisation of the Alps is impossible to ascertain, as many of the records of such species are from the late 20th century.

In relation to the diagrammatic conceptualisation of Matthews *et al*. (2013), the results from analysing the compositional dynamics of the study areas suggest that a non-directional saltatory trajectory (Figure 7.1,d) characterises the behaviour of the plant communities. That is, sites over time cluster in a defined area of dissimilarity space with no tendency for a directional trajectory out of that space, but with varying degrees of initial difference between pre-fire and post-fire samples. However, the two study areas exhibit this in different ways. Although there was a greater initial difference in composition between low and high severity sites within vegetation types in the Brindabella study area due to fire severity, there was no such fire severity related pattern detected for the Burrinjuck study area. However, the effects of sampling season and rainfall pre- and post-fire in the Burrinjuck study area (Chapter 6) caused a significant pre- and post-fire compositional difference, but this was not due to fire severity. The Burrinjuck study area also has a higher proportion of exotic species in the regional species pool compared with the Brindabella study area, which is influenced by both fire and rainfall, and these factors far outweigh any effect of fire severity. The composition of the vegetation types in the study areas and also the composition of each site can be regarded as exhibiting stability, but this stability is a quasi-equilibrium, as composition does fluctuate with fire and rainfall disturbances around a core set of species which give sites and vegetation types a stable long-term identity over time. While post-fire secondary succession occurs, it essentially conforms to Egler’s initial floristics model where species that were on site pre-fire re-emerge post-fire either from seed banks or from resprouting. It is the bud bank and soil seed bank component of the total vascular plant species pool that creates transient dynamics in composition and in apparent species richness (Chapter

6), in response to fire and rainfall cues, but there is no evidence from the study areas that this leads to any directional compositional change.

There is evidence from coastal heathland and woodland communities in the Sydney Basin that fire frequency may account for approximately 60% of variation in floristic composition within communities, and that the effects of fire intensity may account for 11% of the floristic variation (Morrison *et al.*, 1995; Morrison, 2002). As the fire frequency in subalpine, montane and tablelands plant communities in the Australian Alps in many areas is significantly lower than that found in the Sydney Basin (Zylstra, 2006), and given the general lack of fire-killed understoreys in the Alps, the effect of variation in fire frequency is more difficult to quantify for these communities. However, the small influence of fire intensity on floristic composition in the Sydney Basin communities up to two years post-fire supports the findings from this study that fire severity has some influence on floristic composition in the first few years post-fire. However, the current study has shown that this effect is transient in the communities studied and that within 10 years post-fire any difference in floristic composition between low and high severity sites is not detectable. Although no comparable studies to the present research exist in the Australian Alps, Knox and Clarke (2012) studied high severity fire in montane communities on the Northern Tablelands of NSW and found no shifts in floristic composition of forests burned at high severity and that the long-term absence of fire up to 30 years in sclerophyllous communities saw no shift to more mesic communities. These patterns are also consistent with Bassett *et al*. (2017) who found that there was no structural convergence between plant communities in gullies and slopes in lower montane forest ecosystems in Victoria burnt at high severity.

More broadly, the results of the current study can be compared to those from long-term studies after large landscape fires, which are few in number. Abella and Fornwalt (2015) investigated 10 years of understorey recovery in relation to long-term dynamics after a ‘mega-fire’ in Colorado, USA. Their analyses found a rapid recovery of plant cover post-fire, with increases in vascular plant species richness and a dominance in the recovery by native species, many of which were ‘legacy’ species (Dale *et al.*, 2005b) which were present before the fire and recovering either by resprouting or from seed banks. However, there was a significant impact on the fire-killed conifer canopy, *Pinus ponderosa* and overall, high severity fire resulted in reduced resistance and reduced resilience compared to low severity fire (Abella and Fornwalt, 2015). Romme *et al*. (2011) summarised 20 years of post-fire research on the ecosystems of the Yellowstone area after the 1988 fires, and their overall conclusion across a range of biotic and abiotic impacts was that the Yellowstone ecosystem was very resilient to fire, suggesting that the ecological effects and responses represent typical fire effects and responses occurring from the Holocene to the present (i.e. the fires were within the Historic Range of Variability). Depending on the ecosystem studied, there is therefore some variability in ecosystem recovery in relation to high severity fire. Reconsidering Pickett and McDonnell's 'law of vegetation dynamics' (Pickett and McDonnell, 1989):

(1) if a site becomes available, or
(2) if species become differentially available at that site, or
(3) if species perform differentially at that site, then
(4) composition and/or the structure of vegetation will change through time.

The key factors in whether vegetation changes after high severity fire event depend on the differential availability and performance of species pre- and post-fire. In the case of the ecosystems studied in this research, there appears to be no differential availability and no obvious differences in performance across the wide range of vascular plant species and vegetation types investigated, except for the slower recovery of some fire-killed canopy dominants as detailed in Chapters 4 and 5. The overall performance of the ecosystems studied can be assessed in relation to the ecosystem stability measures of Pimm (1984) Whitford *et al*. (1999), being:

Stability: the return to an initial state following a disturbance
Resistance: the degree to which a variable has moved from a pre-disturbance state
Resilience: the time taken for a variable to return to a pre-disturbance state
Persistence: the time a variable lasts before it is changed to a new value

In relation to high severity fire events, the Brindabella and Burrinjuck ecosystems display:

- high relative **stability**, with some transient change occurring within the bounds of a confined area of dissimilarity space
- a high **resistance** to permanent compositional change despite initial transient changes
- rapid rates of **resilience**, with ecosystems recovering in <10 years post-fire
- high **persistence**, with sites not undergoing permanent change to new values over a 10 year period

Although vegetation structure is affected by high severity fire it is not apparent that such structural dynamics in the study area affect composition, even where structural recovery may take a longer period of time to recover than floristic composition.

Even though the fire event of 2003 was a large landscape fire that burnt at high severity in many areas, it did not fundamentally change the composition of the plant communities found in the study areas. Although there are changes in vascular plant species richness (Chapter 6) and to plant community composition post-fire in relation to the overall effects of fire and rainfall, fire severity per se is not a major driver of ecosystem changes in the study areas. The implication of the responses found in the study area reinforces the idea that the type of fire event seen in 2003 is not unprecedented, which is consistent with the simple hypothesis of White and Jentsch (2001) that disturbances with historical precedence are more likely to be responded to by an adapted biota than are novel disturbances.

The implication of this lack of directional compositional change is that ecosystems experiencing infrequent large landscape fires at high severity have evolved traits that enable persistence over long periods of evolutionary time. This gives weight to the proposition that it is not fire severity per se or a single fire event that is likely to lead to directional shifts in community composition, but the interaction of fire severity with fire frequency which can lead to structural and compositional change, especially in those communities dominated by fire-killed vascular plant species.

7.5 Conclusion

Hypothesis 2, that: *High severity fire is associated with a greater change in plant community floristic composition as compared with low severity fire and such change occurs for a longer period of time. At the upper end of fire severity, threshold effects induce permanent compositional change in plant communities* was only partially supported by the analyses. In the Brindabella study area, there was greater change in plant community composition in areas affected by high severity fire and this effect lasted for some years post-fire but became non significant within 10 years post-fire No threshold effects due to high severity fire were detected that induced permanent compositional change in the Brindabella sites over the time period of the research. No short term or permanent changes were detected in the Burrinjuck study area, but the influence of season of sampling influenced community composition. The analyses undertaken do not support the proposition that high severity fire results in significant or permanent change to plant community composition as compared to low severity fire in subalpine, montane and tablelands plant communities. Essentially, although some floristic change can be detected pre- and post-fire, these changes in floristic composition are transient rather than permanent.

While in some plant communities there were initially greater differences between low and high severity sites, these differences diminish over time such that by 10 years post-fire, there is no difference in compositional difference between low and high severity sites. This is not to say that there was no compositional flux over time both in relation to fire and climate, but that for the duration of the sampling from 1997–2012, there was a core ‘compositional identity’ to all sites. Neither the sites, nor the vegetation types within which they form a component, moved in a directional trajectory toward a permanent change in composition.

The lack of long-term directional change in the study areas suggests that infrequent high severity fire events are part of a long-term fire regime to which vascular plant species have adapted over time. However, structure, vascular plant species richness and plant community composition are known to be affected by changes in fire frequency, and also the increase in frequency of high severity fire events. The overall implications of my research and in particular, the implications of an increasing frequency of high severity fire events under a changing climate, are discussed in Chapter 8.

Chapter 8: Synthesis of Research

8.1 Research Summary

My research set out to address two fundamental questions:

1) What is the response to high severity fire of vascular plant species found in subalpine, montane and tablelands plant communities?

2) Does high severity fire result in significant or permanent change to vascular plant species richness and plant community composition as compared to low severity fire in subalpine, montane and tablelands plant communities?

To explore this question I proposed two hypotheses:

Hypothesis 1*: High severity fire is associated with a greater increase in plant species richness within plant communities post-fire as compared to low severity fire and the duration of this effect is positively associated with fire severity.*

Hypothesis 2*: High severity fire is associated with a greater change in plant community floristic composition as compared with low severity fire and such change occurs for a longer period of time. At the upper end of fire severity, threshold effects induce permanent compositional change in plant communities.*

In relation to the first research question, the fire responses of over 400 vascular plant species in 21 plant communities were documented, with 24% of species being fire-killed and responding via canopy and soil seed banks and 76% via resprouting. There was no loss of plant species from the study areas and all vascular plant species recovered rapidly post-fire. Vegetation structure was affected by fire severity, but it is fire severity interacting with fire frequency that has the potential to change vegetation structure, particularly via the loss of structurally dominant obligate seeder species. While the majority of the 23 rare or structurally important fie-killed species recovered rapidly post-fire, two structural dominants, *Eucalyptus delegatensis* subsp. *delegatensis* and *Leptospermum micromyrtus* remain potentially vulnerable to a second fire in the short to medium terms as they are slow to re-accumulate canopy seed banks. However, in the case of *Eucalyptus delegatensis* subsp. *delegatensis,* precocious early maturing individuals may buffer this species against loss from short inter-fire intervals.

In regard to the second research question and changes in vascular plant species richness post-fire, Hypothesis 1 was not supported. The major influence on vascular plant species richness was a general increase in species richness post-fire irrespective of severity, with a decline associated with time since fire. However, this peak and decline response of vascular plant species richness post-fire is modified and varies over time in relation to rainfall, decreasing in drier years and returning to higher levels during years of average or above average rainfall. Vegetation type is also a major influence on vascular plant species richness with variation over time both within and between vegetation types in relation to time since fire and rainfall.

In relation to the second research question and plant community dynamics, Hypothesis 2 is partially supported, but only for the Brindabella study area and not for the Burrinjuck study area. While there was an initial difference in dissimilarity between low and high severity sites in the Brindabella sites and greater dissimilarity in the short to medium term, this dissimilarity diminished over time and there was no evidence of permanent compositional change in high severity sites, as the difference between pre- and post-fire and low and high severity was not significant 9 years after the 2003 fire event. The significant response immediately post-fire was partially due to differences in species composition pre- and post-fire but was also due to changes in the abundances of species pre- and post-fire. In high severity sites, there was an increase in the density of many dominant shrub species adding to differences in the short term. As shrub layers senesced, both changes in composition and changes in abundance reduced and by approximately 10 years post-fire any compositional difference was not detectable. The importance of season of sampling is also crucial in interpreting long-term analyses of pre- and post-disturbance monitoring data. For example, as the original pre-fire sampling in the Burrinjuck study area was undertaken in late summer-early autumn rather than spring, initial pre- to post-fire differences were significant but were due to this seasonal sampling difference rather than to fire severity.

In summary, high severity fire, as compared to low severity fire, in subalpine, montane and tablelands ecosystems in the Australian Alps of southeastern Australia has only a transient effect and then, only in some plant communities, with vascular plant species richness varying over time irrespective of whether fire occurred at low or high severity and at 9 years post-fire there was no evidence of significant permanent change in the composition of the plant communities in either the Brindabella or Burrinjuck study areas.

8.2 Implications of the Research

8.2.1 Long Term Monitoring and Sampling

This research has shown how long-term data sets and monitoring are critical to an understanding of how species richness, community composition and structure change over time in relation to distinguishing between the influences of fire, irrespective of severity, from other background environmental influences such as rainfall. Long term pre- and post-fire monitoring in the two study areas shows that because of rapid post-fire recovery of individuals present pre-fire, plant community composition is robust, even after high severity fire. While there are some initial compositional differences post-fire between low and high severity sites in some communities, these differences diminish rapidly with time since fire. However, because of the interaction of an overall time since fire decline in apparent vascular plant species richness - irrespective of fire severity - with variability in rainfall over time, species richness and community composition will nonetheless vary from year to year. Vegetation structure is affected by fire severity, but it is fire severity interacting with fire frequency that has the potential to change vegetation structure, particularly via the loss of structurally dominant obligate seeder species.

This research has shown that the floristic composition that characterises plant communities in the study areas is buffered against change by the recovery of a persistent local (site) and regional (study area) vascular plant species pool which has been selected for over long periods of time. Occasional high severity fire in plant communities should therefore be seen in the broader context of the longer evolutionary history of drought and fire in Australia, which have driven plant adaptations over millennia (Hill and Jordan, 2016). The ability of plant species to rapidly recover in situ is driven primarily by the resprouting traits that the majority of species exhibit, but even where species are reliant on canopy and soil seed banks, recovery of pre-fire species is also rapid for the majority of species studied. Although fire severity does influence plant species richness and plant community composition, it sits within a matrix of other major environmental influences such as climate.

The results have implications not only for pre- and post-fire sampling but also for vegetation sampling and monitoring in general, as it is clear that any site sample will only reflect the apparent vascular plant species richness and plant community composition at a point in time and will not reflect the total potential species richness of a site. This is particularly important when considering the survey and establishment of long-term monitoring sites, with sequences of samples taken over time, but where the initial sample becomes a reference point. The full expression of vascular plant species composition and richness at a particular site will only be known as a result of many samples reflecting the effects of variation in climatic and disturbance variables. Any starting sample cannot be regarded as an absolute measure of the state of an ecosystem but is only one sample in time across a broad range of potential states. Even when controlled for spatial variation, site samples are only snapshots in time in relation to patterns of vascular plant species richness and community composition. While Historical Range of Variability approaches aim to define historical compositional or structural bounds to variation in communities to assist in determining directional rather than transient change, for many communities, there are at best only very broad structural historical descriptions with no detailed associated compositional details to define historical bounds (Doherty, 2015).

Nonetheless, long term monitoring has to start at a point in time and therefore to make the most of data collected, post-fire monitoring once initiated should be designed to be able to detect short term transient dynamics from long term directional change. Trajectory analysis as used in this study is one approach that can track compositional dynamics over time and therefore can be used to detect compositional shifts. However, the ability to interpret changes in composition will also rely on a basic understanding of species' response to fire, both to events and to changes in components of the fire regime. Over time, integrating species level and community level ecological knowledge can provide conceptual models that give predictive ability for managers in the form of, for example, decision thresholds for intervention (van Wilgen *et al.*, 1998; van Wilgen *et al.*, 2007) or predictive state and transition models (Colloff *et al.*, 2016), all forming part of a ‘learning by doing’ approach (Walters and Holling, 1990).

8.2.2 Environmental and Geographical Relevance

The current research has focused on areas in the northern parts of the Australian Alps. Similar subalpine, montane and tablelands plant communities occur throughout the southern extent of the Australian Alps through Kosciuszko National Park and into Victoria (Coates, 2015; Forbes *et al.*, 1982; Gellie, 2005; Gullan *et al.*, 1985). Although considerable past research has focused on the alpine communities of the NSW and in Victoria, there is still al lack of post-fire studies in the subalpine, montane and tablelands plant communities of both states. Although there is an expectation that the fire response of species will be similar in similar environments, there is nonetheless some variability in response in at least some species over their geographical ranges (Gill and Bradstock, 1992; Vivian *et al.*, 2010). There is also still ambiguity in the literature as to whether fire response traits in some species also vary with fire severity. At this point in time, the post-fire recovery of plant communities in more southern parts of the Australian Alps, dominated by the same species, can be assumed to be similar and at least at the plant community level as there is at least some summary information for southern communities as to their likely response to both low and high severity fire (Chambers, 1977; Cheal, 2010). The current study has added to the growing body of evidence that high severity fire events should not be regarded as being inherently more damaging than low severity fire events in many Australian ecosystems (Bradstock, 2008).

8.2.3 Changing Fire Regimes

Although species will successfully recover from relatively infrequent fire events because of their re-sprouting or re-seeding traits, there are limits to recovery if inter-fire intervals become too short and either seed banks or bud banks become exhausted (Keith, 1996). Depending on the specific population ecology of each species, if only a small number of species are affected, this is unlikely to lead to major compositional change in plant communities. Major structural and compositional shifts are more likely to occur if structurally dominant species are unable to recover and the overstorey-understorey dynamic is fundamentally changed (Specht and Morgan, 1981). The interval-dependent eucalypt *Eucalyptus delegatensis* subsp. *delegatensis* has been a focus of concern among forest managers in relation to too-frequent fire return times (Bassett *et al.*, 2015; Fagg *et al.*, 2013). However, the main focus to date has been on the implications of loss of these species either as commercial timber species or else as important structural components of ecosystems, but the effects of the loss of such canopy dominant species in relation to plant community compositional change has not been examined. Although it is possible from the existing literature to develop state-transition models for groups such as fire-killed ash eucalypts based on the expected changes under different fire frequencies, and to model potential structural change, the overall impacts on species richness and plant community composition are unclear (Colloff *et al.*, 2016; McCarthy *et al.*, 1999; Williams *et al.*, 2009). Under more extreme future changes in fire frequency, if large areas of interval-dependent ash eucalypt species fail to regenerate, there may need to be consideration given to planned adaptation interventions, such as re-seeding with resprouter species, if ecosystem services are to be maintained under climate change (Bassett *et al.*, 2015; Doherty *et al.*, 2017b).

While structural change through loss of dominant canopy species due to an increase in fire frequency is essentially predictable for interval-dependent species, there is emerging evidence that recent recurrent fires in southeastern Australia are beginning to impact the ability of some resprouting eucalypt canopy dominants to recover and survive fire. In the Victorian Alps, multiple fire events have caused adult tree death and a lack of recruitment of the resprouting subalpine species *Eucalyptus pauciflora,* resulting in areas that were subalpine woodland effectively becoming grassland (Fairman *et al.*, 2017; Fairman *et al.*, 2018). Such high frequency fire events affecting the ability of resprouting eucalypt species to recover are of great concern as even planned adaptation interventions may be ineffective. It is under these scenarios that overall plant community composition may shift due to the change in structural dominant species with consequent changes to light and water regimes and also potential negative or positive feedbacks with the local fire regime. Ultimately, the effects of the 2003 fires cannot be fully assessed until the next fire event occurs. That is, the long-term dynamics of plant communities are dependent on the sequence of multiple fire events over time, the fire regime, and hence only long-term monitoring of plant populations will give an indication of whether significant change is occurring. However, as time since fire increases and plant populations have re-accumulated their biomass and reached reproductive maturity, the likelihood of significant floristic or structural change decreases. Secondary succession in general, but particularly that occurring in resprouter-dominated communities, in many ways challenges the notion of vegetation dynamics. Simply put, the floristic composition in such communities is not ‘dynamic’, if dynamic refers to permanent change. The concept of a ‘transient dynamic’ seems a more appropriate way to think about how post-fire secondary succession operates, with some flux in species richness, floristic composition and structure post-fire and in relation to rainfall, but with no overall change in plant community affinities. However under changing climatic conditions and associated changes in fire regimes, the possibility of transformational change in forest communities cannot be precluded (Burns *et al.*, 2015; Colloff *et al.*, 2016; Keith and Myerscough, 2016; Mok *et al.*, 2012).

Peet’s (Peet, 1992) differentiation of primary succession being a reflection of long-term environmental change and secondary succession reflecting population processes can be used to frame how ecosystems may change over the long term. If the situation represented by the response after the 2003 fires can be regarded as transient dynamics, then this can be conceptualised as post-fire structure and composition returning to their pre-fire state rapidly after episodic fires, even allowing for some slight changes in structure and floristics over the longer term, essentially a long-term primary succession (Figure 8.1a). As the effects of climate change begin to impact on ecosystems through changes in temperature and rainfall, there may be detectable changes in floristic composition or vegetation structure as a transition begins from one plant community to another, but with some similarity to the previous community in the medium term (Figure 8.1b). However, as climate interacts with fire regimes in more extreme ways, with for example increases in the frequency of large- area high severity fires, one can conceptualise moving to abrupt event driven change, where plant communities become significantly different and novel ecosystems emerge after extreme fire events (Figure 8.1c).

Given the longevity of eucalypts - many hundreds of years - and the strong resprouting ability of most taxa, there may be some buffering effect against rapid change, at least in structure if not floristics (Doherty *et al.*, 2017b). However any move from transient dynamics to abrupt change may not necessarily move through an interim period of more accelerated and gradual change, as can be seen in the deaths of *Eucalyptus pauciflora* in the Victorian Alps after the accumulated impact of multiple fire events in rapid succession over a relatively short time period (Fairman *et al.*, 2016). Of course, there is uncertainty in relation to the degree to which these rapid changes are unidirectional and permanent, but if they are a sign of what may occur in the Australian Alps as temperature increases and precipitation decreases, then the resprouting *Eucalyptus pauciflora* plant communities may have as much potential for significant structural and floristic change as the fire-killed *Eucalyptus delegatensis* subsp. *delegatensis* plant communities.

8.2.4 Future Fire Management

The rapid and full recovery of the plant species and plant communities found in the Brindabella and Burrinjuck study areas after the 2003 fire event has enabled reserve managers to focus rehabilitation interventions on communities such as the subalpine and alpine bogs further south of the study areas, that were greatly impacted by the 2003 fires (Good, 2004). Overall, the subalpine, montane and tablelands forest and woodland communities studied recover rapidly after high severity fire and appear to be highly resilient. As fire has generally been infrequent in many parts of the study area, greater than 60 years in many parts of Brindabella and 30 years in Burrinjuck, the impact of recurrent fire and the interaction of planned and unplanned fire are focus areas which will benefit from the monitoring of the existing site network into the future. Post-2003, there is a greater focus on planned fires for fuel reduction both in the study areas and in adjacent parts of the Australian Capital Territory.

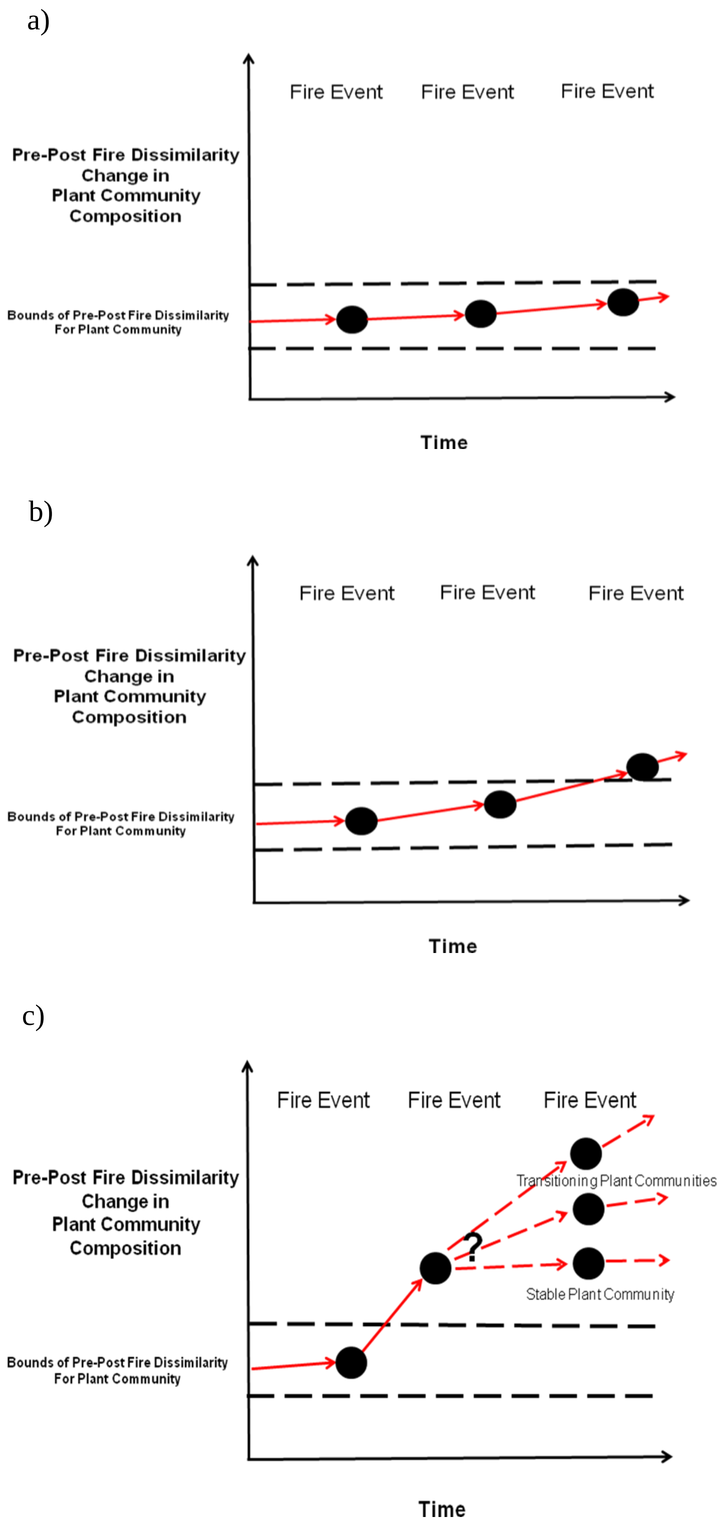


Figure 8.1: Potential changes in plant community composition as measured by post-fire dissimilarity, with increasing frequency and intensity of fire events. a) transient dynamics against a background of slow gradual change; b) transition from transient dynamics to accelerated medium-term climate-driven change; c) move from climate-driven change to rapid event-driven change with threshold effects and state-transition (e.g. climate-fire regime interaction).

Given the known plant and community responses to high severity fire, there is unlikely to be any greater impact from lower severity fires, in and of themselves as fire events. However, there is likely to be an increase in fire frequency in some areas compared to past historical intervals and it is important that the interaction between recent past high severity fire and more frequent low severity fire be monitored as an adaptive management process (England *et al.*, 2004; Gill, 2008; Kitchin, 2008). While fire agencies generally use time since fire as a guide to inter-fire interval management, it is not sufficiently informative on its own for ecological fire management and can only be a meaningful measure when it is coupled with, and qualified by, fire severity data supported by plant, animal and environmental response data (Bassett *et al.*, 2017; Gordon *et al.*, 2017). Additionally, the long-term effects of the interaction between fire frequency and fire intensity on fuel dynamics remains poorly known for many vegetation types and is fundamental to an understanding of how vegetation dynamics and fuels will interact under a changing climate (Collins *et al.*, 2019; Ruthrof *et al.*, 2016).

8.3 Further Research

Although there have been significant studies in southeastern and southwestern Australia on the structural dynamics of dominant canopy species after recent high intensity large landscape fire events including 2003 and after (Etchells *et al.*, 2020; Fairman *et al.*, 2017; Lindenmayer *et al.*, 2011; Smith *et al.*, 2016), assessing post-fire compositional dynamics using full floristics at a site is less studied in Australia than for example, in the USA. The research undertaken after the 2003 fires in the Brindabella and Burrinjuck study areas has shown the importance of detailed floristic data collected over many years, but even there, 10 years post-fire data is still short-term in relation to the lifespan of many of the plants studied. Unfortunately, there was a lost opportunity after the 2003 fires to set up long term monitoring in other vegetation types across the Alps and also more generally in southeastern Australia, irrespective of whether pre-fire measurements had existed prior to the fires.

The implications of a lack of long term monitoring have been highlighted again recently by the extensive fires that occurred in southeastern Australian in the summer of 2019-2020. The extent, if not the severity, of these ‘mega-fires’ is regarded as unprecedented (Australian Academy of Science, 2021; Nolan *et al.*, 2020) and across a range of heath, dry sclerophyll forest, wet sclerophyll forest and rainforest communities the same questions on the impacts of high severity fire are being asked in the absence of long-term monitoring. While it is to be expected that there will be a flurry of short-term research projects based on post-fire responses as a result of these fires, time will tell if any long-term research projects result. Nonetheless, based on the Brindabella and Burrinjuck study area results, it would be predicted that most, if not all of the impacted vegetation communities will recover to their pre-fire state with the majority of species resprouting after fire and those that are fire-killed recovering from canopy and soil seed banks.

Such mega-fire events are predicted to increase with the effects of climate change (Adams, 2013; Millar and Stephenson, 2015) and as and when they do, more accurate measures of carbon flux become necessary for carbon accounting. In contrast to many forested ecosystems in the northern hemisphere which are dominated by coniferous vegetation which is fire-killed during intense wildfire events, significant areas of Australia's forest and woodlands are structurally dominated by eucalypts and understorey species which will resprout after high severity fire. Although carbon is lost from the soil, understorey and canopy during such intense wildfire events, the resprouting capabilities within these forested ecosystems enable significant areas to recover biomass at a rapid rate post-fire and may provide a buffering mechanism against the direct and indirect effects of climate change.

Taken at the broader ecosystem level, the rapid recovery of green biomass in resprouting forests and woodlands means that although carbon is lost in the short term during a fire event, it is likely to rapidly recover post-fire as epicormic resprouting reforms a photosynthetic layer, albeit in the short term held along the trunks of trees rather than as an upper canopy. Work on tree water use in dry sclerophyll tablelands forests in Victoria which were resprouting after high severity fire, has shown that water use returns to pre-fire levels within 3 years (Gharun *et al.*, 2013; Nolan *et al.*, 2014; Nolan *et al.*, 2015; Turnbull *et al.*, 2014). There is an implication in this finding that if water use has equilibrated rapidly post-fire, then the green biomass overall is likely to have recovered to levels similar to their pre-fire levels and hence carbon loss from such forest ecosystems after high severity fire is only transient, rapidly recovering as a function of the resprouting species characterising the plant community.

This very rapid recovery of leaf area in these forests raises broad issues that need to be addressed regionally and globally when consideration is given to potential carbon loss from wildfire and the rate of carbon sequestration post-fire:

- *is the loss of carbon significantly less in resprouter dominated plant communities as compared to fire-killed communities for a given fire intensity?*
- *is the rate of carbon re-accumulation more rapid in resprouting communities?*
- *can there be an initial increase over pre-fire levels in carbon storage as foliage recovers in resprouting species?*

Answers to these questions would enable more accurate modeling of carbon loss vulnerability classes, when coupled with vegetation community type or functional type mapping into the future. So far, examples of dominant eucalypt canopy species that currently resprout losing that capacity from frequent fire are relatively small in area (Fairman *et al.*, 2018). If large areas of forest that are currently able to resprout did become fire-killed from recurrent fire, then this would have major ecosystem impacts. At this point, there is still debate as to the degree and rate that resprouter forests may be impacted, given the robust resprouting responses of many eucalypts across a range of environments to high severity fire (Collins, 2019). The research undertaken in the Brindabella and Burrinjuck study area has shown that for isolated high severity fire events, the resprouting ability of eucalypt canopy species is impressive, and that even the fire-killed *Eucalyptus delegatensis* subsp. *delegatensis* may yet be able to cope with increases in fire frequency due to its precious flowering ability. The utility of this latter trait has now been put to the test in Namadgi National Park in the ACT, where many areas that burnt at high severity in January 2003 also burnt again at high severity in January 2020. There are now stands of *Eucalyptus delegatensis* subsp. *delegatensis* that have been burnt at high severity twice in 17 years (i.e. stands fire-killed in 2002 and again in 2020. Given the species requires cold stratification and does not germinate until summer-autumn (Chapter 5), the first seedlings if they emerge will only begin to be visible in the autumn of 2021. Further research awaits…..

References

Abella, S. R. and Fornwalt, P. J., 2015. Ten years of vegetation assembly after a North American mega fire, *Global Change Biology,* **21**: 789-802.

Adams, M. A., 2013. Mega-fires, tipping points and ecosystem services: Managing forests and woodlands in an uncertain future, *Forest Ecology and Management,* **294**: 250-261.

Adler, P. B. and Levine, J. M., 2007. Contrasting relationships between precipitation and species richness in space and time, *OIKOS,* **116**: 221-232.

Adomeit, E. M., Austin, M. P., Hutchinson, M. F. and Stein, J. L., 1977. *Annual Mean Rainfall and Temperature Surfaces and Contour Maps of South-eastern Australia. Technical Memorandum 87/15,* CSIRO Division of Water Resources Research, Canberra.

Agee, J. K., 1993. *Fire Ecology of Pacific Northwest Forests,* Island Press, Washington.

Agee, J. K., 1998. The landscape ecology of western forest fire regimes, *Northwest Science,* **72**: 24-34.

Alexander, M. E. and Cruz, M. G., 2012. Interdependencies between flame length and fireline intensity in predicting crown fire initiation and crown scorch height, *International Journal of Wildland Fire,* **21**: 95-113.

Allen, C. D., Macalady, A. K., Chenchouni, H., Bachelet, D., McDowell, N., Vennetier, M., Kitzberger, T., Rigling, A., Breshears, D. D., Hogg, E. H., Gonzalez, P., Fensham, R. J., Zhang, Z., Castro, J., Demidova, N., Lim, J.-H., Allard, G., Running, S. W., Semerci, A. and Cobb, N., 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests, *Forest Ecology and Management,* **259**: 660-684.

Anderson, J. E. and Romme, W. H., 1991. Initial floristics in Lodgepole Pine (*Pinus contorta*) forest following the 1988 Yellowstone fires, *International Journal of Wildland Fire,* **1**(2): 119-124.

Anderson, J. E., Romme, W. H., Meyer, G., Knight, D. H. and Wallace, L., 1999. Yellowstone fires, *Science,* **283**: 175-176.

Archibold, O. W., 1995. *Ecology of World Vegetation,* Chapman & Hall, London.

Arnold, M. L., 2008. *Evolution Through Genetic Exchange,* Cambridge University Press, Cambridge.

Ashton, D. H., 1976. The development of even-aged stands of *Eucalyptus regnans* F. Muell. in Central Victoria, *Australian Journal of Botany,* **24**: 397-414.

Ashton, D. H., 1981a. The ecology of the boundary between *Eucalyptus regnans* F. Muell. and *E. obliqua* L'Herit. in Victoria, *Proceedings of the Ecological Society of Australia,* **11**: 75-94.

Ashton, D. H., 1981b. Fire in tall open-forests (wet sclerophyll forests), In *Fire and the Australian Biota* (Eds, Gill, A. M., Groves, R. H. and Noble, I. R.) Australian Academy of Science, Canberra, pp. 339-366.

Ashton, D. H., 1981c. Tall open-forests, In *Australian Vegetation* (Ed, Groves, R. H.) Cambridge University Press, Cambridge, pp. 121-151.

Ashton, D. H., 2000. The Big Ash forest, Wallaby Creek, Victoria - changes during one lifetime, *Australian Journal of Botany,* **48**: 1-26.

Ashton, D. H. and Martin, D. G., 1996. Regeneration in a pole-stage forest of *Eucalyptus regnans* subjected to different fire intensities in 1982, *Australian Journal of Botany,* **44**(4): 393-410.

Ashton, D. H. and Sandiford, E. M., 1988. Natural hybridisation between *Eucalyptus regnans* F. Muell. and *E. macrorhyncha* F. Muell. in the Cathedral Range, Victoria, *Australian Journal of Botany,* **36**: 1-22.

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

Auld, T. D., Keith, D. A. and Bradstock, R. A., 2000. Patterns of longevity of soil seedbanks in fire-prone communities of south-eastern Australia, *Australian Journal of Botany,* **48**: 539-548.

Auld, T. D. and O'Connell, M. A., 1991. Predicting patterns of post-fire germination in 35 eastern Australian Fabaceae, *Australian Journal of Ecology,* **16**: 53-70.

Austin, M. P., 1977. Use of ordination and other multivariate descriptive methods to study succession, *Vegetatio,* **35**(3): 165-175.

Austin, M. P., 1980a. An exploratory analysis of grassland dynamics: An example of a lawn succession, *Vegetatio,* **43**: 87-94.

Austin, M. P., 1980b. Searching for a model for use in vegetation analysis, *Vegetatio,* **42**: 11-21.

Austin, M. P., 1985. Continuum concept, ordination methods and niche theory, *Annual Review of Ecology and Systematics,* **16**: 39-61.

Austin, M. P., 1987. Models for the analysis of species' response to environmental gradients, *Vegetatio,* **69**: 35-45.

Austin, M. P., 1990. Community theory and competition in vegetation, In *Perspectives on Plant Competition* (Eds, Grace, J. B. and Tilman, D.) Academic Press, San Diego, pp. 215-238.

Austin, M. P., 2005. Vegetation and environment: Discontinuities and continuities, In *Vegetation Ecology* (Ed, van der Maarel, E.) Blackwell Publishing, Oxford, pp. 52-84.

Austin, M. P. and Belbin, L., 1981. An analysis of succession along an environmental gradient using data from a lawn, *Vegetatio,* **46**: 19-30.

Austin, M. P. and Belbin, L., 1982. A new approach to the species classification problem in floristic analysis, *Australian Journal of Ecology,* **7**: 75-89.

Austin, M. P. and Cunningham, R. B., 1981. Observational analysis of environmental gradients, *Proceedings of the Ecological Society of Australia,* **11**: 109-119.

Austin, M. P., Cunningham, R. B. and Good, R. B., 1983a. Altitudinal distribution of several eucalypt species in relation to other environmental factors in southern New South Wales, *Australian Journal of Ecology,* **8**: 169-180.

Austin, M. P., Cunningham, R. B. and Wood, J. T., 1983b. The subgeneric composition of eucalypt forest stands in a region of South-eastern Australia, *Australian Journal of Botany,* **31**: 63-71.

Austin, M. P. and Heyligers, P. C., 1989. Vegetation survey design for conservation: Gradsect sampling of forests in North-eastern New South Wales, *Biological Conservation,* **50**: 13-32.

Austin, M. P. and Smith, T. M., 1989. A new model for the continuum concept, *Vegetatio,* **83**: 35-47.

Austin, M. P. and van Niel, K., 2011. Impact of landscape predictors on climate change modelling of species distributions: A case study with *Eucalyptus fastigata* in southern New South Wales, Australia, *Journal of Biogeography,* **38**: 9-19.

Australian Academy of Science, 2021. *Statement Regarding Australian Bushfires,* Available at: https://www.science.org.au/news-and-events/news-and-media-releases/statement-regarding-australian-bushfires (accessed 27/02/21).

Australian Alps National Parks, 2020. *Australian Alps National Parks,* Available at: https://theaustralianalps.wordpress.com/the-alps-partnership/the-parks/ (accessed 22/12/2020).

Avolio, M. L., Forrestel, E. J., Chang, C. C., La Pierre, K. J., Burghardt, K. T. and Smith, M. D., 2019. Demystifying dominant species, *New Phytologist,* **223**: 1106-1126.

Baeza, M. J., Raventos, J., Escarre, A. and Vallejo, V. R., 2006. Fire risk and vegetation structural dynamics in Mediterranean shrubland, *Plant Ecology,* **187**: 189-201.

Banks, J. C. G., 1989. A history of forest fire in the Australian Alps, In *The Scientific Significance of the Australian Alps. The Proceedings of the First Fenner Conference on the Environment. Canberra, September 1988* (Ed, Good, R.) Australian Alps National Parks Liaison Committee / Australian Academy of Science, Canberra, pp. 265-280.

Banks, S. C., Cary, G. J., Smith, A. L., Davies, I. D., Driscoll, D. A., Gill, A. M., Lindenmayer, D. B. and Peakall, R., 2013. How does ecological disturbance influence genetic diversity?, *TRENDS in Ecology and Evolution,* **28**(11): 670-679.

Barbero, R., Abatzoglou, J. T., Larkin, N. K., Kolden, C. A. and Stocks, B., 2015. Climate change presents increased potential for very large fires in the contiguous United States, *International Journal of Wildland Fire,* **24**: 892-899.

Barker, S., 1988. Population structure of Snow Gum (*Eucalyptus pauciflora* Sieb. ex Spreng.) subalpine woodland in Kosciusko National Park, *Australian Journal of Botany,* **36**: 483-501.

Barnes, B., Sidhu, H. S. and Roxburgh, S. H., 2006. A model integrating patch dynamics, competing species and the intermediate disturbance hypothesis, *Ecological Modelling,* **194**: 414-420.

Barrett, T. W., 2006. Modelling burn severity for the 2003 NSW/ACT wildfires using LANDSAT imagery. Bushfire 2006 Conference, Life in a Fire-Prone Environment: Translating Science into Practice, 6-9 June 2006, Brisbane.

Bassett, M., Leonard, S. W. J., Chia, E. K., Clarke, M. F. and Bennett, A. F., 2017. Interacting effects of fire severity, time since fire and topography on vegetation structure after wildfire, *Forest Ecology and Management,* **396**: 26-34.

Bassett, O. D., Prior, L. D., Slijkerman, C. M., Jamieson, D. and Bowman, D. M. J. S., 2015. Aerial sowing stopped the loss of alpine ash (*Eucalyptus delegatensis*) forests burnt by three short-interval fires in the Alpine National Park, Victoria, Australia, *Forest Ecology and Management,* **342**: 39-48.

Bataineh, A. L., Oswald, B. P., Bataineh, M., Unger, D., Hung, I.-K. and Scognamillo, D., 2006. Spatial autocorrelation and pseudoreplication in fire ecology, *Fire Ecology,* **2**(2): 107-118.

Battaglia, M., 1993. Seed germination physiology of *Eucalyptus delegatensis* R.T. Baker in Tasmania, *Australian Journal of Botany,* **41**: 119-136.

Battaglia, M. and Reid, J. B., 1993. The effect of microsite variation on seed germination and seedling survival of *Eucalyptus delegatensis*, *Australian Journal of Botany,* **41**: 169-181.

Bear, R. and Pickering, C. M., 2006. Recovery of subalpine grasslands from bushfire, *Australian Journal of Botany,* **54**: 451-458.

Beaty, R. M. and Taylor, A. H., 2001. Spatial and temporal variation of fire regimes in a mixed conifer forest landscape, Southern Cascades, California, USA, *Journal of Biogeography,* **28**: 955-966.

Belbin, L., 1991a. The analysis of pattern in bio-survey data, In *Nature Conservation: Cost Effective Biological Surveys and Data Analysis* (Eds, Margules, C. R. and Austin, M. P.) CSIRO, Canberra, pp. 176-190.

Belbin, L., 1991b. Semi-strong Hybrid Scaling, a new ordination algorithm, *Journal of Vegetation Science,* **2**: 491-496.

Belbin, L., 1995. *PATN Users Guide and Technical Reference,* CSIRO Division of Wildlife and Ecology, Canberra.

Belbin, L., 2000. *PATN for Windows. Version 2.0. User Manual,* CSIRO Division of Wildlife and Ecology, Canberra.

Belbin, L., Faith, D. P. and G.W., M., 1992. A comparison of two approaches to beta-flexible clustering, *Multivariate Behavioural Research,* **27**(3): 417-433.

Bellingham, P. J. and Sparrow, A. D., 2000. Resprouting as a life history strategy in woody plant communities, *OIKOS,* **89**(2): 409-416.

Belote, R. T., Jones, R. H. and Wieboldt, T. F., 2012. Compositional stability and diversity of vascular plant communities following logging disturbance in Appalachian forests, *Ecological Applications,* **22**(2): 502-516.

Benscoter, B. W. and Vitt, D. H., 2008. Spatial patterns and temporal trajectories of the bog ground layer along a post-fire chronosequence, *Ecosystems,* **11**: 1054-1064.

Benson, D. H., 1985. Maturation periods for fire-sensitive shrub species in Hawkesbury Sandstone vegetation, *Cunninghamia,* **1**(3): 339-349.

Benson, D. H. and McDougall, L., 1995. Ecology of Sydney plant species Part 3: Dicotyledon families Cabombaceae to Eupomatiaceae, *Cunninghamia,* **4**(2): 217-424.

Benwell, A. S., 2007. Response of rock-outcrop and fringing vegetation to disturbance by fire and drought, *Australian Journal of Botany,* **55**: 736-748.

Boland, D. J., Brooker, M. I. H., Turnbull, J. W. and Kleinig, D. A., 1980. *Eucalyptus Seed,* CSIRO Division of Forest Research, Canberra.

Bolker, B. M., Brooks, M. E., Clark, C. J., Geange, S. W., Poulsen, J. R., Stevens, M. H. H. and White, J.-S. S., 2008. Generalized linear mixed models: A practical guide for ecology and evolution, *TRENDS in Ecology and Evolution,* **24**(3): 127-135.

Bolotin, M., 1975. Photoperiodic induction of precocious flowering in a woody species *Eucalyptus occidentalis* Endl., *Botanical Gazette,* **136**(4): 358-365.

Bond, W. J. and Keeley, J. E., 2005. Fire as a global 'herbivore': The ecology and evolution of flammable ecosystems, *TRENDS in Ecology and Evolution,* **20**(7): 387-394.

Bond, W. J. and Midgley, J. J., 2001. Ecology of sprouting in woody plants: The persistence niche, *TRENDS in Ecology and Evolution*, **16**(1): 45-51.

Bond, W. J. and van Wilgen, B. W., 1996. *Fire and Plants. Population and Community Biology Series 14*, Chapman & Hall, London.

Bond, W. J., Woodward, F. I. and Midgley, G. F., 2005. The global distribution of ecosystems in a world without fire, *New Phytologist*, **165**: 525-538.

Boose, E. R., Foster, D. R. and Fluet, M., 1994. Hurricane impacts to tropical and temperate forest landscapes, *Ecological Monographs*, **64**(4): 369-400.

Borchert, M. I. and Odion, D. C., 1995. Fire intensity and vegetation recovery in chaparral: A review, In *Brushfires in California Wildlands: Ecology and Resource Management* (Eds, Keeley, J. E. and Scott, T.) International Association of Wildland Fire, Fairfield, pp. 91-100.

Bornette, G. and Amoros, C., 1996. Disturbance regimes and vegetation dynamics: Role of floods in riverine wetlands, *Journal of Vegetation Science*, **7**: 615-622.

Bowman, D. M. J. S. and Kirkpatrick, J. B., 1986a. Establishment, suppression and growth of *Eucalyptus delegatensis* R.T. Baker in multiaged forests. I. The effects of fire on mortality and seedling establishment, *Australian Journal of Botany*, **34**: 63-72.

Bowman, D. M. J. S. and Kirkpatrick, J. B., 1986b. Establishment, suppression and growth of *Eucalyptus delegatensis* R.T. Baker in multiaged forests. II. Sapling growth and its environmental correlates, *Australian Journal of Botany*, **34**: 73-80.

Bowman, D. M. J. S. and Kirkpatrick, J. B., 1986c. Establishment, suppression and growth of *Eucalyptus delegatensis* R.T. Baker in multiaged forests. III. Intraspecific allelopathy, competition between adult and juvenile for moisture and nutrients and frost damage to seedlings, *Australian Journal of Botany*, **34**: 81-94.

Bowman, D. M. J. S., Murphy, B. P., Boer, M. M., Bradstock, R. A., Cary, G. J., Cochrane, M. A., Fensham, R. J., Krawchuck, M. A., Price, O. F. and Williams, R. J., 2013. Forest fire management, climate change and the risk of catastrophic carbon losses, *Frontiers in Ecology and the Environment*, **11**(2): 66-68.

Bowman, D. M. J. S., Murphy, B. P., Neyland, D. L. J., Williamson, G. J. and Prior, L. D., 2014. Abrupt fire regime change may cause landscape-wide loss of mature obligate seeder forests, *Global Change Biology*, **20**: 1008-1015.

Bowman, D. M. J. S., Perry, G. W. L. and Marston, J. B., 2015. Feedbacks and landscape-level vegetation dynamics, *TRENDS in Ecology and Evolution*, **30**(5): 255-260.

Bowman, D. M. J. S., Williamson, G. J., Prior, L. D. and Murphy, B. P., 2016. The relative importance of intrinsic and extrinsic factors in the decline of obligate seeder forests, *Global Ecology and Biogeography*, **25**(10): 1166-1172.

Bradshaw, S. D., Dixon, K. W., Hopper, S. D., Lambers, H. and Turner, S. R., 2011. Little evidence for fire-adapted plant traits in Mediterranean climate regions, *Trends in Plant Science*, **16**(2): 69-76.

Bradstock, R. A., 2003. Protection of people and property: Towards an integrated risk management model, In *Australia Burning: Fire Ecology, Policy and Management Issues* (Eds, Cary, G., Lindenmayer, D. and Dovers, S.) CSIRO Publishing, Melbourne, pp. 119-123.

Bradstock, R. A., 2008. Effects of large fires on biodiversity in south-eastern Australia: Disaster or template for diversity?, *International Journal of Wildland Fire*, **17**: 809-892.

Bradstock, R. A., Bedward, M., Scott, J. and Keith, D. A., 1996. Simulation of the effect of spatial and temporal variation in fire regimes on the population viability of *Banksia* species, *Conservation Biology*, **10**(3): 776-784.

Bradstock, R. A., Cohn, J. S., Gill, A. M., Bedward, M. and Lucas, C., 2009. Prediction of the probability of large fires in the Sydney region of south-eastern Australia using fire weather, *International Journal of Wildland Fire*, **18**: 932-943.

Bradstock, R. A., Gill, A. M., Kenny, B. J. and Scott, J., 1998. Bushfire risk at the urban interface estimated from historical weather records: Consequences for the use of prescribed fire in the Sydney region of south-eastern Australia, *Journal of Environmental Management*, **52**: 259-271.

Bradstock, R. A., Hammill, K. A., Collins, L. and Price, O. F., 2010. Effects of weather, fuel and terrain on fire severity in topographically diverse landscapes of south-eastern Australia, *Landscape Ecology,* **25**: 607-619.

Bradstock, R. A. and Myerscough, P. J., 1981. Fire effects on seed release and the emergence and establishment of seedlings in *Banksia ericifolia* L.f., *Australian Journal of Botany,* **29**: 521-531.

Bradstock, R. A. and O'Connell, M. A., 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, R. A., Tozer, M. G. and Keith, D. A., 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, R. A., Williams, J. E. and Gill, A. M. (Eds.), 2002. *Flammable Australia: The Fire Regimes and Biodiversity of a Continent,* Cambridge University Press, Cambridge.

Bray, J. R. and Curtis, J. T., 1957. An ordination of the upland forest communities of southern Wisconsin, *Ecological Monographs,* **27**(4): 325-349.

Brooker, M. I. H. and Kleinig, D. A., 2006. *Field Guide to Eucalypts Volume 1: South-eastern Australia,* Bloomings Books, Melbourne.

Brown, J. K., Arno, S. F., Barrett, S. W. and Menakis, J. P., 1994. Comparing the prescribed natural fire program with presettlement fires in the Selway-Bitterroot Wilderness, *International Journal of Wildland Fire,* **4**(3): 157-168.

Brown, V. K., 1991. The effects of changes in habitat structure during succession in terrestrial communities, In *Habitat Structure: The Physical Arrangement of Objects in Space* (Eds, Bell, S. S., McCoy, E. D. and Mushinsky, H. R.) Chapman & Hall, London, pp. 141-168.

Burdon, J. J. and Pryor, L. D., 1975. Interspecific competition between eucalypt seedlings, *Australian Journal of Botany,* **23**: 225-229.

Bureau of Meteorology, 2020a. *Climate Data Online,* Available at: http://www.bom.gov.au/climate/data/index.shtml (accessed 22/12/2020).

Bureau of Meteorology, 2020b. *Recent rainfall, drought and southern Australia's long-term rainfall decline,* Available at: http://www.bom.gov.au/climate/updates/articles/a010-southern-rainfall-decline.shtml (accessed 22/12/2020).

Burns, E. L., Lindenmayer, D. B., Stein, J., Blanchard, W., McBurney, L., Blair, D. and Banks, S. C., 2015. Ecosystem assessment of mountain ash forest in the Central Highlands of Victoria, south-eastern Australia, *Austral Ecology,* **78**: 386-399.

Burrows, C. J., 1990. *Processes of Vegetation Change,* Unwin and Hyman, London.

Burrows, G. E., 2002. Epicormic strand structure in *Angophora*, *Eucalyptus* and *Lophostemon* (Myrtaceae) - implications for fire resistance and recovery, *New Phytologist,* **153**: 111-131.

Burrows, G. E., 2008. *Syncarpia* and *Tristaniopsis* (Myrtaceae) possess specialised fire-resistant epicormic structures, *Australian Journal of Botany,* **56**: 254-264.

Burrows, G. E., 2013. Buds, bushfires and resprouting in the eucalypts, *Australian Journal of Botany,* **61**: 331-349.

Burton, P. J., Parisien, M.-A., Hicke, J. A., Hall, R. J. and Freeburn, J. T., 2008. Large fires as agents of ecological diversity in the North American boreal forest, *International Journal of Wildland Fire,* **17**: 754-767.

Byram, G. M., 1959. Combustion of forest fuels, In *Forest Fire: Control and Use* (Ed, Davis, K. P.) McGraw-Hill, New York, pp. 61-89.

Cai, W., Borlace, S., Lengaigne, M., van Rensch, P., Collins, M., Vecchi, G., Timmermann, A., Santoso, A., McPhaden, M. J., Wu, L., England, M. H., Wang, G., Guilyardi, E. and Jin, F.-F., 2014. Increasing frequency of extreme El Nino events due to greenhouse warming, *Nature Climate Change,* **4**: 111-116.

Camac, J. S., Williams, R. J., Wahren, C.-H., Morris, W. K. and Morgan, J. W., 2013. Post-fire regeneration in heathland: Does fire severity matter?, *Austral Ecology,* **38**: 199-207.

Campbell, M. L. and Clarke, P. J., 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.

Canham, C. D. and Marks, P. L., 1985. The response of woody plants to disturbance: Patterns of establishment and growth, In *The Ecology of Natural Disturbance and Patch Dynamics* (Eds, Pickett, S. T. A. and White, P. S.) Academic Press, San Diego, pp. 197-216.

Capitanio, R. and Carcaillet, C., 2008. Post-fire Mediterranean vegetation dynamics and diversity: A discussion of succession models, *Forest Ecology and Management,* **255**: 431-439.

Carnahan, J. A., 1986. Vegetation. Chapter 10, In *Australia: A Geography. Volume 1. The Natural Environment* (Ed, Jeans, D. N.) Sydney University Press, Sydney, pp. 260-282.

Carr, S. G. M. and Turner, J. S., 1959a. The ecology of the Bogong High Plains. I. The environmental factors and the grassland communities, *Australian Journal of Botany,* **7**: 12-33.

Carr, S. G. M. and Turner, J. S., 1959b. The ecology of the Bogong High Plains. II. Fencing experiments in grassland C, *Australian Journal of Botany,* **7**: 34-63.

Cary, G. J., 2002. Importance of a changing climate for fire regimes in Australia, In *Flammable Australia: The Fire Regimes and Biodiversity of a Continent* (Eds, Bradstock, R. A., Williams, J. E. and Gill, A. M.) Cambridge University Press, Cambridge, pp. 26-46.

Cary, G. J., Flannigan, M. D., Keane, R. E., Bradstock, R. A., Davies, I. D., Lenihan, J. M., Li, C., Logan, K. A. and Parsons, R. A., 2009. Relative importance of fuel management, ignition management and weather for area burned: Evidence from five landscape-fire-succession models, *International Journal of Wildland Fire,* **18**: 147-156.

Cary, G. J., Keane, R. E., Gardner, R. H., Lavorel, S., Flannigan, M. D., Davies, I. D., Li, C., Lenihan, J. M., Rupp, T. S. and Mouillot, F., 2006. Comparison of the sensitivity of landscape-fire-succession models to variation in terrain, fuel pattern, climate and weather, *Landscape Ecology,* **21**: 121-137.

Cary, G. J., Lindenmayer, D. and Dovers, S. (Eds.), 2003a. *Australia Burning: Fire Ecology, Policy and Management Issues,* CSIRO Publishing, Melbourne.

Cary, G. J., Lindenmayer, D. and Dovers, S., 2003b. Research and policy priorities: A synthesis, In *Australia Burning: Fire Ecology, Policy and Management Issues* (Eds, Cary, G., Lindenmayer, D. and Dovers, S.) CSIRO Publishing, Melbourne, pp. 252-265.

Catling, P. C. and Burt, R. J., 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, P. C. and Newsome, A. E., 1981. Responses of the Australian vertebrate fauna to fire: An evolutionary approach, In *Fire and the Australian Biota* (Eds, Gill, A. M., Groves, R. H. and Noble, I. R.) Australian Academy of Science, Canberra, pp. 273-310.

Chafer, C. J., Noonan, M. and Macnaught, E., 2004. The post-fire measurement of fire severity and intensity in the Christmas 2001 Sydney wildfires, *International Journal of Wildland Fire,* **13**: 227-240.

Chambers, T. C., 1977. *Estimates of Time Required for Recovery of Victorian Plant Communities from Ground and Crown Fires*, Interim Reference Areas Advisory Committee, Land Conservation Council, Melbourne.

Chapin, F. S., Torn, M. S. and Tateno, M., 1996. Principles of ecosystem sustainability, *The American Naturalist,* **148**(6): 1016-1037.

Chapin, F. S. and Van Cleve, K., 1981. Plant nutrient absorption and retention under differing fire regimes, In *Fire Regimes and Ecosystem Properties. Proceedings of the Conference. December 11-15 Honolulu, Hawaii. USDA Forest Service Technical Report WO-26* (Eds, Mooney, H. A., Bonnicksen, T. M., Christensen, N. L., Lotan, J. E. and Reiners, W. A.) USDA Forest Service, Washington, pp. 301-321.

Cheal, D. C., 2010. *Growth Stages and Tolerable Fire Intervals for Victoria's Native Vegetation Data Sets. Fire and Adaptive Management Report No. 84,* Department of Sustainability and Environment, Melbourne.

Chick, M. P., Cohn, J. S., Nitschke, C. R. and York, A., 2016. Lack of soil seedbank change with time since fire: Relevance to seed supply after prescribed burns, *International Journal of Wildland Fire,* **25**: 849-860.

Christensen, N. L., 1991. Wilderness and high intensity fire: How much is enough?, In *High Intensity Fire in Wildlands: Management Challenges and Options. Proceedings of the 17th Tall Timbers Fire Ecology Conference* (Ed, Hermann, S. M.) Tall Timbers Research Station, Tallahassee, pp. 9-24.

Christensen, N. L., Recher, H. F. and Hoare, J., 1981. Responses of open forests (dry sclerophyll forests) to fire regimes, In *Fire and the Australian Biota* (Eds, Gill, A. M., Groves, R. H. and Noble, I. R.) Australian Academy of Science, Canberra, pp. 367-393.

Clark, J. S., 1989. Ecological disturbance as a renewal process: Theory and application to fire history, *OIKOS,* **56**: 17-30.

Clark, J. S., Gill, A. M. and Kershaw, A. P., 2002. Spatial variability in fire regimes: Its effects on recent and past vegetation, In *Flammable Australia: The Fire Regimes and Biodiversity of a Continent* (Eds, Bradstock, R. A., Williams, J. E. and Gill, A. M.) Cambridge University Press, Cambridge, pp. 125-141.

Clark, S. S. and McLoughlin, L. C., 1986. Historical and biological evidence for fire regimes in the Sydney Region prior to the arrival of Europeans: Implications for future bushland management, *The Australian Geographer,* **17**: 101-112.

Clarke, P. J., 2002. Habitat islands in fire-prone vegetation: Do landscape features influence community composition?, *Journal of Biogeography,* **29**: 677-684.

Clarke, P. J. and Martin, A. R. H., 1999. Sphagnum peatlands in Kosciuszko National Park in relation to altitude, time and disturbance, *Australian Journal of Botany,* **47**: 519-536.

Clarke, S. and French, K., 2005. Germination response to heat and smoke of 22 Poaceae species from grassy woodlands, *Australian Journal of Botany,* **53**: 445-454.

Clements, F. E., 1916. *Plant Succession: An Analysis of the Development of Vegetation. Carnegie Institution of Washington Publication 242,* Carnegie Institution of Washington, Washington.

Clements, F. E., 1936. Nature and structure of the climax, *Journal of Ecology,* **24**: 252-284.

Coates, F., 2015. Comparative changes in *Eucalyptus pauciflora* (Myrtaceae) stand structure after bushfires in Victoria, *Cunninghamia,* **15**: 1-11.

Cohn, J. S., Bradstock, R. A. and Burke, S., 2002. Effects of time since fire, topography and resprouting eucalypts on ephemeral understorey species composition, in semi-arid mallee communities in NSW, *Cunninghamia,* **7**(3): 579-600.

Collins, B. M., Everett, R. G. and Stephens, S. L., 2011. Impacts of fire exclusion and recent managed fire on forest structure in old growth Sierra Nevada mixed-conifer forests, *Ecosphere,* **2**(4): Article 51.

Collins, L., 2019. Eucalypt forests dominated by epicormic resprouters are resilient to repeated canopy fire, *Journal of Ecology*: DOI: 10.1111/1365-2745.13227.

Collins, L., Bennett, A. F., Leonard, S. W. J. and Penman, T. D., 2019. Wildfire refugia in forests: Severe fire weather and drought mute the influence of topography and fuel age, *Global Change Biology*: DOI: 10.1111/gcb.14735.

Colloff, M. J., Doherty, M. D., Lavorel, S., Dunlop, M., Wise, R. M. and Prober, S. M., 2016. Adaptation services and pathways for the management of temperate montane forests under transformational climate change, *Climatic Change,* **138**: 267-282.

Connell, J. H., 1978. Diversity in tropical rain forests and coral reefs, *Science,* **199**: 1302-1310.

Connell, J. H., 1979. Tropical rain forests and coral reefs as open non-equilibrium systems, In *Population Dynamics. The 20th Symposium of the British Ecological Society.* (Eds, Anderson, B. M., Turner, B. D. and Taylor, L. R.) Blackwell Scientific Publications, Oxford, pp. 141-163.

Connell, J. H. and Slatyer, R. O., 1977. Mechanisms of succession in natural communities and their role in community stability and organization, *The American Naturalist,* **111**(982): 1119-1144.

Coops, N. C. and Catling, P. C., 1997a. Predicting the complexity of habitat in forests from airborne videography for wildlife management, *International Journal of Remote Sensing,* **18**(12): 2677-2682.

Coops, N. C. and Catling, P. C., 1997b. Utilising airborne multispectral videography to predict habitat complexity in eucalypt forests for wildlife management, *Wildlife Research*, **24**: 691-703.

Costin, A. B., 1954. *A Study of the Ecosystems of the Monaro Region of New South Wales,* NSW Government Printer, Sydney.

Cowling, R. M., Gallien, L., Richardson, D. M. and Ojeda, F., 2018. What predicts the richness of seeder and resprouter species in fire-prone Cape fynbos: Rainfall reliability or vegetation density?, *Austral Ecology*.

Cramsie, J., Pogson, D. J. and Baker, C. J., 1975. *Yass 1:100 000 Geological Sheet,* Geological Survey of NSW, Sydney.

Cremer, K. W., 1962. The effect of fire on eucalypts reserved for seeding, *Australian Forestry,* **26**: 129-154.

Cremer, K. W., Cromer, R. N. and Florence, R. G., 1984. Stand establishment, In *Eucalypts for Wood Production* (Eds, Hillis, W. E. and Brown, A. G.) CSIRO / Academic Press, Sydney, pp. 81-135.

Cunningham, T. M., 1957. Seed production and seed fall of *E. regnans* (F. Muell.), *Australian Forestry,* **21**: 30-39.

Cunningham, T. M. and Cremer, K. W., 1965. Control of the understorey in wet eucalypt forests, *Australian Forestry,* **29**: 4-14.

Dale, M. B., 1989. Dissimilarity for partially ranked data and its application to cover-abundance, *Vegetatio,* **82**: 1-12.

Dale, V. H., Lugo, A. E., MacMahon, J. A. and Pickett, S. T. A., 1998. Ecosystem management in the context of large, infrequent disturbances, *Ecosystems,* **1**: 546-557.

Dale, V. H., Swanson, F. J. and Crisafulli, C. M., 2005a. Disturbance, survival and succession: Understanding ecological responses to the 1980 eruption of Mount St. Helens, In *Ecological Responses to the 1980 Eruption of Mount St. Helens* (Eds, Dale, V. H., Swanson, F. J. and Crisafulli, C. M.) Springer, New York, pp. 3-11.

Dale, V. H., Swanson, F. J. and Crisafulli, C. M. (Eds.), 2005b. *Ecological Responses to the 1980 Eruption of Mount St. Helens,* Springer, New York.

Davidson, N. J. and Reid, J. B., 1989. Response of eucalypt species to drought, *Australian Journal of Ecology,* **14**: 139-156.

DeBano, L. F., Neary, D. G. and Ffolliott, P. F., 1998. *Fire's Effects on Ecosystems,* John Wiley and Sons, New York.

del Moral, R. and Grishin, S. Y., 1999. Volcanic disturbances and ecosystem recovery, In *Ecosystems of Disturbed Ground. Ecosystems of the World 16* Vol. 16 (Ed, Walker, L. R.) Elsevier, Amsterdam, pp. 137-160.

Denslow, J. S., 1980. Patterns of plant species diversity during succession under different disturbance regimes, *Oecologia,* **46**: 18-21.

Despain, D. G. and Romme, W. H., 1991. Ecology and management of high-intensity fires in Yellowstone National Park, In *High Intensity Fire in Wildlands: Management Challenges and Options. Proceedings of the 17th Tall Timbers Fire Ecology Conference* (Ed, Hermann, S. M.) Tall Timbers Research Station, Tallahassee, pp. 43-57.

Diamond, J. M., 1983. Laboratory, field and natural experiments, *Nature,* **304**(18): 586-587.

Doherty, M. D., 1997. *Vegetation Survey and Mapping of Brindabella National Park and Adjacent Vacant Crown Lands. A report prepared for NSW National Parks and Wildlife Service, Queanbeyan District*, CSIRO Division of Wildlife and Ecology, Canberra.

Doherty, M. D., 1998a. *The Conservation Value of Regrowth Native Plant Communities. Report for NSW Scientific Committee,* CSIRO Division of Wildlife and Ecology, Canberra.

Doherty, M. D., 1998b. *Vegetation Survey and Mapping of Burrinjuck Nature Reserve. A report prepared for NSW National Parks and Wildlife Service, Queanbeyan District*, CSIRO Division of Wildlife and Ecology, Canberra.

Doherty, M. D., 2008. Moist fingers in a drying landscape. Effects of fire and climate on native vegetation in the northern parts of the Australian Alps., In *Corridors for Survival in a Changing World. Proceedings of the NPA ACT Symposium Canberra, 9-10 May 2008* (Eds, McCue, K. and Lenz, S.) National Parks Association of the ACT Inc, Canberra, pp. 77-85.

Doherty, M. D., 2015. Book Review: Historical Environmental Variation in Conservation and Natural Resource Management, *Ecological Management and Restoration,* **16**(2): [online] e10-e11.

Doherty, M. D., Gill, A. M., Cary, G. J. and Austin, M. P., 2017a. Seed viability of early maturing alpine ash (*Eucalyptus delegatensis* R.T. Baker subsp. *delegatensis*) in the Australian Alps, southeastern Australia, and its implications for management under changing fire regimes, *Australian Journal of Botany,* **65**: 517-523.

Doherty, M. D., Kearns, A. J., Barnett, G., Sarre, A., Hochuli, D. F., Gibb, H. and Dickman, C. R., 1998. *The Interactions Between Habitat Conditions, Ecosystem Processes and Biodiversity - a Review. Australia: State of the Environment. Technical Paper Series No. 2. Paper 2. Biodiversity,* Environment Australia, Canberra.

Doherty, M. D., Lavorel, S., Colloff, M. J., Williams, K. J. and Williams, R. J., 2017b. Moving from autonomous to planned adaptation in the montane forests of southeastern Australia under changing fire regimes, *Austral Ecology,* **42**(3): 309-316.

Doherty, M. D., Meyers, J. A., Tasker, E. M. and Williams, K. J., 2014. *Vegetation Dynamics and Changing Fire Regimes in Southeast New South Wales: Exploratory Analysis of Fire History and Vegetation Survey Data. NSW Environmental Trust Project 2012/RD/0016. Final Report 10th December 2014*, CSIRO Climate Adaptation Flagship, Canberra.

Doherty, M. D., Wright, G. and McDougall, K. L., 2015. The flora of Kosciuszko National Park, New South Wales: Summary and overview, *Cunninghamia,* **15**: 13-68.

Doyle, K. M., Knight, D. H., Taylor, D. L., Barmore, W. J. and Benedict, J. M., 1998. Seventeen years of forest succession following the Waterfalls Canyon Fire in Grand Teton National Park, Wyoming, *International Journal of Wildland Fire,* **8**(1): 45-55.

Drury, W. H. and Nisbet, I. C. T., 1973. Succession, *Journal of the Arnold Arboretum,* **54**(3): 331-368.

Duff, G. A., Reid, J. B. and Jackson, W. D., 1983. The occurrence of mixed stands of the *Eucalyptus* subgenera *Monocalyptus* and *Symphyomyrtus* in south-eastern Tasmania, *Australian Journal of Ecology,* **8**: 405-414.

Edwards, A. C. and Russell-Smith, J., 2009. Ecological thresholds and the status of fire-sensitive vegetation in western Arnhem land, northern Australia: Implications for management, *International Journal of Wildland Fire,* **18**: 127-146.

Egler, F. E., 1954. Vegetation science concepts I. Initial floristic composition. A factor in old-field vegetation development, *Vegetatio,* **4**: 412-417.

Ellis, R. C. and Lockett, E., 1991. The management of high altitude eucalypt forests in Tasmania, In *Forest Management in Australia* (Eds, McKinnell, F. H., Hopkins, E. R. and Fox, J. E. D.) Institute of Foresters Australia / Surrey Beatty and Sons, Sydney, pp. 131-145.

Ellis, R. C., Mount, A. B. and Mattay, J. P., 1980. Recovery of *Eucalyptus delegatensis* from high altitude dieback after felling and burning the understorey, *Australian Forestry,* **43**(1): 29-35.

Ellis, S., 2004. *Council of Australian Governments National Inquiry into Bushfire Mitigation and Management,* Commonwealth of Australia, Canberra.

Elmqvist, T., Folke, C., Nystrom, M., Peterson, G., Bengtsson, J., Walker, B. and Norberg, J., 2003. Response diversity, ecosystem change, and resilience, *Frontiers in Ecology and the Environment,* **1**(9): 488-494.

Elton, C. S., 1927. *Animal Ecology,* Sedgwick and Jackson, London.

England, J., Doherty, M. D., Keith, H. and Raison, J., 2004. *Ecological Effects of Burning in ACT Vegetation Communities. Client Report for Fire Management Unit, Department of Urban Services,* CSIRO Forestry and Forest Products, Canberra.

Enright, N. J., Fontaine, J. B., Bowman, D. M. J. S., Bradstock, R. A. and Williams, R. J., 2015. Interval squeeze: altered fire regimes and demographic responses interact to threaten woody species persistence as climate changes, *Frontiers in Ecology and Environment,* **13**(5): 265-272.

Enright, N. J., Marsula, R., Lamont, B. B. and Wissel, C., 1998a. The ecological significance of canopy seed storage in fire-prone environments: A model for non-sprouting shrubs, *Journal of Ecology,* **86**: 946-959.

Enright, N. J., Marsula, R., Lamont, B. B. and Wissel, C., 1998b. The ecological significance of canopy seed storage in fire-prone environments: A model for resprouting shrubs, *Journal of Ecology,* **86**: 960-973.

Ernst, A., 1908. *The New Flora of the Volcanic Island of Krakatau,* Cambridge University Press, Cambridge.

Etchells, H., O'Donnell, A. J., McCaw, W. L. and Grierson, P. F., 2020. Fire severity impacts on tree mortality and post-fire recruitment in tall eucalypt forests of southwest Australia, *Forest Ecology and Management,* **459**: Article 117850.

Everham, E. M. and Brokaw, N. V. L., 1996. Forest damage and recovery from catastrophic wind, *The Botanical Review,* **62**(2): 113-185.

Ex, S., Smith, F. W. and Keyser, T. L., 2015. Characterizing crown fuel distribution for conifers in the interior western United States, *Canadian Journal of Forest Research,* **45**: 950-957.

Fagg, P., Lutze, M., Slijkerman, C., Ryan, M. and Bassett, O., 2013. Silvicultural recovery in ash forests following three large bushfires in Victoria, *Australian Forestry,* **76**(3-4): 140-155.

Fairman, T., Bennett, L. T., Tupper, S. and Nitschke, C. R., 2017. Frequent wildfires erode tree persistence and alter stand structure and initial composition of a fire-tolerant sub-alpine forest, *Journal of Vegetation Science,* **28**(6): 1151-1165.

Fairman, T. A., Bennett, L. T. and Nitschke, C. R., 2018. Short-interval wildfires increase likelihood of resprouting failure in fire-tolerant trees, *Journal of Environmental Management,* **231**: 59-65.

Fairman, T. A., Nitschke, C. R. and Bennett, L. T., 2016. Too much too soon? A review of the effects of increasing wildfire frequency on tree mortality and regeneration in temperate eucalypt forests, *International Journal of Wildland Fire,* **25**: 831-848.

Faith, D. P., Humphrey, C. L. and Dostine, P. L., 1991. Statistical power and BACI designs in biological monitoring: Comparative evaluation of measures of community dissimilarity based on benthic macroinvertebrate communities in Rockhole Mine Creek, Northern Territory, Australia, *Australian Journal of Marine and Freshwater Research,* **42**: 589-602.

Faith, D. P., Minchin, P. R. and Belbin, L., 1987. Compositional dissimilarity as a robust measure of ecological distance *Vegetatio,* **69**: 57-68.

Falk, D. A., Miller, C., McKenzie, D. and Black, A. E., 2007. Cross-scale analysis of fire regimes, *Ecosystems,* **10**: 809-823.

Falk, D. A., Watts, A. C. and Thode, A. E., 2019. Scaling ecological resilience, *Frontiers in Ecology and Evolution,* **7**: Article 275.

Fastie, C. L., 1995. Causes and ecosystem consequences of multiple pathways of primary succession at Glacier Bay, Alaska, *Ecology,* **76**(6): 1899-1916.

Fenner, M. and Thompson, K., 2005. *The Ecology of Seeds,* Cambridge University Press, Cambridge.

Fensham, R. J., Freeman, M. E., Laffineur, B., MacDermott, H. J., Prior, L. D. and Werner, P. A., 2017. Variable rainfall has a greater effect than fire on the demography of the dominant tree in a semi-arid *Eucalyptus* savanna, *Austral Ecology,* **42**: 772-782.

Ferguson, I., 2011. Strategic seedbanks to meet fire risks for Victorian ash-type species, *Australian Forestry,* **74**(2): 97-107.

Fielding, J. M., 1956. Notes on the flowering and seeding of *Eucalyptus delegatensis* and *E. fastigata* in the Australian Capital Territory, *Australian Forestry,* **20**: 40-43.

Fisher, J. L., Loneragan, W. A., Dixon, K. W., Delaney, J. and Veneklaas, E. J., 2009. Altered vegetation structure and composition linked to fire frequency and plant invasion in a biodiverse woodland, *Biological Conservation,* **142**: 2270-2281.

Flannigan, M. D. and Wotton, B. M., 2001. Climate, weather and area burned, In *Forest Fires: Behaviour and Ecological Effects* (Eds, Johnson, E. A. and Miyanishi, K.) Academic Press, San Diego, pp. 351-373.

Florentine, S. K., Milberg, P., Gibson, M. and Westbrooke, M., 2008. Post-wildfire colonisation patterns in a *Eucalyptus delegatensis* (Myrtaceae) windthrow site at Snowy River National Park, Victoria, *Australian Forestry,* **71**(1): 48-53.

Florentine, S. K. and Westbrooke, M. E., 2004. Effects of windthrow on a stand of *Eucalyptus delegatensis* (Myrtaceae) and early understorey succession at Snowy River National Park, Victoria, *Australian Forestry,* **67**(1): 54-58.

Fonseca, C. R. and Ganade, G., 2001. Species functional redundancy, random extinctions and the stability of ecosystems, *Journal of Ecology,* **89**: 118-125.

Forbes, S. J., Walsh, N. G. and Gullan, P. K., 1982. Vegetation of East Gippsland, *Muelleria,* **5**(1): 53-113.

Forestry Tasmania, 2010. *Eucalyptus delegatensis Forests. Native Forest Silviculture Technical Bulletin No. 2*, Forestry Tasmania, Hobart.

Fornwalt, P. J., Kaufman, M. R. and Stohlgren, T. J., 2010. Impacts of mixed severity wildfire on exotic plants in a Colorado ponderosa pine - Douglas-fir forest, *Biological Invasions,* **12**: 2683-2695.

Foster, D. R., Knight, D. H. and Franklin, J. F., 1998. Landscape patterns and legacies from large, infrequent forest disturbances, *Ecosystems,* **1**: 497-510.

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

Fox, B. J. and McKay, G. M., 1981. Small mammal responses to pyric successional changes in eucalypt forest, *Australian Journal of Ecology,* **6**: 29-41.

Fox, M. D., 1990. Composition and richness in New South Wales mallee, In *The Mallee Lands: A Conservation Perspective. Proceedings of the National Mallee Conference, Adelaide, April 1989* (Eds, Noble, J. C., Joss, P. J. and Jones, G. K.) CSIRO, Melbourne, pp. 8-11.

Fox, M. D. and Fox, B. J., 1986. The effect of fire frequency on the structure and floristic composition of a woodland understorey, *Australian Journal of Ecology,* **11**: 77-85.

Franklin, J. F., Lindenmayer, D. B., MacMahon, J. A., McKee, A., Magnuson, J., Perry, D. A., Waide, R. and Foster, D. R., 2000. Threads of continuity: Ecosystem disturbances, biological legacies and ecosystem recovery, *Conservation,* **1**(1): 8-16.

Freestone, M., Wills, T. J. and Read, J., 2015. Post-fire succession during the long-term absence of fire in coastal heathland and a test of the chronosequence survey method, *Australian Journal of Botany,* **63**: 572-580.

Frelich, L. E. and Reich, P. B., 1999. Neighbourhood effects, disturbance severity and community stability in forests, *Ecosystems,* **2**(2): 151-166.

Gauch, H. G., 1973. A quantitative evaluation of the Bray-Curtis ordination, *Ecology,* **54**: 829-836.

Gauch, H. G., Chase, G. B. and Whittaker, R. H., 1974. Ordination of vegetation samples by Gaussian species distributions, *Ecology,* **55**: 1382-1390.

Gell, P. A., Stuart, I.-M. and Smith, J. D., 1993. The response of vegetation to changing fire regimes and human activity in East Gippsland, Victoria, Australia, *The Holocene,* **3**(2): 150-160.

Gellie, N. J. H., 2005. Native vegetation of the Southern Forests: South-east Highlands, Australian Alps, South-west Slopes and SE Corner bioregions, *Cunninghamia,* **9**(2): 219-253.

Gharun, M., Turnbull, T. L. and Adams, M. A., 2013. Stand water use status in relation to fire in a mixed species eucalypt forest, *Forest Ecology and Management,* **304**: 162-170.

Gilbert, J. M., 1959. Forest succession in the Florentine Valley, Tasmania, *Papers and Proceedings of the Royal Society of Tasmania,* **93**: 129-151.

Gill, A. M., 1975. Fire and the Australian flora: A review, *Australian Forestry,* **38**: 4-25.

Gill, A. M., 1978. Crown recovery of *Eucalyptus dives* following wildfire, *Australian Forestry,* **41**(4): 207-214.

Gill, A. M., 1980. Restoration of bark thickness after fire and mechanical injury in a smooth-barked eucalypt, *Australian Forest Research,* **10**: 311-319.

Gill, A. M., 1981a. Adaptive responses of Australian vascular plant species to fires, In *Fire and the Australian Biota* (Eds, Gill, A. M., Groves, R. H. and Noble, I. R.) Australian Academy of Science, Canberra, pp. 243-272.

Gill, A. M., 1981b. Coping with fire, In *The Biology of Australian Plants* (Eds, Pate, J. S. and McComb, A. J.) University of Western Australia Press, Nedlands, pp. 65-87.

Gill, A. M., 1981c. Fire adaptive traits of vascular plants, In *Fire Regimes and Ecosystem Properties. Proceedings of the Conference. December 11-15 Honolulu, Hawaii. USDA Forest Service*

Technical Report WO-26 (Eds, Mooney, H. A., Bonnicksen, T. M., Christensen, N. L., Lotan, J. E. and Reiners, W. A.) USDA Forest Service, Washington, pp. 208-230.

Gill, A. M., 1981d. Post-settlement fire history in Victorian landscapes, In *Fire and the Australian Biota* (Eds, Gill, A. M., Groves, R. H. and Noble, I. R.) Australian Academy of Science, Canberra, pp. 77-98.

Gill, A. M., 1997. Eucalypts and fires: Interdependent or independent?, In *Eucalypt Ecology: Individuals to Ecosystems* (Eds, Williams, J. E. and Woinarski, J. C. Z.) Cambridge University Press, Cambridge, pp. 151-167.

Gill, A. M., 1999. Biodiversity and bushfires: An Australia-wide perspective on plant-species changes after a fire event, In *Australia's Biodiversity - Responses to Fire. Plants, Birds and Invertebrates. Biodiversity Technical Paper, No. 1.* (Eds, Gill, A. M., Woinarski, J. C. Z. and York, A.) Department of the Environment and Heritage, Environment Australia, Canberra, pp. 9-53.

Gill, A. M., 2006. *Bibliography of the 2003 South-eastern Australian Bushfires and their Effects*.

Gill, A. M., 2008. *Underpinnings of Fire Management for Biodiversity Conservation in Reserves. Fire and Adaptive Management Report No. 73,* Department of Sustainability and Environment, Melbourne.

Gill, A. M. and Allan, G., 2008. Large fires, fire effects and the fire-regime concept, *International Journal of Wildland Fire,* **17**: 688-695.

Gill, A. M. and Ashton, D. H., 1968. The role of bark type in relative tolerance to fire of three central Victorian eucalypts, *Australian Journal of Botany,* **16**: 491-498.

Gill, A. M. and Bradstock, R. A., 1992. A national register for the fire responses of plant species, *Cunninghamia,* **2**(4): 653-660.

Gill, A. M. and Bradstock, R. A., 2003. Fire regimes and biodiversity: A set of postulates, In *Australia Burning: Fire Ecology, Policy and Management Issues* (Eds, Cary, G., Lindenmayer, D. and Dovers, S.) CSIRO Publishing, Melbourne, pp. 15-25.

Gill, A. M. and Catling, P. C., 2002. Fire regimes and biodiversity of forested landscapes of southern Australia, In *Flammable Australia: The Fire Regimes and Biodiversity of a Continent* (Eds, Bradstock, R. A., Williams, J. E. and Gill, A. M.) Cambridge University Press, Cambridge, pp. 351-369.

Gill, A. M., Good, R., Kirkpatrick, J. B., J., L., Mansergh, I. and Norris, R., 2004. *Beyond the Bushfires 2003. Environmental Issues in the Australian Alps*, Australian Alps National Parks Liaison Committee, Canberra.

Gill, A. M. and Moore, P. H. R., 1994. Some ecological research perspectives on the disastrous Sydney fires of January 1994., In *Proceedings of the Second International Conference on Forest Fire Research, Coimbra, Portugal, 21-24 November, 1994*, Vol. 1, L.08 (Ed, Viegas, D. X.) University of Coimbra, Coimbra, pp. 63-72.

Gill, A. M. and Moore, P. H. R., 1996. Regional and historical fire weather patterns pertinent to the January 1994 Sydney bushfires, *Proceedings of the Linnean Society of New South Wales,* **116**: 27-36.

Gill, A. M., Moore, P. H. R. and Martin, W. K., 1994. *Bibliography of Fire Ecology in Australia (Including Fire science and Fire Management). Edition 4,* NSW National Parks and Wildlife Service, Hurstville.

Gill, A. M. and Nicholls, A. O., 1989. Monitoring fire-prone flora in reserves for nature conservation, In *Fire Management on Nature Conservation Lands: Proceedings of a National workshop, Busselton, Western Australia, October 1987.* (Eds, Burrows, N., McCaw, L. and Friend, G.) Department of Conservation and Land Management, Perth, pp. 137-151.

Gill, A. M., Stephens, S. L. and Cary, G. J., 2013. The worldwide "wildfire" problem, *Ecological Applications*, **23**(2): 438-454.

Gill, A. M. and Williams, J. E., 1996. Fire regimes and biodiversity: The effects of fragmentation of southeastern Australian eucalypt forests by urbanisation, agriculture and pine plantations, *Forest Ecology and Management,* **85**: 261-278.

Gleason, H. A., 1917. The structure and development of the plant association, *Bulletin of the Torrey Botanical Club,* **44**: 463-481.

Gleason, H. A., 1926. The individualistic concept of the plant association, *Bulletin of the Torrey Botanical Club,* **53**: 7-26.

Glenn-Lewin, D. C., Peet, R. K. and Veblen, T. T. (Eds.), 1992. *Plant Succession: Theory and Prediction. Population and Community Biology Series 11,* Chapman & Hall, London.

Glenn-Lewin, D. C. and van der Maarel, E., 1992. Patterns and processes of vegetation dynamics, In *Plant Succession: Theory and Prediction* (Eds, Glenn-Lewin, D. C., Peet, R. K. and Veblen, T. T.) Chapman & Hall, London, pp. 11-59.

Good, R., 1992. *Kosciusko Heritage: The Conservation Significance of Kosciusko National Park,* National Parks and Wildlife Service of New South Wales, Hurstville.

Good, R., 2004. Rehabilitating fire-damaged wetlands in the Snowy Mountains, *Australasian Plant Conservation,* **12**: 3-4.

Gordon, C. E., Price, O. F., Tasker, E. M. and Denham, A. J., 2017. *Acacia* shrubs respond positively to high severity wildfire: Implications for conservation and fuel hazard management, *Science of the Total Environment,* **575**: 858-868.

Gosper, C. R., Yates, C. J., Cook, G. D., Harvey, J. M., Liedloff, A. C., McCaw, L., Thiele, K. R. and Prober, S. M., 2018. A conceptual model of vegetation dynamics for the unique obligate-seeder eucalypt woodlands of south-western Australia, *Austral Ecology,* **43**: 681-695.

Gould, J. S., McCaw, W. L., Cheney, N. P., Ellis, P. F., Knight, I. K. and Sullivan, A. L., 2007. *Project Vesta - Fire in Dry Eucalypt Forest: Fuel Structure, Fuel Dynamics and Fire Behaviour,* ENSIS-CSIRO / Department of Environment and Conservation, Canberra / Perth.

Green, K. and Sanecki, G. M., 2006. Immediate and short-term responses of bird and mammal assemblages to a subalpine wildfire in the Snowy Mountains, Australia, *Austral Ecology,* **31**(6): 673-681.

Greene, R. H., 1993. Application of repeated measures designs in environmental impact and monitoring studies, *Austral Ecology,* **18**(1): 81-98.

Griffin, G. F., 1992. Will it burn - should it burn? Management of the spinifex grasslands of inland Australia, In *Desertified Grasslands: Their Biology and Management* (Ed, Chapman, G. P.) Academic Press, London, pp. 63-76.

Grime, J. P. and Hillier, S. H., 2000. The contribution of seedling regeneration to the structure and dynamics of plant communities, ecosystems and larger units of the landscape, In *Seeds: The Ecology of Regeneration in Plant Communities* (Ed, Fenner, M.) C.A.B. International, Wallingford, pp. 361-374.

Groffman, P. M., Baron, J. S., Blett, T., Gold, A. J., Goodman, I., Gunderson, L. H., Levinson, B. M., Palmer, M. A., Paerl, H. W., Petersen, G. D., Poff, N. L., Rejeski, D. W., Reynolds, J. F., Turner, M. D., Weathers, K. C. and Wiens, J., 2006. Ecological thresholds: The key to successful environmental management or an important concept with no practical value?, *Ecosystems,* **9**: 1-13.

Grose, R. J., 1957. *A study of some factors associated with the natural regeneration of Eucalyptus delegatensis. Forests Commission of Victoria Bulletin No. 4,* Forests Commission of Victoria, Melbourne.

Grose, R. J., 1963. *The Silviculture of Eucalyptus delegatensis. Part 1. Germination and Seed Dormancy. Bulletin 2,* School of Forestry, University of Melbourne, Melbourne.

Grubb, P. J., 1977. The maintenance of species-richness in plant communities: The importance of the regeneration niche, *Biological Reviews,* **52**(1): 107-145.

Grubb, P. J. and Hopkins, A. J. M., 1986. Resilience at the level of the plant community, In *Resilience in Mediterranean-type Ecosystems. Tasks for Vegetation Science 16* (Eds, Dell, B., Hopkins, A. J. M. and Lamont, B. B.) Dr. W. Junk, Dordrecht, pp. 21-38.

Guisan, A. and Zimmermann, N. E., 2000. Predictive habitat distribution models in ecology, *Ecological Modelling,* **135**: 147-186.

Gullan, P. K., Forbes, S. J., Earl, G. E., Barley, R. H. and Walsh, N. G., 1985. Vegetation of South and Central Gippsland, *Muelleria,* **6**(2): 97-145.

Hall, E. A. A., Specht, R. L. and Eardley, C. M., 1964. Regeneration of the vegetation on Koonamore Vegetation Reserve, 1926-1962, *Australian Journal of Botany,* **12**: 205-264.

Halpern, C. B., 1988. Early successional pathways and the resistance and resilience of forest communities, *Ecology,* **69**(6): 1703-1715.

Hamilton, D. I. and Cowley, N. B., 1991. Southern Tablelands eucalypts of New South Wales, In *Forest Management in Australia* (Eds, McKinnell, F. H., Hopkins, E. R. and Fox, J. E. D.) Institute of Foresters Australia / Surrey Beatty and Sons, Sydney, pp. 170-181.

Hammill, K. A. and Bradstock, R. A., 2006. Remote sensing of fire severity in the Blue Mountains: Influence of vegetation type and inferring fire intensity, *International Journal of Wildland Fire,* **15**: 213-226.

Hammill, K. A. and Bradstock, R. A., 2009. Spatial patterns of fire behaviour in relation to weather, terrain and vegetation, *Proceedings of the Royal Society of Queensland,* **115**: 129-135.

Hammill, K. A., Penman, T. D. and Bradstock, R. A., 2016. Responses of resilience traits to gradients of temperature, rainfall and fire frequency in fire-prone, Australian forests: Potential consequences of climate change, *Plant Ecology,* **217**(6): 725-741.

Hanes, T. L., 1971. Succession after fire in the chaparral of southern California, *Ecological Monographs,* **41**(1): 27-52.

Hanson, C. T. and North, M. P., 2009. Post-fire survival and flushing in three Sierra Nevada conifers with high initial crown scorch, *International Journal of Wildland Fire,* **18**: 857-864.

Hargrove, W. W. and Pickering, J., 1992. Pseudoreplication: A *sine qua non* for regional ecology, *Landscape Ecology,* **6**(4): 251-258.

Harper, J. L., 1977. *Population Biology of Plants,* Academic Press, London.

Harrison, G. W., 1979. Stability under environmental stress: Resistance, resilience, persistence and variability, *The American Naturalist,* **113**(5): 659-669.

Harrison, X. A., Donaldson, L., Correa-Cano, M. E., Evans, J., Fisher, D. N., Goodwin, C. E. D., Robinson, B. S., Hodgson, D. J. and Inger, R., 2018. A brief introduction to mixed effects modelling and multi-model inference in ecology, *PeerJ,* **6**: e4794.

Haslem, A., Kelly, L. T., Nimmo, D. G., Watson, S. J., Kenny, S. A., Taylor, R. S., Avitabile, S. C., Callister, K. E., Spence-Bailey, L. M., Clarke, M. F. and Bennett, A. F., 2011. Habitat or fuel? Implications of long-term, post-fire dynamics for the development of key resources for fauna and fire, *Journal of Applied Ecology,* **48**: 247-256.

Hennessy, K., Lucas, C., Nicholls, N., Bathols, J., Suppiah, R. and Ricketts, J., 2005. *Climate Change Impacts on Fire-weather in South-east Australia*, CSIRO Marine and Atmospheric Research / Bushfire CRC / Australian Bureau of Meteorology, Melbourne.

Henri, C., 2003. Fire policy: An insurance perspective, In *Australia Burning: Fire Ecology, Policy and Management Issues* (Eds, Cary, G., Lindenmayer, D. and Dovers, S.) CSIRO Publishing, Melbourne, pp. 162-165.

Heward, H., Smith, A. M. S., Roy, D. P., Tinkham, W. T., Hoffman, C. M., Morgan, P. and Lannom, K. O., 2013. Is burn severity related to fire intensity? Observations from landscape scale remote sensing, *International Journal of Wildland Fire,* **22**: 910-918.

Hill, R. S. and Jordan, G. J., 2016. Deep history of wildfire in Australia, *Australian Journal of Botany,* **64**: 557-563.

Hill, R. S. and Read, J., 1984. Post-fire regeneration of rainforest and mixed forest in Western Tasmania, *Australian Journal of Botany,* **32**: 481-493.

Hinds, W. T., 1984. Towards monitoring of long-term trends in terrestrial ecosystems, *Environmental Conservation,* **11**(1): 11-18.

Hobbs, R. J. and Atkins, L., 1990. Fire-related dynamics of a *Banksia* woodland in south-western Australia, *Australian Journal of Botany,* **38**: 97-110.

Hodgkinson, K. C., 1998. Sprouting success of shrubs after fire: Height-dependent relationships for different strategies, *Oecologia,* **115**: 64-72.

Hoffmann, W. A., Adasme, R., Haridasan, M., De Carvalho, M. T., Geiger, E. L., Pereira, M. A. B., Gotsch, S. G. and Franco, A. C., 2009. Tree topkill, not mortality, governs the dynamics of savanna-forest boundaries under frequent fire in central Brazil, *Ecology,* **90**(5): 1326-1337.

Holling, C. S., 1973. Resilience and stability of ecological systems, *Annual Review of Ecology and Systematics,* **4**: 1-23.

Hope, G., Macphail, M. and Keaney, B., 2006. *Mountain Occupation Project: Mires of the Australian Capital Territory Region. Notes on Comparative Sites for Palaeoenvironmental Reconstruction Students GE3029*, Australian National University, Department of Archaeology and Natural History, Canberra.

Horn, H. S., 1974. The ecology of secondary succession, *Annual Review of Ecology and Systematics,* **5**: 25-37.

House, S. M., 1997. Reproductive biology of eucalypts, In *Eucalypt Ecology: Individuals to Ecosystems* (Eds, Williams, J. E. and Woinarski, J. C. Z.) Cambridge University Press, Cambridge, pp. 30-55.

Hurlbert, S. H., 1984. Pseudoreplication and the design of ecological field experiments, *Ecological Monographs,* **54**(2): 187-211.

Huston, M., 1979. A general hypothesis of species diversity, *The American Naturalist,* **113**(1): 81-101.

Hutchinson, G. E., 1957. Concluding remarks, *Cold Spring Harbour Symposium on Quantitative Biology,* **22**: 415-427.

Hutchinson, G. E., 1959. Homage to Santa Rosalia or why are there so many kinds of animals?, *The American Naturalist,* **93**(870): 145-159.

Ingwersen, F., 1983. Numerical analysis of the timbered vegetation in Tidbinbilla Nature Reserve, ACT, Australia, *Vegetatio,* **51**: 157-179.

Jackson, J. F., Adams, D. C. and Jackson, U. B., 1999. Allometry of constitutive defense: A model and a comparative test with tree bark and fire regime, *The American Naturalist,* **153**: 614-632.

Jackson, W. D., 1968. Fire, air, earth, water - an elemental ecology of Tasmania, *Proceedings of the Ecological Society of Australia,* **3**: 9-16.

Jacobs, M. R., 1955. *Growth Habits of the Eucalypts*, Forestry and Timber Bureau, Canberra.

Johnson, E. A., 1995. *Fire and Vegetation Dynamics: Studies from the North American Boreal Forest*, Cambridge University Press, Cambridge.

Johnson, E. A., Miyanishi, K. and Bridge, R. J., 2001. Wildfire regime in the Boreal Forest and the idea of suppression and fuel build up, *Conservation Biology,* **15**(6): 1554-1557.

Johnston, F. M. and Johnston, S. W., 2003. Weeds set to flourish following fires, *The Victorian Naturalist,* **120**(5): 194-197.

Johst, K. and Huth, A., 2005. Testing the intermediate disturbance hypothesis: When will there be two peaks of diversity?, *Diversity and Distributions,* **11**: 111-120.

Kanowski, P. J., Whelan, R. J. and Ellis, S., 2005. Inquiries following the 2002-2003 Australian bushfires: Common themes and future directions for Australian bushfire mitigation and management, *Australian Forestry,* **68**(2): 76-86.

Keane, R. E., Agee, J. K., Fule, P., Keeley, J. E., Key, C., Kitchen, S. G., Miller, R. and Schulte, L. A., 2008. Ecological effects of large fires on US landscapes: Benefit or catastrophe?, *International Journal of Wildland Fire,* **17**: 696-712.

Keane, R. E., Hessburg, P. F., Landres, P. B. and Swanson, F. J., 2009. The use of historical range and variability (HRV) in landscape management, *Forest Ecology and Management,* **258**: 1025-1037.

Keeley, J. E., 1981. Reproductive cycles and fire regimes, In *Fire Regimes and Ecosystem Properties. Proceedings of the Conference. December 11-15 Honolulu, Hawaii. USDA Forest Service Technical Report WO-26* (Eds, Mooney, H. A., Bonnicksen, T. M., Christensen, N. L., Lotan, J. E. and Reiners, W. A.) USDA Forest Service, Washington, pp. 231-277.

Keeley, J. E., 1991. Resilience to fire does not imply adaptation to fire: An example from the California chaparral, In *High Intensity Fire in Wildlands: Management Challenges and Options. Proceedings of the 17th Tall Timbers Fire Ecology Conference* (Ed, Hermann, S. M.) Tall Timbers Research Station, Tallahassee, pp. 113-119.

Keeley, J. E., 2006. Fire severity and plant age in postfire resprouting of woody plants in sage scrub and chaparral, *Madrono,* **53**(4): 373-379.

Keeley, J. E., 2009. Fire intensity, fire severity and burn severity: A brief review and suggested usage, *International Journal of Wildland Fire,* **18**: 116-126.

Keeley, J. E., Bond, W. J., Bradstock, R. A., Pausas, J. G. and Rundel, P. W., 2012. *Fire in Mediterranean Ecosystems: Ecology, Evolution and Management,* Cambridge University Press, Cambridge.

Keeley, J. E., Brennan, T. and Pfaff, A. H., 2008. Fire severity and ecosystem responses following crown fires in California shrublands, *Ecological Applications,* **18**(6): 69-94.

Keeley, J. E. and Fotheringham, C. J., 2000. Role of fire in regeneration from seed, In *Seeds: The Ecology of Regeneration in Plant Communities* (Ed, Fenner, M.) C.A.B. International, Wallingford, pp. 311-330.

Keeley, J. E. and Fotheringham, C. J., 2001a. Historic fire regime in southern California shrublands, *Conservation Biology,* **15**(6): 1536-1548.

Keeley, J. E. and Fotheringham, C. J., 2001b. History and management of crown-fire ecosystems: A summary and response, *Conservation Biology,* **15**(6): 1561-1567.

Keeley, J. E., Fotheringham, C. J. and Baer-Keeley, M., 2005. Determinants of postfire recovery and succession in Mediterranean-climate shrublands of California, *Ecological Applications,* **15**(5): 1515-1534.

Keeley, J. E. and Pausas, J. G., 2019. Distinguishing disturbance from perturbations in fire-prone ecosystems, *International Journal of Wildland Fire,* **28**: 282-287.

Keeley, J. E. and van Mantgem, P. J., 2008. Seedling communities, In *Seedling Ecology and Evolution* (Eds, Leck, M. A., Parker, V. T. and Simpson, R. L.) Cambridge University Press, Cambridge, pp. 255-273.

Keeley, J. E. and Zedler, P. H., 2009. Large, high-intensity fire events in southern California shrublands: Debunking the fine-grain age patch model, *Ecological Applications,* **19**(1): 69-94.

Keith, D. A., 1996. Fire-driven extinction of plant populations: A synthesis of theory and review of evidence from Australian vegetation, *Proceedings of the Linnean Society of New South Wales,* **116**: 37-78.

Keith, D. A., 2004. *Ocean Shores to Desert Dunes: The Native Vegetation of New South Wales and the ACT,* Department of Environment and Conservation (NSW), Hurstville.

Keith, D. A. and Bradstock, R. A., 1994. Fire and competition in Australian heath: A conceptual model and field investigations, *Journal of Vegetation Science,* **5**: 347-354.

Keith, D. A. and Myerscough, P. J., 2016. Population variation in germination traits and its implications for responses to climate change in a fire-prone plant species complex, *Plant Ecology,* **217**(6): 781-788.

Kenny, B., Sutherland, E., Tasker, L. and Bradstock, R. A., 2003. *Guidelines for Ecologically Sustainable Fire Management. A Project Undertaken for the NSW Biodiversity Strategy,* NSW National Parks and Wildlife Service, Biodiversity Research and Management Division, Bushfire Research Unit, Sydney.

Kenny, S. A., Bennett, A. F., Clarke, M. F. and Morgan, J. W., 2018. Time-since-fire and climate interact to affect the structural recovery of an Australian semi-arid plant community, *Austral Ecology*.

Kirkpatrick, J. B., 1997. Vascular plant-eucalypt interactions, In *Eucalypt Ecology: Individuals to Ecosystems* (Eds, Williams, J. E. and Woinarski, J. C. Z.) Cambridge University Press, Cambridge, pp. 227-245.

Kitajima, K. and Fenner, M., 2000. Ecology of seedling regeneration, In *Seeds: The Ecology of Regeneration in Plant Communities* (Ed, Fenner, M.) C.A.B. International, Wallingford, pp. 331-360.

Kitchin, M., 2008. *Ecological Targets for Planned Fire Management - Deriving Fire Thresholds for the Conservation of Plant Species*, Territory and Municipal Services, Parks, Conservation and Lands, Research and Planning, Canberra.

Knox, K. J. E. and Clarke, P. J., 2012. Fire severity, feedback effects and resilience to alternative community states in forest assemblages, *Forest Ecology and Management,* **265**: 47-54.

Knox, K. J. E. and Clarke, P. J., 2016. Measuring fire severity: Are canopy, understorey and below-ground measures coupled in sclerophyll forest fires?, *Plant Ecology,* **217**(6): 607-615.

Koch, J. M., Ruschmann, A. M. and Morald, T. K., 2009. Effect of time since burn on soil seedbanks in the Jarrah forest of Western Australia, *Australian Journal of Botany,* **57**: 647-660.

Krawchuk, M. A., Meigs, G. W., Cartwright, J. M., Coop, J. D., Davis, R., Holz, A., Kolden, C. A. and Meddens, A. J. H., 2020. Disturbance refugia within mosaics of forest fire, drought, and insect outbreaks, *Frontiers in Ecology and Environment,* **18**(5): 235-244.

Lamont, B. B., 1991. Canopy seed storage and release - what's in a name?, *OIKOS,* **60**(2): 266-268.

Lamont, B. B., Le Maitre, D. C., Cowling, R. M. and Enright, N. J., 1991. Canopy seed storage in woody plants, *The Botanical Review,* **57**(4): 277-317.

Laughlin, D. C., Bakker, J. D., Stoddard, M. T., Daniels, M. L., Springer, J. D., Gildar, C. N., Green, A. M. and Covington, W. W., 2004. Toward reference conditions: Wildfire effects on flora in an old-growth ponderosa pine forest, *Forest Ecology and Management,* **199**: 137-152.

Laughlin, D. C. and Grace, J. B., 2006. A multivariate model of plant species richness in forested systems: Old-growth montane forests with a long history of fire, *OIKOS,* **114**: 60-70.

Lauvaux, C. A., Skinner, C. N. and Taylor, A. H., 2016. High severity fire and mixed conifer forest-chaparral dynamics in the southern Cascade Range, USA, *Forest Ecology and Management,* **363**: 74-85.

Lavorel, S., Colloff, M. J., McIntyre, S., Doherty, M. D., Murphy, H. T., Metcalfe, D. J., Dunlop, M., Williams, R. J., Wise, R. M. and Williams, K. J., 2015. Ecological mechanisms underpinning climate adaptation services, *Global Change Biology,* **21**(1): 12-31.

Lawes, M. J. and Clarke, P. J., 2011. Ecology of plant resprouting: Populations to community responses in fire-prone ecosystems, *Plant Ecology,* **212**: 1937-1943.

Lee, Y. and Nelder, J. A., 1996. Hierarchical generalized linear models, *Journal of the Royal Statistical Society Series B (Methodological),* **58**(4): 619-678.

Lee, Y. and Nelder, J. A., 2001. Hierarchical generalised linear models: A synthesis of generalised linear models, random-effects models and structured dispersions, *Biometrika,* **88**(4): 987-1006.

Leigh, J. H., Wimbush, D. J., Wood, D. H., Holgate, M. D., Slee, A. V., Stanger, M. G. and Forrester, R. I., 1987. Effects of rabbit grazing and fire on a subalpine environment. I. Herbaceous and shrubby vegetation, *Australian Journal of Botany,* **35**: 433-464.

Leigh, J. H., Wood, D. H., Slee, A. V. and Holgate, M. D., 1991. The effects of burning and simulated grazing on productivity, forage quality, mortality and flowering of eight subalpine herbs in Kosciusko National Park, *Australian Journal of Botany,* **39**: 97-118.

Leonard, J., 2003. People and property: A researcher's perspective, In *Australia Burning: Fire Ecology, Policy and Management Issues* (Eds, Cary, G., Lindenmayer, D. and Dovers, S.) CSIRO Publishing, Melbourne, pp. 103-112.

Levin, S. A. and Paine, R. T., 1974. Disturbance, patch formation and community structure, *Proceedings of the National Academy of Sciences,* **71**(7): 2744-2747.

Li, J., Loneragan, J. A., Duggin, J. A. and Grant, C. D., 2004. Issues affecting the measurement of disturbance response patterns in herbaceous vegetation - a test of the intermediate disturbance hypothesis, *Plant Ecology,* **172**: 11-26.

Lindenmayer, D. B., Hobbs, R. J., Likens, G. E., Krebs, C. J. and Banks, S. C., 2011. Newly discovered landscape traps produce regime shifts in wet forests, *Proceedings of the National Academy of Sciences,* **108**(38): 15887-15891.

Lucas, C., Hennessy, K., Mills, G. and Bathols, J., 2007. *Bushfire Weather in Southeast Australia: Recent Trends and Projected Climate Change Impacts. Consultancy Report Prepared for the Climate Institute of Australia,* Bushfire CRC / Australian Bureau of Meteorology Melbourne.

Luke, R. H. and McArthur, A. G., 1978. *Bushfires in Australia,* Department of Primary Industry and Timber Bureau. CSIRO Division of Forest Research. Australian Government Publishing Service, Canberra.

Lunn, T. J., Gerwin, M., Buettel, J. C. and Brook, B. W., 2018. Impact of intense disturbance on the structure and composition of wet-eucalypt forests: A case study from the Tasmanian 2016 fires, *PLoS ONE,* **13**(7): [online] e0200905.

Lunt, I. D. and Morgan, J. W., 1999. Effect of fire frequency on plant composition at the Laverton North Grassland Reserve, Victoria, *The Victorian Naturalist,* **116**(3): 84-90.

Lunt, I. D., Prober, S. M. and Morgan, J. W., 2012. How do fire regimes affect ecosystem structure, function and diversity in grasslands and grassy woodlands of southern Australia?, In

Flammable Australia: Fire Regimes, Biodiversity and Ecosystems in a Changing World (Eds, Bradstock, R. A., Gill, A. M. and Williams, R. J.) CSIRO Publishing, Melbourne, pp. 253-270.

MacDonald, R. L., Chen, H. Y. H., Bartels, S. F., Palik, B. J. and Prepas, E. E., 2015. Compositional stability of boreal understorey vegetation after overstorey harvesting across a riparian ecotone, *Journal of Vegetation Science,* **26**: 733-741.

Maconochie, J. R., 1982. Regeneration of arid zone plants: a floristic survey, In *Evolution of the Flora and Fauna of Arid Australia* (Eds, Barker, W. R. and Greenslade, P. J. M.) Peacock Publications / Australian Systematic Botany Society / ANZAAS South Australian Division, Adelaide, pp. 141-144.

Malanson, G. P., 1984. Intensity as a third factor of disturbance regime and its effect on species diversity, *OIKOS,* **43**(3): 411-413.

Malanson, G. P. and Trabaud, L., 1987. Ordination analysis of components of resilience of *Quercus coccifera* garrigue, *Ecology,* **68**(3): 463-472.

Mallen-Cooper, M., Nakagawa, S. and Eldridge, D. J., 2019. Global meta-analysis of soil-disturbing vertebrates reveals strong effects on ecosystem patterns and processes, *Global Ecology and Biogeography,* **28**: 661-679.

Marchant, R. and Dean, J., 2014. A long-term study of the factors that influence compositional stability of stream invertebrates, *Inland Waters,* **4**: 113-120.

Matthews, J. A., 1999. Disturbance regimes and ecosystem response on recently-deglaciated substrates, In *Ecosystems of Disturbed Ground. Ecosystems of the World 16* Vol. 16 (Ed, Walker, L. R.) Elsevier, Amsterdam, pp. 17-37.

Matthews, W. J., Marsh-Matthews, E., Cashner, R. C. and Gelwick, F., 2013. Disturbance and trajectory of change in a stream fish community over four decades, *Oecologia,* **173**: 955-969.

May, R. M., 1973. *Stability and Complexity in Model Ecosystems. Monographs in Population Biology 6,* Princeton University Press, Princeton.

McCarthy, M. A., Gill, A. M. and Lindenmayer, D. B., 1999. Fire regimes in mountain ash forest: Evidence from forest age structure, extinction models and wildlife habitat, *Forest Ecology and Management,* **124**: 193-203.

McDougall, K. L., 2007. Grazing and fire in two subalpine peatlands, *Australian Journal of Botany,* **55**: 42-47.

McDougall, K. L., Walsh, N. G. and Wright, G. T., 2015. Recovery of treeless subalpine vegetation in Kosciuszko National Park after the landscape-scale fire of 2003, *Australian Journal of Botany,* **63**: 597-607.

McIntosh, R. P., 1987. *The Background of Ecology: Concept and Theory,* Cambridge University Press, Cambridge.

McIntyre, S. and Lavorel, S., 1994. Predicting richness of native, rare and exotic plants in response to habitat and disturbance variables across a variegated landscape, *Conservation Biology,* **8**(2): 521-531.

McLeod, R., 2003. *Inquiry into the Operational Response to the January 2003 Bushfires in the ACT*, ACT Government, Canberra.

Merritt, D. J., Turner, S. R., Clarke, S. and Dixon, K. W., 2007. Seed dormancy and germination stimulation syndromes for Australian temperate species, *Australian Journal of Botany,* **55**: 336-344.

Meyn, A., White, P. S., Buhk, C. and Jentsch, A., 2007. Environmental drivers of large, infrequent wildfires: The emerging conceptual model, *Progress in Physical Geography,* **31**(3): 287-312.

Millar, C. I. and Stephenson, N. L., 2015. Temperate forest health in an era of emerging megadisturbance, *Science,* **349**(6250): 823-826.

Miller, C. and Urban, D. L., 1999. Forest pattern, fire, and climatic change in the Sierra Nevada, *Ecosystems,* **2**: 76-87.

Miller, T. E., 1982. Community diversity and interactions between the size and frequency of disturbance, *The American Naturalist,* **120**(4): 533-536.

Mok, H.-F., Arndt, S. A. and Nitschke, C. R., 2012. Modelling the potential impact of climate variability and change in the temperate forests of south-eastern Australia, *Global Change Biology,* **18**: 1053-1072.

Monamy, V. and Fox, B. J., 2000. Small mammal succession is determined by vegetation density rather than time elapsed since disturbance, *Austral Ecology,* **25**: 580-587.

Moreira, F., Catry, F., Duarte, I., Acacio, V. and Silva, J. S., 2008. A conceptual model of sprouting responses in relation to fire damage: An example with cork oak (*Quercus suber* L.) trees in Southern Portugal, *Plant Ecology,* **201**(1): 77-85.

Moreno, J. M. and Oechel, W. C., 1989. A simple method for estimating fire intensity after a burn in California chaparral, *Acta Oecologica: Oecologia Plantarum,* **10**: 57-68.

Morgan, J. W. and Nield, C., 2011. Contrasting effects of fire severity on regeneration of the dominant woody species in two coastal plant communities at Wilsons Promontory, Victoria, *Cunninghamia,* **12**(1): 53-60.

Morgan, P., Aplet, G. H., Haufler, J. B., Humphries, H. C., Moore, M. M. and Wilson, D., 1994. Historical range of variability: A useful tool for evaluating ecosystem change, *Journal of Sustainable Forestry,* **2**: 87-111.

Morgan, P., Hardy, C. C., Swetnam, T. W., Rollins, M. G. and Long, D. G., 2001. Mapping fire regimes across time and space: Understanding coarse and fine-scale pattern, *International Journal of Wildland Fire,* **10**: 329-342.

Morgan, P., Keane, R. E., Dillon, G. K., Jain, T. B., Hudak, A. T., Karau, E. C., Sikkink, P. G., Holden, Z. A. and Strand, E. K., 2014. Challenges of assessing fire and burn severity using field measures, remote sensing and modelling, *International Journal of Wildland Fire,* **23**: 1045-1060.

Moritz, M. A., 1997. Analyzing extreme disturbance events: Fire in Los Padres National Forest, *Ecological Applications,* **7**(4): 1252-1262.

Morrison, D. A., 1995. Some effects of low-intensity fires on populations of co-occurring small trees in the Sydney Region, *Proceedings of the Linnean Society of New South Wales,* **115**: 109-119.

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

Morrison, D. A., Cary, G. J., Pengelly, S. M., Ross, D. G., Mullins, B. J., Thomas, C. R. and 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, D. A. and Renwick, J. A., 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.

Murdoch, A. J. and Ellis, R. H., 2000. Dormancy, viability and longevity, In *Seeds: The Ecology of Regeneration in Plant Communities* (Ed, Fenner, M.) C.A.B. International, Wallingford, pp. 183-214.

Mutch, R. W., 1970. Wildland fires and ecosystems - a hypothesis, *Ecology,* **51**(6): 1046-1051.

Myerscough, P. J., Clarke, P. J. and Skelton, N. J., 1996. Plant coexistence in coastal heaths: Habitat segregation in the post-fire environment, *Australian Journal of Ecology,* **21**: 47-54.

Nano, C. E. M., Jobson, P. and Wardle, G. M., 2017. Arid shrublands and open woodlands of inland Australia, In *Australian Vegetation* (Ed, Keith, D. A.) Cambridge University Press, Cambridge, pp. 626-650.

Newman, J. C., 1954. Burning on sub-alpine pastures, *Journal of the Soil Conservation Service of NSW,* **10**(3): 135-140.

Newsome, A. E. and Catling, P. C., 1979. Habitat preferences of mammals inhabiting heathlands of warm temperate coastal, montane and alpine regions of southeastern Australia, In *Heathlands and Related Shrublands of the World, A. Descriptive Studies* (Ed, Specht, R. L.) Elsevier, Amsterdam, pp. 301-316.

Newsome, A. E., McIlroy, J. and Catling, P., 1975. The effects of an extensive wildfire on populations of twenty ground vertebrates in south-east Australia, *Proceedings of the Ecological Society of Australia,* **9**: 107-123.

Nieuwenhuis, A., 1987. The effect of fire frequency on the sclerophyll vegetation of the West Head, New South Wales, *Australian Journal of Ecology,* **12**: 373-385.

Noble, I. R., 1984. Mortality of lignotuberous seedlings of *Eucalyptus* species after an intense fire in montane forest, *Australian Journal of Ecology,* **9**: 47-50.

Noble, I. R. and Slatyer, R. O., 1977. The effect of disturbance on plant succession, *Proceedings of the Ecological Society of Australia,* **10**: 135-145.

Noble, I. R. and Slatyer, R. O., 1980. The use of vital attributes to predict successional changes in plant communities subject to recurrent disturbances, *Vegetatio,* **43**: 5-21.

Noble, J. C., 1989. Fire regimes and their influence on herbage and mallee coppice dynamics, In *Mediterranean Landscapes in Australia: Mallee Ecosystems and their Management* (Eds, Noble, J. C. and Bradstock, R. A.) CSIRO, Melbourne, pp. 168-180.

Noble, J. C. and Bradstock, R. A. (Eds.), 1989. *Mediterranean Landscapes in Australia: Mallee Ecosystems and their Management,* CSIRO, Melbourne.

Nolan, R. H., Boer, M. M., Collins, L., de Dios, V. R., Jenkins, M. E., Kenny, B. and Bradstock, R. A., 2020. Causes and consequences of eastern Australia's 2019–20 season of mega-fires, *Global Change Biology,* **26**: 1039-1041.

Nolan, R. H., Lane, P. N. J., Benyon, R. G., Bradstock, R. A. and Mitchell, P. J., 2014. Changes in evapotranspiration following wildfire in resprouting eucalypt forests, *Ecohydrology,* **7**: 1363-1377.

Nolan, R. H., Lane, P. N. J., Benyon, R. G., Bradstock, R. A. and Mitchell, P. J., 2015. Trends in evapotranspiration and streamflow following wildfire in resprouting eucalypt forests, *Journal of Hydrology,* **524**: 614-624.

NSW Department of Planning Industry and Environment, 2020. *Australian Alps - biodiversity,* Available at: https://www.environment.nsw.gov.au/bioregions/AustralianAlps-Biodiversity.htm (accessed 22/12/2020).

NSW National Parks and Wildlife Service, 2009. *Brindabella National Park and State Conservation Area Plan of Management,* NSW Department of Environment and Climate Change, Sydney.

NSW National Parks and Wildlife Service, 2010. *Burrinjuck Nature Reserve Plan of Management,* NSW Department of Environment, Climate Change and Water, Sydney.

O'Dowd, D. J. and Gill, A. M., 1984. Predator satiation and site alteration following fire: Mass reproduction of Alpine Ash (*Eucalyptus delegatensis*) in southeastern Australia, *Ecology,* **65**(4): 1052-1066.

O'Neill, R. V., DeAngelis, D. L., Waide, J. B. and Allen, T. F. H., 1986. *A Hierarchical Concept of Ecosystems. Monographs in Population Biology 23,* Princeton University Press, Princeton.

Oksanen, L., 2001. Logic of experiments in ecology: Is pseudoreplication a pseudoissue?, *OIKOS,* **94**: 27-38.

Olson, J. S., 1981. Carbon balance in relation to fire regimes, In *Fire Regimes and Ecosystem Properties. Proceedings of the Conference. December 11-15 Honolulu, Hawaii. USDA Forest Service Technical Report WO-26* (Eds, Mooney, H. A., Bonnicksen, T. M., Christensen, N. L., Lotan, J. E. and Reiners, W. A.) USDA Forest Service, Washington, pp. 327-378.

Ooi, M. K. J., 2010. Delayed emergence and post-fire recruitment success: Effects of seasonal germination, fire season and dormancy type, *Australian Journal of Botany,* **58**: 248-256.

Pate, J. S., Froend, R. H., Bowen, B. J., Hansen, A. and Kuo, J., 1990. Seedling growth and storage characteristics of seeder and resprouter species of Mediterranean-type ecosystems of S.W. Australia, *Annals of Botany,* **65**: 585-601.

Pausas, J. G., Bradstock, R. A., Keith, D. A., Keeley, J. E. and The GCTE (Global Change of Terrestrial Ecosystems) Fire Network, 2004. Plant functional traits in relation to fire in crown-fire ecosystems, *Ecology,* **85**(4): 1085-1100.

Pausas, J. G. and Keeley, J. E., 2014. Evolutionary ecology of resprouting and seeding in fire-prone ecosystems, *New Phytologist,* **204**: 55-65.

Pausas, J. G. and Lloret, F., 2007. Spatial and temporal patterns of plant functional types under simulated fire regimes, *International Journal of Wildland Fire,* **16**: 484-492.

Pausas, J. G., Llovet, J., Rodrigo, A. and Vallejo, R., 2008. Are wildfires a disaster in the Mediterranean basin? - a review, *International Journal of Wildland Fire,* **17**: 713-723.

Peet, R. K., 1992. Community structure and ecosystem function, In *Plant Succession: Theory and Prediction* (Eds, Glenn-Lewin, D. C., Peet, R. K. and Veblen, T. T.) Chapman & Hall, London, pp. 103-151.

Penfold, A. R. and Willis, J. L., 1961. *The Eucalypts: Botany, Cultivation, Chemistry and Utilization,* Leonard Hill (Books) / Interscience Publishers

London / New York.

Penman, T. D., Beukers, M., Kavanagh, R. P. and Doherty, M., 2011. Are long-unburnt eucalypt forest patches important for the conservation of plant species diversity?, *Applied Vegetation Science,* **14**: 172-180.

Penman, T. D., Binns, D. L., Brassil, T. E., Shiels, R. J. and Allen, R. M., 2009. Long-term changes in understorey vegetation in the absence of wildfire in south-east dry sclerophyll forests, *Australian Journal of Botany,* **57**: 533-540.

Penman, T. D., Binns, D. L., Shiels, R. J., Allen, R. M. and Kavanagh, R. P., 2008. Changes in understorey plant species richness following logging and prescribed burning in shrubby dry sclerophyll forests of south-eastern Australia, *Austral Ecology,* **33**(2): 197-210.

Perez, B. and Moreno, J. M., 1998. Methods for quantifying fire severity in shrubland-fires, *Plant Ecology,* **139**: 91-101.

Peters, R. H., 1991. *A Critique for Ecology,* Cambridge University Press, Cambridge.

Petraitis, P. S., Latham, R. E. and Niesenbaum, R. A., 1989. The maintenance of species diversity by disturbance, *The Quarterly Review of Biology,* **64**(4): 393-418.

Pickett, S. T. A., 1976. Succession: An evolutionary interpretation, *The American Naturalist,* **110**(971): 107-119.

Pickett, S. T. A., 1980. Non-equilibrium coexistence of plants, *Bulletin of the Torrey Botanical Club,* **107**(2): 238-248.

Pickett, S. T. A. and Cadenasso, M. L., 2005. Vegetation dynamics, In *Vegetation Ecology* (Ed, van der Maarel, E.) Blackwell Publishing, Oxford, pp. 172-198.

Pickett, S. T. A., Cadenasso, M. L. and Meiners, S. J., 2008. Ever since Clements: From succession to vegetation dynamics and understanding to intervention, *Applied Vegetation Science,* **12**: 9-21.

Pickett, S. T. A., Collins, S. L. and Armesto, J. J., 1987. Models, mechanisms and pathways of succession, *The Botanical Review,* **53**(3): 335-371.

Pickett, S. T. A., Kolasa, J., Armesto, J. J. and Collins, S. L., 1989. The ecological concept of disturbance and its expression at various hierarchical levels, *OIKOS,* **54**: 129-136.

Pickett, S. T. A. and McDonnell, M. J., 1989. Changing perspectives in community dynamics: A theory of successional forces, *TRENDS in Ecology and Evolution,* **4**: 241-245.

Pickett, S. T. A. and White, P. S. (Eds.), 1985. *The Ecology of Natural Disturbance and Patch Dynamics,* Academic Press, San Diego.

Pimm, S. L., 1984. The complexity and stability of ecosystems, *Nature,* **307**: 321-326.

Pitman, A. J., Narisma, G. T. and McAneney, J., 2007. The impact of climate change on the risk of forest and grassland fires in Australia, *Climatic Change,* **84**: 383-401.

Pook, E. W., Costin, A. B. and Moore, C. W. E., 1966. Water stress in native vegetation during the drought of 1965, *Australian Journal of Botany,* **14**: 257-267.

Poore, M. E. D., 1955. The use of phytosociological methods in ecological investigations II. Practical issues involved in an attempt to apply the Braun-Blanquet system, *Journal of Ecology,* **43**: 245-269.

Posamentier, H. G., Clark, S. S., Hain, D. L. and Recher, H. F., 1981. Succession following wildfire in coastal heathland (Nadgee Nature Reserve N.S.W.), *Australian Journal of Ecology,* **6**: 165-175.

Probert, R. J., 2000. The role of temperature in the regulation of seed dormancy and germination, In *Seeds: The Ecology of Regeneration in Plant Communities* (Ed, Fenner, M.) C.A.B. International, Wallingford, pp. 261-292.

Pryor, L. D., 1953. Genetic control in *Eucalyptus* distribution, *Proceedings of the Linnean Society of New South Wales,* **78**: 8-18.

Pryor, L. D., Chattaway, M. M. and Kloot, N. H., 1956. The inheritance of wood and bark characters in *Eucalyptus, Australian Journal of Botany,* **4**(3): 216-239.

Pryor, L. D. and Johnson, L. A. S., 1971. *A Classification of the Eucalypts,* ANU Press, Canberra.

Purdie, R. W., 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, R. W., 1977b. Early stages of regeneration after burning in dry sclerophyll vegetation. II. Regeneration by seed germination, *Australian Journal of Botany,* **25**: 35-46.

Purdie, R. W. and Slatyer, R. O., 1976. Vegetation succession after fire in sclerophyll woodland communities in south-eastern Australia, *Australian Journal of Ecology,* **1**: 223-236.

Raison, R. J., 1980. A review of the role of fire in nutrient cycling in Australian native forests and of methodology for studying the fire-nutrient interaction, *Australian Journal of Ecology,* **5**: 15-21.

Raison, R. J., 1981. More on the effects of intense fires on the long-term productivity of forest sites: Reply to comments, *Search,* **12**(1-2): 10-14.

Raison, R. J., Khanna, P. K. and Woods, P. V., 1985. Transfer of elements to the atmosphere during low-intensity prescribed fires in three Australian subalpine eucalypt forests, *Canadian Journal of Forest Research,* **15**: 657-664.

Raison, R. J., Woods, P. V. and Khanna, P. K., 1983. Dynamics of fine fuels in recently burnt eucalypt forests, *Australian Forestry,* **46**(4): 294-302.

Raison, R. J., Woods, P. V. and Khanna, P. K., 1986. Decomposition and accumulation of litter after fire in sub-alpine eucalypt forests, *Australian Journal of Ecology,* **11**: 9-19.

Raunkiaer, C., 1934. *The Life Forms of Plants and Statistical Plant Geography,* Clarendon Press, Oxford.

Recher, H. F., Lunney, D. and Posamentier, H. G., 1975. A grand natural experiment. The Nadgee fire, *Australian Natural History,* **18**(5): 152-163.

Richter, C., Rejmanek, M., Miller, J. E. D., Welch, K. R., Weeks, J. M. and Safford, H. D., 2019. The species diversity x fire severity relationship is hump-shaped in semiarid yellow pine and mixed conifer forests, *Ecosphere,* **10**(10): Article e02882.

Ricklefs, R. E., 1987. Community diversity: Relative roles of local and regional processes, *Science,* **235**: 167-171.

Ricotta, C. and Feoli, E., 2013. Does ordinal cover estimation offer reliable quality data structures in vegetation ecological studies?, *Folia Geobot,* **48**: 437-447.

Riginos, C., Porensky, L. M., Veblen, K. E. and Young, T. P., 2018. Herbivory and drought generate short-term stochasticity and long-term stability in a savanna understory community, *Ecological Applications,* **28**(2): 323-335.

Romme, W. H., Boyce, M. S., Gresswell, R., Merrill, E. H., Minshall, G. W., Whitlock, C. and Turner, M. G., 2011. Twenty years after the 1988 Yellowstone Fires: Lessons about disturbance and ecosystems, *Ecosystems,* **14**(7): 1196-1215.

Romme, W. H., Everham, E. H., Frelich, L. E., Moritz, M. A. and Sparks, R. E., 1998. Are large, infrequent disturbances qualitatively different from small, frequent disturbances?, *Ecosystems,* **1**: 524-534.

Rothermel, R. C. and Deeming, J. E., 1980. *Measuring and Interpreting Fire Behaviour for Correlation with Fire Effects. USDA Forest Service General Technical Report INT-93,* Intermountain Forest and Range Experiment Station, USDA Forest Service, Ogden.

Rowe, J. S., 1983. Concepts of fire effects on plant individuals and species., In *The Role of Fire in Circumpolar Ecosystems* (Eds, Wein, R. W. and Maclean, D. A.) John Wiley and Sons, New York, pp. 135-154.

Roxburgh, S. H., Shea, K. and Wilson, J. B., 2004. The intermediate disturbance hypothesis: Patch dynamics and mechanisms of species coexistence, *Ecology,* **85**(2): 359-371.

Russell-Smith, J., Ryan, P. G. and Cheal, D. C., 2002. Fire regimes and the conservation of sandstone heath in monsoonal northern Australia: Frequency, interval, patchiness, *Biological Conservation,* **104**: 91-106.

Russell, M. J. and Roberts, B. R., 1996. Effects of four low-intensity burns over 14 years on the floristics of a Blackbutt (*Eucalyptus pilularis*) forest in southern Queensland, *Australian Journal of Botany,* **44**: 315-329.

Russell, R. P. and Parsons, R. F., 1978. Effects of time since fire on heath floristics at Wilson's Promontory, southern Australia, *Australian Journal of Botany,* **26**: 53-61.

Ruthrof, K. X., Fontaine, J. B., Matusick, G., Breshears, D. D., Law, D. J., Powell, S. and Hardy, G., 2016. How drought-induced forest die-off alters microclimate and increases fuel loadings and fire potentials, *International Journal of Wildland Fire,* **25**: 819-830.

Rykiel, E. J., 1985. Towards a definition of ecological disturbance, *Australian Journal of Ecology,* **10**: 361-365.

Santana, V. M., Bradstock, R. A., Ooi, M. K. J., Denham, A. J., Auld, T. D. and Baeza, M. J., 2010. Effects of soil temperature regimes after fire on seed dormancy and germination in six Australian Fabaceae species, *Australian Journal of Botany,* **58**: 539-545.

Schoennagel, T., Smithwick, E. A. H. and Turner, M. D., 2008. Landscape heterogeneity following large fires: Insights from Yellowstone National Park, USA, *International Journal of Wildland Fire,* **17**: 742-753.

Schultz, N. L., Reid, N., Lodge, G. and Hunter, J. T., 2014. Seasonal and interannual variation in vegetation composition: Implications for survey design and data interpretation, *Austral Ecology,* **39**: 755-766.

Schwilk, D. W., 2003. Flammability is a niche construction trait: Canopy architecture affects fire intensity, *The American Naturalist,* **162**(6): 725-733.

Schwilk, D. W., Keeley, J. E. and Bond, W. J., 1997. The intermediate disturbance hypothesis does not explain fire and diversity pattern in fynbos, *Plant Ecology,* **132**: 77-84.

Sebire, I. and Fagg, P., 2009. *High Elevation Mixed Species in Victoria's State Forests. Silviculture Reference Manual No. 2,* Department of Sustainability and Environment, Melbourne.

Shafi, M. I. and Yarranton, G. A., 1973. Diversity, floristic richness and species evenness during a secondary (post-fire) succession, *Ecology,* **54**(4): 897-902.

Sharples, J. J., Cary, G. J., Fox-Hughes, P., Mooney, S., Evans, J. P., Fletcher, M.-S., Fromm, M., Grierson, P. F., McRae, R. and Baker, P., 2016. Natural hazards in Australia: Extreme bushfires, *Climatic Change,* **139**: 85-99.

Sharples, J. J., McRae, R. H. D. and Wilkes, S. R., 2012. Wind-terrain effects on the propagation of wildfires in rugged terrain: fire channelling, *International Journal of Wildland Fire,* **21**: 282-296.

Shea, K., Roxburgh, S. H. and Rauschert, E. S. J., 2004. Moving from pattern to process: Coexistence mechanisms under intermediate disturbance regimes, *Ecology Letters,* **7**: 491-508.

Shugart, H. H. and Noble, I. R., 1981. A computer model of succession and fire response of the high-altitude *Eucalyptus* forest of the Brindabella Range, Australian Capital Territory, *Australian Journal of Ecology,* **6**: 149-164.

Silvertown, J., 2004. Plant coexistence and the niche, *TRENDS in Ecology and Evolution,* **19**(11): 605-611.

Simard, A. J., 1991. Fire severity, changing scales and how things hang together, *International Journal of Wildland Fire,* **1**(1): 23-34.

Smith, A. L., Blanchard, W., Blair, D., McBurney, L., Banks, S. C., Driscoll, D. A. and Lindenmayer, D., 2016. The dynamic regeneration niche of a forest following a rare disturbance event, *Diversity and Distributions,* **22**: 457-467.

Smucker, K. M., Hutto, R. L. and Steele, B. M., 2005. Changes in bird abundance after wildfire: Importance of fire severity and time since fire, *Ecological Applications,* **15**(5): 1535-1549.

Sneath, P. H. A. and Sokal, R. R., 1973. *Numerical Taxonomy: The Principles and Practice of Numerical Classification,* W.H. Freeman & Co., San Francisco.

Sousa, W. P., 1984. The role of disturbance in natural communities, *Annual Review of Ecology and Systematics,* **15**: 353-391.

Specht, R. L., 1970. Vegetation, In *The Australian Environment. 4th ed* (Ed, Leeper, G. W.) CSIRO / Melbourne University Press, Melbourne, pp. 44-67.

Specht, R. L., 1981a. Ecophysiological principles determining the biogeography of major vegetation formations in Australia, In *Ecological Biogeography of Australia,* Vol. 41 (Ed, Keast, A.) Dr. W. Junk, The Hague, pp. 299-333.

Specht, R. L., 1981b. Major vegetation formations in Australia, In *Ecological Biogeography of Australia*, Vol. 41 (Ed, Keast, A.) Dr. W. Junk, The Hague, pp. 163-297.

Specht, R. L., 1981c. Responses to fires in heathlands and related shrublands, In *Fire and the Australian Biota* (Eds, Gill, A. M., Groves, R. H. and Noble, I. R.) Australian Academy of Science, Canberra, pp. 395-415.

Specht, R. L., 1983. Foliage projective covers of overstorey and understorey strata of mature vegetation in Australia, *Australian Journal of Ecology,* **8**: 433-439.

Specht, R. L. and Morgan, D. G., 1981. The balance between the foliage projective covers of overstorey and understorey strata in Australian vegetation, *Australian Journal of Ecology,* **6**: 193-202.

Specht, R. L., Rayson, P. and Jackman, M. E., 1958. Dark Island Heath (Ninety-Mile Plain, South Australia). VI. Pyric succession: Changes in composition, coverage, dry weight and mineral nutrient status, *Australian Journal of Botany,* **6**(1): 59-88.

Specht, R. L. and Specht, A., 1989a. Species richness of overstorey strata in Australian plant communities - the influence of overstorey growth rates, *Australian Journal of Botany,* **37**: 321-336.

Specht, R. L. and Specht, A., 1989b. Species richness of sclerophyll (heathy) plant communities in Australia - the influence of overstorey cover, *Australian Journal of Botany,* **37**: 337-350.

Sprugel, D. G., 1991. Disturbance, equilibrium and environmental variability: What is 'natural' vegetation in a changing environment?, *Biological Conservation,* **58**: 1-18.

Stajsic, V., 2010. *Grevillea victoriae* subsp. *brindabella* (Proteaceae), a new subspecies from the Southern Tablelands of New South Wales, *Muelleria,* **28**: 18-28.

Stephens, S. L., Agee, J. K., Fule, P. Z., North, M. P., Romme, W. H. and Swetnam, T. W., 2013. Managing forests and fire in changing climates, *Science,* **342**: 41-42.

Stephenson, N. L., Parsons, D. J. and Swetnam, T. W., 1991. Restoring natural fire to the sequoia-mixed conifer forest: Should intense fire play a role?, In *High Intensity Fire in Wildlands: Management Challenges and Options. Proceedings of the 17th Tall Timbers Fire Ecology Conference* (Ed, Hermann, S. M.) Tall Timbers Research Station, Tallahassee, pp. 321-337.

Strand, E. K., Satterberg, K. L., Hudak, A. T., Byrne, J., Khalyani, A. H. and Smith, A. M. S., 2019. Does burn severity affect plant community diversity and composition in mixed conifer forests of the United States Intermountain West one decade post fire?, *Fire Ecology,* **15**: Article 25.

Strasser, M. J., Pausas, J. G. and Noble, I. R., 1996. Modelling the response of eucalypts to fire, Brindabella Ranges, ACT, *Australian Journal of Ecology,* **21**: 341-344.

Strusz, D. L., 1971. *1:250,000 Geological Series - Explanatory Notes. Canberra Australian Capital Territory and New South Wales,* Department of National Development, Bureau of Mineral Resources, Geology and Geophysics. Australian Government Publishing Service, Canberra.

Svensson, J. R., Lindegarth, M. and Pavia, H., 2009. Equal rates of disturbance cause different patterns of diversity, *Ecology,* **90**(2): 496-505.

Swaine, M. D. and Greig-Smith, P., 1980. An application of principal components analysis to vegetation change in permanent plots, *Journal of Ecology,* **68**: 33-41.

Swetnam, T. W., Baisan, C. H., Caprio, A. C., Brown, P. M., Touchan, R., Anderson, R. S. and Hallett, D. J., 2009. Multi-millennial fire history of the Giant Forest, Sequoia National Park, California, USA, *Fire Ecology,* **5**(3): 120-150.

Takeno, K., 2012. Stress-induced flowering, In *Abiotic Stress Responses in Plants: Metabolism, Productivity and Sustainability* (Eds, Ahmad, P. and Prassad, M. N. V.) Springer, New York, pp. 331-346.

Tansley, A. G., 1935. The use and abuse of vegetational concepts and terms, *Ecology,* **16**(3): 284-307.

Taylor, A. H. and Beaty, R. M., 2005. Climatic influences on fire regimes in the northern Sierra Nevada mountains, Lake Tahoe Basin, Nevada, USA, *Journal of Biogeography,* **32**: 425-438.

Ter Braak, C. J. F., 1986. Canonical correspondence analysis: A new eigenvector technique for multivariate direct gradient analysis, *Ecology,* **67**(5): 1167-1179.

Thackway, R. and Cresswell, I. D. (Eds.), 1995. *An Interim Biogeographic Regionalisation for Australia: A Framework for Setting Priorities in the National Reserve System Cooperative Program. Version 4.0,* Australian Nature Conservation Agency, Canberra.

Thomas, P. B., Morris, E. C. and Auld, T. D., 2007. Response surfaces for the combined effects of heat shock and smoke on germination of 16 species forming soil seed banks in south-east Australia, *Austral Ecology*, **32**(6): 605-616.

Thompson, K., 2000. The functional ecology of soil seed banks, In *Seeds: The Ecology of Regeneration in Plant Communities* (Ed, Fenner, M.) C.A.B. International, Wallingford, pp. 215-236.

Thornton, I., 1996. *Krakatau: The Destruction and Reassembly of an Island Ecosystem,* Harvard University Press, Cambridge.

Tilman, D., 1985. The resource-ratio hypothesis of plant succession, *The American Naturalist,* **125**(6): 827-852.

Trabaud, L. and Lepart, J., 1980. Diversity and stability in garrigue ecosystems after fire, *Vegetatio,* **43**: 49-57.

Turnbull, T. L., Buckley, T. N., Barlow, A. M. and Adams, M. A., 2014. Anatomical and physiological regulation of post-fire carbon and water exchange in canopies of two resprouting *Eucalyptus* species, *Oecologia,* **176**: 333-343.

Turner, M. G., Baker, W. L., Peterson, C. J. and Peet, R. K., 1998. Factors influencing succession: Lessons from large, infrequent natural disturbances, *Ecosystems,* **1**: 511-523.

Turner, M. G. and Dale, V. H., 1998. Comparing large, infrequent disturbances: What have we learned?, *Ecosystems,* **1**: 493-496.

Turner, M. G., Romme, W. H. and Gardner, R. H., 1999. Prefire heterogeneity, fire severity and early postfire plant reestablishment in subalpine forests of Yellowstone National Park, Wyoming, *International Journal of Wildland Fire,* **9**(1): 21-36.

Underwood, A. J., 1991. Beyond BACI: Experimental designs for detecting human environmental impacts on temporal variations in natural populations, *Australian Journal of Marine and Freshwater Research,* **42**: 569-587.

Underwood, A. J., 1992. Beyond BACI: The detection of environmental impacts on populations in the real, but variable, world, *Journal of Experimental Marine Biology and Ecology,* **161**: 145-178.

van Andel, J., 2005. Species interactions structuring plant communities, In *Vegetation Ecology* (Ed, van der Maarel, E.) Blackwell Publishing, Oxford, pp. 238-264.

van der Maarel, E., 1969. On the use of ordination models in phytosociology, *Vegetatio,* **19**: 21-46.

van Hulst, R., 1980. Vegetation dynamics or ecosystem dynamics: Dynamic sufficiency in succession theory, *Vegetatio,* **43**: 147-151.

van Mantgem, P., Schwartz, M. and Keifer, M., 2001. Monitoring fire effects for managed burns and wildfires: Coming to terms with pseudoreplication, *Natural Areas Journal,* **21**(3): 266-273.

van Mantgem, P. J. and Schwilk, D. W., 2009. Negligible influence of spatial autocorrelation in the assessment of fire effects in a mixed conifer forest, *Fire Ecology,* **5**(2): 116-125.

van Wagner, C. E., 1973. Height of crown scorch in forest fires, *Canadian Journal of Forest Research,* **3**: 373-378.

van Wilgen, B. W., Biggs, H. C. and Potgieter, A. L. F., 1998. Fire management and research in the Kruger National Park, with suggestions on the detection of thresholds of potential concern, *Koedoe,* **41**(1): 69-87.

van Wilgen, B. W., Govender, N. and Biggs, H. C., 2007. The contribution of fire research to fire management: A critical review of a long-term experiment in the Kruger National Park, South Africa, *International Journal of Wildland Fire,* **16**: 519-530.

Vankat, J. L., 2013. *Vegetation Dynamics on the Mountains and Plateaus of the American Southwest. Plant and Vegetation Vol. 8,* Springer, New York.

Veblen, T. T. and Ashton, D. H., 1978. Catastrophic influences on the vegetation of the Valdivian Andes, Chile, *Vegetatio,* **36**(3): 149-167.

Vesk, P. A. and Westoby, M., 2004. Sprouting ability across diverse disturbances and vegetation types worldwide, *Journal of Ecology,* **92**: 310-320.

Vines, R. G., 1974. *Weather Patterns and Bush-fire Cycles in Southern Australia. Division of Chemical Technology Technical Paper, No. 2,* CSIRO, Melbourne.

Vivian, L. M., 2010. 'Variation in Fire Response Traits of Plants in Mountainous Plant Communities of South-eastern Australia', Fenner School of Environment and Society, Australian National University, Canberra.

Vivian, L. M., Cary, G. J., Bradstock, R. A. and Gill, A. M., 2008. Influence of fire severity on the regeneration, recruitment and distribution of eucalypts in the Cotter River Catchment, Australian Capital Territory, *Austral Ecology,* **33**(1): 55-67.

Vivian, L. M., Doherty, M. D. and Cary, G. J., 2010. Classifying the fire-response traits of plants: How reliable are species-level classifications?, *Austral Ecology,* **35**(3): 264-273.

Wahren, C.-H., Papst, W. A. and Williams, R. J., 2001. Early post-fire regeneration in subalpine heathland and grassland in the Victorian Alpine National Park, south-eastern Australia *Austral Ecology,* **26**(6): 670-679.

Waide, R. B., Willig, M. R., Steiner, C. F., Mittelbach, G., Gough, L., Dodson, S. I., Juday, G., P. and Parmenter, R., 1999. The relationship between productivity and species richness, *Annual Review of Ecology and Systematics,* **30**: 257-300.

Wakefield, N. A., 1970. Bushfire frequency and vegetational change in south-eastern Australian forests, *The Victorian Naturalist,* **87**: 152-158.

Walker, B., 1992. Biodiversity and ecological redundancy, *Conservation Biology,* **6**(1): 18-23.

Walker, B., 1995. Conserving biological diversity through ecosystem resilience, *Conservation Biology,* **9**(4): 747-752.

Walker, L. R., 2011. Integration of the study of natural and anthropogenic disturbances using severity gradients, *Austral Ecology,* **36**: 916-922.

Walker, L. R. and Willig, M. R., 1999. An introduction to terrestrial disturbances, In *Ecosystems of Disturbed Ground. Ecosystems of the World 16* Vol. 16 (Ed, Walker, L. R.) Elsevier, Amsterdam, pp. 1-16.

Walsh, N. G., Barley, R. H. and Gullan, P. K., 1986. The alpine vegetation of Victoria, excluding the Bogong High Plains region, *Muelleria,* **6**(4): 265-292.

Walsh, N. G. and McDougall, K. L., 2004. Progress in the recovery of the flora of treeless subalpine vegetation in Kosciuszko National Park after the 2003 fires, *Cunninghamia,* **8**(4): 439-452.

Walters, C. J. and Holling, C. S., 1990. Large-scale management experiments and learning by doing, *Ecology,* **71**(6): 2060-2068.

Warcup, J. H., 1980. Effect of heat treatment of forest soil on germination of buried seed, *Australian Journal of Botany,* **28**: 567-571.

Waters, D. A., Burrows, G. E. and Harper, J. D. I., 2010. *Eucalyptus regnans* (Myrtaceae): A fire-sensitive eucalypt with a resprouter epicormic structure, *American Journal of Botany,* **97**(4): 545-556.

Watson, P. and Wardell-Johnson, G. W., 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.

Watson, P. J., Bradstock, R. A. and Morris, E. C., 2009. Fire frequency influences composition and structure of the shrub layer in an Australian subcoastal temperate grassy woodland, *Austral Ecology,* **34**(2): 218-232.

Webb, L. J., 1958. Cyclones as an ecological factor in tropical lowland rainforest, North Queensland, *Australian Journal of Botany,* **6**: 220-228.

Webb, S. L., 1999. Disturbance by wind in temperate-zone forests, In *Ecosystems of Disturbed Ground. Ecosystems of the World 16* Vol. 16 (Ed, Walker, L. R.) Elsevier, Amsterdam, pp. 187-222.

Webb, S. L. and Scanga, S. E., 2001. Windstorm disturbance without patch dynamics: Twelve years of change in a Minnesota forest, *Ecology,* **82**(3): 893-897.

Weedon, J. T., Kowalchuk, G. A., Aerts, R., Freriks, S., Roling, W. F. M. and van Bodegom, P. M., 2017. Compositional stability of the bacterial community in a climate-sensitive sub-Arctic peatland, *Frontiers in Microbiology,* **8**: Article 317.

Werner, P. A. and Franklin, D. C., 2010. Resprouting and mortality of juvenile eucalypts in an Australian savanna: Impacts of fire season and annual sorghum, *Australian Journal of Botany,* **58**: 619-628.

Wesolowski, A., Adams, M. A. and Pfautsch, S., 2014. Insulation capacity of three bark types of temperate *Eucalyptus* species, *Forest Ecology and Management,* **313**: 224-232.

Westman, W. E., 1978. Measuring the inertia and resilience of ecosystems, *BioScience,* **28**(11): 705-710.

Westman, W. E., 1986. Resilience: Concepts and measures, In *Resilience in Mediterranean-type Ecosystems. Tasks for Vegetation Science 16* (Eds, Dell, B., Hopkins, A. J. M. and Lamont, B. B.) Dr. W. Junk, Dordrecht, pp. 5-19.

Westman, W. E. and O'Leary, J. F., 1986. Measures of resilience: The response of coastal sage scrub to fire, *Vegetatio,* **65**: 179-189.

Westoby, M., 1985. Two main relationships among the components of species richness, *Proceedings of the Ecological Society of Australia,* **14**: 103-107.

Whelan, R. J., 1986. Seed dispersal in relation to fire, In *Seed Dispersal* (Ed, Murray, D. R.) Academic Press, Sydney, pp. 237-271.

Whelan, R. J., 1995. *The Ecology of Fire,* Cambridge University Press, Cambridge.

Whelan, R. J. and Muston, R. M., 1991. Fire regimes and management in southeastern Australia, In *High Intensity Fire in Wildlands: Management Challenges and Options. Proceedings of the 17th Tall Timbers Fire Ecology Conference.* (Ed, Hermann, S. M.) Tall Timbers Research Station, Tallahassee, pp. 235-258.

Whelan, R. J., Rodgerson, L., Dickman, C. R. and Sutherland, E. F., 2002. Critical life cycles of plants and animals: Developing a process-based understanding of population changes in fire-prone landscapes, In *Flammable Australia: The Fire Regimes and Biodiversity of a Continent* (Eds, Bradstock, R. A., Williams, J. E. and Gill, A. M.) Cambridge University Press, Cambridge, pp. 94-124.

White, P. S., 1979. Pattern, process and natural disturbance in vegetation, *The Botanical Review,* **45**(3): 229-299.

White, P. S. and Jentsch, A., 2001. The search for generality in studies of disturbance and ecosystem dynamics, *Progress in Botany,* **62**: 399-449.

White, P. S. and Pickett, S. T. A., 1985. Natural disturbance and patch dynamics: An introduction, In *The Ecology of Natural Disturbance and Patch Dynamics* (Eds, Pickett, S. T. A. and White, P. S.) Academic Press, San Diego, pp. 3-13.

Whitford, W. G., Rapport, D. J. and DeSoyza, A. G., 1999. Using resistance and resilience measurements for 'fitness' tests in ecosystem health, *Journal of Environmental Management,* **57**: 21-29.

Whittaker, J. and Mercer, D., 2004. The Victorian bushfires of 2002-03 and the politics of blame: A discourse analysis, *The Australian Geographer,* **35**(3): 259-287.

Whittaker, R. H., 1953. A consideration of climax theory: The climax as a population and pattern, *Ecological Monographs,* **23**(1): 41-78.

Whittaker, R. H., 1956. Vegetation of the Great Smoky Mountains, *Ecological Monographs,* **26**(1): 1-80.

Whittaker, R. H., 1960. Vegetation of the Siskiyou Mountains, Oregon and California, *Ecological Monographs,* **30**(3): 279-338.

Whittaker, R. H., 1965. Dominance and diversity in land plant communities, *Science,* **147**: 250-260.

Whittaker, R. H., 1967. Gradient analysis of vegetation, *Biological Reviews,* **42**: 207-264.

Whittaker, R. H., 1975. *Communities and Ecosystems,* Macmillan Publishing Co., New York.

Wiens, J. A., Hayward, G. D., Safford, H. D. and Giffen, C. M. (Eds.), 2012. *Historical Environmental Variation in Conservation and Natural Resource Management,* Wiley-Blackwell, Chichester.

Williams, J. E., 1990a. The importance of herbivory in the population dynamics of three sub-alpine eucalypts in the Brindabella Range, south-east Australia, *Australian Journal of Ecology,* **15**: 51-55.

Williams, J. E. and Gill, A. M., 1995. *Forest Issues 1. The Impact of Fire Regimes on Native Forests in Eastern New South Wales. Environmental Heritage Monograph Series, No. 2,* NSW National Parks and Wildlife Service, Hurstville.

Williams, J. E., Whelan, R. J. and Gill, A. M., 1994. Fire and environmental heterogeneity in southern temperate forest ecosystems: Implications for management, *Australian Journal of Botany,* **42**: 125-137.

Williams, P. R. and Clarke, P. J., 1997. Habitat segregation by serotinous shrubs in heaths: Post-fire emergence and seedling survival, *Australian Journal of Botany,* **45**: 31-39.

Williams, R. J., 1990b. Cattle grazing within subalpine heathland and grassland communities on the Bogong High Plains: Disturbance, regeneration and the shrub-grass balance, *Proceedings of the Ecological Society of Australia,* **16**: 255-265.

Williams, R. J. and Bradstock, R. A., 2008. Large fires and their ecological consequences: Introduction to the special issue, *International Journal of Wildland Fire,* **17**: 685-687.

Williams, R. J., Bradstock, R. A., Cary, G. J., Enright, N. J., Gill, A. M., Liedloff, A. C., Lucas, C., Whelan, R. J., Andersen, A. N., Bowman, D. M. J. S., Clarke, P. J., Cook, G. D., Hennessy, K. J. and York, A., 2009. *Interactions Between Climate Change, Fire Regimes and Biodiversity in Australia - a Preliminary Assessment. Report to the Department of Climate Change and the Department of the Environment, Water, Heritage and the Arts*, CSIRO / Department of Climate Change / Department of the Environment, Water, Heritage and the Arts, Canberra.

Williams, R. J., Bradstock, R. A., Matthews, S., Price, O. F., Sullivan, A. L. and Watson, P., 2011. *Climate Change, Fire Regimes and Risk in Australian Landscapes: Lessons for Adaptation. Report to the Australian Government Department of Climate Change and Energy Efficiency*, CSIRO Ecosystem Sciences, Darwin.

Williams, R. J., Griffiths, A. D. and Allan, G. E., 2002. Fire regimes and biodiversity in the savannas of northern Australia, In *Flammable Australia: The Fire Regimes and Biodiversity of a Continent* (Eds, Bradstock, R. A., Williams, J. E. and Gill, A. M.) Cambridge University Press, Cambridge, pp. 281-304.

Williams, R. J., Wahren, C.-H., Bradstock, R. A. and Muller, W. J., 2006. Does alpine grazing reduce blazing? A landscape test of a widely-held hypothesis, *Austral Ecology,* **31**(8): 925-936.

Williams, R. J., Wahren, C.-H., Tolsma, A. D., Sanecki, G. M., Papst, W. A., Myers, B. A., McDougall, K. L., Heinze, D. A. and Green, K., 2008. Large fires in Australian alpine landscapes: Their part in the historical fire regime and their impacts on alpine biodiversity, *International Journal of Wildland Fire,* **17**: 793-808.

Willig, M. R. and Walker, L. R., 1999. Disturbance in terrestrial ecosystems: Salient themes, synthesis and future directions, In *Ecosystems of Disturbed Ground. Ecosystems of the World 16* Vol. 16 (Ed, Walker, L. R.) Elsevier, Amsterdam, pp. 747-767.

Wills, T. J. and Read, J., 2002. Effects of smoke on germination of soil-stored seed in a south-eastern Australian sand heathland, *Australian Journal of Botany,* **50**: 197-206.

Wilson, J. B., Gitay, H., Roxburgh, S. H., King, W. M. and Tangney, R. S., 1992. Egler's concept of 'initial floristic composition' in succession - ecologists citing it don't agree what it means, *OIKOS,* **64**(3): 591-593.

Wilson, S. D., 1999. Plant interactions during secondary succession, In *Ecosystems of Disturbed Ground. Ecosystems of the World 16,* Vol. 16 (Ed, Walker, L. R.) Elsevier, Amsterdam, pp. 611-632.

Wimbush, D. J. and Costin, A. B., 1979a. Trends in vegetation at Kosciusko. I. Grazing trials in the subalpine zone, 1957-1971, *Australian Journal of Botany,* **27**(6): 741-787.

Wimbush, D. J. and Costin, A. B., 1979b. Trends in vegetation at Kosciusko. II. Subalpine range transects, 1959-1978, *Australian Journal of Botany,* **27**(6): 789-831.

Wimbush, D. J. and Forrester, R. I., 1988. Effects of rabbit grazing and fire on a subalpine environment. II. Tree vegetation, *Australian Journal of Botany,* **36**: 287-298.

Woods, K. D., 2004. Intermediate disturbance in a late-successional hemlock-northern hardwood forest, *Journal of Ecology,* **92**: 464-476.

Woods, P. V. and Raison, R. J., 1983. Decomposition of litter in sub-alpine forests of *Eucalyptus delegatensis, E. pauciflora* and *E. dives, Australian Journal of Ecology,* **8**: 287-299.

Wouters, M., Tolhurst, K. and Friend, G., 2002. *Analysis of Disturbance by Fire on Public Land in Victoria. Fire Ecology Working Group,* Department of Natural Resources and Environment / Parks Victoria, Melbourne.

Wright, G. T. and Robertson, G., 2014. *Alpine Ash Recovery from the 2003 Landscape Fires, Kosciuszko National Park, NSW*, NSW Office of Environment and Heritage, Queanbeyan.

Zhao, M., A, G., Velicogna, I. and Kimball, J. S., 2017. A global gridded dataset of GRACE drought severity index for 2002-14: Comparison with PDSI and SPEI and a case study of the Australia millennium drought, *Journal of Hydrometeorology,* **18**(8): 2117-2129.

Zylstra, P., 2006. *Fire History of the Australian Alps: Prehistory to 2003*, Australian Alps National Parks Liaison Committee, Canberra.

Appendix 1: Vascular Plant Species Lists for the Brindabella and Burrinjuck Study Areas

Combined listing including species recorded on monitoring sites and incidental observations, supplemented with herbarium records.

Taxa Break Up Treating Reserves as Separate Units:

	Brindabella	**Burrinjuck**
Native Taxa	479	305
Exotic Taxa	97	90
Total Taxa	576	395

83.2% of taxa in Brindabella are native; 16.8% of taxa in Brindabella are exotic.

77.2% of taxa in Burrinjuck are native; 22.8% of taxa in Burrinjuck are exotic.

Taxa Break Up Across Both Reserves Looking At Species in Common and Species Only in Each Reserve:

	Total Both Reserves	**In Common**	**Brindabella Only**	**Burrinjuck Only**
Native Taxa	549	235 (43%)	244 (44%)	70 (13%)
Exotic Taxa	115	72 (62%)	25 (22%)	18 (16%)
Total Taxa	664	307 (46%)	269 (41%)	88 (13%)

82.7% of all taxa across both reserves are native; 17.3% of all taxa across both reserves are exotic.

a) Plant Species List for Brindabella National Park and Bimberi Nature Reserve, NSW

Based on records from Australia's Virtual Herbarium http://avh.chah.org.au/ and collections from 130 long-term vegetation monitoring sites in Brindabella National Park and that part of Bimberi Nature Reserve north of Snowgum Hill.
Species are arranged alphabetically within family, under major taxonomic groupings - lichens, mosses, ferns, gymnosperms, angiosperms (dicotyledons and monocotyledons). Taxonomy corresponds to Rogers and Hafellner (1992) for lichens, Scott (1985) for liverworts, Ramsay (1983) for mosses and PlantNET http://plantnet.rbgsyd.nsw.gov.au for all other taxa.

'*' indicates an exotic species; '?' indicates unconfirmed ID; 'subsp' indicates subspecies and 'var' indicates variety. Where a subspecies or variety is not specified in the list despite the existence of such taxa, it means that the taxon has not been identified to that level.

Lichens

Lecanoraceae
- *Lecanora bicincta* Ram.
- *Lecanora farinacea* Fee
- *Lecanora rupicola* (L.) Zahlbr.

Parmeliaceae
- *Parmelina stevensiana* Elix & Johnston
- *Xanthoparmelia flavescentireagens* (Gyelnik) Gallow.
- *Xanthoparmelia nebulosa* (Kurok. & Filson) Elix & Johnston
- *Xanthoparmelia notata* (Kurok.) Hale
- *Xanthoparmelia rubrireagens* (Gyeln.) Elix & Johnston

Phyllosporaceae
- *Hypocenomyce australis* Timdal

Liverworts

Aneuraceae
- *Riccardia cochleata* (Hook.f. & Tayl.) Kuntze

Cephaloziellaceae

Cephaloziella arctica Bryhn & Douin

Frullaniaceae

Frullania falciloba Lehm.
Frullania pentapleura Tayl.
Frullania probosciphora Tayl.
Frullania rostrata (Hook.f. & Tayl.) Gotts. et al.

Geocalycaceae

Chiloscyphus limosus Carring. & Pears.
Lophocolea semiteres (L. & L.) Mitt.

Lepidoziaceae

Lepidozia obtusifolia Steph.

Metzgeriaceae

Metzgeria decipiens (Mass.) Schiffn. & Gotts.
Metzgeria furcata (L.) Dumort.

Mosses

Amblystegiaceae

Campyliadelphus polygamus (B.S.G.) Kanda
Drepanocladus aduncus (Hedw.) Warnst.

Bartramiaceae

Bartramia papillata Hook.f. & Wils.
Breutelia affinis (Hook.) Mitt.
Breutelia pendula (Sm.) Mitt.
Conostomum curvirostre (Mitt.) Mitt.
Philonotis tenuis (Tayl.) Rchdt.

Brachytheciaceae

Brachythecium paradoxum (Hook.f. & Wils.) Jaeg.
Brachythecium salebrosum (Web. & Mohr) B.S.G.
Rhynchostegium tenuifolium (Hedw.) Jaeg.

Bryaceae

Bryum argenteum Hedw.

- *Bryum billardieri* Schwager. var. *billardieri*
- *Bryum blandum* Hook.f. & Wils. subsp. *blandum*
- *Bryum caespiticium* Hedw.
- *Bryum capillare* Hedw.
- *Bryum pachytheca* C. Muell.
- *Orthodontium lineare* Schwaegr.

Dicranaceae

- *Campylopus catarractilis*
- *Campylopus clavatus* (R. Br.) Hook.f. & Wils.
- *Campylopus introflexus* (Hedw.) Brid.
- *Campylopus pyriformis* (Schultz) Brid.
- *Dicranoloma billardieri* (Brid. ex anon.) Par.
- *Dicranoloma robustum* (H.f. & W.) Par.

Ditrichaceae

- *Ceratodon purpureus* (Hedw.) Brid.
- *Ditrichum difficile* (Dub.) Fleisch.

Fissidentaceae

- *Fissidens asplenioides* Hedw.
- *Fissidens pungens* C. Muell. & Hampe
- *Fissidens taylorii* C. Muell.

Mitteniaceae

- *Mittenia plumula* (Mitt.) Lindb.

Orthotrichaceae

- *Orthotrichum assimile* C. Muell.
- *Orthotrichum tasmanicum* Hook.f. & Wils. var. *tasmanicum*

Plagiotheciaceae

- *Plagiothecium novae-seelandiae* Broth.

Pottiaceae

- *Barbula calycina* Schwaegr.
- *Tortula panduraefolia* (C. Muell. & Hampe) Broth.

Rhizogoniaceae

- *Leptotheca gaudichaudii* Schwaegr.

Sphagnaceae	
Sphagnum sp.	

Ferns

Adiantaceae	
Adiantum aethiopicum L.	**Common Maidenhair**
Cheilanthes austrotenuifolia H.M. Quirk & T.C. Chambers	**Rock Fern**
Cheilanthes sieberi Kunze	
Pellaea falcata (R. Br.) Fee	**Sickle Fern**
Aspleniaceae	
Asplenium flabellifolium Cav.	**Necklace Fern**
Asplenium flaccidum G. Forst. subsp. *flaccidum*	**Weeping Spleenwort**
Asplenium trichomanes L.	**Common Spleenwort**
Blechnaceae	
Blechnum minus (R. Br.) Ettingsh.	**Soft Water Fern**
Blechnum nudum (Labill.) Mett. ex Luers.	**Fishbone Water Fern**
Blechnum patersonii (R. Br.) Mett. subsp. *patersonii*	**Strap Water Fern**
Blechnum wattsii Tindale	**Hard Water Fern**
Doodia caudata (Cav.) R. Br.	**Small Rasp Fern**
Dennstaedtiaceae	
Hypolepis muelleri N.A. Wakef.	**Harsh Ground Fern**
Hypolepis rugulosa (Labill.) J. Sm.	**Ruddy Ground Fern**
Pteridium esculentum (G. Forst.) Cockayne	**Bracken**
Dicksoniaceae	
Dicksonia antarctica Labill.	**Soft Tree-fern**
Dryopteridaceae	
Polystichum proliferum (R. Br.) C. Presl	**Mother Shield Fern**
Pteridaceae	
Anogramma leptophylla (L.) Link	**Annual Fern**

Gymnosperms

Pinaceae	
* *Pinus radiata* D. Don	**Monterey Pine** (Planted at Bull's Head)
* *Pseudotsuga menziesii* (Mirb.) Franco	**Douglas-fir** (Planted at Bull's Head)
Angiosperms	
Dicots	
Amygdalaceae	
* *Prunus armeniaca* L.	**Apricot**
* *Prunus persica* (L.) Batsch	**Peach**
Apiaceae	
* *Conium maculatum* L.	**Hemlock**
Daucus glochidiatus (Labill.) Fischer, C. Meyer & Ave-Lall.	**Native Carrot**
Gingidia harveyana (F. Muell.) Dawson	**Slender Gingidia**
Hydrocotyle algida Wakef.	**Pennywort**
Hydrocotyle laxiflora DC.	**Stinking Pennywort**
Hydrocotyle sibthorpioides Lam.	
Hydrocotyle tripartita R. Br. ex A. Rich.	**Pennywort**
Oreomyrrhis eriopoda (DC.) Hook.f.	**Australian Caraway**
Platysace lanceolata (Labill.) Druce	**Shrubby Platysace**
* *Torilis nodosa* (L.) Gaertn.	**Knotted Hedge-parsley**
Araliaceae	
Astrotricha ledifolia DC.	
Polyscias sambucifolia (Sieber ex DC.) Harms	**Elderberry Panax**
Asteraceae	
* *Achillea millefolium* L.	**Yarrow**
* *Arctotheca calendula* (L.) Levyns	**Capeweed**
Arrhenechthites mixta (A. Rich.) Belcher	**Purple Fireweed**
Bedfordia arborescens Hochr.	**Blanket Leaf**
Brachyscome aculeata (Labill.) Less.	**Hill Daisy**
Brachyscome diversifolia (Hook.) Fisch. & C.A. Mey.	**Large-headed Daisy**

Brachyscome rigidula (DC.) G.L. Davis	**Hairy Cur-leaf Daisy**
Brachyscome scapigera (Sieber ex Spreng.) DC.	**Tufted Daisy**
Brachyscome spathulata Gaudich.	
Calotis scabiosifolia var. *integrifolia* F. Muell. ex Benth.	**Rough Burr-daisy**
* *Carduus pycnocephalus* L.	**Nodding Thistle**
Cassinia aculeata (Labill.) R. Br. subsp. *aculeata*	**Dolly Bush**
Cassinia longifolia R. Br.	
Cassinia quinquefaria R. Br.	
Celmisia longifolia Cass.	**Snow Daisy**
Celmisia tomentella M. Gray & Given	
Centipeda cunninghamii (DC.) A. Braun & Asch.	**Common Sneezeweed**
Centipeda minima (L.) A. Braun & Asch. subsp. *minima*	
Chrysocephalum apiculatum (Labill.) Steetz	**Common Everlasting**
Chrysocephalum semipapposum (Labill.) Steetz	**Clustered Everlasting**
* *Cirsium vulgare* (Savi) Ten.	**Spear Thistle**
* *Conyza albida* Willd. ex Sprengel	**Tall Fleabane**
* *Conyza bonariensis* (L.) Cronquist	**Flaxleaf Fleabane**
Coronidium scorpioides (Labill.) Paul G. Wilson	**Button Everlasting**
Cotula alpina (Hook.f.) Hook.f.	**Alpine Cotula**
Cotula australis (Sieber ex. Sprengel) Hook.f.	**Common Cotula**
Craspedia jamesii J. Everett & Joy Thomps.	
Craspedia paludicola J. Everett & Doust	**Swamp Billy Buttons**
Craspedia variabilis J. Everett & Doust	**Common Billy Buttons**
* *Crepis capillaris* (L.) Wallr.	**Smooth Hawksbeard**
* *Crepis foetida* L. subsp. *vulgaris* (Bisch.) Babc.	**Stinking Hawksbeard**
Cymbonotus lawsonianus Gaudich.	**Bears-ear**
Cymbonotus preissianus Steetz	**Austral Bears-ear**
? *Eclipta* sp.	
Euchiton involucratus (G. Forst.) Holub	**Star Cudweed**
Euchiton japonicus (Thunb.) Holub	**Creeping Cudweed**
Euchiton sphaericus (Willd.) Holub	
* *Hypochoeris glabra* L.	**Smooth Catsear**

* *Hypochoeris radicata* L.	**Catsear**
* *Lactuca serriola* L.	**Prickly Lettuce**
Lagenophora stipitata (Labill.) Druce	**Blue Bottle-daisy**
Leptinella filicula (Hook.f.) Hook.f. ex Benth.	
Leptorhynchos elongatus DC.	**Lanky Buttons**
Leptorhynchos squamatus (Labill.) Less.	**Scaly Buttons**
* *Leucanthemum vulgare* Lam.	**Oxe-eye daisy**
Leucochrysum albicans var. *tricolor* (DC.) Paul G. Wilson	**Hoary Sunray**
Microseris lanceolata (Walp.) Schultz-Bip.	**Murnong / Yam Daisy**
Olearia argophylla (Labill.) F. Muell. ex Benth.	**Native Musk**
Olearia erubescens (Sieber ex DC.) Dippel	**Silky Daisy Bush**
Olearia lirata (Sims) Hutch.	**Snowy Daisy Bush**
Olearia megalophylla (F. Muell.) F. Muell. ex Benth.	**Large-leaf Daisy Bush**
Olearia myrsinoides (Labill.) F. Muell. ex Benth.	**Blush Daisy Bush**
Olearia tenuifolia (DC.) Benth.	**Thin-leaf Daisy Bush**
Ozothamnus stirlingii (F. Muell.) Anderb.	**Ovens Everlasting**
Picris angustifolia subsp. *merxmuelleri* Lack & S. Holzapfel	
* *Picris hieracioides* L.	**Hawkweed Picris**
Podolepis hieracioides F. Muell.	**Long Podolepis**
Podolepis jaceioides (Sims) Voss	**Showy Copper-wire Daisy**
* *Pseudognaphalium luteo-album* (L.) Hilliard & B.L. Burtt	**Jersey Cudweed**
Senecio bathurstianus (DC.) Sch. Bip.	
Senecio bipinnatisectus Belcher	
Senecio diaschides D.G. Drury	
Senecio distalilobatus I. Thomps.	
Senecio gunnii (Hook.f.) Belcher	
Senecio hispidulus A. Rich.	**Hill Fireweed**
Senecio interpositus I. Thomps.	
Senecio linearifolius var. *latifolius* I. Thomps.	
Senecio prenanthoides A. Rich.	
Senecio quadridentatus Labill.	**Cotton Fireweed**
Solenogyne gunnii (Hook.f.) Cabrera	

* *Sonchus asper* subsp. *glaucescens* (Jord.) Ball	**Prickly Sowthistle**
* *Sonchus oleraceus* L.	**Common Sowthistle**
Stuartina hamata Philipson	**Hooked Cudweed**
* *Taraxacum officinale* Weber	**Dandelion**
* *Tolpis barbata* (L.) Gaertn.	**Yellow Hawkweed**
Xerochrysum bracteatum (Vent.) Tzvelev	**Golden Everlasting**
Xerochrysum subundulatum (Sch. Bip.) R.J. Bayer	**Alpine Everlasting**
Xerochrysum viscosum (Sieber ex Spreng.) R.J. Bayer	**Sticky Everlasting**
Boraginaceae	
Cynoglossum australe R. Br.	
Cynoglossum suaveolens R. Br.	
* *Myosotis discolor* Pers.	**Forget-me-not**
* *Myosotis laxa* subsp. *caespitosa* (Schultz) Hyl. ex Nordh.	
Brassicaceae	
? *Barbarea* sp.	
Cardamine gunnii Hewson	
Cardamine paucijuga Turcz.	
Rorippa dictyosperma (Hook.) L.A.S. Johnson	
Rorippa gigantea (Hook.f.) Garn.-Jones	
* *Rorippa nasturtium-aquaticum* (L.) Hayek	**Watercress**
* *Sisymbrium officinale* (L.) Scop.	**Hedge Mustard**
Callitrichaceae	
* *Callitriche stagnalis Scop.*	**Common Starwort**
Campanulaceae	
Wahlenbergia ceracea Lothian	**Waxy Bluebell**
Wahlenbergia communis Carolin	**Tufted Bluebell**
Wahlenbergia gloriosa Lothian	**Royal Bluebell**
Wahlenbergia gracilenta Lothian	**Annual Bluebell**
Wahlenbergia stricta (R. Br.) Sweet subsp. *stricta*	**Tall Bluebell**
Caprifoliaceae	
* *Lonicera japonica* Thunb.	**Japanese Honeysuckle**
Caryophyllaceae	

* *Cerastium glomeratum* Thuill.	**Mouse-ear Chickweed**
* *Moenchia erecta* (L.) P. Gaertn., B. Mey. & Scherb.	**Erect Chickweed**
* *Petrorhagia nanteuilii* (Burnat) P.W. Ball & Heywood	**Proliferous Pink**
* *Polycarpon tetraphyllum* (L.) L.	**Four-leaf Allseed**
* *Sagina apetala* Ard.	**Annual Pearlwort**
Scleranthus biflorus (J.R. Forst. & G. Forst.) Hook.f.	**Two-flowered Knawel**
Scleranthus diander R. Br.	**Tufted Knawel**
* *Silene gallica* L. var. *gallica*	**French Catchfly**
* *Spergularia rubra* (L.) J. Presl & C. Presl	**Sandspurry**
Stellaria flaccida Hook.	
* *Stellaria media* (L.) Vill.	**Common Chickweed**
Stellaria pungens Brongn.	**Prickly Starwort**
Casuarinaceae	
Casuarina cunninghamiana Miq. subsp. *cunninghamiana*	**River Oak**
Chenopodiaceae	
Einadia nutans (R. Br.) subsp. *nutans* A. J. Scott	**Climbing Saltbush**
Clusiaceae	
Hypericum gramineum G. Forst.	**Small St. John's Wort**
Hypericum japonicum Thunb.	
* *Hypericum perforatum* L.	**St. John's Wort**
Convolvulaceae	
Convolvulus angustissimus R. Br.	**Bindweed**
Dichondra repens J.R. Forst. & G. Forst.	**Kidney Weed**
Crassulaceae	
Crassula sieberiana (Schult. & Schult.f.) Druce	**Australian Stonecrop**
Dilleniaceae	
Hibbertia obtusifolia DC.	**Hoary Guinea Flower**
Droseraceae	
Drosera auriculata Backh. ex Planch.	**Sundew**
Drosera peltata Thunb.	**Sundew**
Ericaceae	
Acrothamnus hookeri (Sond.) Quinn	

? *Acrothamnus montanus* (R. Br.) Quinn	
Acrotriche divaricata R. Br.	
Acrotriche serrulata R. Br.	**Honeypots**
Astroloma humifusum (Cav.) R. Br.	**Native Cranberry**
Brachyloma daphnoides (Sm.) Benth.	**Daphne Heath**
Epacris breviflora Stapf	
Epacris microphylla R. Br.	**Coral Heath**
Leucopogon attenuatus A. Cunn.	
Leucopogon fletcheri subsp. *brevisepalus* J.M. Powell	
Leucopogon fraseri A. Cunn.	
Leucopogon gelidus (F. Muell. ex Benth.) Wakef.	
Leucopogon microphyllus var. *pilibundus* (A. Cunn. ex DC.) Benth.	
Leucopogon virgatus (Labill.) R. Br.	
Melichrus procumbens (Cav.) Druce	**Jam Tarts**
Monotoca scoparia (Sm.) R. Br.	
Fabaceae	
Faboideae	
Bossiaea buxifolia A. Cunn.	
Bossiaea foliosa A. Cunn.	**Leafy Bossiaea**
Cullen microcepahalum (Rchb. ex Kunze) J.W. Grimes	**Dusky Scurf-pea**
Daviesia latifolia R. Br.	**Hop Bitter-pea**
Daviesia leptophylla A. Cunn. ex Don	
Daviesia mimosoides subsp. *acris* Crisp	
Daviesia mimosoides R. Br. subsp. *mimosoides*	
Daviesia ulicifolia subsp. *ruscifolia* (A. Cunn. ex Benth.) G. Chandler & Crisp	**Gorse Bitter Pea**
Desmodium varians (Labill.) G. Don.	**Slender Tick-trefoil**
Dillwynia juniperina Lodd.	
Dillwynia phylicoides A. Cunn.	
Dillwynia sericea A. Cunn.	**Showy Parrot-pea**
Glycine clandestina J.C. Wendl.	
Gompholobium huegelii Benth.	**Pale Wedge Pea**
Hardenbergia violacea (Schneev.) Stearn	**False Sarsparilla**

Hovea asperifolia I. Thomps. subsp. *asperifolia*	
Hovea heterophylla A. Cunn. ex Hook.f.	
Hovea purpurea Sweet	
Indigofera australis Willd.	**Australian Indigo**
* *Medicago* sp.	**Medic**
Mirbelia oxyloboides F. Muell.	**Mountain Mirbelia**
Oxylobium ellipticum (Vent.) R. Br.	**Common Shaggy Pea**
Platylobium formosum Sm. subsp. *formosum*	**Handsome Flat Pea**
Podolobium alpestre (F. Muell.) Crisp & P.H. Weston	**Alpine Shaggy Pea**
Pultenaea juniperina Labill.	**Prickly Bush Pea**
Pultenaea polifolia A. Cunn.	**Dusky Bush Pea**
Pultenaea spinosa (DC.) H.B. Will.	**Spiny Bush Pea**
* *Trifolium angustifolium* L.	**Narrow-leaved Clover**
* *Trifolium arvense* L.	**Haresfoot Clover**
* *Trifolium campestre* Schreb.	**Hop Clover**
* *Trifolium dubium* Sibth.	**Yellow Suckling Clover**
* *Trifolium glomeratum L.*	**Clustered Clover**
* *Trifolium repens* L.	**White Clover**
Mimosoideae	
Acacia buxifolia A. Cunn. subsp. *buxifolia*	**Box-leaved Wattle**
Acacia dealbata subsp. *alpina* Tindale & Kodela	**Monaro Silver Wattle**
Acacia falciformis DC.	**Broad-leaved Hickory**
Acacia gunnii Benth.	**Ploughshare Wattle**
Acacia mearnsii De Wild.	**Black Wattle**
Acacia melanoxylon R. Br.	**Blackwood**
Acacia obliquinervia Tindale	**Mountain Hickory**
Acacia pravissima F. Muell.	**Wedge-leaved Wattle**
Acacia rubida A. Cunn.	**Red-leaved Wattle**
Acacia siculiformis A. Cunn. ex Benth.	**Dagger Wattle**
Acacia ulicifolia (Salisb.) Court	**Prickly Moses**
Acacia verniciflua A. Cunn.	**Varnish Wattle**
Gentianaceae	

* *Centaurium erythraea* Rafn	**Common Centaury**
Geraniaceae	
* *Geranium molle* L. subsp. *molle*	**Cranesbill Geranium**
Geranium potentilloides L'Her. & DC. var. *potentilloides*	
Geranium retrorsum L'Her. ex DC	**Common Cranesbill**
Geranium solanderi Carolin var. *solanderi*	**Native Geranium**
Pelargonium australe Willd.	**Native Storksbill**
Goodeniaceae	
Brunonia australis Sm. ex R. Br.	**Blue Pincushion**
Goodenia hederacea Sm. subsp. *hederacea*	**Ivy Goodenia**
Velleia paradoxa R. Br.	**Spur Velleia**
Haloragaceae	
Gonocarpus micranthus Thunb. subsp. *micranthus*	
Gonocarpus tetragynus Labill.	
Haloragis heterophylla Brogn.	**Rough Raspwort**
Myriophyllum simulans Orchard	
Myriophyllum variifolium Hook.f.	
Myriophyllum verrucosum Lindl.	**Red Water-milfoil**
Lamiaceae	
Lycopus australis R. Br.	**Australian Gipsywort**
* *Melissa officinalis* L. subsp. *officinalis*	**Lemon Balm**
Mentha australis R. Br.	**River Mint**
Mentha diemenica Spreng.	**Slender Mint**
* *Mentha pulegium* L.	**Pennyroyal**
Prostanthera lasianthos Labill.	**Victorian Christmas Bush**
Prostanthera rotundifolia R. Br.	**Round-leaved Mint-bush**
Prostanthera ? sp. aff. *cryptandroides*	(Rocky Outcrops near Genges Trig & Waterfall Creek)
Prostanthera ? sp. aff. *rugosa*	(Rocky Outcrops near Genges Trig & Waterfall Creek)
* *Prunella vulgaris* L.	**Self-heal**
Scutellaria humilis R. Br.	**Dwarf Skullcap**
Lauraceae	
Cassytha glabella R. Br.	

Cassytha melantha R. Br.	
Cassytha pubescens R. Br.	
Lentibulariaceae	
Utricularia dichotoma Labill.	**Fairy Aprons**
Linaceae	
Linum marginale A. Cunn. ex Planch.	**Native Flax**
Lobeliaceae	
Isotoma fluviatilis subsp. *australis* McComb	**Swamp Isotome**
Lobelia gibbosa Labill.	**Tall Lobelia**
Pratia pedunculata (R. Br.) Benth.	**Matted Pratia**
Pratia purpurascens (R. Br.) E. Wimm.	**Whiteroot**
Lythraceae	
Lythrum salicaria L.	**Purple Loosestrife**
Malaceae	
* *Malus pumila* Mill.	**Apple**
Malvaceae	
Gynatrix pulchella (Willd.) Alef.	**Hempbush**
* *Modiola caroliniana* (L.) G. Don	**Red-flowered Mallow**
Monimiaceae	
Hedycarya angustifolia A. Cunn.	**Native Mulberry**
Myrtaceae	
Baeckea utilis F. Muell. ex Miq.	**Mountain Baeckea**
Callistemon pallidus (Bonpl.) DC.	**Lemon Bottlebrush**
Callistemon pityoides F. Muell.	**Alpine Bottlebrush**
Callistemon sieberi DC.	**River Bottlebrush**
Calytrix tetragona Labill.	**Common Fringe-myrtle**
Eucalyptus bridgesiana R. Baker	**Apple Box**
Eucalyptus camphora R. Baker subsp. *humeana* L. Johnson & K. Hill	**Mountain Swamp Gum**
Eucalyptus dalrympleana Maiden subsp. *dalrympleana*	**Mountain Gum**
Eucalyptus delegatensis subsp. *delegatensis* R. Baker	**Alpine Ash**
Eucalyptus dives Schauer	**Broad-leaved Peppermint**
Eucalyptus fastigata Deane & Maiden	**Brown Barrel**

Eucalyptus macrorhyncha F. Muell ex Benth.	**Red Stringybark**
Eucalyptus mannifera Mudie subsp. *mannifera*	**Brittle Gum**
Eucalyptus melliodora A. Cunn. ex Schauer	**Yellow Box**
Eucalyptus nortonii (Blakely) L. Johnson	**Large-flowered Bundy**
Eucalyptus pauciflora Maiden & Blakely	**Snow Gum**
Eucalyptus robertsonii Blakely subsp. *robertsonii*	**Robertson’s Peppermint**
Eucalyptus rossii R. Baker & H.G. Smith	**Scribbly Gum**
Eucalyptus rubida subsp. *rubida* Deane & Maiden	**Candlebark**
Eucalyptus stellulata Sieber ex DC.	**Black Sally**
Eucalyptus viminalis Labill.	**Ribbon Gum**
Kunzea ericoides (A. Rich.) J. Thomps.	**Burgan**
Kunzea parvifolia Schauer	**Violet Kunzea**
Leptospermum brevipes F. Muell.	**Slender Tea-tree**
Leptospermum continentale Joy Thomps.	**Prickly Tea-tree**
Leptospermum grandifolium Sm.	**Woolly Tea-tree**
Leptospermum juniperinum Sm.	**Prickly Tea-tree**
Leptospermum micromyrtus Miq.	**Button Tea-tree**
Leptospermum myrtifolium Sieber ex DC.	**Myrtle Tea-tree**
Oleaceae	
* *Ligustrum sinense* Lour.	**Small-leaved Privet**
Onagraceae	
Epilobium billardierianum Ser. subsp. *cinereum* (Rich.) Raven & Engelhorn	
Epilobium gunnianum Hausskn.	
* *Ludwigia palustris* (L.) Elliot	
Orobanchaceae	
Euphrasia collina R. Br. var. *paludosa* (R. Br.) W.R. Barker	**Swamp Eyebright**
* *Orobanche minor* Sm.	**Broomrape**
Oxalidaceae	
* *Oxalis corniculata* L.	**Creeping Oxalis**
Oxalis perennans Haw.	
Oxalis sp.	
Papaveraceae	

* *Papaver dubium* L.	**Longhead Poppy**
Phyllanthaceae	
Poranthera microphylla Brongn.	
Phrymaceae	
* *Erythranthe moschata* (Douglas ex Lindl.) G.L. Nesom	**Musk Monkey-flower**
Picrodendraceae	
Micrantheum hexandrum Hook.f.	
Pittosporaceae	
Billardiera scandens Sm.	**Appleberry**
Bursaria spinosa subsp. *lasiophylla* (E.M. Benn.) L. Cayzer, Crisp & I. Telford	
Bursaria spinosa Cav. subsp. *spinosa*	**Blackthorn**
Cheiranthera linearis A. Cunn. ex Lindl.	**Finger Flower**
Plantaginaceae	
Gratiola peruviana L.	**Australian Brooklime**
* *Linaria pelisseriana* (L.) Mill.	**Pelisser’s Toadflax**
Plantago hispida R. Br.	
* *Plantago lanceolata* L.	**Plantain**
* *Plantago major* L.	**Large Plantain**
Plantago varia R. Br.	
* *Veronica anagallis-aquatica* L.	**Blue Water-Speedwell**
* *Veronica arvensis* L.	**Wall Speedwell**
Veronica calycina R. Br.	**Hairy Speedwell**
Veronica derwentiana Andrews	
Veronica gracilis R. Br.	**Slender Speedwell**
Veronica perfoliata R. Br.	**Digger's Speedwell**
Veronica plebeia R. Br.	**Trailing Speedwell**
Veronica subtilis B.G. Briggs & Ehrend.	
Polygalaceae	
Comesperma retusum Labill.	
Comesperma volubile Labill.	**Love Creeper**
Polygonaceae	
* *Acetosella vulgaris* Fourr.	**Sheep Sorrel**

Persicaria decipiens (R. Br.) K. L. Wilson	**Slender Knotweed**
Persicaria hydropiper (L.) Spach	**Water Pepper**
Persicaria prostrata (R. Br.) Sojak	**Creeping Knotweed**
Polygonum sp.	
Rumex brownii Campd.	**Swamp Dock**
* *Rumex crispus* L.	**Curled Dock**
Portulacaceae	
Neopaxia australasica (Hook.f.) O. Nilsson	
Portulaca sp.	
Primulaceae	
* *Anagallis arvensis* L.	**Scarlet Pimpernel**
Proteaceae	
Banksia marginata Cav.	**Silver Banksia**
Grevillea lanigera A. Cunn. ex R. Br.	**Woolly Grevillea**
Grevillea oxyantha Makinson subsp. *oxyantha*	
Grevillea victoriae F. Muell. subsp. *brindabella* Stajsic	**Baldy Range Grevillea**
Hakea lissosperma R. Br.	**Needle Bush**
Hakea microcarpa R. Br.	**Small-fruited Hakea**
Lomatia myricoides (C.F. Gaertn.) Domin	**River Lomatia**
Persoonia chamaepeuce Lhotsky ex Meisn.	**Dwarf Geebung**
Persoonia rigida R. Br.	
Persoonia subvelutina L.A.S. Johnson	
Ranunculaceae	
Clematis aristata R. Br. ex Ker Gawl.	**Old Man’s Beard**
Clematis leptophylla (F. Muell.) H. Eichler	**Small-leaved Clematis**
Ranunculus amphitrichus Colenso	**Small River Buttercup**
Ranunculus inundatus R. Br. ex DC.	
Ranunculus lappaceus Smith	**Common Buttercup**
* *Ranunculus muricatus* L.	**Sharp Buttercup**
Ranunculus plebeius R. Br. ex DC.	**Forest Buttercup**
Ranunculus pumilio R. Br. ex DC. var. *pumilio*	
Ranunculus scapiger Hook.	

Ranunculus sessiliflorus R. Br. ex DC. var. *sessiliflorus*	**Small-flowered Buttercup**
Resedaceae	
* *Reseda luteola* L.	**Weld**
Rhamnaceae	
Cryptandra amara var. *longifolia* F. Muell. ex Maiden & Betche	**Bitter Cryptandra**
Pomaderris aspera Sieber ex DC.	**Hazel Pomaderris**
? *Pomaderris costata* N.A. Wakef.	
Pomaderris eriocephala N.A. Wakef.	
Pomaderris intermedia Sieber	
Pomaderris phylicifolia subsp. *ericoides* (Maiden & Betche) N.G. Walsh & Coates	**Narrow-leaf Pomaderris**
Pomaderris subcapitata N.A. Wakef.	
Spyridium parvifolium (Hook.) F. Muell.	**Dusty Miller**
Rosaceae	
Acaena echinata Nees	
Acaena novae-zelandiae Kirk	**Bidgee-widgee**
Acaena ovina A. Cunn.	
* *Aphanes arvensis* L.	**Parsley-piert**
Aphanes australiana (Rothm.) Rothm.	
* *Potentilla recta* L.	
* *Rosa rubiginosa* L.	**Sweet Briar**
* *Rubus fruticosus* L. sens. lat.	**Blackberry**
Rubus parvifolius L.	**Native Raspberry**
Rubiaceae	
Asperula conferta Hook.f.	**Common Woodruff**
Asperula gunnii Hook.f.	**Mountain Woodruff**
Asperula scoparia Hook.f.	**Prickly Woodruff**
Coprosma hirtella Labill.	**Coffee-berry**
Coprosma quadrifida (Labill.) Rob.	**Prickly Currant Bush**
* *Galium aparine* L.	**Cleavers**
Galium ciliare Hook.f. subsp. *ciliare*	
Galium gaudichaudii DC. subsp. *gaudichaudii*	**Rough Bedstraw**
Galium polyanthum I. Thomps.	

Galium propinquum A. Cunn.	**Bedstraw**
Leptostigma reptans (F. Muell.) Fosberg	
Nertera granadensis (Mutis) Druce	**Matted Nertera**
Opercularia hispida Spreng.	**Hairy Stinkweed**
* *Sherardia arvensis* L.	**Field Madder**
Rutaceae	
Leionema lamprophyllum subsp. *obovatum* F.M. Anderson	
Phebalium squamulosum subsp. *ozothamnoides* (F. Muell.) Paul G. Wilson	**Alpine Phebalium**
Philotheca myoporoides (DC.) Bayly subsp. *myoporoides*	**Long-leaf Wax Flower**
Salicaceae	
* *Salix alba* L.	**White Willow**
Sambucaceae	
Sambucus australasica (Lindley) Fritsch	**Native Elderberry**
* *Sambucus nigra* L.	**Common Elder**
Santalaceae	
Choretrum pauciflorum DC.	**Dwarf Sour Bush**
Exocarpos cupressiformis Labill.	**Cherry Ballart**
Exocarpos strictus R. Br.	**Dwarf Cherry**
Omphacomeria acerba (R. Br.) A. DC.	
Sapindaceae	
Dodonaea viscosa subsp. *angustissima* (L.f.) J.G. West	**Sticky Hop-bush**
Dodonaea viscosa subsp. *cuneata* (Sm.) J.G. West	**Wedge-leaf Hop-bush**
Dodonaea viscosa subsp. *spatulata* (Sm.) J.G. West	
Scrophulariaceae	
Limosella australis R. Br.	**Australian Mudwort**
* *Verbascum thapsus* L. subsp. *thapsus*	**Aaron's Rod**
* *Verbascum virgatum* Stokes	**Twiggy Mullein**
Solanaceae	
Solanum aviculare G. Forst.	**Kangaroo Apple**
* *Solanum nigrum* L.	**Black-berry Nightshade**
Stackhousiaceae	
Stackhousia monogyna Labill.	**Creamy Candles**

Stackhousia viminea Smith	**Slender Stackhousia**
Sterculiaceae	
Brachychiton populneus (Schott & Endl.) R. Br. subsp. *populneus*	**Kurrajong**
Stylidiaceae	
Stylidium graminifolium Sw. ex Willd.	**Grass Triggerplant**
Thymelaeaceae	
Pimelea curviflora var. *acuta* Threlfall	
Pimelea curviflora var. *sericea* Benth.	
Pimelea glauca R. Br.	**Smooth Rice Flower**
Pimelea ligustrina subsp. *ciliata* Threlfall	
Pimelea linifolia Sm.	**Slender Rice Flower**
Pimelea pauciflora R. Br.	
Pimelea treyvaudii F. Muell. ex Ewart & B. Rees	**Grey Rice Flower**
Tremandraceae	
Tetratheca bauerifolia F. Muell. ex Schuch.	**Heath Pink-bells**
Ulmaceae	
* *Ulmus* sp.	**Elm**
Urticaceae	
Australina pusilla Gaudich	
Urtica incisa Poiret	**Stinging Nettle**
* *Urtica urens* L.	**Small Nettle**
Verbenaceae	
* *Verbena officinalis* L.	**Common Verbena**
Violaceae	
Viola betonicifolia Sm.	**Native Violet**
Viola hederacea Labill.	**Ivy-leaved Violet**
Winteraceae	
Tasmannia xerophila (P. Parm.) M. Gray subsp. *xerophila*	**Alpine Pepperbush**

Monocots

Anthericaceae

Arthropodium milleflorum (DC.) J.F. Macbr.	**Pale Vanilla Lily**
Arthropodium minus R. Br.	
Arthropodium sp. B sensu Harden (1993)	
Thysanotus patersonii R. Br.	**Twining Fringe Lily**
Thysanotus tuberosus R. Br. subsp. *tuberosus*	**Common Fringe-lily**
Asphodelaceae	
Bulbine bulbosa (R. Br.) Haw.	**Native Leek**
Bulbine glauca (Raf.) E.M. Watson	**Rock Lily**
Colchicaceae	
Wurmbea dioica (R. Br.) F. Muell. susbsp. *dioica*	**Early Nancy**
Cyperaceae	
Baumea rubiginosa (Spreng.) Boeck.	
Carex appressa R. Br.	**Tall Sedge**
Carex breviculmis R. Br.	
Carex fascicularis Sol. ex Boott	**Tassel Sedge**
Carex gaudichaudiana Kunth	
Carex inversa R. Br.	**Knob Sedge**
Carex polyantha F. Muell.	
* *Cyperus eragrostis* Lam.	**Umbrella Sedge**
Cyperus trinervis R. Br.	
Eleocharis gracilis R. Br.	
Gahnia subaequiglumis S.T. Blake	**Bog Saw-sedge**
Isolepis fluitans (L.) R. Br.	**Floating Club-rush**
Isolepis gaudichaudiana Kunth	**Benambra Club-sedge**
Isolepis hookeriana Boeck.	
Isolepis inundata R. Br.	
Lepidosperma laterale R. Br.	
Lepidosperma gunnii Boeck.	
Schoenus apogon Roem. & Schult.	**Common Bog-rush**
Schoenus ericetorum R. Br.	**Heath Bog-rush**
Scirpus sp.	
Hypoxidaceae	

Hypoxis hygrometrica Labill. var. *hygrometrica*	**Golden Weather-grass**
Juncaceae	
Juncus falcatus E. Mey.	
Juncus holoschoenus R. Br.	
Juncus phaeanthus L.A.S. Johnson	
Juncus planifolius R. Br.	
Juncus sandwithii Lourteig	
Juncus subsecundus N.A. Wakef.	
Luzula australasica subsp. *dura* (Edgar) M.E. Jansen	
Luzula flaccida (Buchenau) Edgar	
Lomandraceae	
Lomandra filiformis subsp. *coriacea* A.T. Lee	**Wattle Mat-rush**
Lomandra filiformis (Thunb.) Britten subsp. *filiformis*	**Wattle Mat-rush**
Lomandra longifolia Labill.	**Spiny-headed Mat-rush**
Lomandra multiflora (R. Br.) Britten subsp. *multiflora*	**Many-flowered Mat-rush**
Luzuriagaceae	
Drymophila cyanocarpa R. Br.	**Turquoise Berry**
Orchidaceae	
Acianthus sp.	
Arthrochilus huntianus (F. Muell.) Blaxell	**Elbow Orchid**
Caladenia alpina R.S. Rogers	**Alpine Caladenia**
Caladenia capillata D.L. Jones	
Caladenia carnea R. Br.	**Pink Fingers**
Caladenia catenata (Sm.) Druce	**White Caladenia**
Caladenia cucculata Fitz.	**Hooded Caladenia**
Caladenia tentaculata Schltdl.	**Green Comb Spider Orchid**
Caladenia gracilis R. Br.	**Musky Caladenia**
Chiloglottis pluricallata D. L. Jones	
Chiloglottis valida D.L. Jones	**Large Bird Orchid**
Corybas sp.	**Helmet Orchid**
Cyrtostylis reniformis R. Br.	**Gnat Orchid**
Dipodium punctatum (Sm.) R. Br.	**Hyacinth Orchid**

Dipodium roseum D. L. Jones & M.A. Clem.	**Hyacinth Orchid**
Diuris lanceolata Lindl.	**Large Golden Moths**
Diuris maculata Sm.	**Spotted Doubletail**
Diuris sulphurea R. Br.	**Tiger Orchid**
Eriochilus cucculatus (Labill.) Rchb.f.	**Parsons Bands**
Gastrodia sesamoides R. Br.	**Potato Orchid**
Genoplesium nudum (Hook.f.) D.L. Jones & M.A. Clem.	**Tiny Midge Orchid**
Glossodia major R. Br.	**Waxlip Orchid**
Microtis unifolia (G. Forst.) Rchb.f.	**Common Onion Orchid**
Prasophyllum brevilabre (Lindl.) Hook.f.	**Short-lipped Leek Orchid**
Pterostylis coccina Fitzg.	**Scarlet Greenhood**
Pterostylis curta R. Br.	**Blunt Greenhood**
Pterostylis decurva R.S. Rogers	**Summer Greenhood**
Pterostylis fischii Nicholls	**Fisch’s Greenhood**
Pterostylis furcata Lindl.	**Sickle Greenhood**
Pterostylis longifolia R. Br.	**Tall Greenhood**
Pterostylis nutans R. Br.	**Nodding Greenhood**
Pterostylis pedunculata R. Br.	**Maroonhood**
Spiranthes australis (R. Br.) Lindl.	**Ladies’ Tresses**
Thelymitra megcalyptra Fitz.	**Scented Sun Orchid**
Thelymitra nuda R. Br.	**Plain Sun Orchid**
Thelymitra pauciflora R. Br.	**Slender Sun Orchid**
Thelymitra simulata D.L. Jones & M.A. Clem.	**Collared Sun Orchid**
Philydraceae	
Philydrum lanuginosum Banks & Sol. ex Gaertn.	**Woolly Waterlily**
Phormiaceae	
Dianella revoluta R. Br. var. *revoluta*	
Dianella tasmanica Hook.f.	**Tasman Flax-lily**
Stypandra glauca R. Br.	**Nodding Blue Lily**
Poaceae	
…*Agrostis bettyae* S.W.L. Jacobs	
* *Agrostis gigantea* Roth	**Redtop Bent**

* *Aira caryophyllea* L.	**Silvery Hairgrass**
* *Aira cupaniana* Guss.	**Silvery Hairgrass**
* *Aira elegantissima* Schur	**Delicate Hairgrass**
* *Anthoxanthum odoratum* L.	**Sweet Vernal Grass**
Austrostipa ramosissima (Trin.) S.W.L. Jacobs & J. Everett	**Stout Bamboo Grass**
Austrostipa rudis subsp. *nervosa* (Vickery) S.W.L. Jacobs & J. Everett	
Austrostipa rudis (Spreng.) S.W.L. Jacobs & J. Everett subsp. *rudis*	
Austrostipa scabra subsp. *falcata*	**Speargrass**
* *Briza minor* L.	**Shivery Grass**
* *Bromus brevis* Steud.	
* *Bromus catharticus* Vahl	**Prairie Grass**
* *Bromus diandrus* Roth	**Great Brome**
* *Bromus hordeaceus* L.	**Soft Brome**
* *Bromus rubens* L.	**Red Brome**
Deyeuxia accedens Vickery	
Deyeuxia brachyathera (Stapf) Vickery	
Deyeuxia contracta (F. Muell. ex Hook.f.) Vickery	
Deyeuxia gunniana (Nees) Benth.	
Deyeuxia monticola (Roem. & Schult.) Vickery var. *monticola*	
Deyeuxia nudiflora Vickery	**Climbing Bent Grass**
Deyeuxia parviseta Vickery var. *parviseta*	
Deyeuxia quadriseta (Labill.) Benth.	
Deyeuxia rodwayi Vickery	
Dichelachne crinita (L.) Hook.f.	**Longhair Plumegrass**
Dichelachne hirtella N.G. Walsh	**Plumegrass**
Dichelachne inaequiglumis (Hack. ex Cheesman) Edgar & Connor	
Dichelachne micrantha (Cav.) Domin	**Shorthair Plumegrass**
Dichelachne parva B.K. Simon	
Dichelachne rara (R. Br.) Vickery	
Dichelachne sieberiana Trin. & Rupr.	
Echinopogon ovatus(G. Forst.) P. Beauv.	**Forest Hedgehog Grass**
Elymus scaber (R. Br.) A. Love	**Common Wheatgrass**

Hemarthria uncinata R. Br. var. *uncinata*	**Matgrass**
Hierochloe redolens (J. Vahl) Roem. & Schult.	**Sweet Holygrass**
* *Holcus lanatus* L.	**Yorkshire Fog**
Hookerochloa eriopoda (Vickery) S.W.L. Jacobs	**Snow Fescue**
Hookerochloa hookeriana (Hook.f.) E.B. Alexeev	**Hooker's Fescue**
* *Hordeum leporinum* Link	**Barley Grass**
Lachnagrostis aemula (R. Br.) Trin.	**Blowngrass**
Lachnagrostis aequata (Nees) S.W.L. Jacobs	
Lachnagrostis filiformis (G. Forst.) Trin.	
Microlaena stipoides (Labill.) R. Br. var. *stipoides*	**Weeping Grass**
Panicum sp.	
Phragmites australis (Cav.) Trin. ex Steud.	**Common Reed**
* *Poa annua* L.	**Winter Grass**
Poa clivicola Vickery	**Fine-leaved Snow Grass**
Poa helmsii Vickery	**Broad-leaved Snowgrass**
Poa hookeri Vickery	
Poa induta Vickery	
Poa labillardierei var. *labillardierei* Steud.	**Tussock**
Poa phillipsiana Vickery	
Poa saxicola R. Br.	**Rock Poa**
Poa sieberiana var *cyanophylla* Vickery	**Snowgrass**
Poa sieberiana var *hirtella* Vickery	**Snowgrass**
Poa sieberiana Spreng. var *sieberiana*	**Snowgrass**
Poa tenera F. Muell. ex Hook.f.	**Slender Tussock-grass**
Rytidosperma auriculatum (J.M. Black) Connor & Edgar	L**obed Wallaby Grass**
Rytidosperma erianthum (Lindl.) Connor & Edgar	
Rytidosperma pallidum (R. Br.) A.M. Humphreys & H.P. Linder	**Redanther Wallaby Grass**
Rytidosperma pilosum (R. Br.) Connor & Edgar	**Smooth-flowered Wallaby Grass**
Rytidosperma racemosum (R. Br.) Connor & Edgar var. *racemosum*	
Themeda triandra Forssk.	**Kangaroo Grass**
* *Triticum aestivum* L.	**Wheat**
* *Vulpia bromoides* (L.) Gray	**Squirrel Tail Fescue**

* *Vulpia muralis* (Kunth) Nees	
* *Vulpia myuros* (L.) C.C. Gmel.	**Rat’s Tail Fescue**
Restionaceae	
Baloskion australe (R. Br.) B.G. Briggs & L.A.S. Johnson	
Empodisma minus (Hook.f.) L.A.S. Johnson & D.F. Cutler	**Spreading Rope-rush**

b) Plant Species List for Burrinjuck Nature Reserve, NSW

Based on records from Australia’s Virtual Herbarium http://avh.chah.org.au/ and collections from 33 long-term vegetation monitoring sites in Burrinjuck Nature Reserve. Species are arranged alphabetically within family, under major taxonomic groupings - ferns, gymnosperms, angiosperms (dicotyledons and monocotyledons). Taxonomy corresponds to PlantNET http://plantnet.rbgsyd.nsw.gov.au.

‘*’ indicates an exotic species; ‘?’ indicates unconfirmed ID; ‘subsp’ indicates subspecies and ‘var’ indicates variety. Where a subspecies or variety is not specified in the list despite the existence of such taxa, it means that the taxon has not been identified to that level.

Ferns

Adiantaceae	
Adiantum aethiopicum L.	**Common Maidenhair**
Cheilanthes austrotenuifolia H.M. Quirk & T.C. Chambers	**Rock Fern**
Cheilanthes sieberi Kunze	
Pellaea falcata (R. Br.) Fee	**Sickle Fern**
Aspleniaceae	
Asplenium flabellifolium Cav.	**Necklace Fern**
Pleurosorus rutifolius (R. Br.) Fee	**Bristly Cloak Fern**
Dennstaedtiaceae	
Pteridium esculentum (G. Forst.) Cockayne	**Bracken**
Ophioglossaceae	
Ophioglossum sp.	**Adders Tongue**
Pteridaceae	
Pteris tremula R. Br.	**Tender Brake**

Gymnosperms

Pinaceae	
* *Pinus radiata* D. Don	**Monterey Pine**

Angiosperms

Dicots

Amaranthaceae	
Alternanthera denticulata R. Br.	**Lesser Joyweed**
Apiaceae	
Daucus glochidiatus (Labill.) Fisch., C.A. Mey. & Ave-Lall.	**Native Carrot**
Hydrocotyle laxiflora DC.	**Stinking Pennywort**
Hydrocotyle sibthorpioides Lam.	
Hydrocotyle tripartita R. Br. ex A. Rich.	**Pennywort**
Oreomyrrhis eriopoda (DC.) Hook.f.	**Australian Caraway**
Araliaceae	
Astrotricha ledifolia DC.	
Asteraceae	
Ammobium craspedioides Benth.	
* *Arctotheca calendula* (L.) Levyns	**Capeweed**
Brachyscome diversifolia (Hook.) Fisch. & C.A. Mey.	**Large-headed Daisy**
Brachyscome graminea (Labill.) F. Muell.	**Grass Daisy**
Brachyscome spathulata Gaudich.	
Calotis scabiosifolia var. *integrifolia* F. Muell. ex Benth.	**Rough Burr-daisy**
* *Carduus nutans* L subsp. *nutans*	**Nodding Thistle**
* *Carduus pycnocephalus* L.	**Slender Thistle**
Cassinia aculeata (Labill.) R. Br. subsp. *aculeata*	**Dolly Bush**
Cassinia laevis R. Br.	**Cough Bush**
Cassinia longifolia R. Br.	
Cassinia uncata A. Cunn. ex DC	**Sticky Cassinia**
Centipeda cunninghamii (DC.) A. Braun & Asch.	**Common Sneezeweed**
Centipeda minima (L.) A. Braun & Asch. subsp. *minima*	
Chrysocephalum apiculatum (Labill.) Steetz	**Common Everlasting**
Chrysocephalum semipapposum (Labill.) Steetz	**Clustered Everlasting**
* *Chondrilla juncea* L.	**Skeleton Weed**
* *Cirsium vulgare* (Savi) Ten.	**Spear Thistle**
* *Conyza bonariensis* (L.) Cronquist	**Flaxleaf Fleabane**

Coronidium scorpioides (Labill.) Paul G. Wilson	**Button Everlasting**
Cotula australis (Sieber ex Spreng.) Hook.f.	**Common Cotula**
Craspedia paludicola J. Everett & Doust	**Swamp Billy Buttons**
* *Crepis foetida* L. subsp. *foetida*	**Stinking Hawksbeard**
Cymbonotus lawsonianus Gaudich.	**Bears-ear**
Cymbonotus preissianus Steetz	**Austral Bears Ear**
Epaltes australis Less.	**Spreading Nut-heads**
Euchiton japonicus (Thunb.) Holub	**Creeping Cudweed**
* *Hypochaeris glabra* L.	**Smooth Catsear**
* *Hypochaeris radicata* L.	**Catsear**
Isoetopsis graminifolia Turcz.	**Grass Cushions**
* *Lactuca serriola* L.	**Prickly Lettuce**
Lagenophora stipitata (Labill.) Druce	**Blue Bottle-daisy**
* *Leucanthemum vulgare* Lam.	**Oxeye Daisy**
Microseris lanceolata (Walp.) Sch. Bip.	**Murnong / Yam Daisy**
Olearia erubescens (Sieber ex DC.) Dippel	**Pink-tip Daisy Bush**
Pseudognaphalium luteoalbum (L.) Hilliard & B.L. Burtt	**Jersey Cudweed**
Senecio bathurstianus (DC.) Sch. Bip.	
Senecio diaschides D.G. Drury	
Senecio distalilobatus I. Thomps.	
? *Senecio garlandii* F. Muell. Ex Belcher	**Woolly Ragwort**
Senecio hispidulus A. Rich.	**Hill Fireweed**
Senecio prenanthoides A. Rich.	
Senecio quadridentatus Labill.	**Cotton Fireweed**
Solenogyne sp.	
* *Silybum marianum* (L.) Gaertn.	**Variegated Thistle**
* *Sonchus asper* subsp. *glaucescens* (Jord.) Ball	**Prickly Sowthistle**
* *Sonchus oleraceus* L.	**Common Sowthistle**
Stuartina hamata Philipson	**Hooked Cudweed**
* *Taraxacum officinale* Weber	**Dandelion**
* *Tolpis barbata* (L.) Gaertn.	**Yellow Hawkweed**
Triptilodiscus pygmaeus Turcz.	**Common Sunray**

Xerochrysum bracteatum (Vent.) Tzvelev	**Golden Everlasting**
Xerochrysum viscosum (Sieber ex Spreng.) R.J. Bayer	**Sticky Everlasting**
Boraginaceae	
Cynoglossum australe R. Br.	
Cynoglossum suaveolens R. Br.	
* *Echium plantagineum* L.	**Paterson's Curse**
* *Echium vulgare* L.	**Vipers Bugloss**
* *Myosotis discolor* Pers.	**Forget-me-not**
* *Myosotis laxa* subsp. *caespitosa* (Schultz) Hyl. Ex Nordh.	
Brassicaceae	
* *Capsella bursa-pastoris* (L.) Medik.	**Shepherd's Purse**
Cardamine paucijuga Turcz.	
Rorippa sp.	
* *Sisymbrium officinale* (L.) Scop.	**Hedge Mustard**
* *Sisymbrium orientale* L.	**Indian Hedge Mustard**
Callitrichaceae	
* *Callitriche stagnalis* Scop.	**Common Starwort**
Campanulaceae	
Wahlenbergia communis Carolin	**Tufted Bluebell**
Wahlenbergia gracilenta Lothian	**Annual Bluebell**
Wahlenbergia stricta R. Br. subsp. *stricta*	**Tall Bluebell**
Caryophyllaceae	
* *Cerastium glomeratum* Thuill.	**Mouse-ear Chickweed**
* *Moenchia erecta* (L.) P. Gaertn., B. Mey. & Scherb.	**Erect Chickweed**
* *Petrorhagia nanteuilii* (Burnat) P.W. Ball & Heywood	**Proliferous Pink**
* *Polycarpon tetraphyllum* (L.) L.	**Four-leaf Allseed**
* *Sagina apetala* Ard.	**Annual Pearlwort**
* *Silene gallica* L.	**French Catchfly**
* *Spergularia rubra* (L.) J. Presl & C. Presl	**Sandspurry**
Stellaria flaccida Hook.	
* *Stellaria media* (L.) Vill.	**Common Chickweed**
Stellaria pungens Brongn.	**Prickly Starwort**

Casuarinaceae	
Allocasuarina verticillata (Lam.) L.A.S. Johnson	**Drooping Sheoak**
Casuarina cunninghamiana Miq. subsp. *cunninghamiana*	**River Oak**
Clusiaceae	
Hypericum gramineum G. Forst.	**Small St. John’s Wort**
* *Hypericum perforatum* L.	**St. John’s Wort**
Convolvulaceae	
Convolvulus erubescens Sims	**Blushing Bindweed**
Dichondra repens J.R. Forst. & G. Forst.	**Kidney Weed**
Crassulaceae	
Crassula sieberiana (Schult. & Schult.f.) Druce	**Australian Stonecrop**
Dilleniaceae	
Hibbertia incana (Lindl.) Toelken	
Hibbertia obtusifolia DC.	**Hoary Guinea Flower**
Droseraceae	
Drosera auriculata Backh. ex Planchon	
Drosera peltata Thunb.	
Ericaceae	
Acrotriche serrulata R. Br.	**Honeypots**
Astroloma humifusum (Cav.) R. Br.	**Native Cranberry**
Brachyloma daphnoides (Sm.) Benth.	**Daphne Heath**
? *Leucopogon ericoides* (Smith) R. Br.	**Pink Beard Heath**
Leucopogon fletcheri subsp. *brevisepalus* J.M. Powell	
Melichrus urceolatus R. Br.	**Urn Heath**
Monotoca scoparia (Sm.) R. Br.	
Euphorbiaceae	
* *Euphorbia peplus* L.	**Petty Spurge**
Fabaceae	
Faboideae	
Bossiaea buxifolia A. Cunn.	
Bossiaea foliosa A. Cunn.	**Leafy Bossiaea**
Cullen microcepahalum (Rchb. ex Kunze) J.W. Grimes	**Dusky Scurf-pea**

Daviesia leptophylla A. Cunn. ex Don	
Daviesia mimosoides R. Br. subsp. *mimosoides*	
Desmodium varians (Labill.) G. Don	**Slender Tick-trefoil**
Dillwynia juniperina Lodd.	
Dillwynia phylicoides A. Cunn.	
Dillwynia sericea A. Cunn.	**Showy Parrot Pea**
Glycine clandestina J.C. Wendl.	
Gompholobium huegelii Benth.	**Pale Wedge Pea**
Hardenbergia violacea (Schneev.) Stearn	**False Sarsparilla**
Hovea heterophylla A. Cunn. ex Hook.f.	
Indigofera adesmiifolia A. Gray	
Indigofera australis Willd.	**Australian Indigo**
* *Medicago* sp.	
Platylobium formosum Smith subsp. *formosum*	
Pultenaea juniperina Labill.	**Prickly Bush Pea**
Pultenaea polifolia A. Cunn.	**Dusky Bush Pea**
Pultenaea procumbens A. Cunn.	**Heathy Bush Pea**
Pultenaea spinosa (DC.) H.B. Will.	**Spiny Bush Pea**
* *Trifolium arvense* L.	**Haresfoot Clover**
* *Trifolium campestre* Schreb.	**Hop Clover**
* *Trifolium dubium* Sibth.	**Yellow Suckling Clover**
* *Trifolium glomeratum* L.	**Clustered Clover**
* *Trifolium repens* L.	**White Clover**
Mimosoideae	
Acacia buxifolia A. Cunn. subsp. *buxifolia*	**Box-leaved Wattle**
Acacia dealbata subsp. *alpina* Tindale & Kodela	**Monaro Silver Wattle**
Acacia falciformis DC.	**Broad-leaved Hickory**
Acacia gunnii Benth.	**Ploughshare Wattle**
Acacia implexa Benth.	**Hickory Wattle**
Acacia mearnsii De Wild.	**Black Wattle**
Acacia melanoxylon R. Br.	**Blackwood**
Acacia penninervis Sieber ex DC.	**Mountain Hickory**

Acacia rubida A. Cunn.	**Red-leaved Wattle**
Acacia ulicifolia (Salisb.) Court	**Prickly Moses**
Acacia verniciflua A. Cunn.	**Varnish Wattle**
Fumariaceae	
* *Fumaria* sp.	**Fumitory**
Gentianaceae	
* *Centaurium erythraea* Rafn	**Common Centaury**
Geraniaceae	
* *Erodium moschatum* (L.) L’Her.	**Musky Crowfoot**
* *Geranium molle* L. subsp. *molle*	**Cranesbill Geranium**
Geranium potentilloides L'Her. & DC. var *potentilloides*	
Geranium retrorsum L’Her. ex DC.	**Common Cranesbill**
Geranium solanderi Carolin var. *solanderi*	**Native Geranium**
Pelargonium australe Willd.	**Native Storksbill**
Goodeniaceae	
Brunonia australis Smith ex R. Br.	**Blue Pincushion**
Dampiera purpurea R. Br.	
Haloragaceae	
Gonocarpus elatus (A. Cunn. ex. Fenzl) Orchard	
Gonocarpus tetragynus Labill.	
Haloragis heterophylla Brongn.	**Rough Raspwort**
Haloragis serra Brongn.	
Lamiaceae	
Lycopus australis R. Br.	**Australian Gipsywort**
Mentha diemenica Spreng.	**Slender Mint**
* *Prunella vulgaris* L.	**Self-heal**
Scutellaria humilis R. Br.	**Dwarf Skullcap**
Westringia eremicola A. Cunn. ex Benth.	**Slender Westringia**
Lauraceae	
Cassytha pubescens R. Br.	
Lobeliaceae	
Pratia sp. A. sensu Harden (1992)	

Loranthaceae	
Amyema pendulua (Sieber ex Spreng.) Tieg. subsp. *pendula*	
Malvaceae	
Gynatrix pulchella (Willd.) Alef.	**Hempbush**
* *Modiola caroliniana* (L.) G. Don	**Red-flowered Mallow**
Brachychiton populneus (Schott & Endl.) R. Br. subsp. *populneus*	**Kurrajong**
Myrsinaceae	
* *Anagallis arvensis* L.	**Scarlet Pimpernel**
Myrtaceae	
Baeckea utilis F. Muell. ex Miq.	**Mountain Baeckea**
Callistemon sieberi DC.	**River Bottlebrush**
Calytrix tetragona Labill.	**Common Fringe-myrtle**
Eucalyptus blakelyi Maiden	**Blakely's Red Gum**
Eucalyptus bicostata Maiden, Blakely & J.H. Simmonds	**Eurabbie**
Eucalyptus bridgesiana R.T. Baker	**Apple Box**
Eucalyptus dives Schauer	**Broad-leaved Peppermint**
Eucalyptus goniocalyx F. Muell. ex Miq.	**Bundy**
Eucalyptus macrorhyncha F. Muell ex Benth.	**Red Stringybark**
Eucalyptus mannifera Mudie subsp. *mannifera*	**Brittle Gum**
Eucalyptus melliodora A. Cunn. ex Schauer	**Yellow Box**
Eucalyptus polyanthemos Schauer subsp. *polyanthemos*	**Red Box**
Eucalyptus robertsonii Blakely subsp. *robertsonii*	**Robertson’s Peppermint**
Eucalyptus rossii R.T. Baker & H.G. Sm.	**Inland Scribbly Gum**
Eucalyptus rubida H. Deane & Maiden subsp. *rubida*	**Candlebark**
Eucalyptus viminalis Labill.	**Ribbon Gum**
Kunzea ericoides (A. Rich.) Joy Thomps.	**Burgan**
Leptospermum continentale Joy Thomps.	**Prickly Tea-tree**
Leptospermum juniperinum Sm.	**Prickly Tea-tree**
Leptospermum multicaule A. Cunn.	**Silver Tea-tree**
Leptospermum myrtifolium Sieber ex DC.	**Myrtle Tea-tree**
Onagraceae	
Epilobium billardierianum subsp. *cinereum* (A. Rich.) P.H. Raven & Engelhorn	

Orobanchaceae	
* *Orobanche minor* Sm.	**Broomrape**
* *Parentucellia latifolia* (L.) Caruel	**Red Bartsia**
Oxalidaceae	
* *Oxalis corniculata* L.	
Oxalis perennans Haw.	
Oxalis sp.	
Phyllanthaceae	
Poranthera microphylla Brongn.	
Picrodendraceae	
Micrantheum hexandrum Hook.f.	
Pittosporaceae	
Billardiera scandens Sm.	**Hairy Appleberry**
Bursaria spinosa Cav. subsp. *spinosa*	**Blackthorn**
Cheiranthera linearis A. Cunn. ex Lindl.	**Finger Flower**
Rhytidosporum procumbens (Hook.) F. Muell.	
Plantaginaceae	
Gratiola peruviana L.	**Australian Brooklime**
* *Linaria pelisseriana* (L.) Mill.	**Pelisser’s Toadflax**
Plantago debilis R. Br.	
Plantago gaudichaudii Barneoud	**Narrow Plantain**
Plantago hispida R. Br.	
* *Plantago lanceolata* L.	**Plantain**
* *Plantago major* L.	**Large Plantain**
Plantago varia R. Br.	
* *Veronica arvensis* L.	**Wall Speedwell**
Veronica calycina R. Br.	**Hairy Speedwell**
Veronica plebeia R. Br.	**Trailing Speedwell**
Polygonaceae	
* *Acetosella vulgaris* Fourr.	**Sheep Sorrel**
Persicaria prostrata R. Br.	**Creeping Knotweed**
Polygonum sp.	

Rumex brownii Campd.	**Swamp Dock**
Portulacaceae	
Calandrinia eremaea Ewart	
Montia fontana L.	
Proteaceae	
Grevillea polybractea H.B. Will.	**Crimson Grevillea**
Hakea decurrens R.Br subsp. *decurrens*	
Lomatia myricoides (C.F. Gaertn.) Domin	**River Lomatia**
Persoonia rigida R. Br.	
Ranunculaceae	
Clematis aristata R. Br. ex Ker Gawl.	**Old Man’s Beard**
Clematis leptophylla (F. Muell.) H. Eichler	**Small-leaved Clematis**
Ranunculus lappaceus Sm.	**Common Buttercup**
* *Ranunculus muricatus* L.	**Sharp Buttercup**
Ranunculus pumilio R. Br. ex DC. var. *pumilio*	
Ranunculus sessiliflorus R. Br. ex DC. var. *sessiliflorus*	
Ranunculus plebeius R. Br. ex DC.	**Forest Buttercup**
Rhamnaceae	
Pomaderris ? *andromedifolia* A. Cunn.	
Pomaderris betulina subsp. *actensis* N.G. Walsh & Coates	
Pomaderris eriocephala N.A Wakef.	
Pomaderris subcapitata N.A. Wakef.	
Spyridium parvifolium (Hook.) F. Muell.	**Dusty Miller**
Rosaceae	
Acaena echinata Nees	
Acaena novae-zelandiae Kirk	**Bidgee-widgee**
Acaena ovina A. Cunn.	
* *Aphanes arvensis* L.	**Parsley-piert**
Aphanes australiana (Rothm.) Rothm.	
* *Rosa rubiginosa* L.	**Sweet Briar**
* *Rubus fruticosus* L. *sens. lat*.	**Blackberry**
Rubus parvifolius L.	**Native Raspberry**

Rubiaceae	
Asperula conferta Hook.f.	**Common Woodruff**
Asperula pusilla Hook.f	**Alpine Woodruff**
Asperula scoparia Hook.f.	**Prickly Woodruff**
Coprosma quadrifida (Labill.) Rob.	**Prickly Currant Bush**
* *Galium aparine* L.	**Goosegrass**
Galium gaudichaudii DC.	**Rough Bedstraw**
* *Galium murale* (L.) All.	**Small Bedstraw**
Galium propinquum A. Cunn.	**Bedstraw**
Opercularia diphylla Gaertn.	
Opercularia hispida Spreng.	**Hairy Stinkweed**
Opercularia varia Hook.f.	**Variable Stinkweed**
Pomax umbellata (Gaertn.) Sol. ex A. Rich	
* *Sherardia arvensis* L.	**Field Madder**
Rutaceae	
Correa reflexa (Labill.) Vent var. *reflexa*	**Common Correa**
Salicaceae	
* *Salix fragilis* L.	**Crack Willow**
Santalaceae	
Exocarpos cupressiformis Labill.	**Cherry Ballart**
Sapindaceae	
Dodonaea boroniifolia G. Don	**Fern-leaf Hop-bush**
Dodonaea viscosa subsp. *angustissima* (DC.) J.G. West	**Narrow-leaved Hopbush**
Scrophulariaceae	
* *Verbascum thapsus* L. subsp. *thapsus*	**Aaron's Rod**
* *Verbascum virgatum* Stokes	**Twiggy Mullein**
Solanaceae	
Solanum linearifolium Geras. ex Symon	**Mountain Kangaroo Apple**
* *Solanum nigrum* L.	**Black-berry Nightshade**
Stackhousiaceae	
Stackhousia monogyna Labill.	**Creamy Candles**
Stylidiaceae	

Stylidium graminifolium Sw. ex Willd.	**Grass Triggerplant**
Thymelaeaceae	
Pimelea curviflora var. *sericea* Benth.	
Pimelea linifolia Sm.	**Slender Rice Flower**
Pimelea treyvaudii F. Muell. ex Ewart & B. Rees	**Grey Rice-flower**
Tremandraceae	
Tetratheca bauerifolia F. Muell. ex Schuch.	**Heath Pink-bells**
Urticaceae	
Urtica incisa Poir.	**Stinging Nettle**
Verbenaceae	
* *Verbena bonariensis* L.	**Purpletop**
* *Verbena officinalis* L.	**Common Verbena**
Violaceae	
Viola betonicifolia Sm.	**Native Violet**
Viola hederacea Labill.	**Ivy-leaved Violet**

Monocots

Anthericaceae	
Arthropodium milleflorum (DC.) J.F. Macbr.	**Pale Vanilla Lily**
Arthropodium minus R. Br.	
Dichopogon strictus (R. Br.) Baker	**Chocolate Lily**
Thysanotus patersonii R. Br.	**Twining Fringe-lily**
Thysanotus tuberosus R. Br. subsp. *tuberosus*	**Common Fringe-lily**
Tricoryne elatior R. Br.	**Yellow Autumn-lily**
Asphodelaceae	
Bulbine bulbosa (R. Br.) Haw.	**Native Leek**
Bulbine glauca (Raf.) E.M. Watson	**Rock Lily**
Colchicaceae	
Burchardia umbellata R. Br.	**Milkmaids**
Wurmbea biglandulosa (R. Br.) T.D. Macfarl. subsp. *biglandulosa*	
Wurmbea dioica (R. Br.) F. Muell. subsp. *dioica*	**Early Nancy**

Cyperaceae	
Bolboschoenus fluviatilis (Torr.) Sojak	**Marsh Club-rush**
Carex appressa R. Br.	**Tall Sedge**
Carex inversa R. Br.	
Isoloepis hookeriana Boeckeler	
Lepidosperma gunnii Boeckeler	
Lepidosperma laterale R. Br.	
Schoenus apogon Roem. & Schult.	
Schoenus ericetorum R. Br.	**Heath Bog-rush**
Scirpus sp.	
Hypoxidaceae	
Hypoxis hygrometrica Labill. var. *hygrometrica*	**Golden Weather-grass**
Juncaceae	
Juncus bufonius L.	**Toad Rush**
Juncus planifolius R. Br.	
Juncus subsecundus N.A. Wakef.	
Juncus usitatus L.A.S. Johnson	
Luzula flaccida (Buchenau) Edgar	
Lomandraceae	
Lomandra filiformis subsp. *coriacea* A.T. Lee	**Wattle Mat-rush**
Lomandra filiformis (Thunb.) Britten subsp. *filiformis*	**Wattle Mat-rush**
Lomandra longifolia Labill.	**Spiny-headed Mat-rush**
Lomandra multiflora (R. Br.) Britten subsp. *multiflora*	**Many-flowered Mat-rush**
Orchidaceae	
Acianthus exsertus R. Br.	**Mosquito Orchid**
Caladenia capillata D.L. Jones (syn. *Caladenia tentaculata* Tate)	**Green Comb**
Caladenia carnea R. Br.	**Pink Fingers**
Caladenia gracilis R. Br.	**Musky Caladenia**
Caleana major R. Br.	**Large Duck Orchid**
Chiloglottis trapeziformis Fitzg.	**Broad-lip Bird Orchid**
Cyrtostylis reniformis R. Br.	**Gnat Orchid**
Dipodium roseum D. L. Jones & M.A. Clem.	**Hyacinth Orchid**

Diuris sulphurea R. Br.	**Tiger Orchid**
Diuris semilunulata Messmer	
Genoplesium sp.	
Microtis unifolia (G. Forst.) Rchb.f.	**Common Onion Orchid**
Paracaleana minor (R. Br.) Blaxell	**Small Duck Orchid**
Prasophyllum brevilabre (Lindl.) Hook.f.	**Short-lipped Leek Orchid**
Pterostylis cynocephala Fitzg.	**Swan Greenhood**
Pterostylis furcata Lindl.	**Sickle Greenhood**
Pterostylis hamata Blackmore & Clemesha	**Hooked Greenhood**
Pterostylis nutans R. Br.	**Nodding Greenhood**
Pterostylis pedunculata R. Br.	**Maroonhood**
Thelymitra pauciflora R. Br.	**Slender Sun Orchid**
Phormiaceae	
Dianella revoluta R. Br. var. *revoluta*	
Stypandra glauca R. Br.	**Nodding Blue Lily**
Poaceae	
* *Aira caryophyllea* L.	**Silvery Hairgrass**
* *Aira elegantissima* Schur	**Delicate Hairgrass**
* *Anthoxanthum odoratum* L.	**Sweet Vernal Grass**
Austrostipa densiflora (Hughes) S.W.L. Jacobs & J. Everett	
Austrostipa scabra var. *falcata* (Hughes) S.W.L. Jacobs & J. Everett	
* *Briza maxima* L.	**Quaking Grass**
* *Briza minor* L.	**Shivery Grass**
* *Bromus diandrus* Roth	**Great Brome**
* *Bromus molliformis* Jn. Lloyd	**Soft Brome**
Deyeuxia quadriseta (Labill.) Benth.	
Deyeuxia rodwayi Vickery	
Dichelachne crinita (L.) Hook.f.	**Longhair Plumegrass**
Dichelachne inaequiglumis (Hack. ex Cheeseman) Edgar & Connor	
Dichelachne micrantha (Cav.) Domin	**Shorthair Plumegrass**
Dichelachne rara (R. Br.) Vickery	
Echinopogon ovatus (G. Forst.) P. Beauv.	**Forest Hedgehog Grass**

Elymus scaber (R. Br.) A. Love	**Common Wheatgrass**
* *Holcus lanatus* L.	**Yorkshire Fog**
* *Hordeum leporinum* Link	**Barley Grass**
Lachnagrostis aemula (R. Br.) Trin.	**Blowngrass**
Lachnagrostis filiformis (G. Forst.) Trin.	
Microlaena stipoides (Labill.) R. Br. var. *stipoides*	**Weeping Grass**
Panicum effusum R. Br.	**Hairy Panic**
Pentapogon quadridus (Labill.) Baill.	**Fiveawn Speargrass**
* *Poa annua* L.	**Winter Grass**
Poa labillardierei Steud. var. *labillardierei*	**Tussock**
Poa sieberiana var. *cyanophylla* Vickery	
Poa sieberiana var *hirtella* Vickery	
Poa sieberiana Spreng. var *sieberiana*	**Snowgrass**
Rytidosperma erianthum (Lindl.) Connor & Edgar	
Rytidosperma fulvum (Vickery) A.M. Humphreys & H.P. Linder	
Rytidosperma pallidum (R. Br.) A.M. Humphreys & H.P. Linder	**Redanther Wallaby Grass**
Rytidosperma pilosum (R. Br.) Connor & Edgar	
Rytidosperma racemosum (R.Br.) Connor & Edgar var. *racemosum*	
Themeda triandra Forssk.	**Kangaroo Grass**
* *Triticum aestivum* L.	**Wheat**
* *Vulpia bromoides* (L.) Gray	**Squirrel Tail Fescue**
* *Vulpia muralis* (Kunth) Nees	
* *Vulpia myuros* (L.) C.C. Gmel.	**Rat’s Tail Fescue**
Xanthorrhoeaceae	
Xanthorrhoea glauca subsp. *angustifolia* D.J. Bedford	

Appendix 2: Vegetation Type Descriptions for the Brindabella and Burrinjuck Study Areas

Vegetation Type 1: *Eucalyptus dives - Eucalyptus macrorhyncha - Eucalyptus mannifera* (+/- *Eucalyptus rubida*) Open Forest (Species Rich)

Brindabella Sites: 1, 3, 70, 71, 73, 74, 75, 76, 79, 80, 82, 84, 85, 89, 93, 94, 96, 98, 101, 106, 110, 111, 123, 125, 126

Altitude Range: 500 – 1150 m

Lithology: Sediments and Volcanics

Position in Landscape: Crests and Slopes

Structure: Low Open Forest or Open Forest from 10 – 15 m in height

Upper Stratum: *Eucalyptus dives* and *E. mannifera;* occasional *E. macrorhyncha* and *E. rubida*.

Mid Stratum: Sparse to absent shrub layer with occasional thickets of *Cassinia longifolia*, *Cassinia aculeata* and *Daviesia mimosoides* subsp. *mimosoides*.

Lower Stratum: *Rytidosperma pallidum* and *Poa sieberiana*, grass layer; diverse array of herbs and small shrubs including *Platylobium formosum* ssp. *formosum*, *Dianella revoluta*, *Hibbertia obtusifolia*, *Monotoca scoparia*, *Hardenbergia violacea*, *Gonocarpus tetragynus*, *Persoonia chamaepeuce*, *Tetratheca bauerifolia*, *Lomandra longifolia* and *Stylidium graminifolium.*

Vegetation Type 2: *Eucalyptus macrorhyncha - Eucalyptus nortonii* Low Open Forest

Brindabella Sites: 8, 128, 129

Altitude Range: 550 - 800 m

Lithology: Arenite, Siltstone, Shale

Position in Landscape: Slopes

Structure: Low Open Forest or Woodland from 8 – 10 m in height

Upper Stratum: *Eucalyptus macrorhyncha* and *E. nortonii*; occasional *E. rossii* and *E. dives*.

Mid Stratum: Very open with few shrubs; some scattered *Dillwynia juniperina* and *Acacia ulicifolia.*

Lower Stratum: Sparse grassy layer of *Rytidosperma pallidum* or *Poa sieberiana* with *Lomandra longifolia* and *Stylidium graminifolium.*

Vegetation Type 3: *Eucalyptus dives - Eucalyptus mannifera - Eucalyptus macrorhyncha* Open Forest (Species Poor)

Brindabella Sites: 57, 60

Altitude Range: 900 – 1000 m

Lithology: Volcanics

Position in Landscape: Crests, Upper Slopes

Structure: Open Forest from 12 - 15 m in height

Upper Stratum: *Eucalyptus dives* and *E. mannifera*; some *E. macrorhyncha* and *E. bridgesiana.*

Mid Stratum: *Cassinia longifolia* and *Dodonaea viscosa.*

Lower Stratum: scattered grasses in low abundance with *Lomandra longifolia* and *Stylidium graminifolium.*

Vegetation Type 4: *Eucalyptus bridgesiana - Eucalyptus melliodora* Open Forest

Brindabella Sites: 130

Altitude Range: 400 - 500 m

Lithology: Black Shale and Mudstone

Position in Landscape: Lower Slopes and Flats

Structure: Open Forest up to 20 m in height

Upper Stratum: *Eucalyptus melliodora* and *E. bridgesiana.*

Mid Stratum: *Brachychiton populneus.*

Lower Stratum: Mixture of native and exotic forbs and herbs.

Vegetation Type 5: *Eucalyptus dalrympleana - Eucalyptus dives* (+/- *Eucalyptus pauciflora)* Open Forest

Brindabella Sites: 5, 10, 12, 14, 17, 19, 21, 23, 30, 33, 34, 36, 47, 52, 55, 58, 59, 62, 64, 67, 100, 107, 108, 117, 120

Altitude Range: 750 – 1300 m

Lithology: Sediments and Volcanics

Position in Landscape: Crests and Slopes

Structure: Open Forest from 15 - 25 m in height

Upper Stratum: *Eucalyptus dalrympleana* and *E. dives*; occasional *E. pauciflora* and *E. robertsonii*.

Mid Stratum: variable with scattered *Acacia dealbata* or a dense cover of *Daviesia mimosoides* subsp. *mimosoides.*

Lower Stratum: *Poa sieberiana* dominant with occasional *Rytidosperma pallidum* and *Lomandra longifolia*. *Hibbertia obtusifolia, Monotoca scoparia, Coprosma hirtella, Lomandra longifolia* and *Dianella tasmanica* form the common low shrub species.

Vegetation Type 6: *Eucalyptus dalrympleana - Eucalyptus robertsonii* Tall Open Forest

Brindabella Sites: 2, 4, 6, 42, 78, 88, 95, 97, 112, 119, 122

Altitude Range: 700 – 1200 m

Lithology: Sediments and Volcanics

Position in Landscape: Higher Undulating Terrain, Protected Slopes and Gullies

Structure: Tall Open Forest between 20 and 35 m in height

Upper Stratum: *Eucalyptus robertsonii* and *E. dalrympleana* with occasional *E. pauciflora.*

Mid Stratum: *Acacia dealbata* or *A. melanoxylon* as a scattered shrub layer.

Lower Stratum: *Poa sieberiana*, *Pteridium esculentum* and *Coprosma hirtella* forming a moderate cover. A diversity of herbs, forbs and grasses including *Lomandra longifolia*, *Stylidium graminifolium*, *Acaena novae-zelandiae*, *Dichondra repens*, *Microlaena stipoides*, *Hibbertia obtusifolia*, *Dianella revoluta* and *Gonocarpus tetragynus*.

Vegetation Type 7: *Eucalyptus dalrympleana - Eucalyptus delegatensis* Tall Open Forest

Brindabella Sites: 15, 38, 46, 65, 68

Altitude Range: 1100 – 1300 m

Lithology: Sediments and Volcanics

Position in Landscape: Protected Slopes and Gullies above 1100 m

Structure: Tall Open Forest from 30 - 35 m in height

Upper Stratum: *Eucalyptus delegatensis* with scattered *E. dalrympleana;* stands north of Piccadilly Circus may also contain *E. fastigata* or *E. robertsonii*. Mixed stands of *E. delegatensis* and *E. fastigata* north of Mt. Coree are unusual because these species form distinctly separate vegetation types further south and are generally clearly differentiated, with *E. delegatensis* occurring in pure stands at higher altitude above the limits of *E. fastigata.*

Mid Stratum: May contain a shrub layer of *Acacia dealbata*, *Coprosma hirtella*, *Oxylobium ellipticum* or *Daviesia mimosoides*.

Lower Stratum: *Poa sieberiana* forming up to 50% cover with *Lomandra longifolia* and a variety of herbs with high cover including *Stylidium graminifolium*, *Viola hederacea* and *Geranium* spp.

Vegetation Type 8: *Eucalyptus dalrympleana - Eucalyptus pauciflora* Open Forest

Brindabella Sites: 16, 24, 25, 29, 31, 32, 39, 41, 43, 44, 45, 48, 49, 61, 63, 90, 91, 103, 104, 105, 113, 114, 115, 116, 127

Altitude Range: 1150 – 1450 m

Lithology: Volcanics

Position in Landscape: Crests, Slopes (particularly Upper Slopes), occasionally Gullies

Structure: Low Open Woodland up to 12 m in height to Open Forest up to 25 m in height, depending on local site conditions

Upper Stratum: Eucalyptus dalrympleana with scattered *E. pauciflora* in protected positions and *E. pauciflora* with the occasional *E. dalrympleana* in more exposed positions. Factors such as site exposure and soil depth determine the height and overall structure of this vegetation type, but the understorey floristics are less variable in relation to these factors. Also contains occasional *E. viminalis.*

Mid Stratum: scattered *Acacia dealbata* and *Coprosma hirtella*.

Lower Stratum: *Poa sieberiana* and *Lomandra longifolia* with scattered herbs including *Stellaria pungens, Asperula scoparia, Acaena novae-zelandiae* and *Stylidium graminifolium. Leucopogon hookeri* occurs more commonly at the higher elevation sites within this type, as *does Brachycome aculeata* and *Scleranthus diander*.

Vegetation Type 9: *Eucalyptus camphora* subsp. *humeana* Open Forest

Brindabella Sites: 7

Altitude Range: 650 -750 m

Lithology: Sediments

Position in Landscape: Flats adjacent to creeks at lower elevations

Structure: Open Forest of to 25 m in height

Upper Stratum: *Eucalyptus camphora* subsp. *humeana.*

Mid Layer: not present.

Lower Stratum: complex lower layer consisting of *Poa sieberiana* and *Carex appressa* over a herb layer of *Acaena novae-zelandiae*, *Hydrocotyle laxiflora*, *Dichondra repens* and *Rubus parvifolius*. A number of species found at this site were not located elsewhere in the study area including *Lycopus australis*, *Veronica gracilis*, *Daucus glochidiatus* and *Gratiola peruviana.*

Vegetation Type 10: *Eucalyptus viminalis - Acacia melanoxylon* Open Forest

Brindabella Sites: 9, 18, 35, 37, 51, 54, 56, 77, 81, 83, 86, 87, 99, 124

Altitude Range: 650 – 1050 m

Lithology: Sediments and Volcanics

Position in Landscape: Minor and major Creeklines

Structure: Tall Open Forest up to 35 m in height

Upper Stratum: E. viminalis, with occasional *E. dalrympleana* or *E. robertsonii.*

Mid Stratum: *Acacia melanoxylon* and *Pomaderris aspera* to and occasionally *Bedfordia arborescens*, often with *Coprosma quadrifida*, *Cassinia longifolia* and *Lomatia myricoides.*

Lower Stratum: *Poa helmsii* and *Poa labillardierei* subsp. *labillardierei* with *Pteridium esculentum*, *Carex appressa*, *Adiantum aethiopicum*, *Rubus parvifolius* and *Microlaena stipoides.*

Vegetation Type 11: *Casuarina cunninghamiana* Tall Open Forest

Brindabella Sites: Site 72

Altitude Range: 400 – 500 m

Lithology: Volcanics

Position in Landscape: Primarily Goodradigbee River and lower parts of tributaries

Structure: Tall Open Forest up to 30 m in height

Upper Stratum: *Casuarina cunninghamiana.*

Mid Stratum: Kunzea ericoides and *Leptospermum brevipes* with occasional *Callistemon sieberi*. However, the lower layers in many stretches are dominated by the Blackberry, *Rubus fruticosus sens. lat.*

Lower Stratum: Microlaena stipoides forms a distinctive lower layer with other grasses and forbs.

Vegetation Type 12: *Eucalyptus dalrympleana - Eucalyptus fastigata* Tall Open Forest

Brindabella Sites: 11, 13, 20, 27, 28, 40, 50, 53, 66, 69, 92, 102, 109, 118

Altitude Range: 900 – 1200 m

Lithology: Sediments and Volcanics

Position in Landscape: Creeks and Protected Slopes

Structure: Tall Open Forest to 35m in height

Upper Stratum: *Eucalyptus fastigata*, *E. dalrympleana* or *E. viminalis*.

Mid Stratum: *Acacia melanoxylon*, *Bedfordia arborescens* and *Pomaderris aspera* forming a dense layered mid-layer. In protected upper creek lines *Olearia argophylla, Olearia stellulata, Hedycarya angustifolia* and *Dicksonia antarctica* are often present.

Lower Stratum: *Poa helmsii* and *Poa labillardierei* with *Pteridium esculentum, Carex appressa* and occasionally *Blechnum nudum*.

Vegetation Type 13: *Leptospermum micromyrtus* Montane Rocky Heath

Brindabella Sites: 22

Altitude Range: 1300 - 1400 m

Lithology: Volcanics

Position in Landscape: Exposed rocky outcrops on ridgelines

Structure: Closed shrubland to 1.5 m in height

Upper Stratum: *Leptospermum micromyrtus.* Other outcrops to the east are dominated by *Leptospermum brevipes*, *Leionema lamprophyllum* and *Micrantheum hexandrum* in the vicinity of Two Sticks Road and by *Prostanthera rotundifolia* and *Kunzea parvifolia* along the Baldy Range.

Mid Stratum: low abundance of *Phebalium squamulosum* ssp. *ozothamnoides* and *Callistemon pallidus.*

Lower Layer: occasional *Bulbine glauca.*

Vegetation Type 14: *Leptospermum myrtifolium – Baeckea utilis* Swamp

Brindabella Sites: 26, 121

Altitude Range: 750 – 1050 m

Lithology: Volcanics

Position in Landscape: Low lying sediment basins in creeks

Structure: Open to Closed Shrubland to 2.5 m

Upper Stratum: *Leptospermum myrtifolium* and *Baeckea utilis,* occasional *Leptospermum continentale* and *Leptospermum juniperinum.*

Mid Stratum: Epacris breviflora.

Lower Stratum: Restionaceae (*Empodisma minus* and *Baloskion australe*) and Cyperaceae (*Baumea rubiginosa*).

Vegetation Type 15: *Eucalyptus bridgesiana* Open Forest

Burrinjuck Sites: 1, 2, 5, 9

Altitude Range: 350 – 600 m

Lithology: Shale and Tuff

Position in Landscape: Eastern Creeks

Structure: Open Forest from 18 - 25 m in height

Upper Stratum: *Eucalyptus bridgesiana* with occasional *E. macrorhyncha* and *E. goniocalyx.*

Mid Stratum: Acacia dealbata subsp. *alpina.*

Lower Stratum: Acaena novae-zelandiae, Geranium spp., *Urtica incisa, Pteridium esculentum* and *Carex appressa.*

Vegetation Type 16: *Eucalyptus bicostata – E. robertsonii – E. dalrympleana* Tall Open Forest

Burrinjuck Sites: 13, 15, 16, 17, 18, 22, 24

Altitude Range: 550 – 950 m

Lithology: Volcanics

Position in Landscape: Western Creeks; and undulating terrain at higher altitude

Structure: Tall Open Forest 20 – 35 m in height

Upper Stratum: *Eucalyptus bicostata* and *E. robertsonii* with occasional *E. bridgesiana* in creeks; *E. dalrympleana* and *E. viminalis* at higher altitude.

Mid Stratum: Acacia dealbata subsp. *alpina.*

Lower Stratum: Poa sieberiana, Acaena novae-zelandiae, Glycine clandestina and Stellaria pungens.

Vegetation Type 17: *Eucalyptus bridgesiana – E. melliodora* Woodland / Open Forest

Burrinjuck Sites: 31, 32

Altitude Range: 550 – 750 m

Lithology: Volcanics

Position in Landscape: Western creeks

Structure: Woodland - Open Forest 10 -15 m in height

Upper Stratum: *Eucalyptus bridgesiana* and *E. melliodora* occasional *E. rubida.*

Mid Stratum: Leptospermum juniperinum, Acacia mearnsii and *Acacia dealbata* subsp. *alpina* with occasional *Kunzea ericoides* and *Callistemon sieberi.*

Lower Stratum: Microlaena stipoides and *Poa sieberiana.*

Vegetation Type 18: *Calytrix tetragona – Westringia eremicola* Rocky Outcrop Heath

Burrinjuck Sites: 27

Altitude Range: 750 -850 m

Lithology: Rhyolite

Position in Landscape: Crests and west facing upper slopes

Structure: Low Shrubland from 0.5 to 1.0 m in height

Upper Stratum: *Calytrix tetragona* and *Westringia eremicola*. Other rocky outcrops further to the west of the Reserve are dominated by dense stands of *Allocasuarina verticillata.*

Mid Stratum: not present.

Lower Stratum: Melichrus urceolatus and *Dichelachne crinita.*

Vegetation Type 19: *Eucalyptus goniocalyx – E. macrorhyncha* Open Forest

Burrinjuck Sites: 3, 4, 6, 8, 12, 21, 29

Altitude Range: 400 – 600 m

Lithology: Tuff

Position in Landscape: Crests and Slopes

Structure: Open Forest from 15 – 18 m in height

Upper Stratum: *Eucalyptus goniocalyx* and *E. macrorhyncha* with occasional *E. dives.*

Mid Stratum: *Cassinia longifolia, Acacia rubida.*

Lower Stratum: *Poa sieberiana, Hibbertia obtusifolia, Lomandra filiformis, Gonocarpus tetragynus* and *Lomandra multiflora*.

Vegetation Type 20: *Eucalyptus dives – E. macrorhyncha – E. mannifera* Open Forest

Burrinjuck Sites: 7, 10, 14, 19, 20, 23, 25, 26, 28, 30, 33

Altitude Range: 450 – 900 m

Lithology: Volcanics

Position in Landscape: Mid-slopes

Structure: Open Forest 15 – 20 m in height

Upper Stratum: *Eucalyptus dives, E. macrorhyncha* and *E. mannifera.*

Mid Stratum: sparse.

Lower Stratum: Poa sieberiana, Platylobium formosum, Rytidosperma pallidum and *Lomandra longifolia.*

Vegetation Type 21: *Eucalyptus rossii* Low Open Forest

Burrinjuck Sites: 11

Altitude Range: 500 – 600 m

Lithology: Tuff

Position in Landscape: Mid-slopes

Structure: Open Forest from 10 -12 m in height

Upper Stratum: *Eucalyptus rossii.*

Mid Stratum: *Acacia buxifolia* and *Daviesia leptophylla.*

Lower Stratum: Dampiera purpurea and *Rhytidosporum procumbens.*

Appendix 3: Fire Responses of Vascular Plant Species in the Brindabella and Burrinjuck Study Areas

Fire response categories based on Gill and Bradstock (1992)

* = exotic taxon; sp. = undetermined taxon; subsp. = subspecies; var. = variety

GENUS	**SPECIES**	**GILL & BRADSTOCK FIRE RESPONSE CATEGORY**						
		1	**2**	**3**	**4**	**5**	**6**	**7**
**Acetosella*	*vulgaris*				4			
**Aira*	*caryophyllea*		2					
**Aira*	*cupaniana*		2					
**Aira*	*elegantissima*		2					
**Aira*	sp.		2					
**Anagallis*	*arvensis*		2					
**Anthoxanthum*	*odoratum*				4			
**Arctotheca*	*calendula*		2					
**Briza*	*maxima*		2					
**Briza*	*minor*		2					
**Bromus*	*brevis*		2					
**Bromus*	*diandrus*		2					
**Bromus*	*mollis*		2					
**Carduus*	*nutans*		2					
**Carduus*	*pycnocephalus*		2					
**Centaurium*	*erythraea*		2					
**Cerastium*	*glomeratum*		2					
**Cirsium*	*vulgare*		2					
**Conium*	*maculatum*				4			
**Conyza*	*bonariensis*		2					
**Crepis*	*capillaris*		2					
**Crepis*	*foetida*		2					
**Echium*	*plantagineum*		2					
**Echium*	*vulgare*		2					
**Erodium*	*moschatum*		2					
**Euphorbia*	*peplus*		2					
**Galium*	*aparine*		2					
**Geranium*	*molle*		2					
**Holcus*	*lanatus*		2					
**Hordeum*	*leporinum*		2					
**Hypochoeris*	*glabra*		2					
**Hypochoeris*	*radicata*				4			
**Lactuca*	*serriola*		2					
**Leucanthemum*	*vulgare*		2		4			
**Linaria*	*pelisseriana*		2					
**Medicago*	sp.		2					
**Mentha*	*pulegium*				4			
**Modiola*	*caroliniana*				4			
**Myosotis*	*discolor*		2					
**Orobanche*	*minor*				4			
**Papaver*	*dubium*		2					
**Parentucellia*	*latifolia*				4			
**Pinus*	*radiata*	1						
**Plantago*	*lanceolata*				4			

**Poa*	*annua*		2					
**Polycarpon*	*tetraphyllum*		2					
**Prunella*	*vulgaris*				4			
**Pseudognaphalium*	*luteoalbum*				4			
**Ranunculus*	*amphitrichus*				4			
**Ranunculus*	*muricatus*		2					
**Rorippa*	*nasturtium-aquaticum*		2					
**Rosa*	*rubiginosa*				4			
**Rubus*	*fruticosus*				4			
**Sagina*	*apetala*							
**Sherardia*	*arvensis*		2					
**Silene*	*gallica*		2					
**Silybum*	*marianum*		2					
**Sisymbrium*	*officinale*		2					
**Sisymbrium*	*orientale*		2					
**Solanum*	*nigrum*		2					
**Sonchus*	*asper*		2					
**Sonchus*	*oleraceus*		2					
**Spergularia*	*rubra*		2					
**Stellaria*	*media*		2					
**Taraxacum*	*officinale*		2		4			
**Tolpis*	*umbellata*		2					
**Trifolium*	*angustifolium*		2					
**Trifolium*	*arvense*		2					
**Trifolium*	*campestre*		2					
**Trifolium*	*dubium*		2					
**Trifolium*	*repens*		2					
**Triticum*	*aestivum*		2					
**Verbascum*	*thapsus*				4			
**Verbascum*	*virgatum*				4			
**Veronica*	*anagallis-aquatica*		2					
**Veronica*	*arvensis*		2					
**Vulpia*	*bromoides*		2					
**Vulpia*	*muralis*		2					
**Vulpia*	*myuros*		2					
Acacia	*buxifolia*				4			
Acacia	*dealbata* subsp. *alpina*				4			
Acacia	*falciformis*				4			
Acacia	*gunnii*				4			
Acacia	*implexa*				4			
Acacia	*mearnsii*				4			
Acacia	*melanoxylon*				4			
Acacia	*pravissima*				4			
Acacia	*rubida*				4			
Acacia	*ulicifolia*				4			
Acacia	*verniciflua*				4			
Acaena	*echinata*				4			
Acaena	*novae-zelandiae*				4			
Acaena	*ovina*				4			
Acianthus	*exsertus*				4			
Acianthus	*fornicatus*				4			
Acrotriche	*serrulata*				4			
Adiantum	*aethiopicum*				4			
Agrostis	*bettyae*		2					
Allocasuarina	*verticillata*	1						
Ammobium	*craspedioides*				4			
Aphanes	*arvensis*		2					
Aphanes	*australiana*		2					

Arrhenechthites	*mixta*				4			
Arthropodium	*milleflorum*				4			
Arthropodium	*minus*				4			
Asperula	*conferta*				4			
Asperula	*pusilla*		2					
Asperula	*scoparia*				4			
Asplenium	*flabellifolium*				4			
Asplenium	*trichomanes*		2					
Astroloma	*humifusum*				4			
Astrotricha	*ledifolia*				4			
Australina	*pusilla*				4			
Austrostipa	*rudis*				4			
Austrostipa	*scabra* subsp. *falcata*				4			
Baeckea	*utilis*					5		
Baloskion	*australe*				4			
Banksia	*marginata*	1						
Baumea	*rubiginosa*				4			
Bedfordia	*arborescens*					5		
Billardiera	*scandens*				4			
Blechnum	*minus*				4			7
Blechnum	*nudum*				4			7
Blechnum	*wattsii*				4			7
Bossiaea	*buxifolia*				4			
Bossiaea	*foliosa*				4	5		
Brachychiton	*populneus*				4			
Brachyloma	*daphnoides*					5		
Brachyscome	*aculeata*				4			
Brachyscome	*scapigera*				4			
Brachyscome	*spathulata*				4			
Bracteantha	*bracteata*		2					
Brunonia	*australis*				4			
Bulbine	*bulbosa*				4			
Bulbine	*glauca*				4			
Burchardia	*umbellata*				4			
Bursaria	*spinosa* subsp. *lasiophylla*					5		
Bursaria	*spinosa* subsp. *spinosa*					5		
Caesia	sp.				4			
Caladenia	*carnea*				4			
Caladenia	*gracilis*				4			
Caladenia	*tentaculata*				4			
Caleana	*major*				4			
Callistemon	*pallidus*					5		
Callistemon	*pityoides*					5		
Callistemon	*sieberi*					5	6	
Callitriche	*stagnalis*				4			
Calochilus	*robertsonii*				4			
Calotis	*scabiosifolia*				4			
Calytrix	*tetragona*		2					
Carex	*appressa*				4			
Cassinia	*aculeata*		2					
Cassinia	*laevis*				4	5	6	
Cassinia	*longifolia*					5	6	
Cassytha	*pubescens*				4			
Casuarina	*cunninghamiana*						6	
Celmisia	*asteliifolia*				4			
Celmisia	*longifolia*				4			
Cheilanthes	*austrotenuifolia*				4			
Cheilanthes	*sieberi*				4			

Cheiranthera	*linearis*				4			
Chiloglottis	*trapeziiformis*				4			
Chiloglottis	*valida*				4			
Choretrum	*pauciflorum*				4			
Chrysocephalum	*apiculatum*				4			
Chrysocephalum	*semipapposum*				4			
Clematis	*aristata*				4			
Clematis	*leptophylla*				4			
Convolvulus	*erubescens*				4			
Coprosma	*hirtella*				4			
Coprosma	*quadrifida*				4			
Coronidium	*scorpioides*				4			
Corybas	sp.				4			
Craspedia	*variabilis*				4			
Crassula	*sieberiana*		2					
Cullen	*microcephalum*				4			
Cymbonotus	*lawsonianus*				4			
Cynoglossum	*australe*				4			
Cynoglossum	*suaveolens*				4			
Cyrtostylis	*reniformis*				4			
Cyrtostylis	*trapeziiformis*				4			
Dampiera	*purpurea*				4			
Daucus	*glochidiatus*		2					
Daviesia	*leptophylla*				4			
Daviesia	*mimosoides* subsp. *acris*	1						
Daviesia	*mimosoides* subsp. *mimosoides*				4	5		
Daviesia	*ulicifolia* subsp. *ruscifolia*					5		
Desmodium	*varians*				4			
Deyeuxia	*quadriseta*				4			
Deyeuxia	*reflexa*				4			
Deyeuxia	*rodwayi*				4			
Dianella	*laevis*				4			
Dianella	*revoluta* var. *revoluta*				4			
Dianella	*tasmanica*				4			
Dichelachne	*crinita*				4			
Dichelachne	*inaequiglumis*				4			
Dichelachne	*micrantha*				4			
Dichelachne	*rara*				4			
Dichelachne	*sciurea*				4			
Dichelachne	*sieberiana*				4			
Dichondra	*repens*				4			
Dichopogon	*strictus*				4			
Dicksonia	*antarctica*				4			7
Dillwynia	*phylicoides*		2					
Dillwynia	*sericea*		2					
Dillwynia	*sieberi*		2					
Dipodium	*roseum*				4			
Diuris	*lanceolata*				4			
Diuris	*maculata*				4			
Diuris	*sulphurea*				4			
Dodonaea	*boroniifolia*		2					
Dodonaea	*viscosa* subsp. *angustissima*		2					
Drosera	*auriculata*		2					
Drosera	*spatulata*		2					
Drymophila	*cyanocarpa*				4			
Echinopogon	*ovatus*				4			
Einadia	*nutans* subsp. *nutans*				4			
Elymus	*scaber*				4			

Empodisma	*minus*				4			
Epacris	*breviflora*					5		
Epacris	*microphylla*				4			
Epilobium	*billardierianum* subsp. *cinereum*		2					
Eriochilus	*cucculatus*				4			
Eucalyptus	*bicostata*					5	6	
Eucalyptus	*bridgesiana*					5	6	
Eucalyptus	*camphora* var. *humeana*					5	6	
Eucalyptus	*dalrympleana subsp. dalrympleana*					5	6	
Eucalyptus	*delegatensis* subsp. *delegatensis*	1						
Eucalyptus	*dives*					5	6	
Eucalyptus	*fastigata*					5	6	
Eucalyptus	*goniocalyx*					5	6	
Eucalyptus	*macrorhyncha*					5	6	
Eucalyptus	*mannifera* subsp. *mannifera*					5	6	
Eucalyptus	*melliodora*					5	6	
Eucalyptus	*nortonii*					5	6	
Eucalyptus	*pauciflora*					5	6	
Eucalyptus	*polyanthemos* subsp. *polyanthemos*					5	6	
Eucalyptus	*robertsonii* subsp. *robertsonii*					5	6	
Eucalyptus	*rossii*					5	6	
Eucalyptus	*rubida* subsp. *rubida*					5	6	
Eucalyptus	*stellulata*					5	6	
Eucalyptus	*viminalis* subsp. *viminalis*					5	6	
Euchiton	*involucratus*				4			
Euchiton	*japonicus*				4			
Euchiton	*sphaericus*				4			
Exocarpos	*cupressiformis*				4		6	
Exocarpos	*strictus*				4	5		
Galium	*gaudichaudii*				4			
Galium	*propinquum*				4			
Gastrodia	*sesamoides*				4			
Genoplesium	*nudum*				4			
Geranium	*retrorsum*				4			
Geranium	*solanderi* var. *solanderi*				4			
Gingidia	*harveyana*				4			
Glossodia	*major*				4			
Glycine	*clandestina*				4			
Gompholobium	*huegelii*				4			
Gonocarpus	*elatus*				4			
Gonocarpus	*micranthus*				4			
Gonocarpus	*tetragynus*				4			
Goodenia	*hederacea* subsp. *hederacea*				4			
Gratiola	*peruviana*				4			
Grevillea	*lanigera*				4			
Grevillea	*oxyantha* subsp. *oxyantha*		2					
Grevillea	*victoriae* subsp. *brindabella*		2					
Gynatrix	*pulchella*		2					
Hakea	*decurrens* subsp. *decurrens*				4			
Hakea	*lissosperma*				4			
Hakea	*microcarpa*					5		
Hardenbergia	*violacea*				4			
Hedycarya	*angustifolia*					5		
Hibbertia	*obtusifolia*				4			
Hookerochloa	*eriopoda*				4			
Hookerochloa	*hookeriana*				4			
Hovea	*asperifolia* subsp. *asperifolia*		2					
Hovea	*heterophylla*				4			

Hydrocotyle	*laxiflora*				4			
Hydrocotyle	*sibthorpioides*				4			
Hydrocotyle	*tripartita*				4			
Hypericum	*gramineum*				4			
Hypolepis	*muelleri*				4			
Hypolepis	*rugulosa*				4			
Hypoxis	*hygrometrica*				4			
Indigofera	*adesmiifolia*				4			
Indigofera	*australis*				4	5		
Isoetopsis	*graminifolia*		2					
Isolepis	*gaudichaudiana*				4			
Isolepis	*hookeriana*				1			
Juncus	*bufonius*				4			
Juncus	*subsecundus*				4			
Kunzea	*ericoides*					5		
Kunzea	*parvifolia*					5		
Lachnagrostis	*aemula*		2					
Lagenophora	*stipitata*				4			
Leionema	*lamprophyllum* subsp. *obovatum*		2					
Lepidosperma	*laterale*				4			
Lepidosperma	*lineare*				4			
Leptinella	*filicula*		2					
Leptospermum	*brevipes*					5		
Leptospermum	*grandifolium*	1						
Leptospermum	*juniperinum*					5		
Leptospermum	*micromyrtus*	1						
Leptospermum	*multicaule*					5		
Leptospermum	*myrtifolium*					5		
Leucopogon	*attenuatus*					5		
Leucopogon	*fraseri*				4			
Leucopogon	*gelidus*				4			
Leucopogon	*juniperinus*				4			
Leucopogon	*virgatus*					5		
Lobelia	*dentata*				4			
Lobelia	*gibbosa*				4			
Lomandra	*filiformis* var. *coriacea*				4			
Lomandra	*filiformis* var. *filiformis*				4			
Lomandra	*longifolia* var. *longifolia*				4			
Lomandra	*multiflora* subsp. *multiflora*				4			
Lomatia	*myricoides*					5		
Luzula	*flaccida*				4			
Lycopus	*australis*				4			
Melichrus	*urceolatus*					5		
Mentha	*diemenica*				4			
Micrantheum	*hexandrum*		2					
Microlaena	*stipoides* var. *stipoides*				4			
Microseris	*lanceolata*				4			
Microtis	*unifolia*				4			
Mirbelia	*oxyloboides*					5		
Monotoca	*scoparia*					5		
Montia	*fontana*				4			
Neopaxia	*australasica*				4			
Olearia	*argophylla*					5	6	
Olearia	*erubescens*				4			
Olearia	*lirata*		2					
Olearia	*megalophylla*				4	5		
Omphacomeria	*acerba*				4			
Opercularia	*diphylla*				4			

Opercularia	*hispida*				4			
Ophioglossum	*lusitanicum*				4			
Oreomyrrhis	*eriopoda*				4			
Oxalis	*corniculata*				4			
Oxalis	*perennans*				4			
Oxylobium	*ellipticum*				4			
Ozothamnus	*stirlingii*		2					
Panicum	*effusum*		2					
Pelargonium	*australe*				4			
Pellaea	*falcata*				4			
Persoonia	*chamaepeuce*				4	5		
Persoonia	*rigida*				4			
Phebalium	*squamulosum* subsp. *ozothamnoides*		2					
Picris	*angustifolia* subsp. *merxmuelleri*		2					
Pimelea	*curviflora* var. *sericea*				4			
Pimelea	*linifolia*				4			
Pimelea	*treyvaudii*		2					
Plantago	*debilis*				4			
Plantago	*major*				4			
Plantago	*varia*				4			
Platylobium	*formosum* subsp. *formosum*				4			
Platysace	*lanceolata*				4			
Poa	*helmsii*				4			
Poa	*hookeri*				4			
Poa	*induta*				4			
Poa	*labillardierei* var. *labillardierei*				4			
Poa	*phillipsiana*				4			
Poa	*sieberiana* var. *cyanophylla*				4			
Poa	*sieberiana* var. *hirtella*				4			
Poa	*sieberiana* var. *sieberiana*				4			
Podolepis	*hieracioides*				4			
Podolepis	*jaceoides*				4			
Podolobium	*alpestre*				4			
Polyscias	*sambucifolia*				4			
Polystichum	*proliferum*							7
Pomaderris	*aspera*					5	6	
Pomaderris	*eriocephala*				4			
Pomaderris	*subcapitata*				4			
Pomax	*umbellata*							
Poranthera	*microphylla*		2					
Prasophyllum	*brevilabre*				4			
Pratia	*pedunculata*				4			
Prostanthera	*lasianthos*		2					
Prostanthera	*rotundifolia*		2					
Prostanthera	sp. Undescribed		2					
Pteridium	*esculentum*				4			
Pterostylis	*curta*				4			
Pterostylis	*furcata*				4			
Pterostylis	*nutans*				4			
Pterostylis	*pedunculata*				4			
Pultenaea	*cunninghamii*				4	5		
Pultenaea	*juniperina*				4			
Pultenaea	*procumbens*				4			
Pultenaea	*spinosa*				4			
Ranunculus	*inundatus*				4			
Ranunculus	*lappaceus*				4			
Ranunculus	*muralis*		2					
Ranunculus	*plebeius*				4			

Ranunculus	*pumilio* var. *pumilio*		2					
Ranunculus	*sessiliflorus* var. *sessiliflorus*		2					
Rorippa	*dictyosperma*		2					
Rorippa	*gigantea*				4			
Rubus	*parvifolius*				4			
Rumex	*brownii*				4			
Rytidosperma	*pallidum*				4			
Rytidosperma	*racemosum*				4			
Schoenus	*apogon*				4			
Schoenus	*ericetorum*				4			
Scirpus	sp.				4			
Scleranthus	*diander*				4			
Senecio	*bathurstianus*				4			
Senecio	*bipinnatisectus*				4			
Senecio	*diaschides*		2					
Senecio	*gunnii*				4			
Senecio	*lineariifolius* var. *latifolius*				4			
Senecio	*quadridentatus*		2					
Solanum	*aviculare*		2					
Stackhousia	*monogyna*				4			
Stellaria	*flaccida*		2					
Stellaria	*pungens*				4			
Stuartina	*hamata*				4			
Stylidium	*graminifolium*				4			
Stypandra	*glauca*				4			
Tetratheca	*bauerifolia*				4			
Thelymitra	*nuda*				4			
Thelymitra	*pauciflora*				4			
Themeda	*triandra*				4			
Thysanotus	*patersonii*				4			
Thysanotus	*tuberosus*				4			
Triptilodiscus	*pygmaeus*		2					
Urtica	*incisa*				4			
Velleia	*paradoxa*				4			
Veronica	*calycina*				4			
Veronica	*derwentiana*				4			
Veronica	*perfoliata*				4			
Viola	*betonicifolia*				4			
Viola	*hederacea*				4			
Wahlenbergia	*communis*				4			
Wahlenbergia	*gloriosa*				4			
Wahlenbergia	*gracilis*				4			
Wahlenbergia	*stricta* subsp. *stricta*				4			
Westringia	*eremicola*		2					
Wurmbea	*biglandulosa*				4			
Wurmbea	*dioica*				4			
Xanthorrhoea	*glauca* subsp. *angustifolia*							7
Xerochrysum	*bracteatum*				4			
Xerochrysum	*viscosum*				4			

Appendix 4: Selected Pre- and Post-fire Photo Sequences, by Vegetation Type, in the Brindabella and Burrinjuck Study Areas 1996-2012

Vegetation Type 1: *Eucalyptus dives* - *Eucalyptus macrorhyncha* - *Eucalyptus mannifera* (+/- *Eucalyptus rubida*) Open Forest (Species Rich)

a) Example of recovery after low-moderate severity fire in 2003 = Site Brin_126

Pre-fire | 1 Year Post-fire | 9 Years Post-fire

16/04/1997 | 09/12/2003 | 12/11/2012

b) Example of recovery after high-very high severity fire in 2003 = Site Brin_093

Pre-fire | 1 Year Post-fire | 9 Years Post-fire

02/04/1997 | 15/12/2003 | 13/11/2012

Vegetation Type 2: *Eucalyptus macrorhyncha - Eucalyptus nortonii* Low Open Forest

a) Example of site of this vegetation type, remaining unburnt in 2003 = Site Brin_129

Pre-fire 1 Year Post-fire 9 Years Post-fire

12/05/1997 25/11/2003 12/11/2012

b) Example of recovery after high-very high severity fire in 2003 = Site Brin_128

Pre-fire 1 Year Post-fire 9 Years Post-fire

12/05/1997 25/11/2003 12/11/2012

Vegetation Type 3: *Eucalyptus dives - Eucalyptus mannifera - Eucalyptus macrorhyncha* Open Forest (Species Poor)

a) Example of recovery after low-moderate severity fire in 2003 = Site Brin_060

Pre-fire 1 Year Post-fire 9 Years Post-fire

25/02/1997 18/01/2003 01/11/2012

b) Example of recovery after high-very high severity fire in 2003 = Site Brin_057

Pre-fire 1 Year Post-fire 9 Years Post-fire

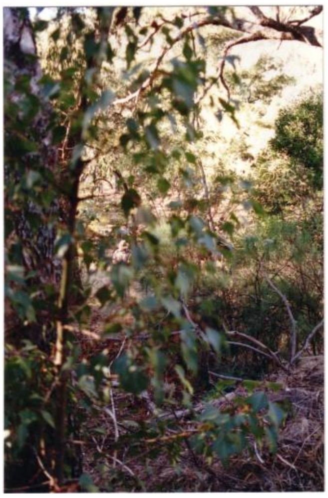
25/02/1997 18/11/2003 01/11/2012

Vegetation Type 4: *Eucalyptus bridgesiana - Eucalyptus melliodora* Open Forest

a) Example of recovery after low-moderate severity fire in 2003 = No Sites

b) Example of recovery after high-very high severity fire in 2003 = Site Brin_130

Pre-fire



12/05/1997

1 Year Post-fire

25/11/2003

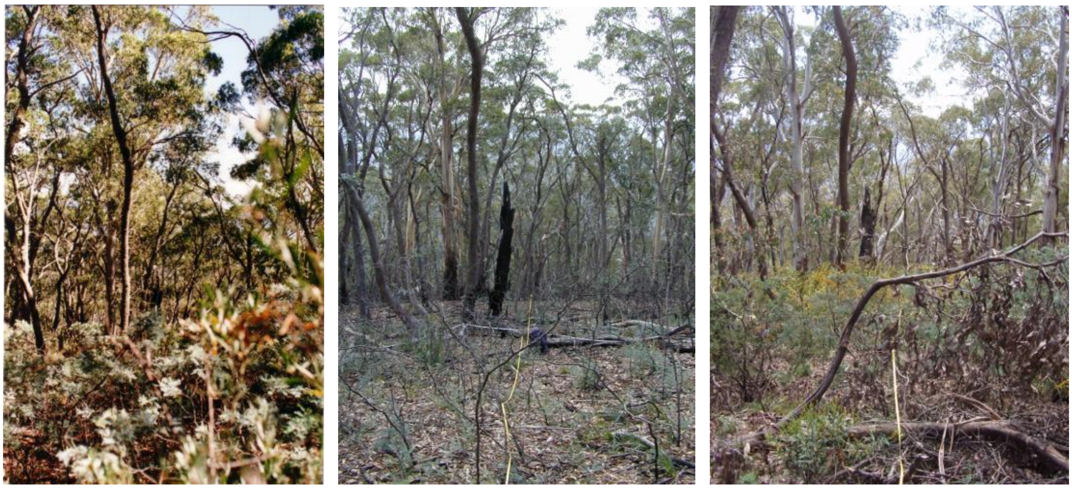
9 Years Post-fire

Not subsequently resampled

Vegetation Type 5: *Eucalyptus dalrympleana* - *Eucalyptus dives* (+/- *Eucalyptus pauciflora)* Open Forest

a) Example of recovery after low-moderate severity fire in 2003 = Site Brin_067

Pre-fire 1 Year Post-fire 9 Years Post-fire



21/03/1997 29/03/2004 15/11/2012

b) Example of recovery after high-very high severity fire in 2003 = Brin_Site 064

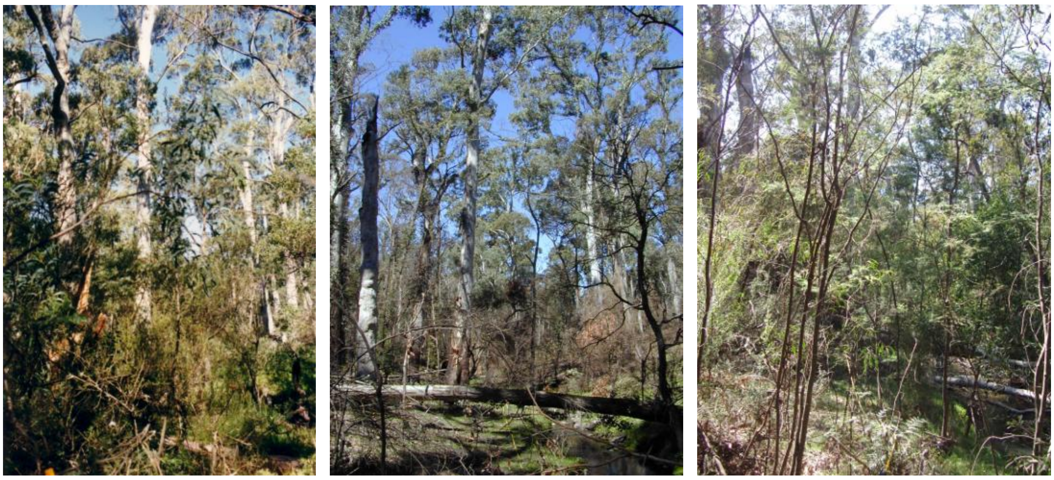
Pre-fire 1 Year Post-fire 9 Years Post-fire

19/03/1997 16/03/2004 02/11/2012

Vegetation Type 6: *Eucalyptus dalrympleana - Eucalyptus robertsonii* Tall Open Forest

a) Example of recovery after low-moderate severity fire in 2003 = Site Brin_006

Pre-fire 1 Year Post-fire 9 Years Post-fire



27/11/1996 03/11/2003 31/10/2012

b) Example of recovery after high-very high severity fire in 2003 = Site Brin_002

Pre-fire 1 Year Post-fire 9 Years Post-fire

27/11/1996 03/11/2003 31/10/2012

Vegetation Type 7: *Eucalyptus dalrympleana - Eucalyptus delegatensis* Tall Open Forest

a) Example of recovery after low-moderate severity fire in 2003 = Site Brin_068

Pre-fire 1 Year Post-fire 9 Years Post-fire

21/03/1997 29/03/2003 15/11/2012

b) Example of recovery after high-very high severity fire in 2003 = Site Brin_065

Pre-fire 1 Year Post-fire 9 Years Post-fire

19/03/1997 31/03/2004 02/11/2012

Vegetation Type 8: *Eucalyptus dalrympleana - Eucalyptus pauciflora* Open Forest

a) Example of recovery after <u>low-moderate severity</u> fire in 2003 = Site Brin_090

Pre-fire 1 Year Post-fire 9 Years Post-fire

27/03/1997 08/11/2004 15/11/2012

b) Example of recovery after <u>high-very high severity</u> fire in 2003 = Site Brin_091

Pre-fire 1 Year Post-fire 9 Years Post-fire

27/03/1997 04/11/2003 08/11/2012

Vegetation Type 9: *Eucalyptus camphora* subsp. *humeana* Open Forest

a) Example of recovery after low-moderate severity fire in 2003 = No Sites

b) Example of recovery after high-very high severity fire in 2003 = Site Brin_007

Pre-fire 1 Year Post-fire 9 Years Post-fire

27/11/1996 03/11/2003 31/10/2012

Vegetation Type 10: *Eucalyptus viminalis - Acacia melanoxylon* Open Forest

a) Example of recovery after <u>low-moderate severity</u> fire in 2003 = Site Brin_037

Pre-fire 1 Year Post-fire 9 Years Post-fire

18/02/1997 11/11/2003 09/11/2012

b) Example of recovery after <u>high-very high severity</u> fire in 2003 = Site Brin_056

Pre-fire 1 Year Post-fire 9 Years Post-fire

24/02/1997 10/11/2003 01/11/2012

Vegetation Type 11: *Casuarina cunninghamiana* Tall Open Forest

a) Example of recovery after low-moderate severity fire in 2003 = No Sites

b) Example of recovery after high-very high severity fire in 2003 = Site Brin_072

Pre-fire

24/03/1997

1 Year Post-fire

10/12/2003

9 Years Post-fire

13/11/2012

Vegetation Type 12: *Eucalyptus dalrympleana - Eucalyptus fastigata* Tall Open Forest

a) Example of recovery after low-moderate severity fire in 2003 = Site Brin_118

Pre-fire
1 Year Post-fire
9 Years Post-fire

11/04/1997
01/12/2003
08/11/2012

b) Example of recovery after high-very high severity fire in 2003 = No Sites

Vegetation Type 13: *Leptospermum micromyrtus* Montane Rocky Heath

a) Example of recovery after low-moderate severity fire in 2003 = No Sites

b) Example of recovery after high-very high severity fire in 2003 = Site Brin_022

Pre-fire | 1 Year Post-fire | 9 Years Post-fire

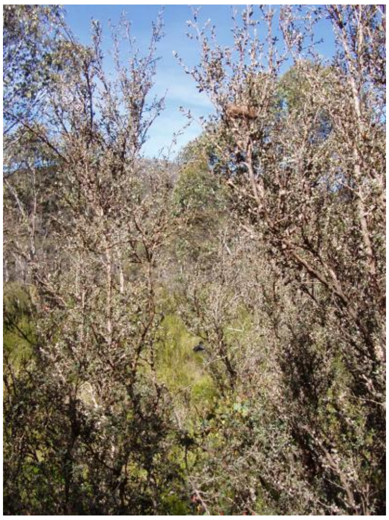
14/02/1997 | 04/11/2003 | 14/11/2012

Vegetation Type 14: *Leptospermum myrtifolium – Baeckea utilis* Swamp

a) Example of recovery after low-moderate severity fire in 2003 = No Sites

b) Example of recovery after high-very high severity fire in 2003 = Site Brin_026

Pre-fire 1 Year Post-fire 9 Years Post-fire



14/02/1997 06/01/2004 13/11/2012

Vegetation Type 15: *Eucalyptus bridgesiana* Open Forest

a) Example of recovery after low-moderate severity fire in 2003 = Site Bur_09

Pre-fire — 1 Year Post-fire — 9 Years Post-fire

25/03/1998 — 29/10/2003 — 05/11/2012

b) Example of recovery after high-very high severity fire in 2003 = Site Bur_05

Pre-fire — 1 Year Post-fire — 9 Years Post-fire

20/03/1998 — 30/10/2003 — 05/11/2012

Vegetation Type 16: *Eucalyptus bicostata – E. robertsonii – E. dalrympleana* Tall Open Forest

a) Example of recovery after low-moderate severity fire in 2003 = Site Bur_13

Pre-fire 1 Year Post-fire 9 Years Post-fire

25/03/1998 28/10/2003 06/11/2012

b) Example of recovery after high-very high severity fire in 2003 = Site Bur_24

Pre-fire 1 Year Post-fire 9 Years Post-fire

30/03/1998 27/10/2003 06/11/2012

Vegetation Type 17: *Eucalyptus bridgesiana – E. melliodora* Woodland / Open Forest

a) Example of recovery after low-moderate severity fire in 2003 = Site Bur_31

Pre-fire | 1 Year Post-fire | 9 Years Post-fire

31/03/1998 | 27/10/2003 | 06/11/2012

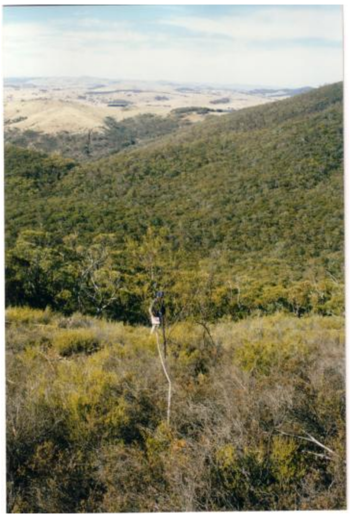
b) Example of recovery after high-very high severity fire in 2003 = No Sites

Vegetation Type 18: *Calytrix tetragona – Westringia eremicola* Rocky Outcrop Heath

a) Example of recovery after low-moderate severity fire in 2003 = No Sites

b) Example of recovery after high-very high severity fire in 2003 = Site Bur_27

Pre-fire



30/03/1998

1 Year Post-fire

23/10/2003

9 Years Post-fire

06/11/2012

Vegetation Type 19: *Eucalyptus goniocalyx – E. macrorhyncha* Open Forest

a) Example of recovery after low-moderate severity fire in 2003 = Site Bur_21

Pre-fire 1 Year Post-fire 9 Years Post-fire

30/03/1998 30/10/2003 06/11/2012

b) Example of recovery after high-very high severity fire in 2003 = Site Bur_06

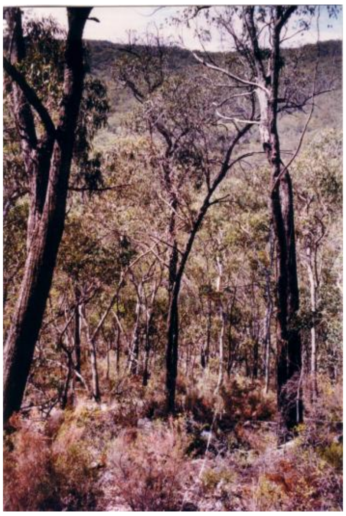
Pre-fire 1 Year Post-fire 9 Years Post-fire

20/03/1998 30/10/2003 05/11/2012

Vegetation Type 20: *Eucalyptus dives – E. macrorhyncha – E. mannifera* Open Forest

a) Example of recovery after low-moderate severity fire in 2003 = Site Bur_10

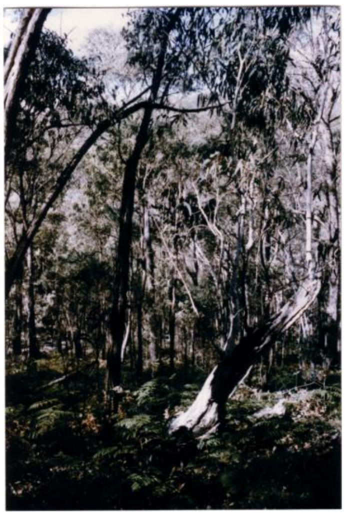
Pre-fire 1 Year Post-fire 9 Years Post-fire



25/03/1998 29/10/2003 05/11/2012

b) Example of recovery after high-very high severity fire in 2003 = Site Bur_19

Pre-fire 1 Year Post-fire 9 Years Post-fire



26/03/1998 28/10/2003 06/11/2012

Vegetation Type 21: *Eucalyptus rossii* Low Open Forest

a) Example of recovery after low-moderate severity fire in 2003 = Site Bur_11

Pre-fire 1 Year Post-fire 9 Years Post-fire

25/03/1998 29/10/2003 05/11/2012

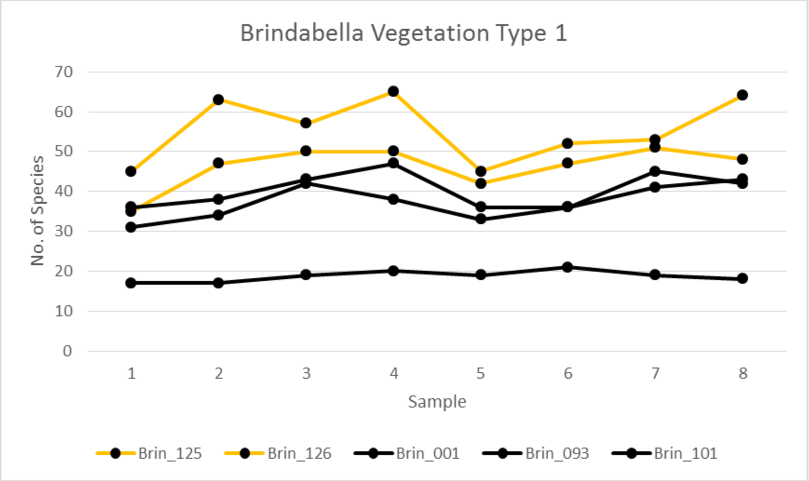
b) Example of recovery after high-very high severity fire in 2003 = No Sites

Appendix 5: Changes in Plant Species Richness Over Time for Ongoing Sampling of 52 Sites From 1996 to 2012 in the Brindabella and Burrinjuck Study Areas, in Relation to Fire Severity Category

Vegetation types are sequenced from 1 to 21 across both Brindabella (14 types) and Burrinjuck (7 Types). See Table 3.1 and Appendix 2 for descriptions of vegetation types. Note that some communities did not experience both low and high severity fire and this is noted for each diagram.

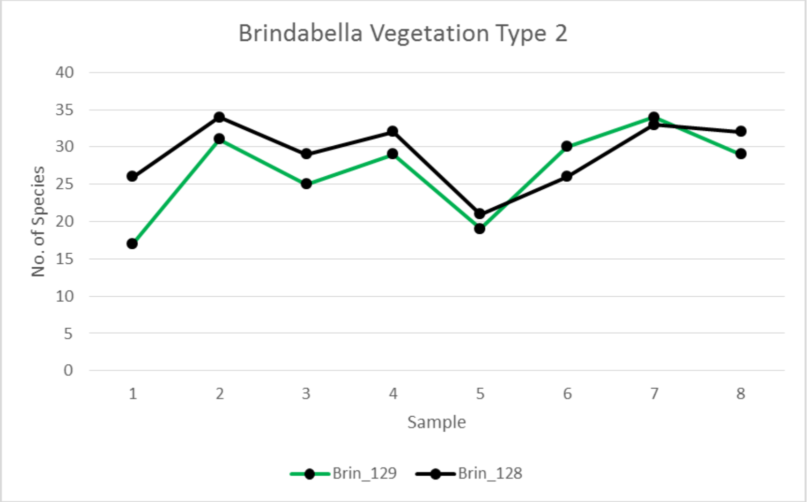
Vegetation Type 1: *Eucalyptus dives - Eucalyptus macrorhyncha - Eucalyptus mannifera* (+/- *Eucalyptus rubida*) Open Forest (Species Rich)

Brin_125/126 = Low-Moderate Severity; Brin_001/093/101 = High-Very High Severity



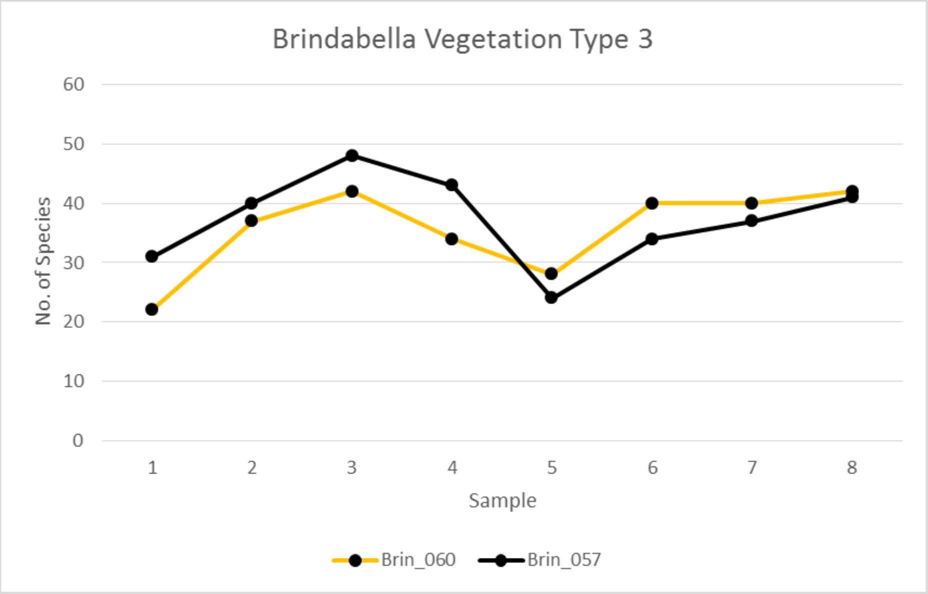
Vegetation Type 2: *Eucalyptus macrorhyncha - Eucalyptus nortonii* Low Open Forest

Brin_129 = Unburnt; Brin_128 = High-Very High Severity



Vegetation Type 3: *Eucalyptus dives - Eucalyptus mannifera - Eucalyptus macrorhyncha* Open Forest (Species Poor)

Brin_060 = Low-Moderate Severity; Brin_057 = High-Very High Severity

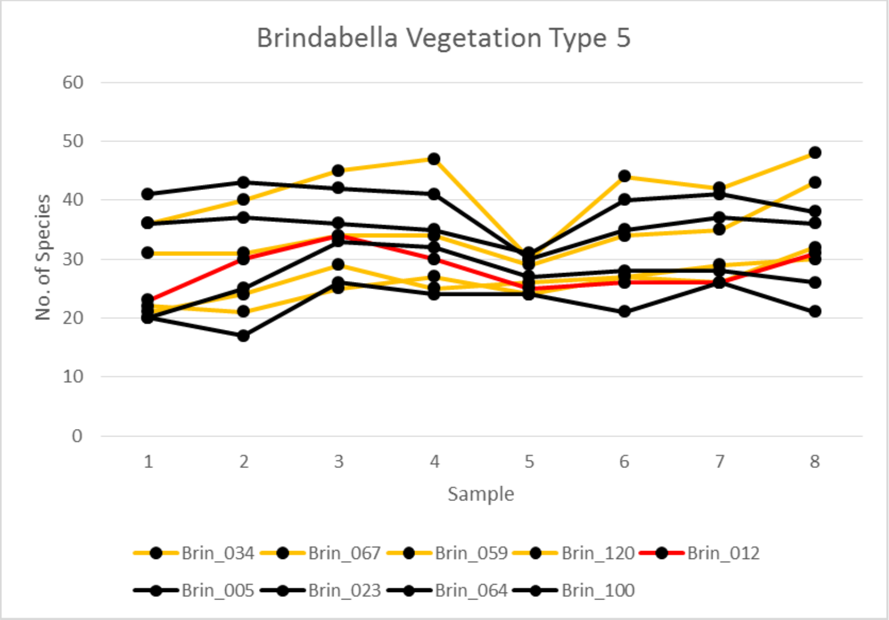


Vegetation Type 4: *Eucalyptus bridgesiana - Eucalyptus melliodora* Open Forest

Not subsequently sampled after the initial post-fire sample in 2003

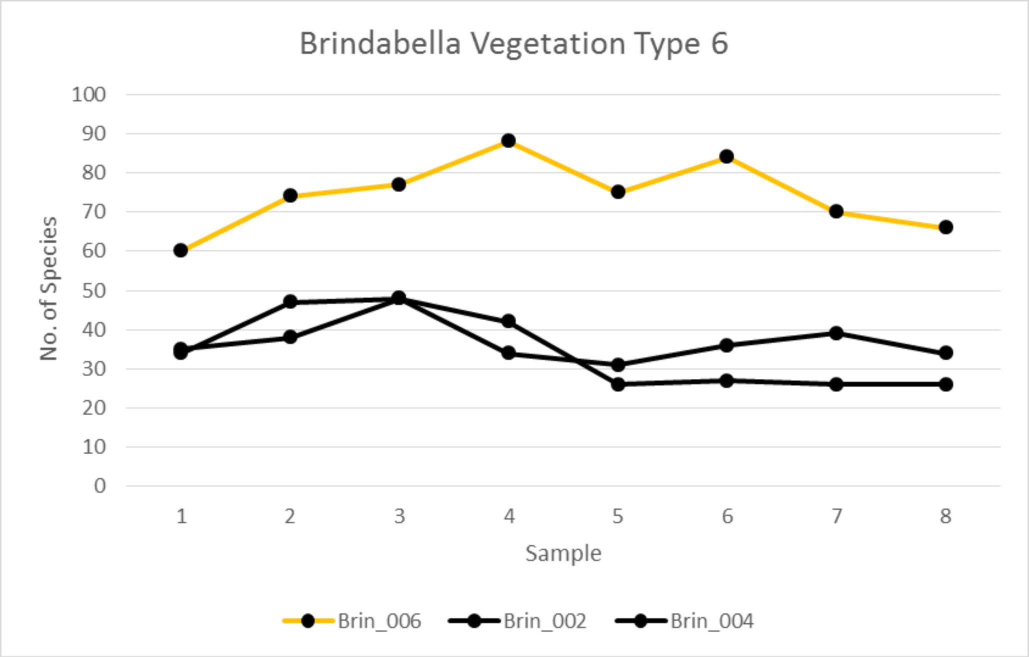
Vegetation Type 5: *Eucalyptus dalrympleana - Eucalyptus dives* (+/- *Eucalyptus pauciflora)* Open Forest

Brin_034/067/059/120/ = Low-Moderate Severity; Brin_012 = Moderate Severity;
Brin_005/023/064/100 = High-Very High Severity



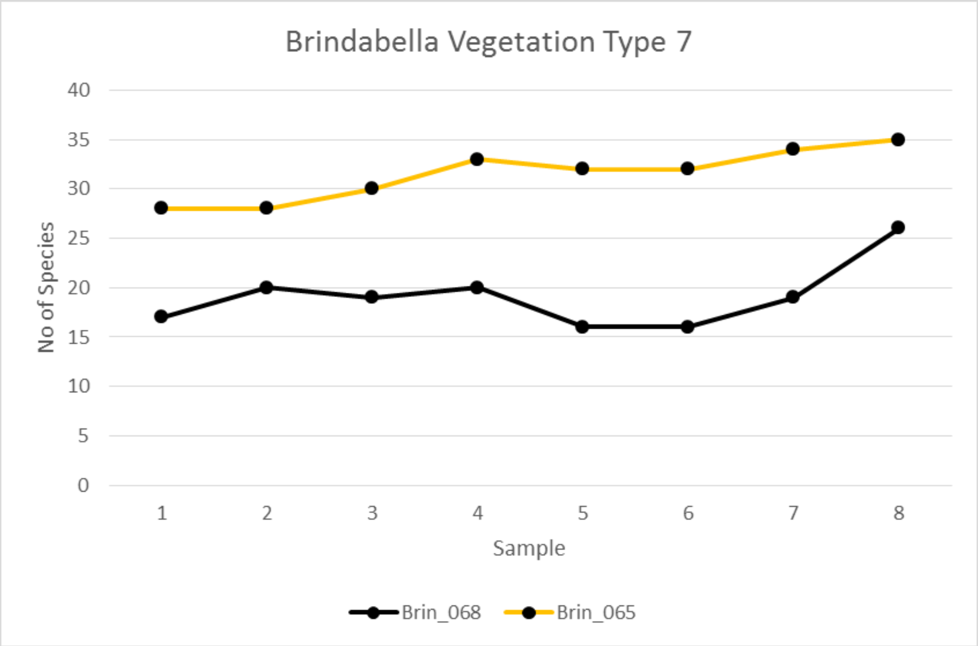
Vegetation Type 6: *Eucalyptus dalrympleana - Eucalyptus robertsonii* Tall Open Forest

Brin_006 = Low-Moderate Severity; Brin_002/004 = High-Very High Severity



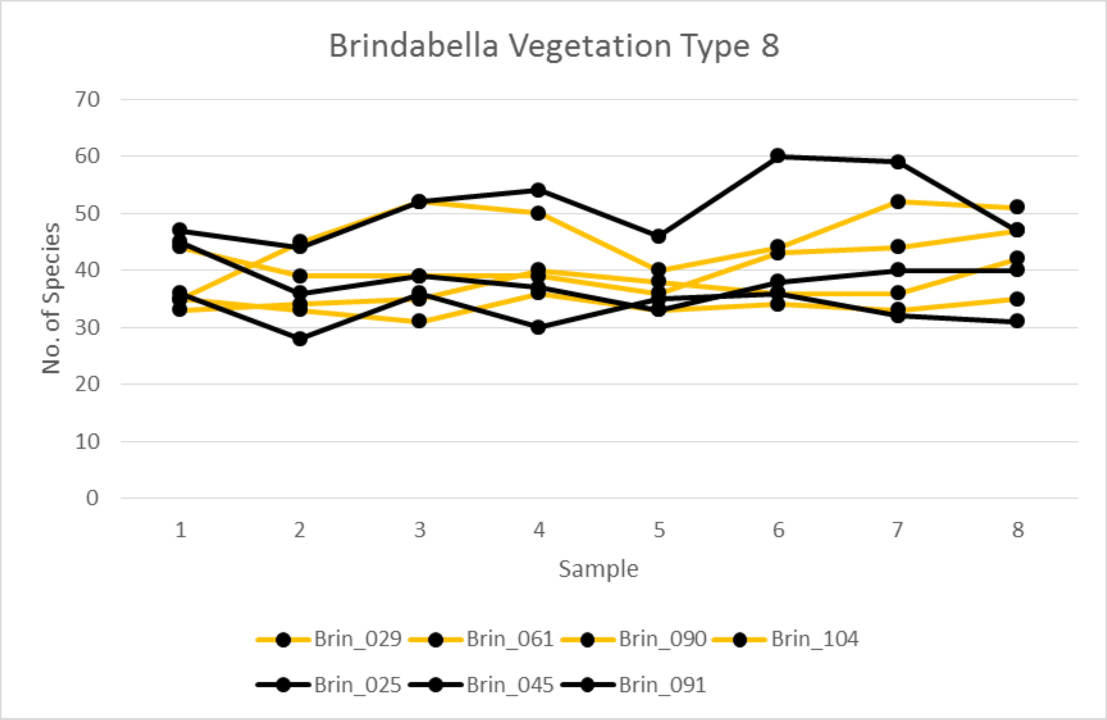
Vegetation Type 7: *Eucalyptus dalrympleana - Eucalyptus delegatensis* Tall Open Forest

Brin_068 = Low-Moderate Severity; Brin_065 = High-Very High Severity



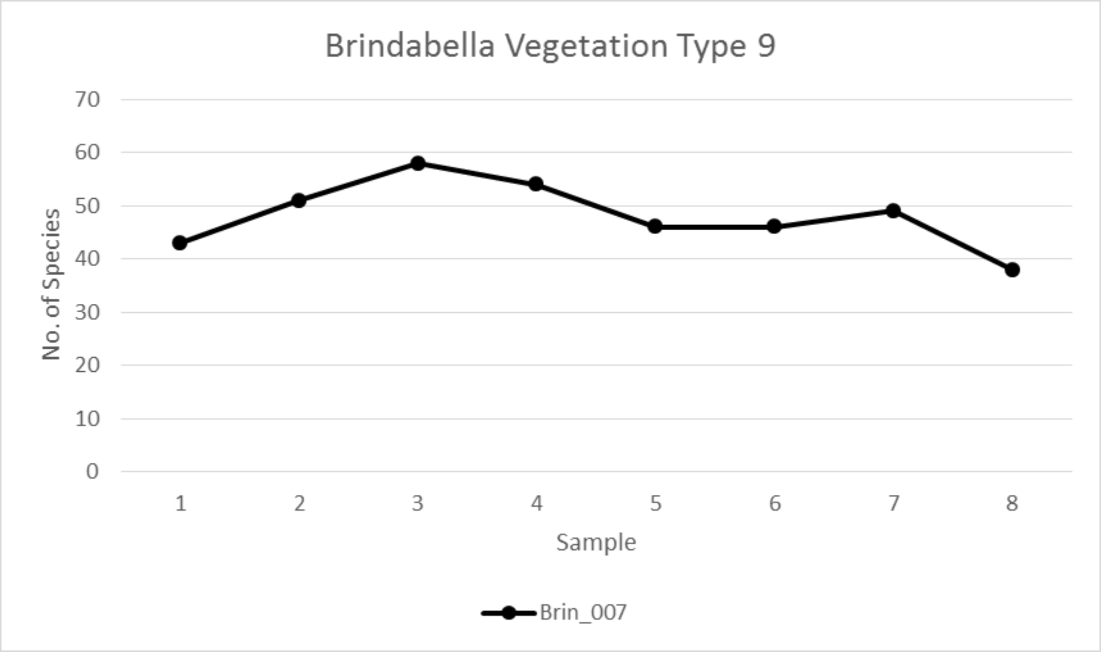
Vegetation Type 8: *Eucalyptus dalrympleana - Eucalyptus pauciflora* Open Forest

Brin_029/061/090/104 = Low-Moderate Severity; Brin_025/045/091 = High-Very High Severity



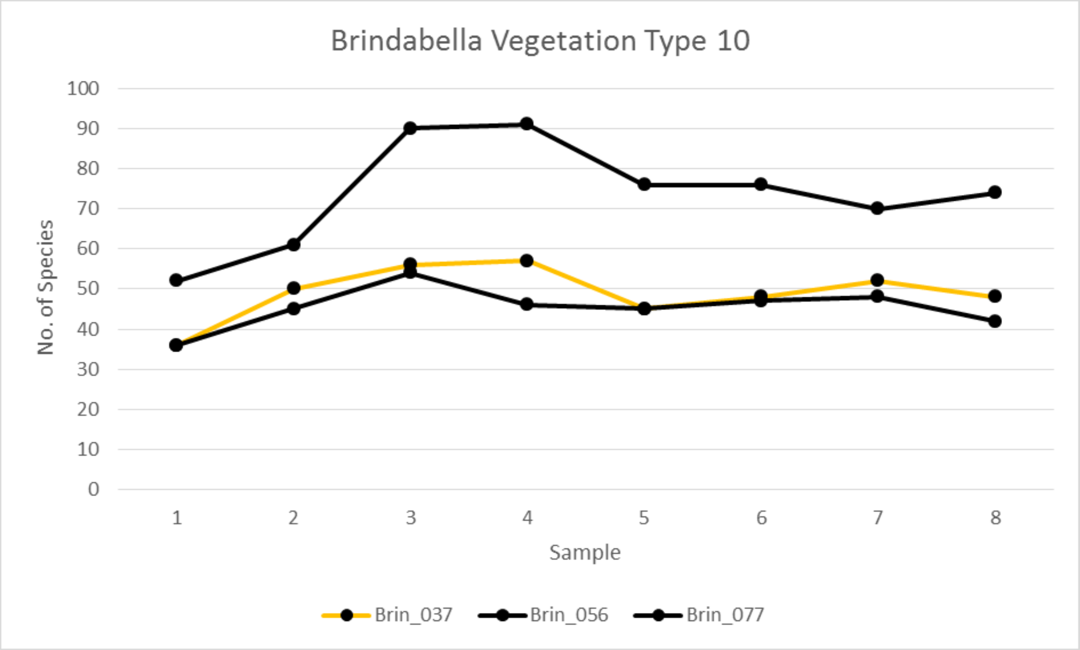
Vegetation Type 9: *Eucalyptus camphora* subsp. *humeana* Open Forest

Brin_007 = High-Very High Severity



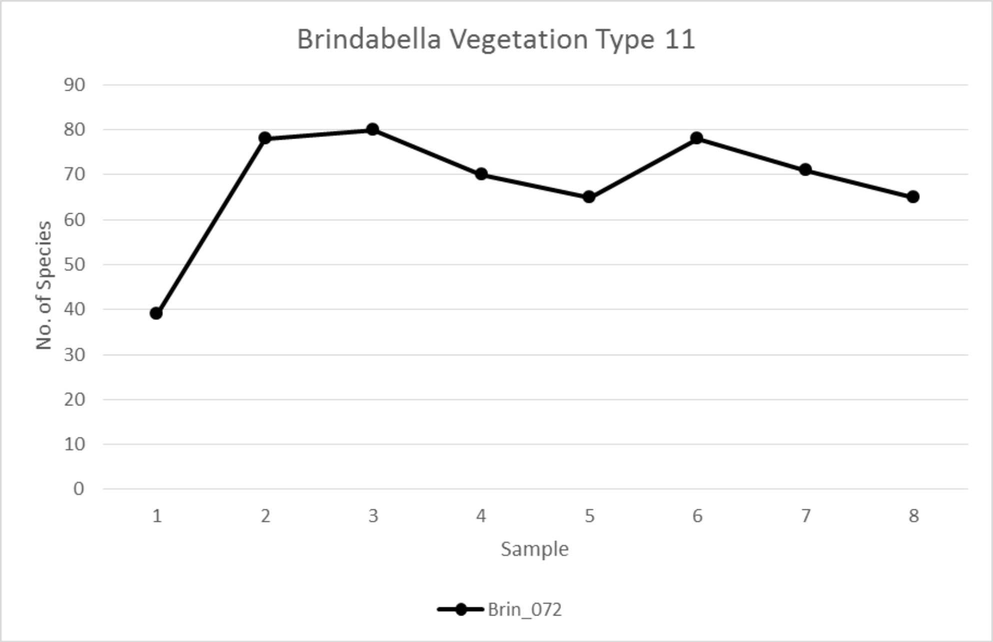
Vegetation Type 10: *Eucalyptus viminalis - Acacia melanoxylon* Open Forest

Brin_037 = Low-Moderate Severity; Brin_056/077 = High-Very High Severity



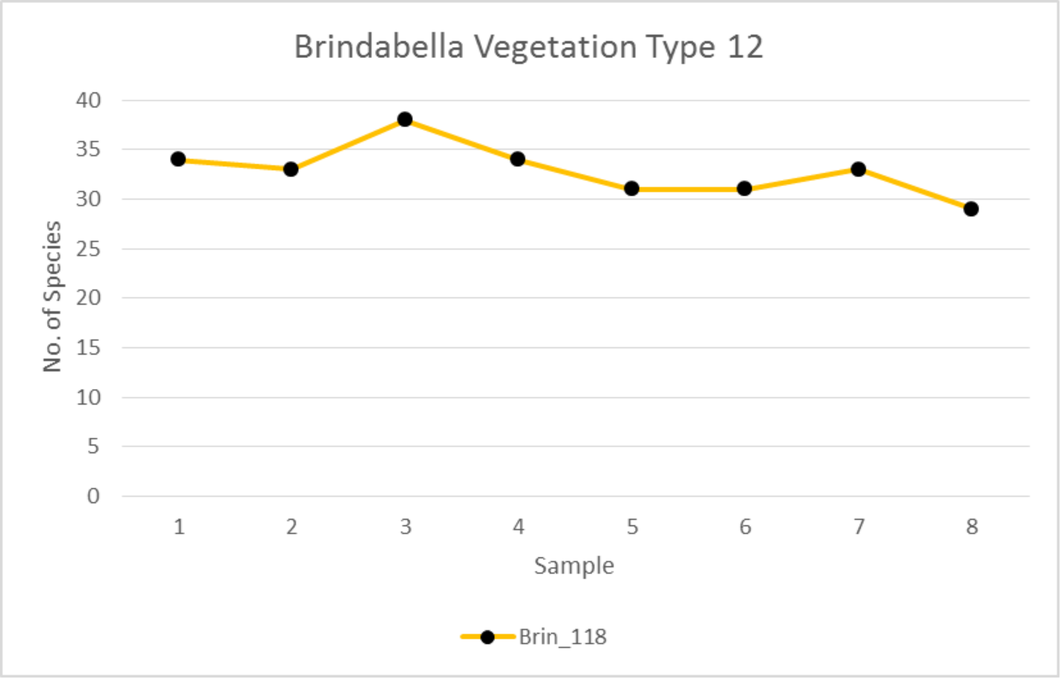
Vegetation Type 11: *Casuarina cunninghamiana* Tall Open Forest

Brin_072 = High-Very High Severity



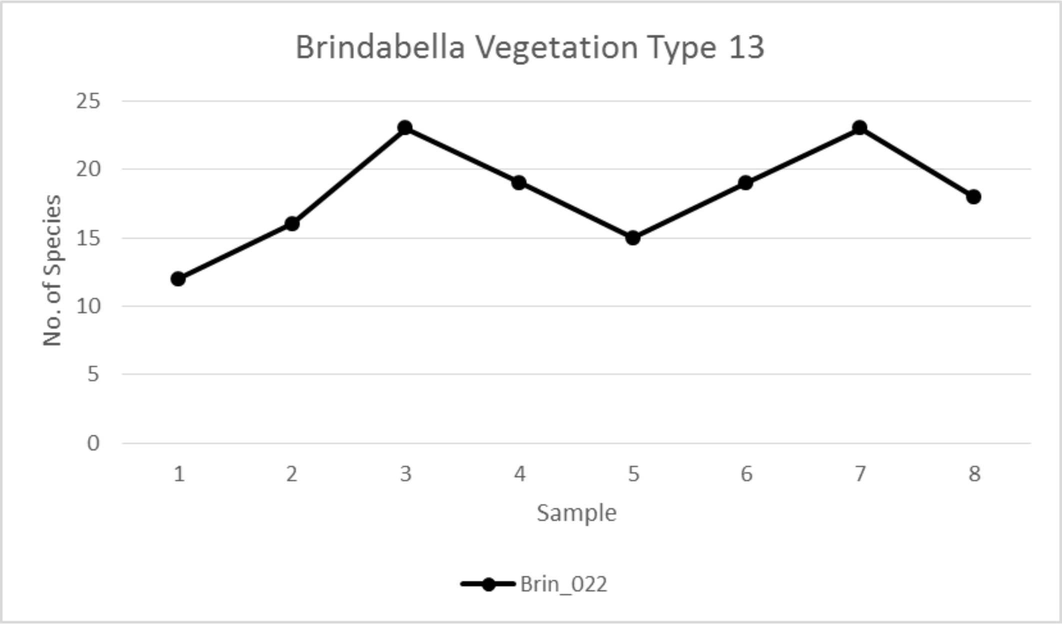
Vegetation Type 12: *Eucalyptus dalrympleana - Eucalyptus fastigata* Tall Open Forest

Brin_118 = Low-Moderate Severity



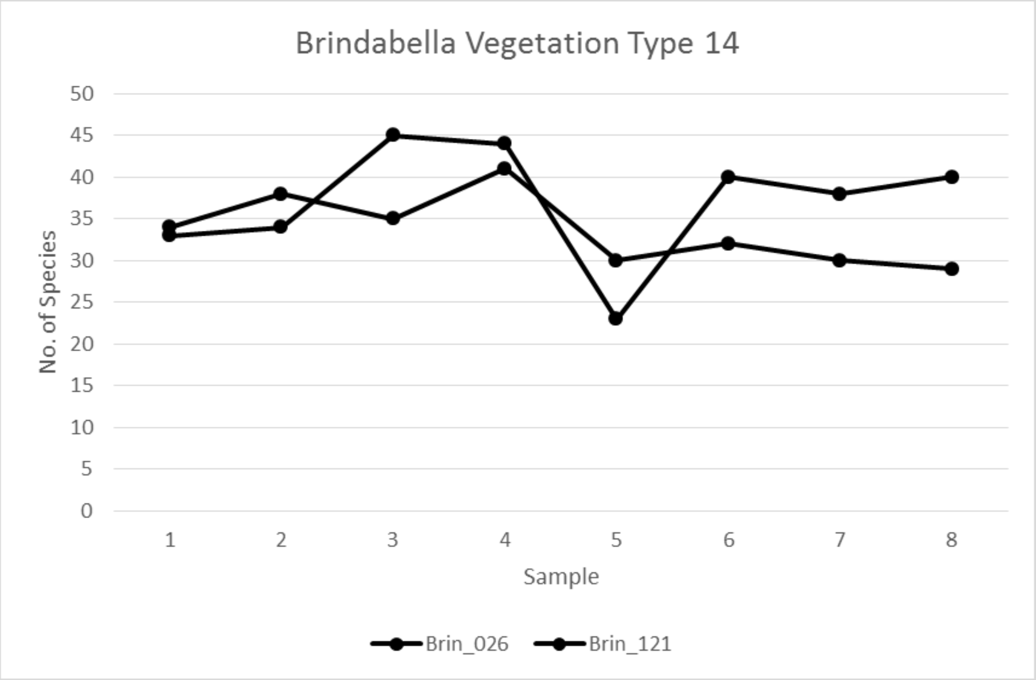
Vegetation Type 13: *Leptospermum micromyrtus* Montane Rocky Heath

Brin_022 = High-Very High Severity



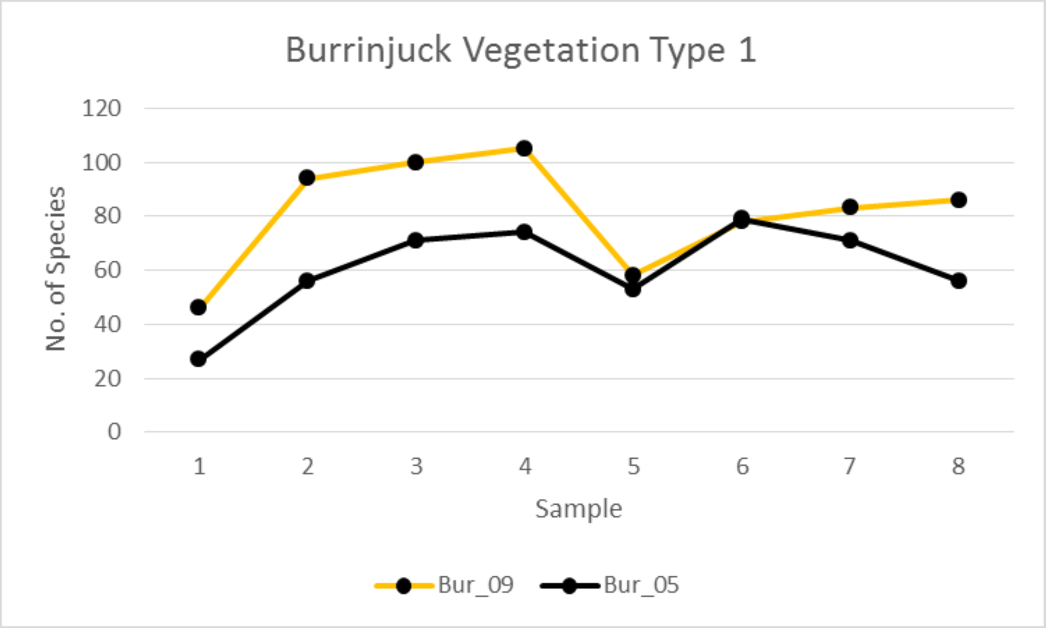
Vegetation Type 14: *Leptospermum myrtifolium – Baeckea utilis* Swamp

Brin_026/121 = High-Very High Severity



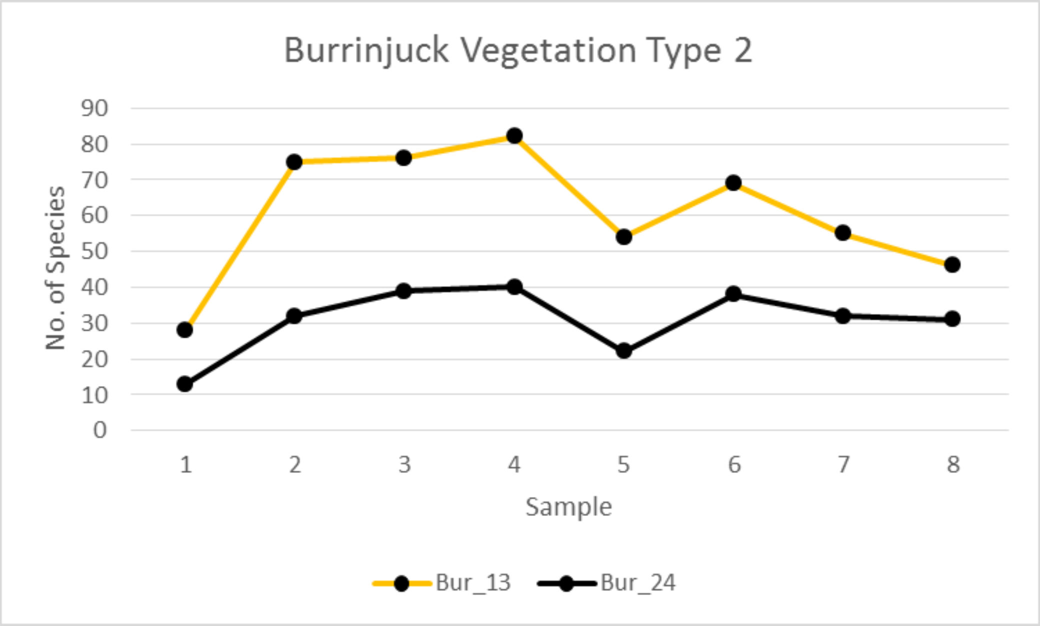
Vegetation Type 15: *Eucalyptus bridgesiana* Open Forest

Bur_09 = Low-Moderate Severity; Bur_05 = High-Very High Severity



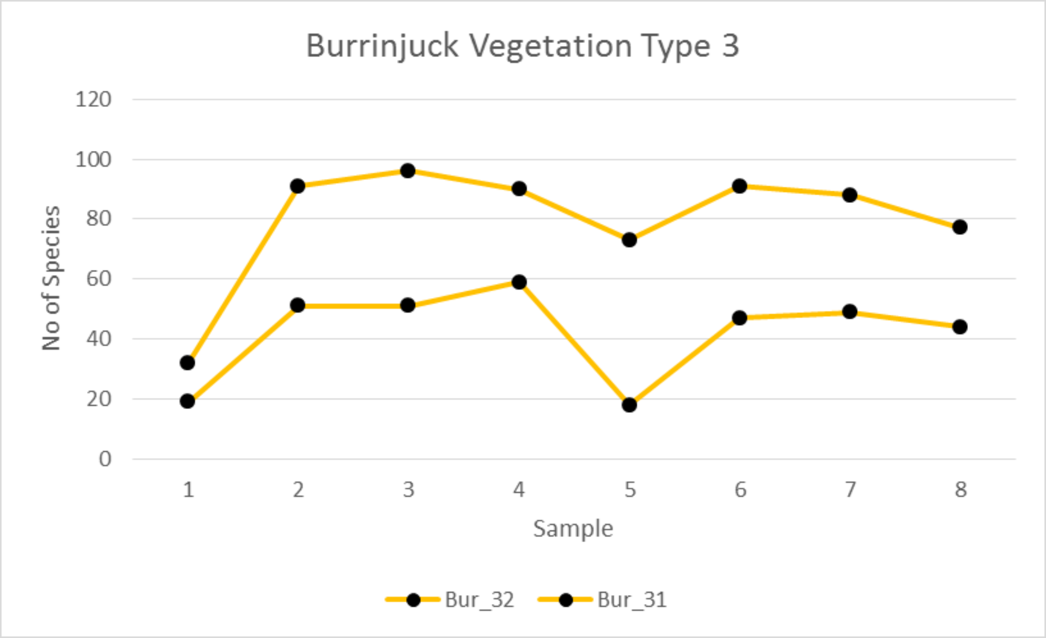
Vegetation Type 16: *Eucalyptus bicostata – E. robertsonii – E. dalrympleana* Tall Open Forest

Bur_13 = Low-Moderate Severity; Bur_24 = High-Very High Severity



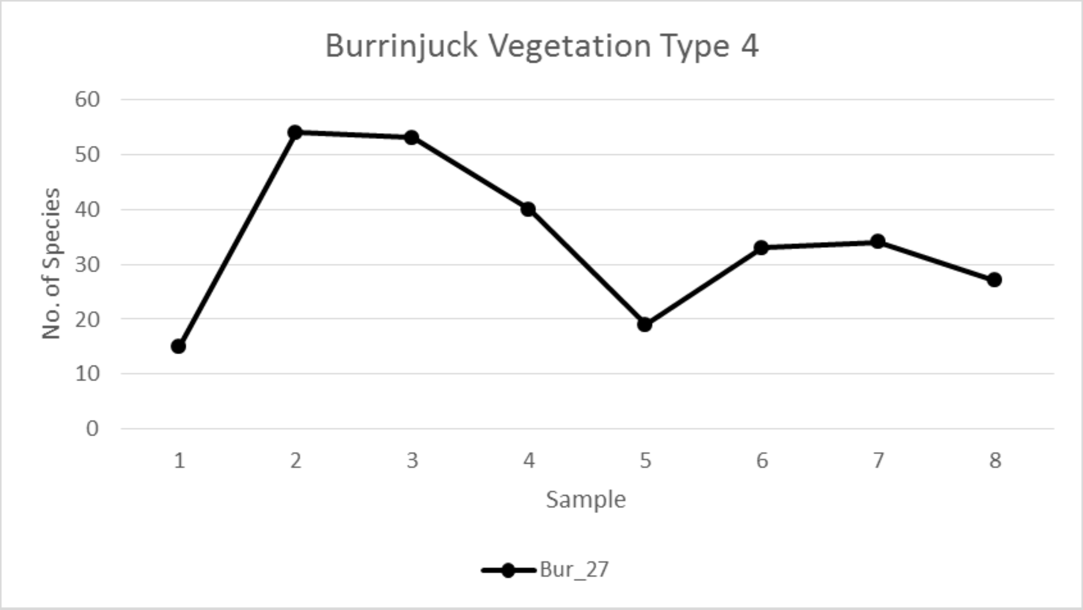
Vegetation Type 17: *Eucalyptus bridgesiana – E. melliodora* Woodland / Open Forest

Bur_32/31 = Low-Moderate Severity



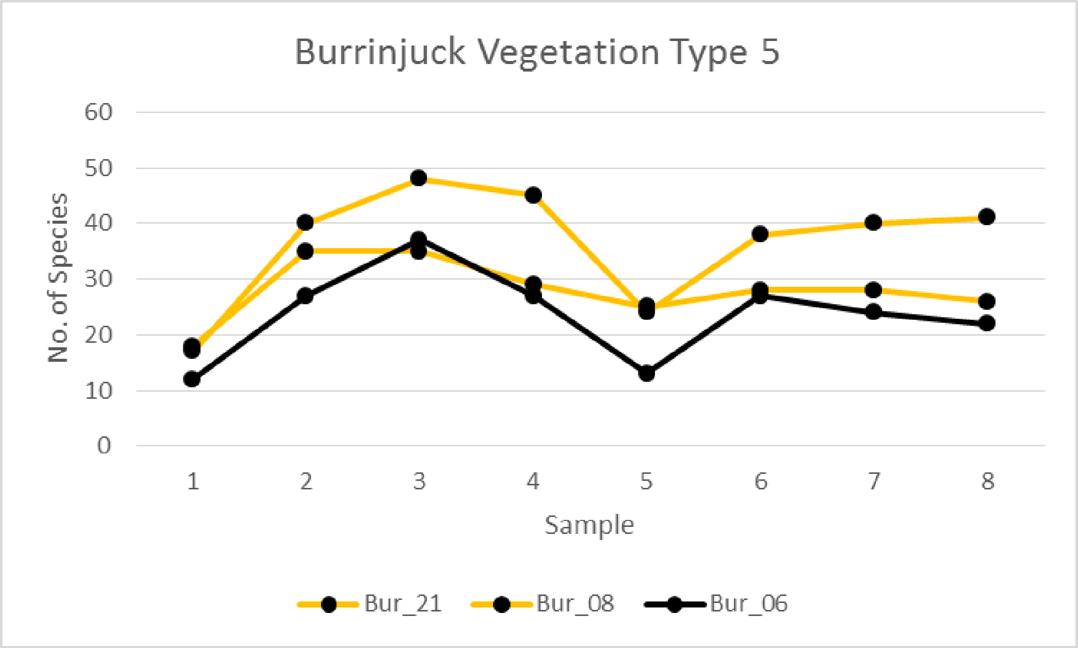
Vegetation Type 18: *Calytrix tetragona – Westringia eremicola* Rocky Outcrop Heath

Bur_27 = High-Very High Severity



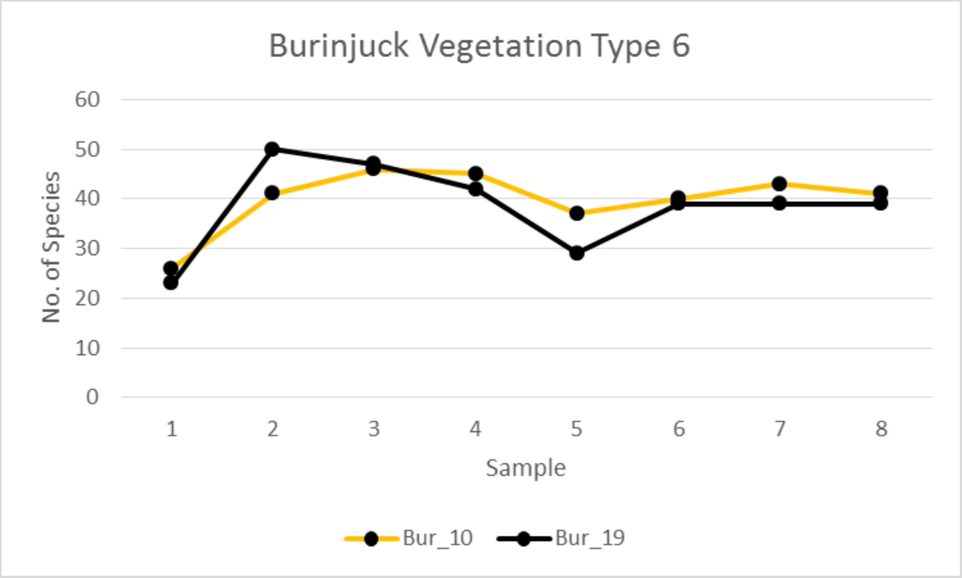
Vegetation Type 19: *Eucalyptus goniocalyx – E. macrorhyncha* Open Forest

Bur_21/08 = Low-Moderate Severity; Bur_06 = High-Very High Severity



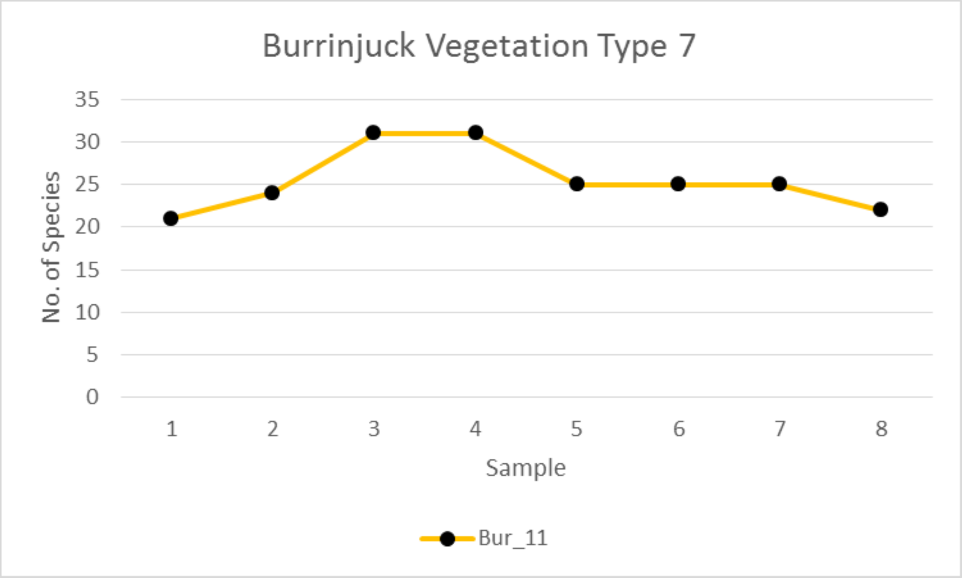
Vegetation Type 20: *Eucalyptus dives – E. macrorhyncha – E. mannifera* Open Forest

Bur_10 = Low-Moderate Severity; Bur_19 = High-Very High Severity



Vegetation Type 21: *Eucalyptus rossii* Low Open Forest

Bur_11 = Low-Moderate Severity



Appendix 6: Brindabella GLMM Runs

a) First Model Run: Brindabella All Terms Using Overstorey and Understorey Cover

Sequentially adding terms to fixed model

Fixed term	Wald statistic	n.d.f.	F statistic	d.d.f.	F pr
Habitat_Complexity	3.22	1	3.22	399.1	0.073
Number_of_Strata	1.72	1	1.72	409.0	0.191
Overstorey_Cover_log10_1	41.65	1	41.65	396.0	<0.001
Overstorey_Height 1	2.60	1	12.60	450.1	<0.001
Rainfall_log10	57.06	1	57.06	352.1	<0.001
Sample_Season	7.44	2	3.72	403.0	0.025
Scorch_Prop	8.64	1	8.64	443.5	0.003
Time_Since_Fire_log10	36.17	1	36.17	421.6	<0.001
Understorey_Cover_log10_	112.39	1	12.39	418.9	<0.001
Vegetation_Type	85.99	13	6.61	102.9	<0.001

Dropping individual terms from full fixed model

Fixed term	Wald statistic	n.d.f.	F statistic	d.d.f.	F pr
Habitat_Complexity	8.38	1	8.38	399.1	0.004
Number_of_Strata	0.88	1	0.88	409.0	0.348
Overstorey_Cover_log10_1	1.29	1	1.29	396.0	0.256
Overstorey_Height	0.95	1	0.95	450.1	0.330
Rainfall_log10	30.92	1	30.92	352.1	<0.001
Sample_Season	4.60	2	2.30	403.0	0.101
Scorch_Prop	0.99	1	0.99	443.5	0.320
Time_Since_Fire_log10	34.81	1	34.81	421.6	<0.001
Understorey_Cover_log10_1	8.55	1	8.55	418.9	0.004
Vegetation_Type	85.99	13	6.61	102.9	<0.001

b) Second Model Run: Brindabella All terms using Total Cover

Sequentially adding terms to fixed model

Fixed term	Wald statistic	n.d.f.	F statistic	d.d.f.	F pr
Habitat_Complexity	2.76	1	2.76	396.8	0.098
Number_of_Strata	1.70	1	1.70	410.7	0.193
Overstorey_Height	6.22	1	6.22	454.4	0.013
Rainfall_log10	72.55	1	72.55	352.9	<0.001
Sample_Season	16.69	2	8.34	402.9	<0.001
Scorch_Prop	23.92	1	23.92	447.0	<0.001
Time_Since_Fire_log10	37.33	1	37.33	420.7	<0.001
Total_Cover_log10_1	1.87	1	1.87	400.8	0.172
Vegetation_Type	86.19	13	6.63	103.0	<0.001

Dropping individual terms from full fixed model

Fixed term	Wald statistic	n.d.f.	F statistic	d.d.f.	F pr
Habitat_Complexity	9.00	1	9.00	396.8	0.003
Number_of_Strata	0.04	1	0.04	410.7	0.850
Overstorey_Height	0.24	1	0.24	454.4	0.628
Rainfall_log10	29.24	1	29.24	352.9	<0.001
Sample_Season	4.65	2	2.33	402.9	0.099
Scorch_Prop	0.05	1	0.05	447.0	0.828
Time_Since_Fire_log10	31.85	1	31.85	420.7	<0.001
Total_Cover_log10_1	1.31	1	1.31	400.8	0.254
Vegetation_Type	86.19	13	6.63	103.0	<0.001

c) Third Model Run: Brindabella Using Only Significant Terms

Sequentially adding terms to fixed model

Fixed term	Wald statistic	n.d.f.	F statistic	d.d.f.	F pr
Vegetation_Type	107.59	13	8.28	100.8	<0.001
Time_Since_Fire_log10	83.67	1	83.67	377.1	<0.001
Rainfall_log10	43.53	1	43.53	352.9	<0.001
Habitat_Complexity	15.04	1	15.04	401.8	<0.001
Understorey_Cover_log10_1	10.40	1	10.40	435.1	0.001

Dropping individual terms from full fixed model

Fixed term	Wald statistic	n.d.f.	F statistic	d.d.f.	F pr
Vegetation_Type	83.75	13	6.44	100.8	<0.001
Time_Since_Fire_log10	81.00	1	81.00	377.1	<0.001
Rainfall_log10	39.26	1	39.26	352.9	<0.001
Habitat_Complexity	7.34	1	7.34	401.8	0.007
Understorey_Cover_log10_1	10.40	1	10.40	435.1	0.001

Appendix 7: Burrinjuck GLMM Runs

a) First Model Run: Burrinjuck All Terms Using Overstorey and Understorey Cover

Sequentially adding terms to fixed model

Fixed term	Wald statistic	n.d.f.	F statistic	d.d.f.	F pr
Habitat_Complexity	0.93	1	0.93	125.5	0.336
Number_of_Strata	13.92	1	13.92	124.3	<0.001
Overstorey_Cover_log10_1	77.52	1	77.52	125.2	<0.001
Overstorey_Height	40.05	1	40.05	61.1	<0.001
Rainfall_log10	92.23	1	92.23	107.7	<0.001
Sample_Season	131.36	1	131.36	127.9	<0.001
Scorch_Prop	1.16	1	1.16	110.2	0.285
Time_Since_Fire_log10	14.64	1	14.64	120.8	<0.001
Understorey_Cover_log10_1	4.57	1	4.57	119.2	0.035
Vegetation_Type	17.19	6	2.86	22.6	0.032

Dropping individual terms from full fixed model

Fixed term	Wald statistic	n.d.f.	F statistic	d.d.f.	F pr
Habitat_Complexity	18.11	1	18.11	125.5	<0.001
Number_of_Strata	0.00	1	0.00	124.3	0.995
Overstorey_Cover_log10_1	0.68	1	0.68	125.2	0.412
Overstorey_Height	1.58	1	1.58	61.1	0.213
Rainfall_log10	19.80	1	19.80	107.7	<0.001
Sample_Season	28.39	1	28.39	127.9	<0.001
Scorch_Prop	0.93	1	0.93	110.2	0.337
Time_Since_Fire_log10	10.12	1	10.12	120.8	0.002
Understorey_Cover_log10_1	3.75	1	3.75	119.2	0.055
Vegetation_Type	17.19	6	2.86	22.6	0.032

b) Second Model Run: Burrinjuck All Terms Using Total Cover

Sequentially adding terms to fixed model

Fixed term	Wald statistic	n.d.f.	F statistic	d.d.f.	F pr
Habitat_Complexity	1.28	1	1.28	122.6	0.261
Number_of_Strata	14.29	1	14.29	125.5	<0.001
Overstorey_Height	48.68	1	48.68	66.7	<0.001
Rainfall_log10	121.57	1	121.57	109.4	<0.001
Sample_Season	166.08	1	166.08	129.0	<0.001
Scorch_Prop	0.46	1	0.46	99.5	0.497
Time_Since_Fire_log10	15.10	1	15.10	118.6	<0.001
Total_Cover_log10_1	0.51	1	0.51	119.9	0.477
Vegetation_Type	16.97	6	2.83	23.9	0.032

Dropping individual terms from full fixed model

Fixed term	Wald statistic	n.d.f.	F statistic	d.d.f.	F pr
Habitat_Complexity	14.17	1	14.17	122.6	<0.001
Number_of_Strata	0.23	1	0.23	125.5	0.631
Overstorey_Height	1.51	1	1.51	66.7	0.224
Rainfall_log10	21.37	1	21.37	109.4	<0.001
Sample_Season	25.29	1	25.29	129.0	<0.001
Scorch_Prop	0.51	1	0.51	99.5	0.478
Time_Since_Fire_log10	13.30	1	13.30	118.6	<0.001
Total_Cover_log10_1	0.55	1	0.55	119.9	0.460
Vegetation_Type	16.97	6	2.83	23.9	0.032

<u>c) Third Model Run</u>: Burrinjuck Using Only Significant Terms

Sequentially adding terms to fixed model

Fixed term	Wald statistic	n.d.f.	F statistic	d.d.f.	F pr
Sample_Season	288.52	1	288.52	124.8	<0.001
Habitat_Complexity	23.46	1	23.46	137.1	<0.001
Time_Since_Fire_log10	22.51	1	22.51	116.8	<0.001
Rainfall_log10	18.93	1	18.93	111.7	<0.001

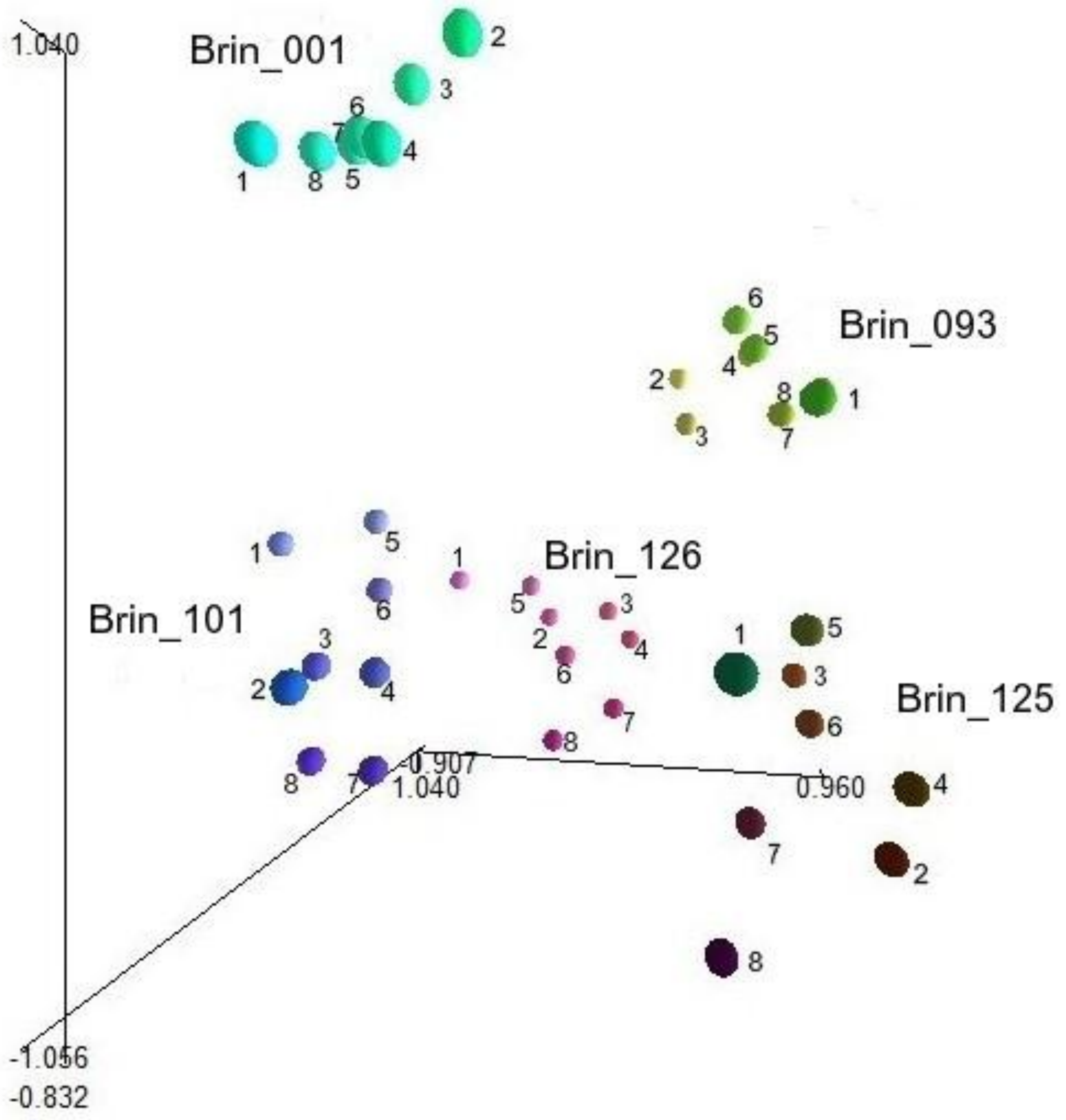
Dropping individual terms from full fixed model

Fixed term	Wald statistic	n.d.f.	F statistic	d.d.f.	F pr
Sample_Season	39.09	1	39.09	124.8	<0.001
Habitat_Complexity	21.13	1	21.13	137.1	<0.001
Time_Since_Fire_log10	28.20	1	28.20	116.8	<0.001
Rainfall_log10	18.93	1	18.93	111.7	<0.001

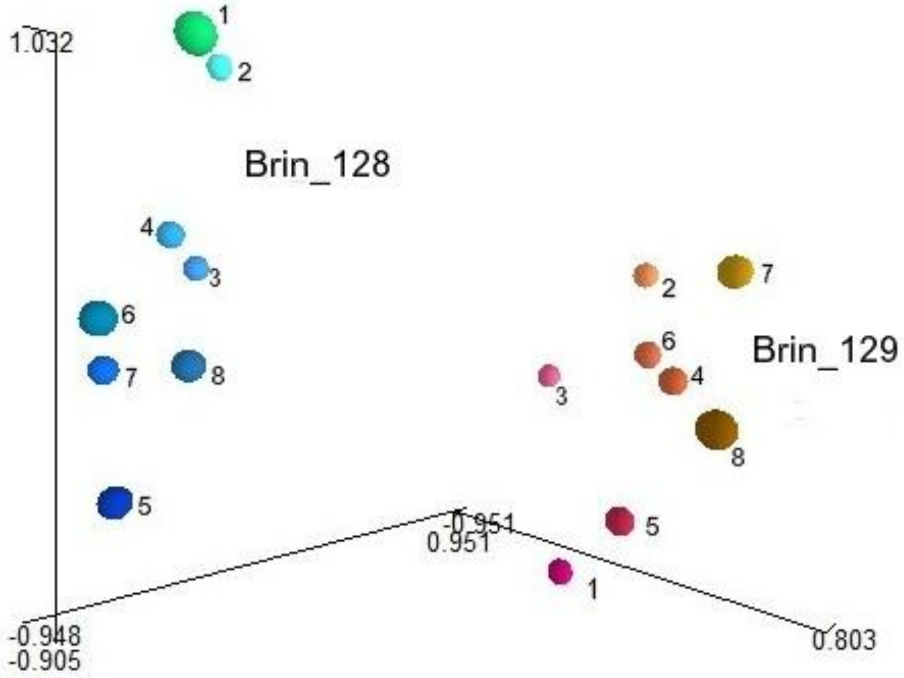
Appendix 8: SSH Ordination Site-Time Trajectory Diagrams for Ongoing Sampling of 52 Sites From 1996 to 2012 in the Brindabella and Burrinjuck Study Areas, by Vegetation Type

Site clusters labelled as either ‘Brin_XXX’ or ‘Burr_XX’. Time samples numbered from 1 to 8 and in similar colour tones. The scale on the ordination axes is derived from standardized association measure distances. The ranges appear as thin black lines and the range values are displayed at each end.

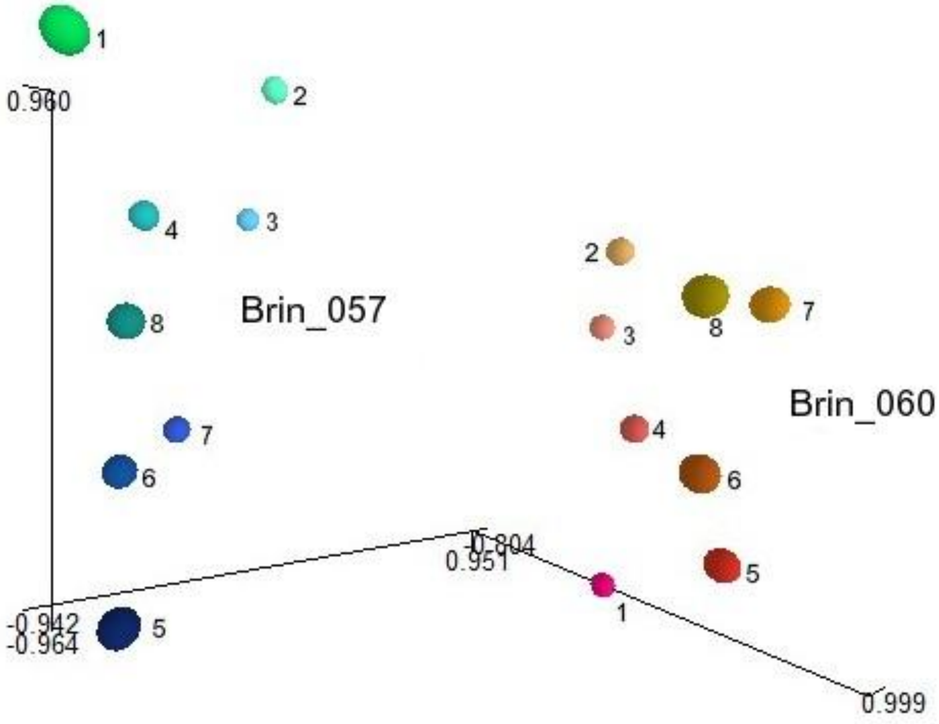
Vegetation Type 1: *Eucalyptus dives - Eucalyptus macrorhyncha - Eucalyptus mannifera* (+/- *Eucalyptus rubida*) Open Forest (Species Rich)



Vegetation Type 2: ***Eucalyptus macrorhyncha - Eucalyptus nortonii*** **Low Open Forest**



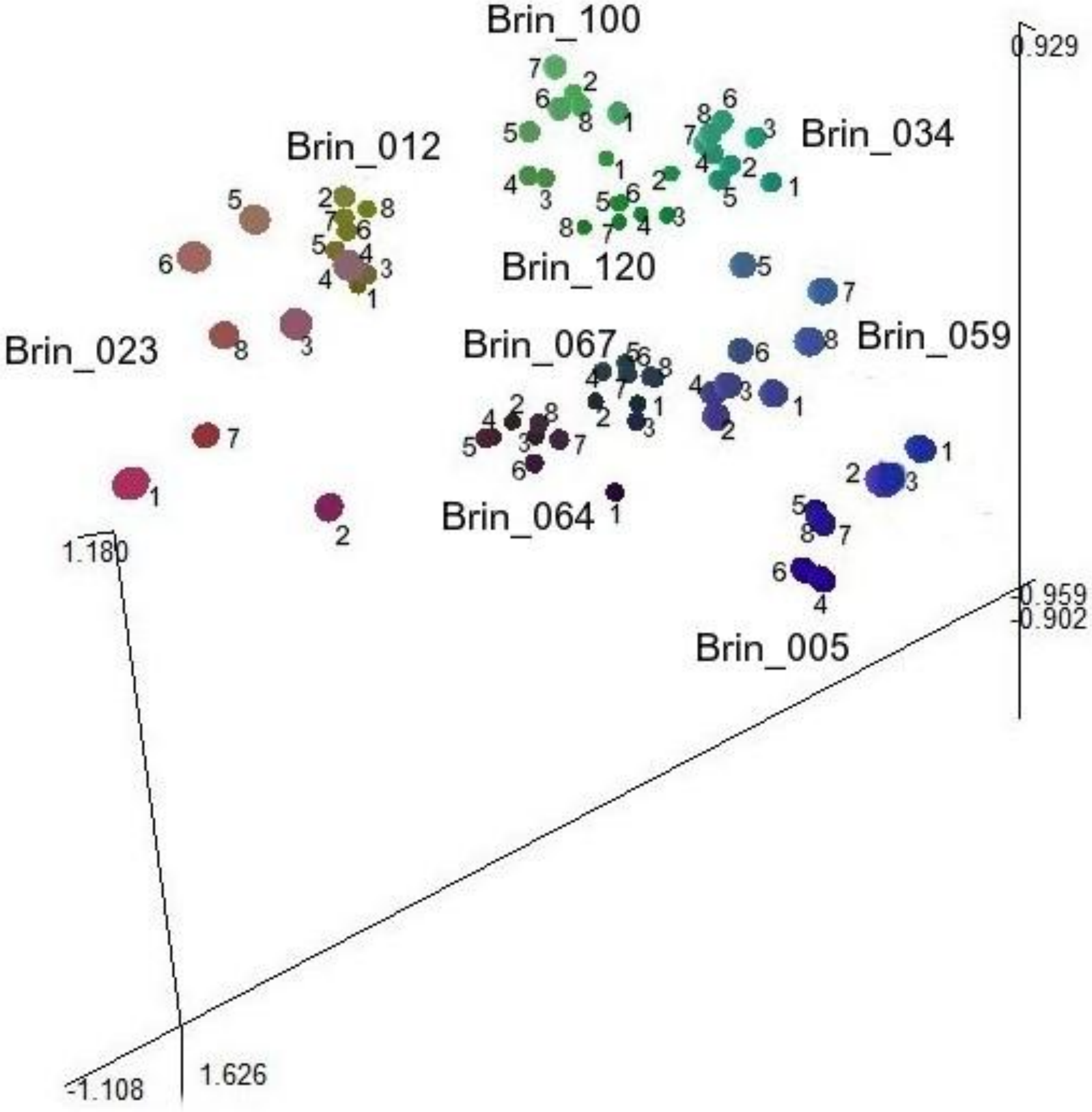
Vegetation Type 3: ***Eucalyptus dives - Eucalyptus mannifera - Eucalyptus macrorhyncha*** **Open Forest (Species Poor)**



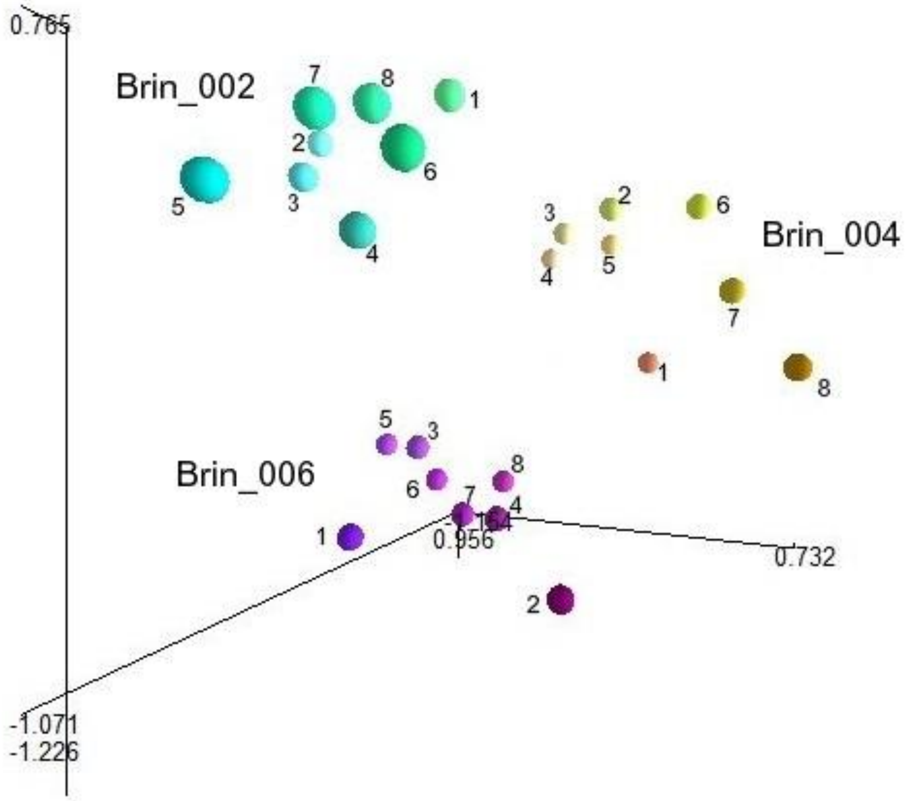
Vegetation Type 4: *Eucalyptus bridgesiana - Eucalyptus melliodora* Open Forest

Not subsequently sampled after the initial post-fire sample in 2003

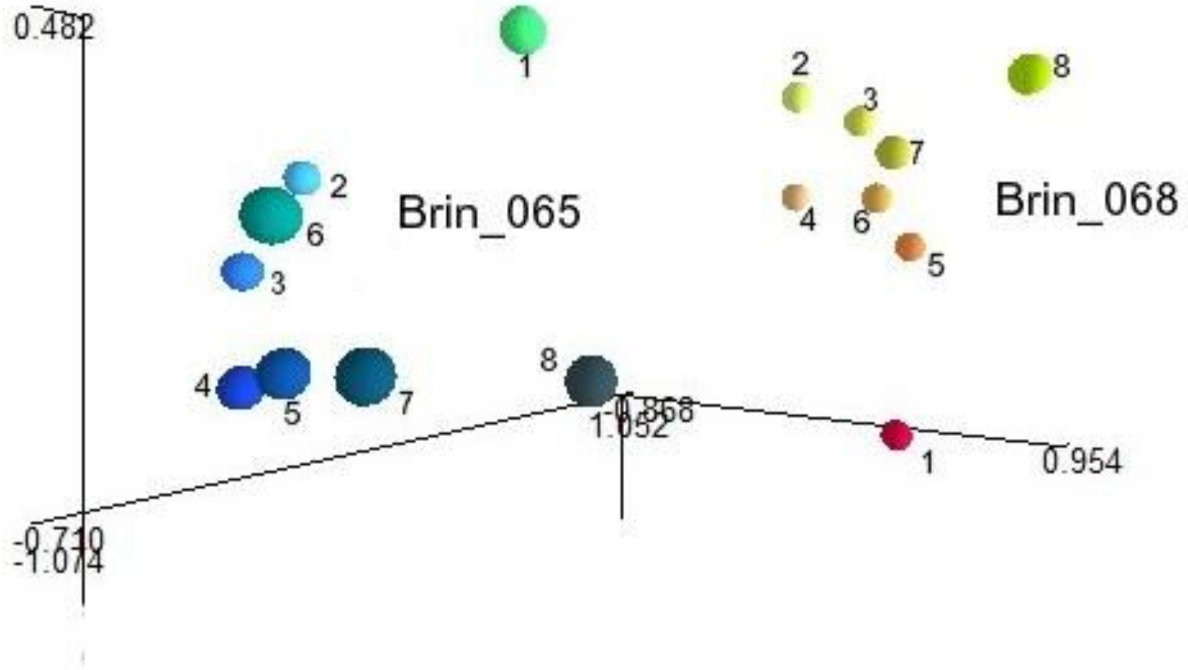
Vegetation Type 5: *Eucalyptus dalrympleana - Eucalyptus dives* (+/- *Eucalyptus pauciflora)* Open Forest



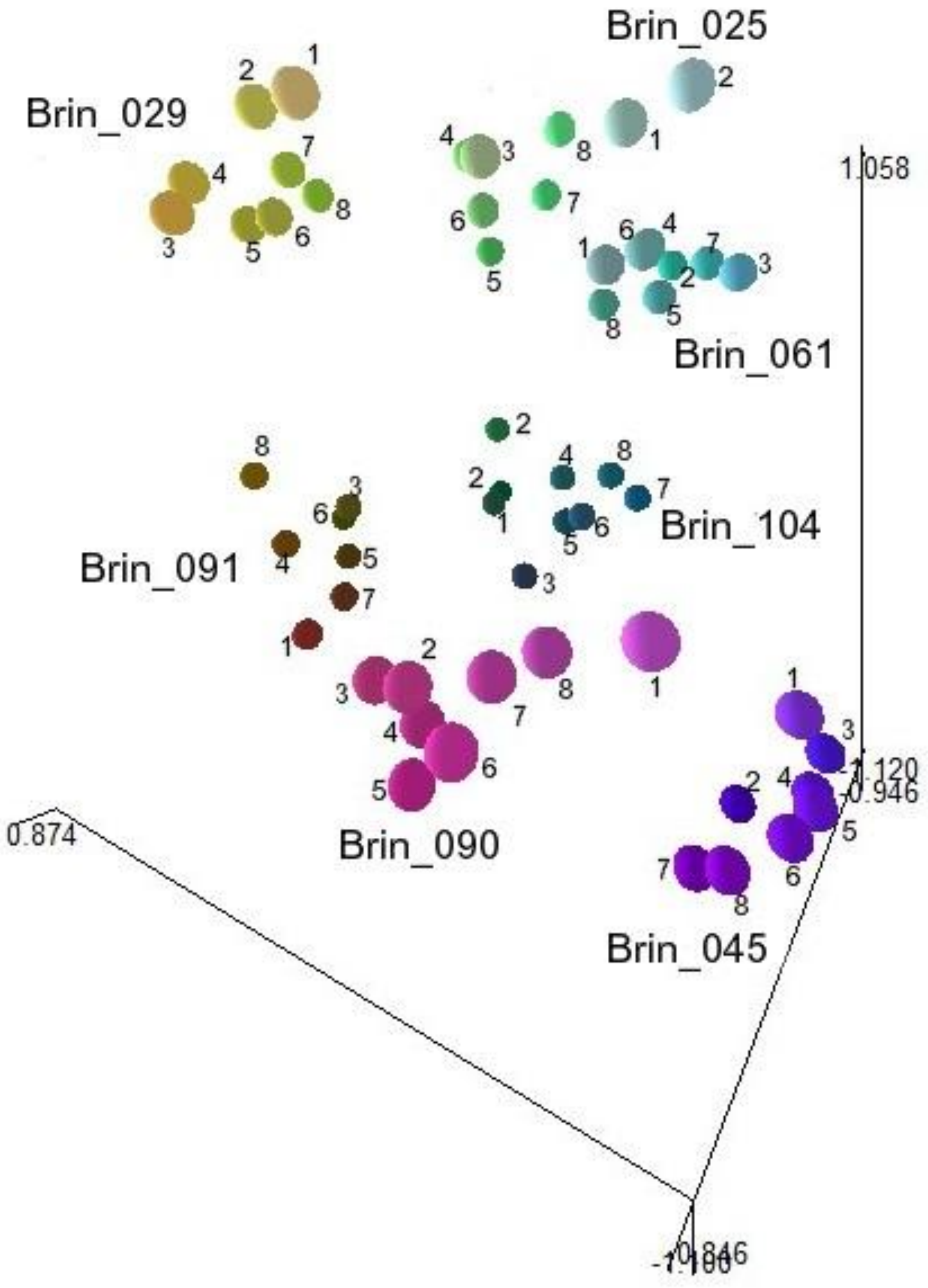
Vegetation Type 6: *Eucalyptus dalrympleana - Eucalyptus robertsonii* Tall Open Forest



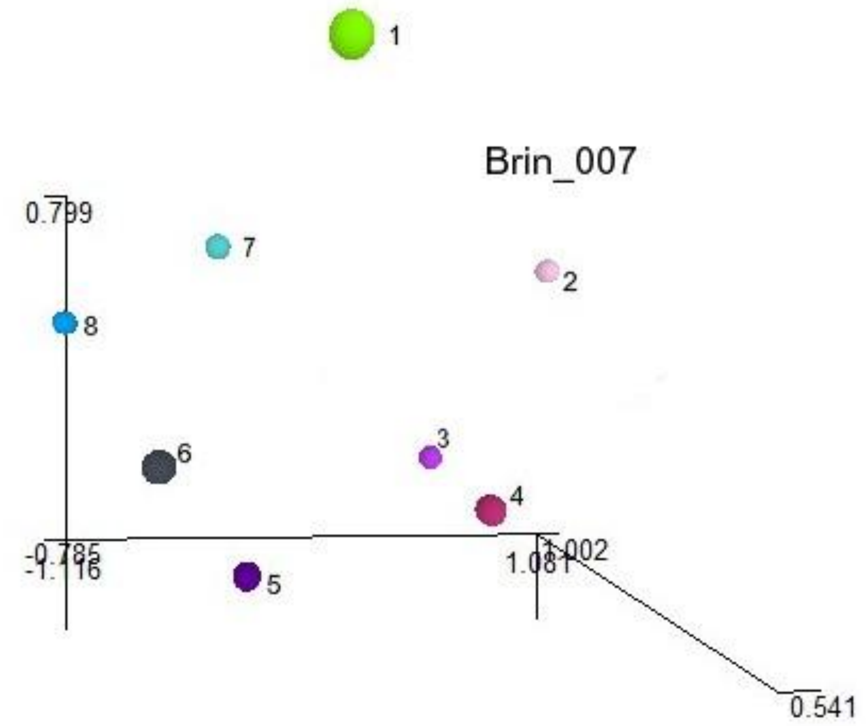
Vegetation Type 7: *Eucalyptus dalrympleana - Eucalyptus delegatensis* Tall Open Forest



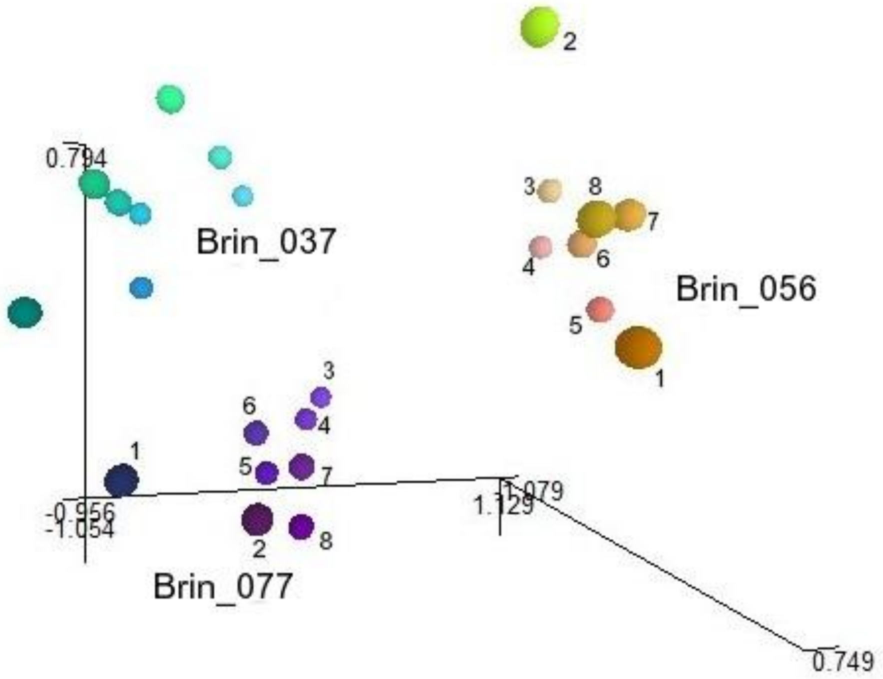
Vegetation Type 8: ***Eucalyptus dalrympleana*** **-** ***Eucalyptus pauciflora*** **Open Forest**



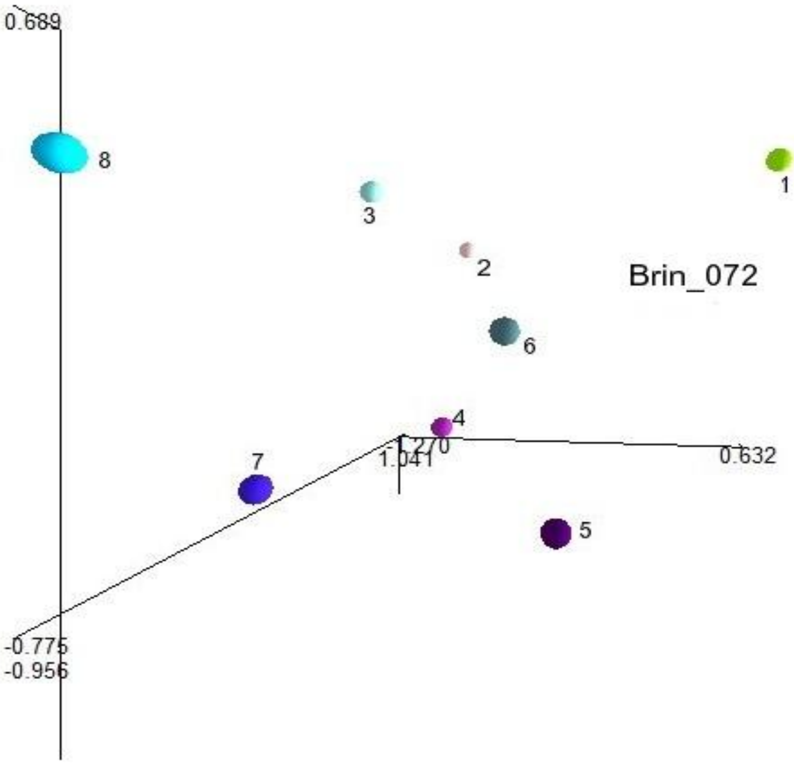
Vegetation Type 9: ***Eucalyptus camphora*** **subsp.** ***humeana*** **Open Forest**



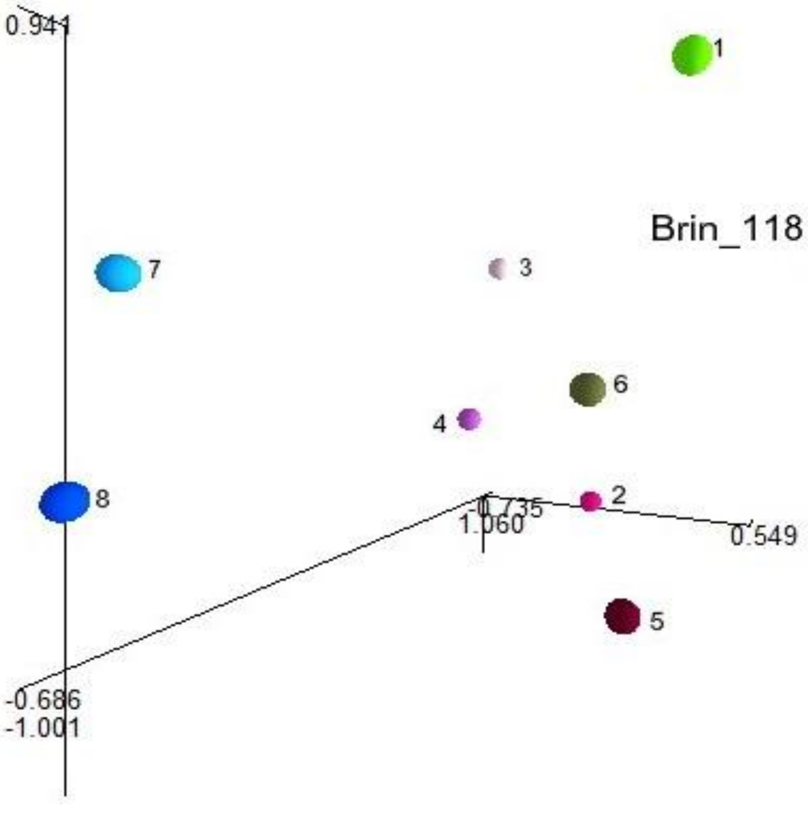
Vegetation Type 10: ***Eucalyptus viminalis - Acacia melanoxylon*** **Open Forest**



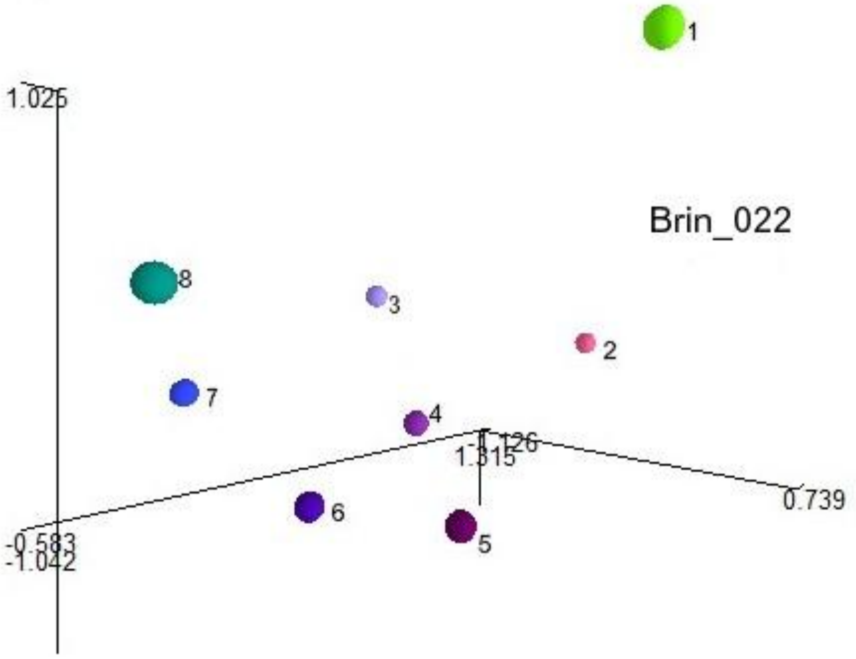
Vegetation Type 11: ***Casuarina cunninghamiana*** **Tall Open Forest**



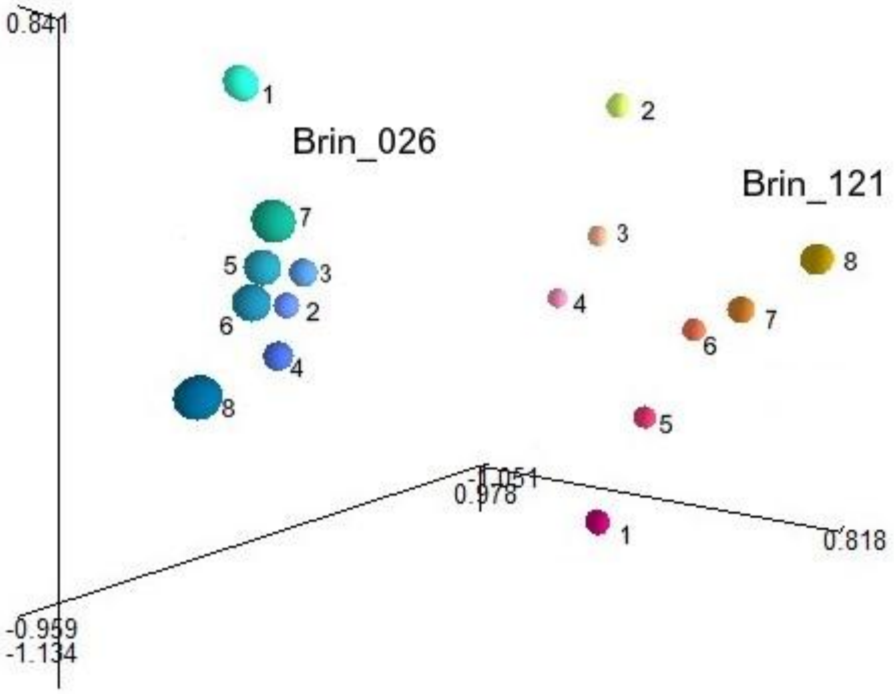
Vegetation Type 12: ***Eucalyptus dalrympleana - Eucalyptus fastigata*** **Tall Open Forest**



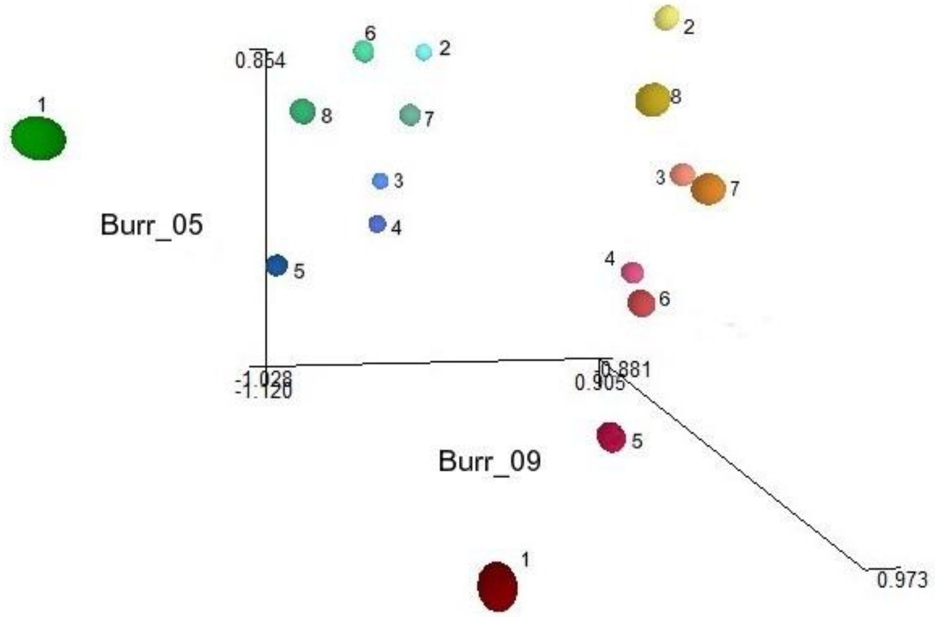
Vegetation Type 13: *Leptospermum micromyrtus* Montane Rocky Heath



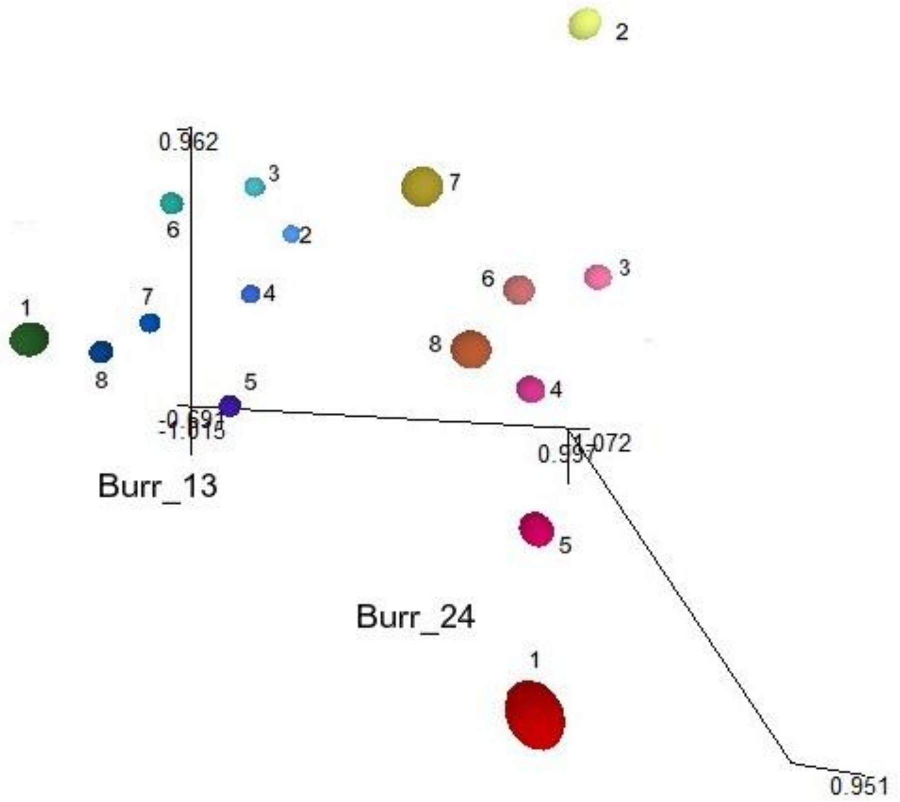
Vegetation Type 14: *Leptospermum myrtifolium – Baeckea utilis* Swamp



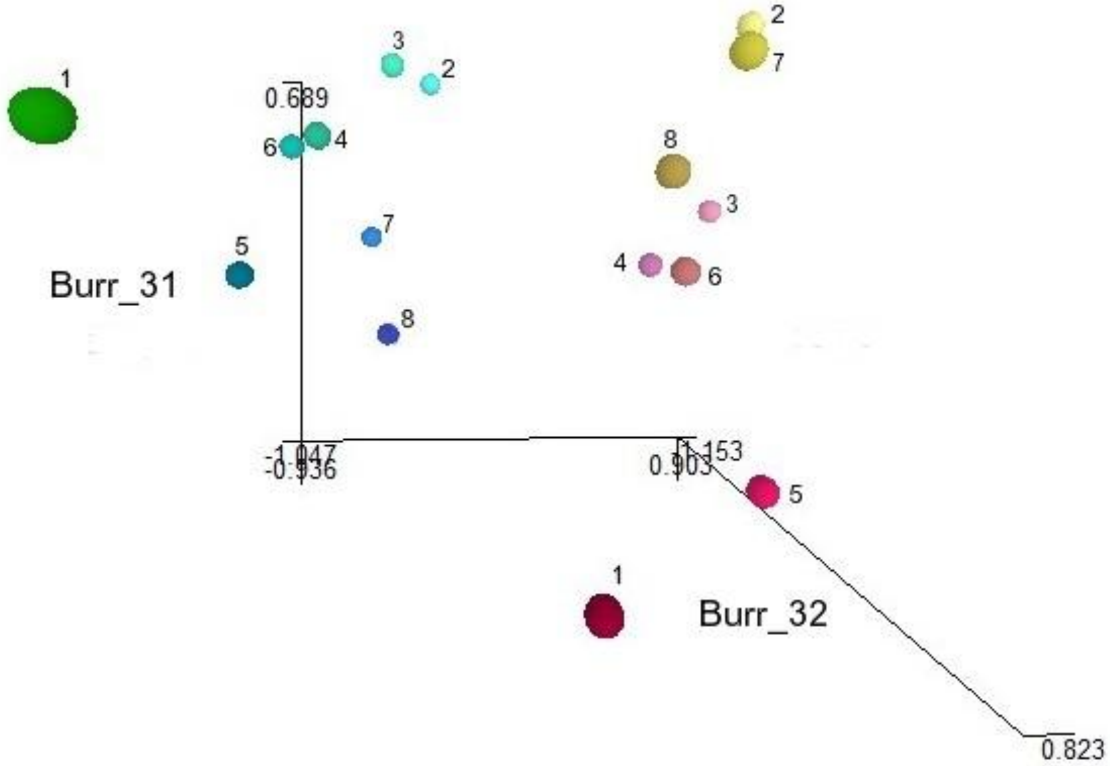
Vegetation Type 15: *Eucalyptus bridgesiana* Open Forest



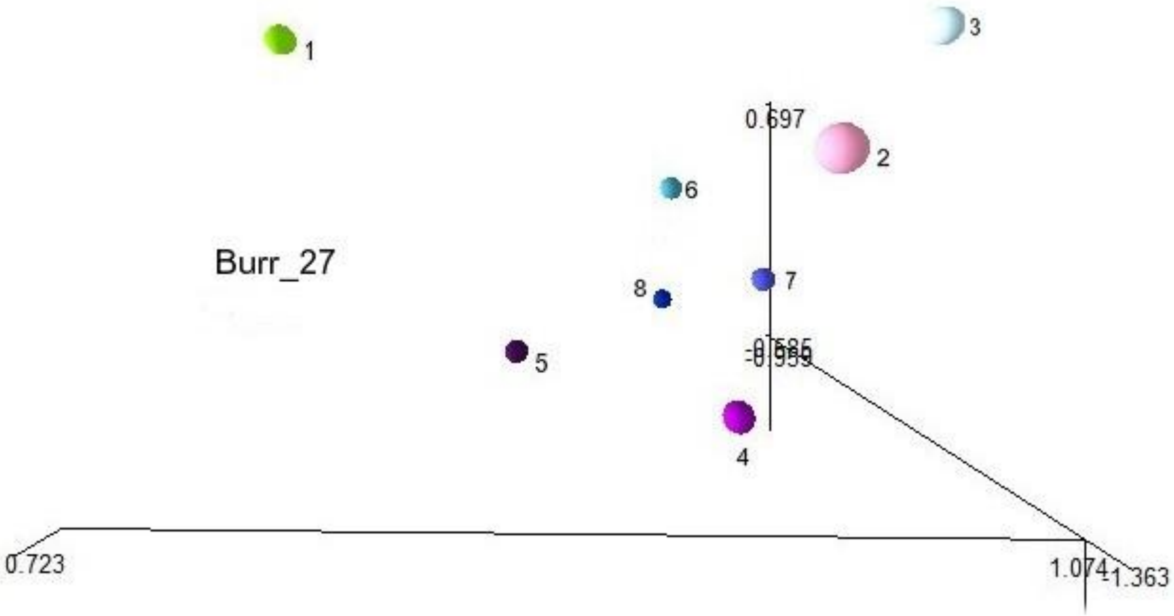
Vegetation Type 16: *Eucalyptus bicostata – E. robertsonii – E. dalrympleana* Tall Open Forest



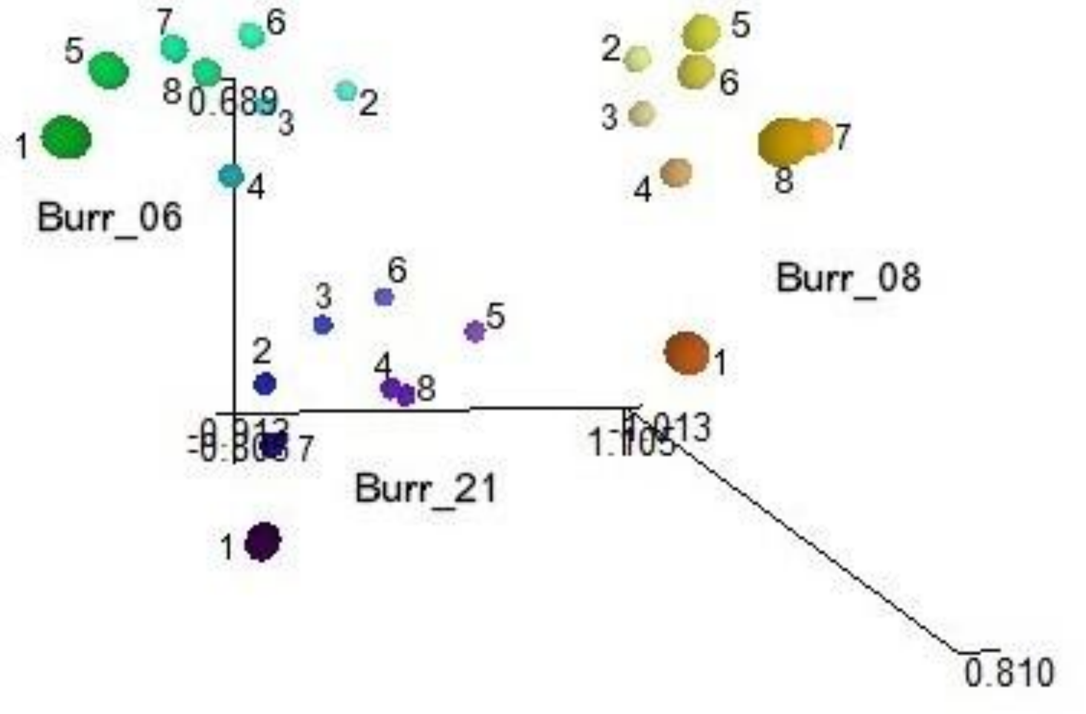
Vegetation Type 17: *Eucalyptus bridgesiana – E. melliodora* Woodland / Open Forest



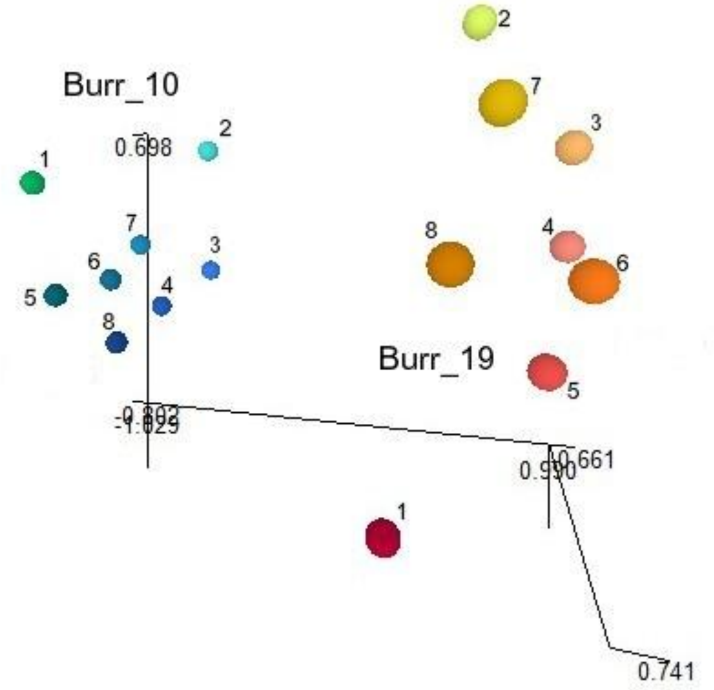
Vegetation Type 18: *Calytrix tetragona – Westringia eremicola* Rocky Outcrop Heath



Vegetation Type 19: ***Eucalyptus goniocalyx – E. macrorhyncha*** **Open Forest**



Vegetation Type 20: ***Eucalyptus dives – E. macrorhyncha – E. mannifera*** **Open Forest**



Vegetation Type 21: *Eucalyptus rossii* Low Open Forest