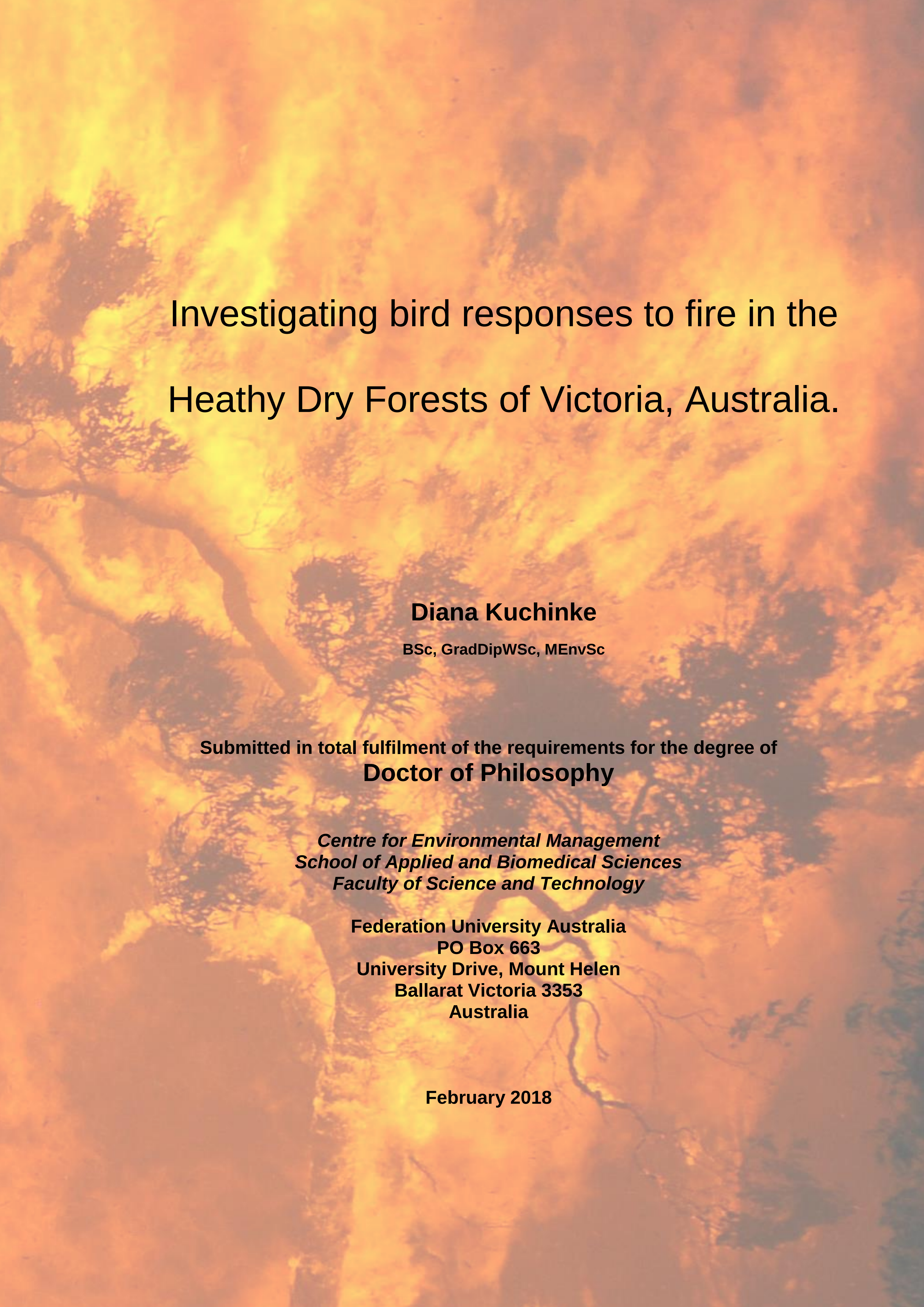

Investigating bird responses to fire in the Heathy Dry Forests of Victoria, Australia.

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BSc, GradDipWSc, MEnvSc

Submitted in total fulfilment of the requirements for the degree of
Doctor of Philosophy

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Statement of Authorship and Originality

Except where explicit reference is made in the text of the thesis, this thesis contains no material published elsewhere or extracted in whole or part from a thesis by which I have qualified for or been awarded another degree or diploma. No other person's work has been relied upon or used without due acknowledgement in the main text and in the list of references in this thesis. No editorial assistance has been received in the production of the thesis without due acknowledgement. Except where duly referred to, this thesis does not include material with copyright provisions or requiring copyright approvals.

Diana Kuchinke

15 February 2018

Resubmitted 14 Dec 2018

To my mentor, my brother Christopher

…now you can read it…

Preface

The 84 study sites in this thesis form part of the long-term monitoring project *Fire In Temperate Forest Landscapes,* by Dr Grant Palmer of Federation University.

Part of this thesis is an investigation into bird responses to prescribed burns of different severities. This section was structured as a Before-After Control-Impact study. Field work for this thesis began in the winter of 2012, after prescribed burns were conducted on sites in March/April of the same year. Dr Palmer had monitored the sites in the spring/summer of 2010, and his dataset was used as the pre-burn data for modelling.

This thesis has many references to individual avian species. The scientific names for all birds observed in the Heathy Dry Forests are listed in Table 3.2. Throughout the thesis, birds are referred to by common names.

The Heathy Dry Forest sites reviewed in this thesis have a maximum 30% canopy. Therefore, comparisons are made between the sites that form this research and ‘woodland’.

All photos within this thesis were taken by me, unless otherwise acknowledged.

Acknowledgements

I cry reading Attenborough and Flannery. Therefore, it stands to reason that with this undertaking, the process was always going to be emotionally charged. Indeed, it was. Especially in this last year of writing. In fact, over the last twelve months I have had to search deep within to produce a document worthy of submitting as a Doctoral Thesis, amongst tears and frustrations. The person who has walked me through this and to whom I am forever grateful is my Principal Supervisor, Prof Peter Gell. Thank you for continually challenging me. Thank you for your patience. Thank you for your many careful, thorough and thought-provoking suggestions. Thank you for taking me in and out the front door.

At an ESA conference some time back (I've been to a few over the seven years it has taken to complete this), I was asked who my supervisors were. A wise man (called Alan) whispered through the crowded heads "cast of thousands". He was correct. There were many people that played pivotal roles at numerous stages of this PhD process, a handful of whom I wish to thank here.

To my Associate Supervisor, Dr Grant Palmer, thank you for guiding me through the field work process. I would never have learned all the bird calls, in such a short time, if it weren't for our review of hundreds of recordings. To Richard Loyn, my External Co-Supervisor, thank you for sharing with me some of your incredible and quite remarkable knowledge about birds.

I returned to studies after some years away - years I spent in breeding mode. In the first year of my PhD, one of the most significant and relevant exercises was that of the weekly induction meeting with Prof Jim Sillitoe. A direct result of the meetings and numerous talks with Jim, was the creation of a project design. Over this time, I also spent valuable weekly hours with Dr Janis Webb, a master of the English language. Thank you, Jim and Janis, for your time and effort.

A couple of years later, with two years of bird data, I worked through 510 pages of Quinn and Keough. Then I decided that I didn't have a clue. I knocked on the door of Assoc Prof Alan York's Melbourne Uni Creswick Campus and screamed for help. Thank you, Alan (the wise man), for welcoming me into your fold, for your support and suggestions. To Dr Julian di Stefano, my deepest and sincerest of thanks. It is only really you and I that know how far I have come since my scanning of Quinn and Keough. Your ability to guide me through the thick, murky waters of statistics was truly remarkable. It is because of you that I can model, code and then interpret results – across programs. To Dr Holly Sitters, thank you for your ability to make 'R' processes appear seamless. I only needed one 11-hour Google search stint on an error message to understand how much time and trauma your help was saving me.

The thesis write-up time is one of extreme pain. It includes long periods of anguish, along with many hours of self-doubt. Considerable time is spent looking at an empty screen, Facebook pages and The Age. There is one partner in crime, who throughout these many months, I could not have done without. Thank you to Inka Veltheim for your friendship and support, for our many emails, our extraordinary number of hours on the phone during which we worked through ideas, reworked ideas, summarized ideas, reviewed new ideas, laughed and cried - with and without wine.

Every PhD candidate needs a mentor. I have an incredible brother who has been there in the background through-out this journey. Thank you, Christopher, for your constant support. I know that, as an atmospheric physicist PhD and ocean optics Research Fellow, you have ecology-envy. I am glad I have been able to do something about scratching that itch for you. Even-though you are yet to read this thesis, your efforts in reading my PhD journey communications can best be summed up in approximate numerical terms. If we assume five emails per day is reduced to two per day (to allow for missed days over seven years), and that 50% were PhD specific, that is 2555 emails you read. Thank you.

To my husband, Ian. You are my rock. Thank you for your unwavering love and support as I moved through this time-honoured and selfish pursuit. There were many days/weeks/months I spent alone, at my desk and in my head. And all the stories about candidates being hard to live with are true. Now you can come home from working interstate. I have morphed back into a human.

To my two quite remarkable and incredible sons, Saxon and Kael, thank you for your support and understanding, when I spent so much time over this last seven years doing my thing. You now no longer need to mimic “I need to get this chapter finished”. Ultimately, all that I do to help conserve birds is so that your grandchildren can enjoy them, as we do. This thesis and my love, to you both.

Lastly, I would like to acknowledge those that provided funding top-ups, in addition to my APA Scholarship. I was a recipient of a Collaborative Research Network grant and further, a TREE grant from the Association for Fire Ecology (USA). Both these income streams enabled me to present at numerous overseas and Australian conferences at various stages over the last seven years.

I was supported by an Australian Government Research Training Program (RTP) Fee-Offset Scholarship through Federation University Australia.

“And that” she said, “Is a wrap”. *mic drop

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Abstract

Globally, forest birds are under pressure, from agriculture and urban development fragmenting the landscape. Adding to these pressures, changes in the patterns of global climate drivers give rise to an increase in the frequency of extreme weather events. In Victoria, Australia, changing weather conditions are resulting in increases in the frequency and extent of bushfires. Furthermore, prescribed burning is applied to the landscape in attempts to not only ameliorate the impacts from bushfire, but as part of a 'pyrodiversity begets biodiversity' protocol. These pressures all impact birds by reducing resources for: food, nesting and protection from predators. This thesis investigated bird responses to fire in the Heathy Dry Forests of Victoria, against variables of time-since-fire and fire frequency. Bird responses were modelled in terms of: community, foraging guilds and individual species. As a community, birds showed a resilience to both time since fire and fire frequency. Responses by foraging guilds and individual species highlighted some different responses. The common species from each foraging guild showed responses that broadly represent their guild. This thesis highlighted that an effective tool in adaptive management is to predict the trends of our common forest birds, as surrogates for entire bird communities, not just for fire responses, but for a broader reflection on the health of the landscape. The modelling of one species, the Laughing Kookaburra, showed a response to both time since fire and fire frequency, with a reduced abundance in post-fire new-growth vegetation. As this species is noted as being in decline down the east coast of Australia, it is flagged in this thesis as a species of concern. Further, this thesis investigated alpha and beta responses by the forest birds to prescribed burns of different severities. Results suggest that forest birds show little response to small

prescribed burns in the landscape, regardless of severity. This may be a result of Heathy Dry Forests' rapid regeneration post-fire. However, one species that exhibits site fidelity, the White-throated Treecreeper, left areas impacted by high severity prescribed burns. The White-throated Treecreeper's response flagged the importance of tree hollows being maintained in the landscape, essential for the species that require hollows for roosting and nesting.

Plate 1 ‘Black Saturday’ bushfire, Victoria, Australia. February 2009.

Source: AAP

1 Introduction

Generations of biologists, ecologists, ornithologists and nature-lovers have been inspired by the forest birds of the south-east region of Australia. However, in the years since European settlement, human needs in this region have driven landscape-changes on an immense scale. These changes have impacted avian assemblages to such a degree that many reports of avian species' declines have been published. In fact, these forests contain some of our most recognized avian species. Yet, research on these birds lags well behind the rate at which the forest landscape is changing form and diminishing in area and extent.

This is a study of bird responses to aspects of the fire regime, in Heathy Dry Forests of Victoria, Australia. The central region of Victoria is a highly fragmented landscape, and subject to a drying climate over many decades. So, factors other than fire may impact bird responses. These forests represent an important case study to consider multiple drivers of biodiversity decline worldwide, with a focus on informing avian conservation in these, the most flammable forests on Earth.

1.1 Birds in decline in temperate regions

1.1.1 A global biodiversity crisis

Currently, there is a wave of anthropogenic biodiversity loss (Dirzo et al., 2014), with unprecedented rates of extinction being documented (Ceballos et al., 2015, McCallum, 2015, Ceballos et al., 2017). The recent total extinctions of vertebrate species, if at the same rate as that prevailing over the last 2 million years, should have taken approximately 10,000 years, not one century (Ceballos et al., 2015).

Hundreds of species are being driven to extinction every year (Dirzo et al., 2014, Maxwell et al., 2016, Young et al., 2016, Ceballos et al., 2017), with major declines in temperate regions (Ceballos et al., 2017). While some authors have, within the last two decades, suggested a ‘sixth mass extinction’ is inevitable (Leakey, 1996, Thomas, 2007); this is now an accepted concept (Barnosky et al., 2011, Dirzo et al., 2014, McCallum, 2015, Ceballos et al., 2017).

‘Global defaunation’, the extinction of faunal species and populations and the decline in abundance of individuals within populations (Young et al., 2016), is a pervasive component of the ‘sixth mass extinction’ (Dirzo et al., 2014). Predictions on extinction rates can vary, based on different analysis methods, and because, for some species, data may be deficient (McCallum, 2015, Ceballos et al., 2017). Nonetheless, with the large percentage of decreasing species in temperate latitudes, it is clear the imperative is to halt the declining trend. This is a challenging issue when population extirpation can occur without there being clear evidence flagging that the entire species may be retracting in range; i.e. a species may be extinct in one location, however abundant in another (Hobbs and Mooney, 1998).

1.1.2 Temperate forests under pressure

Some of the most significant large-scale land conversion from forests and woodlands to agricultural and urban landscapes can be seen in the temperate regions around the globe (Hobbs and Yates, 2000, Majer et al., 2010, Sleeter et al., 2012). Contemporary land conversion continues in some regions unabated, driven largely by human population growth and expansion (Sleeter et al., 2012). As modifications continue across landscapes, the challenges are to manage for species' protection, when the focus in a region may be to accommodate an expanding human population.

Following European settlement, large tracts of the fertile regions of southern Australia were converted to pastures, or cleared for crops or livestock grazing (Prober and Thiele, 1995). Prior to European settlement, 32% of Victoria consisted of temperate woodlands, however by 1987, 92% of Victoria's woodlands had been cleared (Lunt and Bennett, 2000), resulting in threats to both vegetation persistence (Reid and Landsberg, 2000, Clarke, 2000) and arthropod abundances (Abensperg-Traun M. et al., 2000, Majer et al., 2000).

The extent to which this land-use change occurred has resulted in one of the most significant changes across the Australian landscape since European settlement (Hobbs and Yates, 2000). These extensive modifications to the landscape have resulted in profound changes for birds (Bennett and Radford, 2010). Although birds are highly mobile and exhibit large temporal and spatial abundance fluctuations (MacNally, 1996, Manning et al., 2007, Lindenmayer and Cunningham, 2011) many species have been reported to be in severe decline in Australia (Recher, 1999, MacNally et al., 2009, Bennett and Watson, 2011, Ford, 2011, Watson, 2011). These declines result from reductions to geographical ranges (Franklin et al., 1989) and regional extirpation (Ford et al., 2009).

Species persist and reproduce within their 'realized niche' (Hutchinson, 1957); the total range of conditions under which a population or a species survives. Intrinsic complexities arise from multiple disturbances, that occur both spatially and temporally. Combining these factors, a species will only continue to survive if its tolerable limits are not exceeded. If tolerable limits for a species are exceeded, the result will be a shift in range or even local extirpation, or in extreme cases, species extinction.

1.1.3 Avian species in decline in south-eastern Australia

Australia's bird taxa are slipping towards extinction (Recher, 1999, Lindenmayer, 2007, Bennett and Watson, 2011, Ford, 2011, Szabo et al., 2012, Birdlife Australia, 2015, Ceballos et al., 2017). Whilst the global Red List Index (RLI) has recently flagged the eastern ecoregion of Australia as being of concern (Ceballos et al., 2017), the National Index recognizes that the entire east coast of Australia is a highly diverse habitat (Birdlife Australia, 2015) and so has split the region into the "East Coast" and the "South-eastern Mainland". The *2015 State of Australia's Birds* report highlights that the south-eastern mainland, of predominantly eucalypt woodland, is a highly modified landscape and that the indices for most bird groups in this region have been declining since 2006. In fact, 22 of the 86 species listed, showed significant decreases over the reporting period, suggesting a pattern of concern for this group. Birdlife Australia stress that further research on the south-eastern region is imperative (Birdlife Australia, 2015).

Twenty of the 26 bird species considered as declining species in the southern region of Australia are insectivorous (Watson, 2011). As 14 of these species are primarily ground foragers (Razeng and Watson, 2012), landscape changes are critical causes of this decline. Drought conditions greatly impact bird survival (MacNally et al., 2009). Further, it has been argued that only half of Australia's terrestrial birds will survive the next one hundred years if factors impacting their decline are not addressed – factors that include inappropriate fire regimes (Recher, 1999).

Ford et al. (2001) reviewed the reasons why so many of southern Australia's woodland birds have declined. Described as a seminal work for woodland birds (Bennett and Watson, 2011), this research outlines the impacts on birds from habitat loss, fragmentation, land degradation, competition and predation. Authors noted a wide range of foraging guilds as being in decline – the granivores, nectarivores, raptors, ground-nesters and ground-foraging insectivores. A subsequent update on this assessment is "The Action Plan for Australian Birds 2010", which highlights that land clearance and habitat degradation may continue to cause species' declines for decades, and that fire is a threatening process (Garnett et al., 2011). Whilst fires were once less frequent in a highly fragmented landscape (Ford et al., 2001), much has changed in recent decades in terms of fire weather and the fire regime. Bushfire frequency continues to increase in Australia (Fairman et al., 2016) and in Victoria, fire is applied to the landscape in the form of prescribed burns, at a frequency greater than ever before (Teague, 2010).

With an increase in temperature being one of the most important factors resulting in an increase in bushfire activity (Flannigan et al., 2009), impacts on forest birds are therefore numerous and complex. Studies have shown that an increase in precipitation is a crucial trigger for breeding timing (Baker, 1939, Johnston, 1954, James and Shugart, 1974); a drying climate therefore may result in young hatching at times when food supply is not at its maximum. Under current climate change scenarios, southern Australian temperatures are expected to increase 1-5 degrees by 2070 (Commonwealth Scientific and Industrial Research Organization., 2007, Timbal and Jones, 2008). This may result in species breeding when food supply is not optimum. MacNally et al. (2009) showed that approximately two-thirds of the bird species in a highly fragmented woodland and forest region of central Victoria, Australia, have declined by what they argue is the result of a changing climate creating a marked decrease in rainfall in the region. They suggest that species' decline reflects an on-going and consistent erosion to their resource base; decline further exacerbated by bushfires across spring and summer months and prescribed burns being carried out throughout the year.

In more recent times, Ford (2011) summarized the extensive body of work and advancements in our understanding of the woodland bird crisis by arguing that the focus had shifted to one of process. Furthermore, he suggested that to reverse woodland bird decline, effective management requires science based on a whole of landscape approach at regional and national scales. Clearly, this shift of focus now must include impacts from a changing fire regime.

Underpinning a whole of landscape approach are climate variables. An integral component of climate processes that are changing, is a variation to rainfall levels. This has implications for birds, beyond just an adequate supply of drinking water. Water storage in soils that are heavily grazed, or of low-level structure (hence poor quality), may move from within the soil to surface reservoirs, impacting below-ground decomposer communities, leading to fewer ground-dwelling invertebrate prey for birds (Watson, 2011). Further, not all woodland soils are the same, with many lacking in structure (Watson, 2011). This is the case for the Heathy Dry Forests in Victoria (Cheal, 2010). Low rainfall will therefore have profound effects on invertebrates, necessary prey for insectivorous birds. This renders the Heathy Dry Forests extremely sensitive to drought conditions, especially with the prolonged rainfall anomalies that Victoria is currently experiencing (Bureau of Meteorology, 2016). In fact, recent research on modelling fire regime factors with faunal species and vegetation abundances in species distribution models, showed that the two strongest influences on the fire regime, for both vertebrate and plant species in the Victorian foothills, were rainfall and temperature (Kelly et al., 2017). These two factors had the highest influence on species distributions and when modelled, sometimes showed marked variation in the responses of species (Kelly et al., 2017).

This has major implications for the avifauna in the woodlands of south-eastern Australia. Woodlands contain almost one-third of the threatened ecological communities listed under the Federal Government's *Environmental Protection Biodiversity Conservation Act 1999* (Lindenmayer, 2007). With extensive land clearing, heavy modification and high levels of degradation within woodlands and forests, many woodland birds have lost much of their range in eastern Australia (Lindenmayer, 2007). So, climate and human driven changes to fire regimes will likely act as an additional threat to forest and woodland birds that are already impacted by large scale vegetation clearance.

1.2 Fire as a driver of change

1.2.1 Fire in the landscape

Climate, along with factors such as edaphic conditions, will influence the distribution of vegetation species. Following these first order controls, a second order control is fire disturbance, which can homogenize the landscape and so create large, uniform age stands of fire-adapted vegetation (Bendix and Cowell, 2010). However, variations in the terrain and vegetation may lead to heterogeneity in a landscape (Brotons et al., 2004, Leonard et al., 2013). As these factors will affect fire spread, creating different burn severities and intensities, heterogeneity arises even within a single fire. Furthermore, fire can fragment the habitat into mosaics, due to the variations in the frequency and intensity of fire events (Stuart-Smith et al., 2002).

Fire has been a common feature of many Australian biomes since possibly the Oligocene (Holdgate et al., 2014) and much of the continent's contemporary flora and fauna exhibit adaptations to survive fire, and to persist and even flourish. Fire is inextricably linked to the distribution and ecology of many vegetation species right across the Australian landscape, in that vegetation supplies biomass as fire fuel (Bradstock, 2010).

A fire will result in complex interactions, involving many factors, impacting biota. Effects may be both direct (smoke, temperature and heat resulting in emigration or mortality), and indirect (by altering the environment such that it has limited resources for species to persist). Fire in the landscape may also result in species declining some time after the event, as a result of starvation or predation (Tilman et al., 1994, Whelan et al., 2010). Here, it is the 'fire regime' that needs to be considered - the timing, size, intensity, severity and frequency of fires in a location, as well as the interval of time between fire events (Gill, 1975). This is important not just for the continuing regeneration of vegetation, but also for the persistence of faunal communities (Hobbs, 2010, Keith et al., 2010). The current impact of a changing fire regime on Australia’s flora and fauna is profound (Woinarski and Recher, 1997, Recher et al., 2009, Bowman et al., 2012, Enright et al., 2012, Gill, 2012, Recher and Davis Jr, 2013, Lindenmayer et al., 2014), with a significant cross-section of Australia’s biodiversity under threat, including more than fifty bird species (Lindenmayer, 2007).

1.2.2 Changes to fire weather pushing temperate forests to tipping points

High inter-annual variability in fire weather in Australia has been linked to climate drivers (Williamson et al., 2016). Therefore, dealing with high variability in data when modelling creates challenges when attempting to predict seasonal patterns to fire weather (Williamson et al., 2016). Nonetheless, what has been argued is that, across south-eastern Australia, there has been an increase in recent decades in the severity of weather conditions conducive to fire (Bradstock et al., 2014). Furthermore, there has been an increase in the frequency of mega-scale bushfires in Victoria (Fairman et al., 2016).

As bushfires increase in severity and frequency, there is the risk and likelihood of the elimination of some vegetation species from the landscape (Bowman et al., 2014). As weather patterns change, ecosystems and species may be pushed to tolerable limits. Laurance et al. (2011) compiled research from various authors on Australian ecosystems vulnerable to tipping points. They list temperate eucalypt forests as particularly vulnerable. While a changing climate will predispose ecosystems to tipping points, vulnerable ecosystems are being impacted by multiple drivers (Laurance et al., 2011) and even-though different drivers of species decline are often modelled and managed independently, there is mounting evidence that complex, synergistic effects can occur between drivers of native species decline (Didham et al., 2007). Furthermore, it is argued that relationships between drivers, with the combined effects of climate change, can result in challenges when attempting to determine causes of range decline and regional extinctions (Thomas et al., 2006).

1.2.3 Determining fire history guiding understanding on climate drivers

There is a need for long-term data on fire history to contextualize the present climate-fire relationship. Recent studies have linked a positive Southern Annular Mode (SAM) to an increase in fire frequency, by examining charcoal deposits in western Tasmania from the last 1000 years and arguing for a relationship between fire activity and SAM, at centennial time-scales (Mariani and Fletcher, 2016).

The link between fire activity and SAM phases is not limited to the temperate forests of Tasmania. A study based on the conifer forests of Patagonia was based on 42 fire history sites of 600 fire-scarred trees, from which researchers determined that the years of widespread fire were associated with positive phases of SAM (Holz and Veblen, 2011).

Combining these studies, it is now suggested that the SAM is an important driver of fire around the entire southern hemisphere (Mariani and Fletcher, 2016). In fact, it has been found, on reconstructing an annual average SAM index since AD 1000, that the long-term mean of the SAM index is currently at its highest positive value for at least the last 1000 years, and that with continued increases in greenhouse gases, the prediction is that SAM will move further towards positive phases over the next 100 years (Abram et al., 2014).

1.2.4 Changes to the fire regime in Mediterranean regions

Mediterranean regions around the globe have hot summers and wet winters, the winter rain providing for vegetation growth, which dries out in the summer months to become fuel for fire. The physiographic nature of the vegetation that persists in Mediterranean regions is very similar at the same latitude around the globe. These fire-prone landscapes can be seen in the Mediterranean Macchia, the South African Fynbos, the Chaparral of California and Chile, along with the eastern deciduous forests across the central portion of eastern North America, and the Kwongan and Mallee of south-west and south-eastern Australia respectively. The eucalypt forests of the south-east region of Australia afford even greater fire risk than the Kwongan or Mallee vegetation, as, while they share the feature of volatile oils, they grow fast and tall.

As global climate systems increase in variability, fire regimes change, resulting in complex climate-system changes impacting vegetation in Mediterranean landscapes around the globe (Pausas and Vallejo, 1999). Climate drivers are precipitating changes to the fire regime in Australia’s south-eastern region, with unknown consequences on forest biodiversity (Sala et al., 2000, Gil-tena et al., 2009).

Researchers assessing the complex interactions between climate, fire and the landscape in Mediterranean regions are faced with challenges when interpreting bird responses. In the region south-west of Barcelona, the landscape is continually being modified by fire and changes to agricultural practices, and this has resulted in bird biodiversity being maintained (Brotons et al., 2004). Determining the impacts of a changing fire regime on birds in this landscape is challenging because several factors e.g. agricultural and urban developments as well as fire events, will all impact biodiversity in the region. A North American symposium collectively reviewed fire across forty ecosystems in the region (Saab and Powell, 2005). This research described how changes in spatial patterns and frequency are linked to changes in drought conditions, highlighting the importance of climate in explaining variations in fire frequency and extent over time. Furthermore, the Mediterranean eastern and central US region have had both bushfire occurring, and prescribed fires applied to these regions, to restore and maintain oak savannas. Therefore, a greater understanding of the fire regime in these regions has been flagged as imperative (Artman et al., 2005).

A decrease in rainfall in the summer months is a primary factor predicted to impact Mediterranean regions around the world (Giorgi and Lionello, 2008). Furthermore, researchers argue that different environmental conditions along climate gradients will impact fire fuel moisture, hence flammability, and that fires will occur at different intervals and spatial scales (Kelly et al., 2017). These factors combined will in turn influence vegetation growth and so impact on birds.

It is the Mediterranean regions around the globe that will possibly experience the greatest impacts to levels of biodiversity, a result of substantial impacts from the drivers of biodiversity change (Sala et al., 2000) such as fire (Mouillot and Field, 2005).

1.2.5 Global climate changes and how they impact fire weather in south-eastern Australia

The southern region of Australia is known as one of the most flammable places on earth (Adams and Attiwill, 2011), due to its unique meteorological and biophysical arrangement (Teague, 2010). Key climate drivers contribute to, and drive, patterns of bushfire across the region.

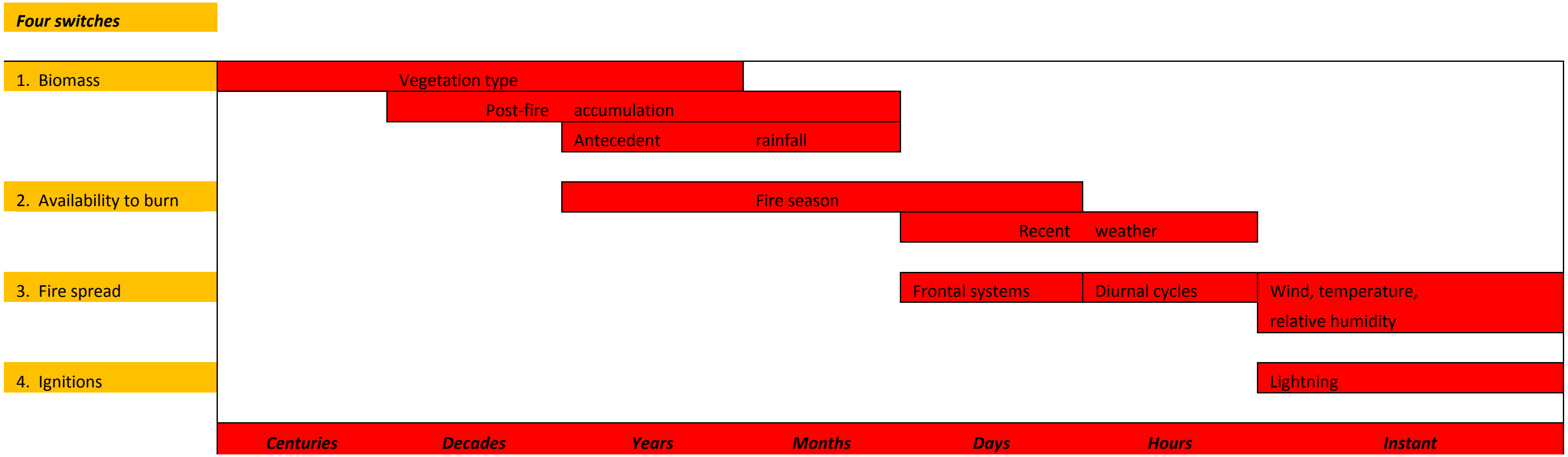
Fire weather impacts aspects of the fire regime (Bradstock, 2010). The global atmospheric pressures, sea surface temperatures and variation in winds, all give rise to bushfire weather in the south-east region of Australia. As changes occur to fire weather components over both the short and long terms, then so too will bushfire frequency and severity rates change. It is assumed by some researchers that the main driver of the fire regime is climate – in that while fire-spread is determined by the features of the landscape, fire-size is dictated mostly by climate variables (Brotons et al., 2013).

In regions of the globe where temperatures are increasing, there is an increased pressure on the biota, as they must adapt to changing conditions (Birdlife International, 2008). In Australia, different climate forecasts all arrive at a collective conclusion - that the number of hot days across the country will increase, which will not only extend drought periods but also increase the frequency of high fire-danger days (Steffen et al., 2017). Both these factors combined will result in an increase in bushfires in areas that have fire fuel load and weather conditions conducive to fire.

1.2.6 The climate drivers in south-east Australia

It is a challenge for researchers to create a framework that effectively identifies all drivers of fire in the landscape, however, Bradstock (2010) provided a conceptual framework to define processes that limit fire. The model incorporates key drivers, as on/off switches, being the influences of biogeographic factors: 1) biomass growth; 2) the availability of vegetation for burning, measured in terms of drought conditions altering the moisture status of vegetation; 3) ambient fire weather; and 4) ignition, from lightning and anthropogenic sources. Murphy et al. (2011) and Williamson et al. (2016) added to this framework a range of time-scales over which the four switches will operate, to impact the factors that are fundamental to the fire regime. Over-arching this, in south-east Australia, four key climate drivers can be added to the framework: the Pacific Decadal Oscillation (PDO), the El Niño Southern Oscillation (ENSO), the Indian Ocean Dipole (IOD), and the Southern Annular Mode (SAM) (Fig 1.1).

These four drivers, driven by winds and temperature variations that impact levels of rainfall, alter in frequency and intensity – and across time-scales. They are also the key elements to consider when attempting to predict future bushfire patterns across this region.



Four switches

1. Biomass
2. Availability to burn
3. Fire spread
4. Ignitions

Vegetation type

Post-fire accumulation

Antecedent rainfall

Fire season

Recent weather

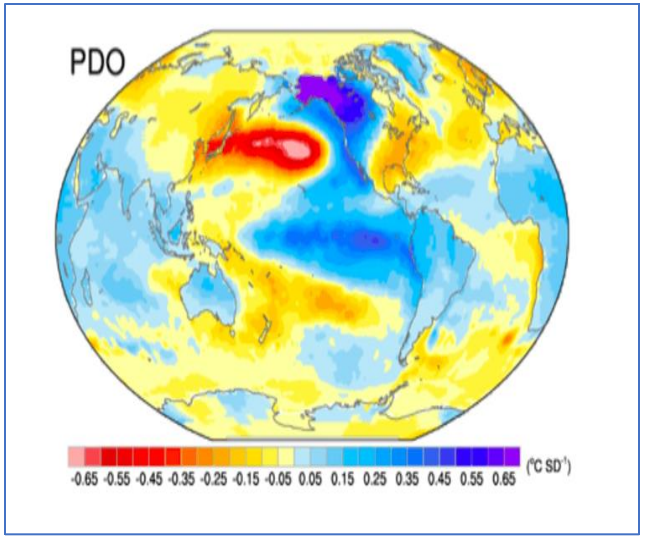
Frontal systems

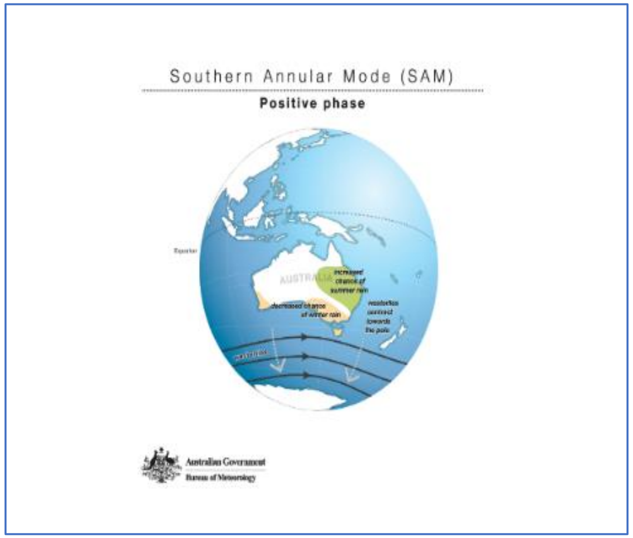
Diurnal cycles

Wind, temperature, relative humidity

Lightning

Centuries ***Decades*** ***Years*** ***Months*** ***Days*** ***Hours*** ***Instant***





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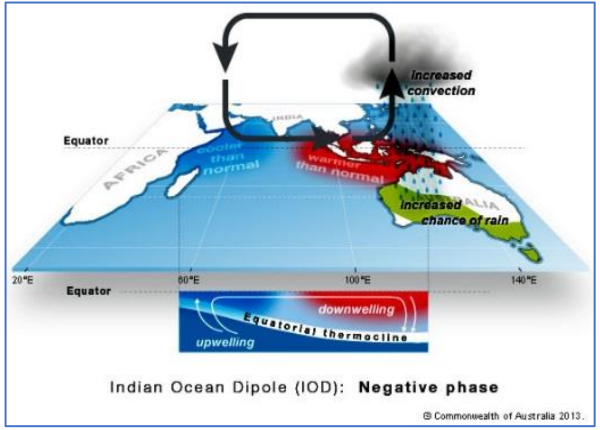


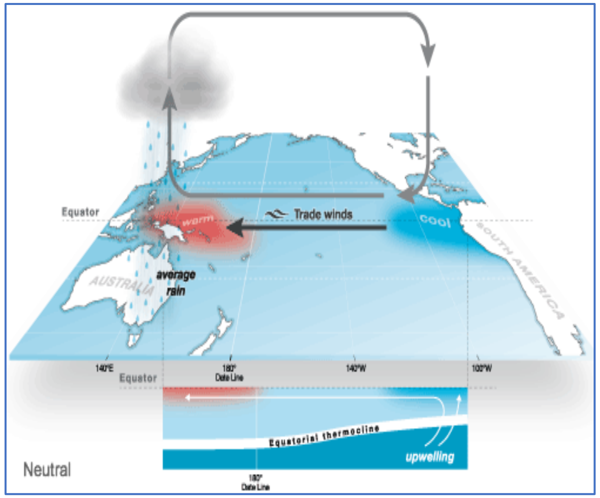


Figure 1.1 Four climate drivers impacting the south-eastern region of Australia, over time.
Framework of four switches taken from Bradstock (2010), time-scales and factors impacted from Murphy et al., (2011).
Four climate drivers impacting the region: Pacific Decadal Oscillation (Trenberth, 2014), Southern Annular Mode, Indian Ocean Dipole and the El Niño Southern Oscillation (Bureau of Meteorology, 2017).

The complexities and the inter-connectedness of these four climate drivers can be illustrated with weather events in the south-east Australian region over recent years. Compounding the effects of El Niño during the Millennium Drought (1996 – 2010) (Gergis et al., 2012), was a positive SAM, which was in place for large parts of the years 1997 – 2010 (Bureau of Meteorology, 2017). In an area where the endemism of many flora and fauna taxa is high, the continued years of rainfall deficiencies resulted in many species being at risk because of restrictions to both geographical and climatic range (Hennessy et al., 2007).

Rains in 2010 and 2011 marked the end of the Millennium Drought across Australia, however Victoria can be seen to be on a drying trend that has continued, from the 1970’s through to 2016 (Fig 1.2), with rainfall anomalies in the years since 2010 (Fig 1.3).

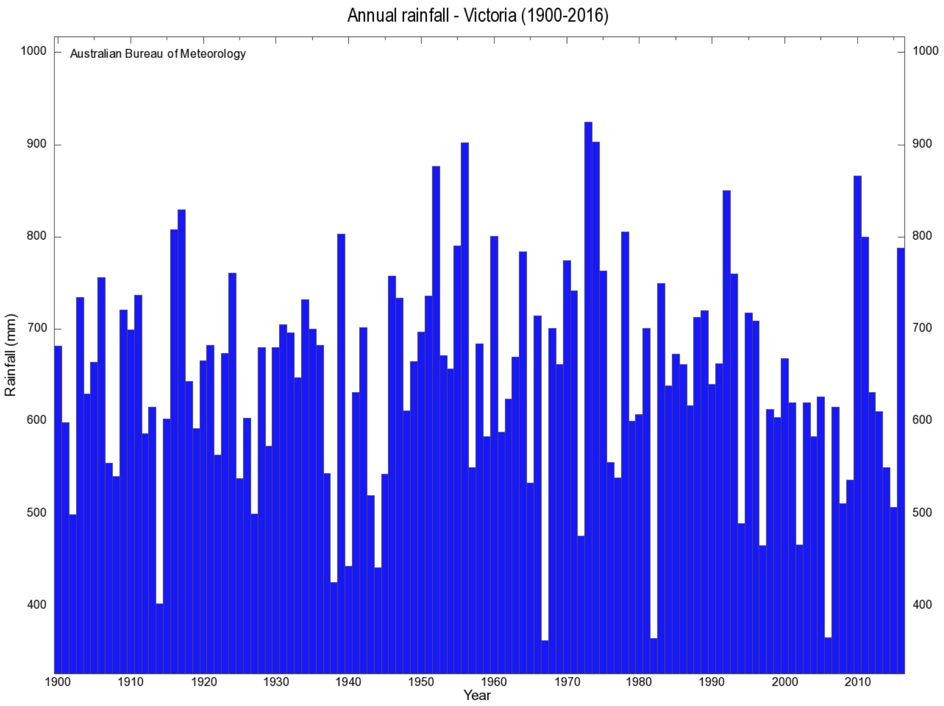


Figure 1.2		Annual rainfall Victoria, Australia (1900-2016).

Source: Bureau of Meteorology (2017).

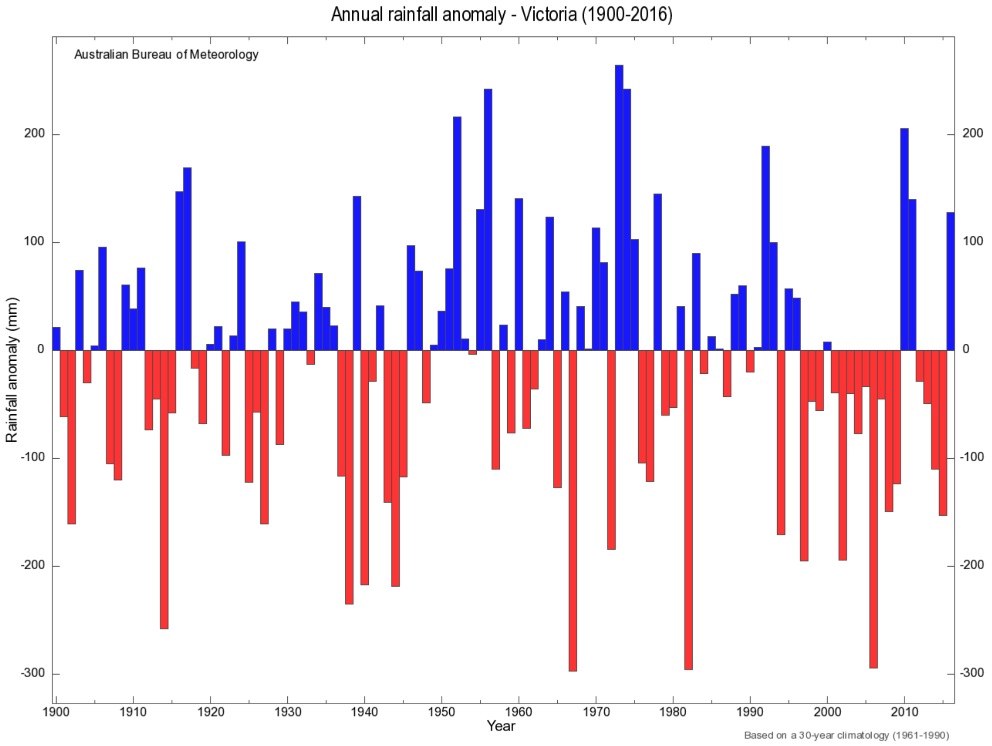


Figure 1.3 Rainfall anomalies Victoria, Australia (1900-2016).

Source: Bureau of Meteorology (2017).

In 2010 a negative IOD operated in concert with an ENSO La Niña phase. Warmer than average sea surface temperatures in the Indian Ocean, plus a break in the El Niño drought, resulted in increased moisture moving across Australia. A negative IOD was seen again more recently in 2016, accompanied by an El Niño phase until May 2016, which was then followed by an ENSO neutral phase through to December 2016 (Bureau of Meteorology, 2017). As 2016 progressed, there was a shift in Australia’s climate from ‘warm/dry’ to ‘cool/wet’ conditions. This shift resulted in well above average rainfall falling on most of the east coast of Australia later in the year (Bureau of Meteorology, 2017). However, even though the 2016 rains extended from the north-western region of western Australia, across the central region of Australia to the east, and into South Australia, parts of the western and central regions of Victoria received little of this well-above average rainfall. The weather system that may have been impacting this is the SAM. The poleward shift of this wind-stream resulted in drying across the southern parts of mainland Australia and Tasmania. As the SAM has switched between positive and negative phases over the twelve month period from mid-2016 to mid-2017 (Bureau of Meteorology, 2017), this goes some way to explaining how the more recent negative IOD, during which rains would have normally been experienced across most of Victoria, resulted in less rainfall than expected. Even though species are mostly well-adapted to short-term variability in climate, they are not necessarily adapted to longer-term shifts in mean climate and the increased frequency and intensity of extreme events (Hennessy et al., 2007).

1.2.7 Anthropogenic changes to Australia's fire regime

While many believe that Aborigines burned bush on a regular basis (Jones, 1969, Gammage, 2011), it is questionable whether the frequency and extent matches that of burns, both bushfire and prescribed, of present times. In fact, what has been highlighted is that distinct changes in fire regimes have occurred with the arrival of Europeans approximately 200 years ago (Lindenmayer, 2007, Mooney et al., 2010). Bickford et al., (2008) have shown in a region of South Australia that sedimentary records indicate a change has occurred in the vegetation species composition. It is suggested this occurred during early European settlement (Bickford et al., 2008). Fire regimes have been significantly altered with the length of the fire season extended, resulting in more extensive burning, more frequent fires and significant changes in the spatial patterns of fires (Steffen et al., 2017). This is in contrast with the patchy fire regime carried out by the indigenous people prior to European settlement (Gammage, 2011).

1.2.8 The political context behind current prescribed burning protocol in Victoria, Australia

Prescribed burning, as a means of managing vegetation growth processes and fire fuel loads, has global application across different landscapes. Mediterranean land managers across southern Europe apply prescribed burns to ameliorate the effects of bushfire and for managing habitats for grazing and wildlife (Fernandes et al., 2013). South African conservation areas of savanna ecosystems use prescribed burns to conserve biodiversity (Brockett et al., 2001); in the USA, prescribed burns are applied to remnant prairie patches to restore ecosystem integrity (Brudvig et al., 2007) and in the US eastern deciduous forests, to improve the sustainability of oak forests (Artman et al., 2005).

Victoria has a prescribed burning program implemented by government agencies as per the *Code of Practice for Bushfire Management on Public Land* (Department of Sustainability and Environment, 2012). The operational and planning principles are underpinned by two objectives: 1) “To minimise the impact of major bushfires on human life, communities, essential and community infrastructure, industries, the economy and the environment. Human life will be afforded priority over all other considerations” and 2) “To maintain or improve the resilience of natural ecosystems and their ability to deliver services such as biodiversity, water, carbon storage and forest products” (Department of Sustainability and Environment, 2012).

Prescribed burning is a practice that has evolved over time in Australia and was reinforced after the Victorian bushfires of 1939, 1983 and 2009, where all three of the major fire events were instrumental in the evolution of the prescribed burning process. Beginning in 1940, the Stretton enquiry into the 1939 fires responded with a policy of fire suppression. Over time, the prescribed burning protocol was modified to the point of creating, as a legal base, The Forests Act 1958 (Department of Sustainability and Environment, 2009). In 2010, the Royal Commission investigating the Black Saturday bushfires of 2009 developed Recommendation 56, which suggested an increase in the level of prescribed burns in the Victorian landscape (Teague, 2010). It was recommended that an increase in burns, via an area-based quota of prescribed burns, be applied to the State, in addition to the area burnt in bushfires. This recommendation was subsequently legislated by the State Government. In 2015, the area-burn protocol was replaced with a strategy-burn policy - "..strategic bushfire management plans that outline activities to implement the bushfire management strategies.." (Department of Environment Land Water and Planning, 2015). The intention is to manage the landscape for risk to human life and assets, as opposed to burning the landscape to achieve an area-burn quota.

Of importance is the point that crises can impact evidence-based policy. The Royal Commission, in investigating the bushfires of 2009, examined each of the fires that occurred on Black Saturday (Teague, 2010). Their investigations included the effects of prescribed burns on two of the locations that burned that day. The Kilmore-East fire was not impeded by a prescribed burn that was carried out in that location more than three years prior, whereas the Kinglake area had been burned by a bushfire in 2006 and the rate and spread of the Black Saturday fire was considered to have been greatly reduced. The Royal Commission determined that, on Black Saturday, the effects of the bushfires were limited by prior bushfires or one-year old fuel reduction burns. However, a recommendation to increase the area managed across Victoria with prescribed burns was presented and the State Government of Victoria then implemented a burn program, to meet the stipulated burn quota. This is evidence that a crisis may drive a policy that protects people first and demotes ecological values. Ever since the Stretton enquiry into the 1939 bushfires (Stretton, 1939) Victorians have grappled with the issue of increasing prescribed burns (Ellis et al., 2004, Pyne, 2006, Attiwill and Adams, 2013). Some people questioned the potential effectiveness of such an increase in the fuel-reduction target after the 2009 bushfires, thereby there was a divide in opinions between the community, ecologists and key land-management agencies (Attiwill and Adams, 2013).

1.3 Aspects of the fire regime that limit bird resources

Evidence highlights that bushfires are occurring more frequently, with both the frequency and severity of fires expected to increase (Fairman et al., 2016, Steffen et al., 2017). The application of prescribed burns is an attempt to reduce the impacts from severe, extreme and extensive bushfires. Driscoll et al. (2010) highlight that knowing how species respond to fire regimes is crucial and that sustainable management requires knowledge based on “(i) a mechanistic understanding of species’ responses to fire regimes; (ii) knowledge of how the spatial and temporal arrangement of fires influences the biota; and (iii) an understanding of interactions of fire regimes with other processes”. The synergies that occur, because of a changing climate, add a level of complexity to these processes.

Whilst much ecological fire management in Australia is built on the ‘pyrodiversity begets biodiversity’ premise (Parr and Andersen, 2006), meeting the needs of flora will not necessarily result in the needs of fauna being met (Clarke, 2008). Habitat requirements for avifauna include structural features such as hollows, which are not necessarily relevant to conserving floral diversity (Loyn, 2012). Further to this, birds have territory, nesting and food requirements (Legge and Cockburn, 2000, Debus, 2006) and these resources may become limited as fires vary in frequency and severity in an already fragmented landscape.

Literature on the effects of fire on bird communities is extensive (Woinarski, 1999, Saab and Powell, 2005, Leidolf and Bissonette, 2009), and avian responses to fire in the south-east region of Australia continue to be a topic of widespread interest.

In the forests of south-eastern Australia, a landscape impacted by a dynamic changing climate, recent research has often been focused on different vegetation types. Eucalypt Foothills Forests are a major vegetation category across Victoria (south-west, central and north-east regions), and similar to Heathy Dry Forests in as much as Eucalypt Foothills Forests are Heathy Dry Forests with wetter, denser gullies (Cheal, 2010). There has been considerable research on the Eucalypt Foothills Forests (Loyn et al., 2003, Loyn and McNab, 2015, Haslem et al., 2016, Leonard et al., 2016, Kelly et al., 2017). In recent years there have been extensive efforts to examine bird responses to multiple fire events, with two major works considering multiple faunal species (Leonard et al., 2016, Kelly et al., 2017). Most recently Kelly et al. (2017) published extensive work on birds, mammal and vegetation distributions based on factors related to the fire regime. These comprehensive assessments highlight the importance of moving from research based on single fire events to more comprehensive fire regime assessments, as we move into an era where the elements of a fire regime are rapidly changing, and modelling becomes more complex. Further studies based on south-east Australian landscapes have focused on both the semi-arid region (Brown et al., 2009, Taylor et al., 2012, Taylor et al., 2013, Watson et al., 2012) and more recent studies have investigated fire impacts on avian species in mixed vegetation of damp scrub, heathland and tall mixed woodland in this region (Sitters et al., 2014a, Sitters et al., 2014b, Sitters et al., 2015).

Despite numerous studies, limited research has been carried out on the impact of fire on the birds of the Heathy Dry Forests across western and central Victoria. As this region is heavily impacted by rainfall anomalies and is subject to climate drivers that may impact to a greater degree than they do across the eastern or northern regions of the State (Bureau of Meteorology, 2018), it is imperative for current research of bird responses to fire to be directed to this vegetation type. Furthermore, there is uncertainty regarding the characteristics of planned burns that may influence avian diversity (Sitters et al., 2014a).

What is known of bird responses to fire is that fire will directly impact birds, resulting in flight or death (Reilly, 1991, Gill and Catling, 2002); and indirectly, by altering the habitat resources associated with vegetation structure (Loyn et al., 2003). The vegetation provides the environment for courtship, cues for breeding and sites for nesting; along with predator protection, and food and shelter from inclement weather (MacArthur and MacArthur, 1961, Wiens and Rotenberry, 1981). Hence, birds are sensitive to disturbance within their habitat and, as such, are highly responsive to spatiotemporal environmental change (Loyn and McNabb, 2015, Leonard et al., 2016, Kelly et al., 2017).

As the body of literature around birds and fire research grows, highlighting the complexities inherent in systems impacted by a changing climate, there is a need to explore whether the observations hold for the flammable Heathy Dry Forests, given the importance of this habitat for bird conservation; the “Victorian temperate-woodland bird community” is listed as a threatened community and is therefore protected under the Flora and Fauna Guarantee Act 1988 (Department of Environment Land Water and Planning, 2017). In a decade underpinned by a fire regime that is rapidly changing in terms of not only bushfire frequency but the extent of the application of prescribed burns, further research is crucial. The combination of a deliberate fire management regime, under changing natural fire climate, risks the ongoing security of fire sensitive communities, including forest birds.

A meta-analysis of the fire-bird literature in the United States highlighted the importance of the fire regime in its impacts on population changes in bird species (Fontaine and Kennedy, 2012). Furthermore, recent contributions to our understanding of fire regimes have greatly enhanced our understanding of forest ecosystems (Bradstock, 2012, Fernandes et al., 2013, Kelly et al., 2017). However, Mediterranean landscapes are in a constant state of flux, due not only to regional climate drivers, but as a result of constant land-use change

(Fernandes et al., 2013). Continually updating bird data records, in regions heavily fragmented by numerous drivers of change, is therefore crucial. The challenge lies with working with short-term datasets, as longer and more detailed databases are required to understand change across landscapes where climate is variable.

For avian communities, it is important to continue increasing the level of understanding on where birds of an assemblage feed and forage, and where, when and how they nest. This may vary spatially, with changes in elevation and sub-surface geology, and temporally with seasonal changes and migration. In the event of a fire, prescribed burn or bushfire, there needs to be a greater understanding of the immediate and post-fire responses of birds – the effective connectivity between suitable vegetation patches and how far the affected species will disperse. Therefore, the age and structural components of the vegetation are fundamental to our understanding on bird responses to fire. It has been highlighted that relatively little is known of the relationship between either individual fires or fire regimes and woodland fauna in Australia (Bradstock et al., 2010). But, we know that heterogeneity in the landscape is a key factor in maintaining bird biodiversity (Brotons et al., 2004). So, while individual fire events in Mediterranean regions may not necessarily have negative impacts on birds, it is further aspects of the fire regime such as the severity of fires or the impacts from frequent fires, changing as climate changes, that may be critical to bird biodiversity (Moreira and Russo, 2007, Gil-tena et al., 2009).

The combination of all factors impacting fire regimes is complex (DellaSala et al., 2004). Fire behaviour changes over time and as a result policy changes that were intended to ameliorate the impacts of severe fires may instead result in significant risks to the integrity of ecosystems (DellaSala et al., 2004, Teague, 2010).

1.4 Forest bird responses to the most recent fire

Ecological succession describes the sequential processes of changes in plant species growth and development following a disturbance such as fire (Clements, 1916, Connell and Slatyer, 1977). Further, it provides a theoretical framework for investigating changes to faunal diversity in a landscape impacted by disturbance. The 'habitat accommodation' model of post-fire succession suggests that species respond to local habitat conditions that have been altered by factors such as fire. It states that species will reach peak abundance levels when the habitat fulfils their requirements, and leave or decrease in abundance when the habitat becomes unsuitable (Fox, 1982). As there is an uneven application across the landscape of fire, both spatially and temporally, successional stages create a mosaic of vegetation ages in any given area. Therefore, variable fire regimes will benefit bird biodiversity to the degree that the vegetation species present will provide resources specific to each successional stage (Bradstock et al., 2005, Parr and Andersen, 2006).

The south-east Australian woodlands are not ecosystems in equilibrium, but rather, heterogeneous, open systems which are in a dynamic, non-equilibrium state (Levin, 1999, Phillips, 2004, Moore et al., 2009). Disturbances can significantly enhance environmental heterogeneity at multiple scales and woodland ecosystems are influenced by the attributes of the disturbance regime. In the case of fire, the regime of this disturbance is characterized by the climate, topography, substrate and biota in the region; it varies spatially and temporally; is dependent on specificity (the relationships between the type of disturbance and the attributes of affected sites, including species present, the seral stage of the vegetation community and its location); and it will vary in magnitude (including intensity, severity and frequency).

Brotons et al. (2004) highlighted the importance of maintaining heterogeneity in the landscape for maintaining bird diversity in this region of warm summers (Piñol et al., 1998) and increasing fire frequency. They argue that in this region, either bushfire or prescribed burns are favourable, (to maintain landscape heterogeneity), provided they are of a moderate size. Indeed, they further suggest that, in Mediterranean regions where bushfires have drastically fragmented the landscape, the fragments are essential for avian species to persist. In south-eastern Australia, fire disturbance occurs naturally as bushfires, and anthropogenically as prescribed burns, leading to a great degree of unpredictability over area, space and time of burn (Moore et al., 2009).

Despite a widespread advocacy for creating a heterogeneous landscape, to maintain bird diversity (Brockett et al., 2001, Brotons et al., 2005, Brudvig et al., 2007), further research on avian communities clearly illustrates that the extent and mix of vegetation age classes is necessarily specific to faunal species and geographic locations. Research in the Mallee region of Australia highlighted how, in that semi-arid area, management strategies that promote mid to older vegetation are of a greater benefit to bird species; burning to create a diversity of vegetation age-classes would negatively affect more birds than it would aid (Watson et al., 2012, Taylor et al., 2012, Taylor et al., 2013). However, avifaunal thresholds within fire-age classes have rarely been examined; much empirical research has been based on only vegetation thresholds (Driscoll et al., 2010).

There is a paucity of long term bird data within fire research (Ford et al., 2001, Bennett and Watson, 2011). Furthermore, much research is based on short time frames of two to three years of data collection. Therefore, a great deal of the modelling of bird responses to changes in the vegetation is in terms of time-since-fire; where an interval of time has lapsed since the last disturbance and the vegetation develops across seral stages. Whilst the inter-fire interval (the time between fire events) is a strong influence on flora growth and development (Enright et al., 2015), time since the last fire often strongly influences the distributions of the more mobile species such as mammals and birds (Catling et al., 2001, Kelly et al., 2017). This is because time-since-fire influences ground cover, and bark and canopy levels in the trees (Haslem et al., 2016) and thereby impacts food, nesting and protection resources for birds.

While recent studies have shown that habitat variables (such as the extent and number of species forming ground cover), can provide greater predictive power than relying on fire variables alone (Sitters et al., 2014b), time-since-fire is useful in its utility as a predictor variable, as birds forming part of a guild will respond to multiple vegetation features across a location. In fact, Kelly et al. (2017) highlight the importance of time-since-fire in their recent research in Victorian Foothills Forests. They found that there were distinct preferences by some species to both older and younger vegetation age classes. An example was the Flame Robin (*Petroica phoenicea*), whose populations, peaked in probability of occurrence in early successional stage vegetation. This response aligned with earlier research where this species has been reported at higher frequencies following several large fires in south-eastern Australia (Loyn, 1997, Lindenmayer et al., 2014, Loyn and McNabb, 2015).

Since 1999, fire management planning in Victoria, Australia has moved towards integrating ecological guidelines into the fire management process (MacHunter et al., 2009, Cheal, 2010). In 2006 a project was initiated assigning tolerable fire intervals to vegetation throughout Victoria. The document known as *Growth stages and tolerable fire intervals for Victoria's native vegetation data sets* (Cheal, 2010) groups similar ecological vegetation classes across the State of Victoria into ecological vegetation divisions – EVC's into EVD's - and details the growth stages, post-fire, for each EVD. Research can vary in its form of time-since-fire measure (Di Stefano et al., 2011, Sitters et al., 2014a) based largely on the broad-scale vegetation type. Therefore, while acknowledging that much valuable research considers vegetation age with a continuous 'years since fire' measure, it may be useful to consider that the Cheal classification system may be appropriate to use for research across a region, a further tool for research comparisons between areas. Reviewing the current research published for Victoria, it would seem that Cheal's classification has not been validated for its utility in measuring faunal responses and biodiversity changes. As more emphasis is placed on incorporating faunal changes in the landscape into management policy and procedures, it is prudent that the classification system currently accepted for broad vegetation age classification in the Victorian region, be reviewed in terms of bird responses.

1.5 An increase in fire frequency and what this means for forest birds

1.5.1 Predictions on future fire activity

Research on a global historical reconstruction and interpolation of fire events back to the 1900’s was based on fourteen regions of the world (Mouillot and Field, 2005). Temperate forests were described for Europe, the United States and Australia. The authors argued that, despite an increase in investment in fire prevention in the southern European region, fires there have increased in frequency from the 1960’s. Further, they highlighted how fires in the western United States have increased in frequency from the 1970’s, possibly as a result from fuel accumulation over 40 years of fire suppression. For Australia, the authors discuss the changing fire regime, a result of European settlement, as opposed to Aboriginal burning practices. The brief reference to fire management policy changes fails to adequately reflect the complexities surrounding fire in this temperate landscape. Nonetheless, their research highlights fire frequency increases across temperate regions right around the world (Mouillot and Field, 2005).

It is predicted that in Australia, the Forest Fire Danger Index (FFDI) (an index incorporating a drought factor, created by McArthur in the 1960’s) for the south-eastern region will increase strongly by 2100 (Clarke et al., 2011). Every model simulation produced in this research generated scenarios suggesting that fire seasons will start earlier in the year, leading to a longer overall season, accompanied by more severe fire weather, with the biggest increases in FFDI consistently driven by temperature increases.

Victoria, Australia has experienced an increase in ‘mega-fires’ (Attiwill and Adams, 2013), those fires that burn more than 100,000 hectares (Cruz et al., 2012). From 2003 through to 2014, 4.3 million hectares of eucalypt forest burned in Victoria, equivalent in extent to the cumulative area burnt in the region in the 50 years prior (Fairman et al., 2016). The FFDI increased markedly in the period from 1973 to 2010 (Clarke et al., 2013), and is projected to increase further in the coming decades, with the largest increases noted during the autumn and spring seasons (Clarke et al., 2013). This will result in an even greater increase in the frequency of bushfires (Bradstock, 2010, Clarke et al., 2013, Flannigan et al., 2009). The recent increase in bushfire activity in Victoria resulted in 350,000 hectares of forest burning twice, with a short interval between fires. A short fire interval between severe fires may impact both the obligate seeders that need fire to regenerate and the fire tolerant re-sprouters (Fairman et al., 2016). Furthermore, this may result in shifts from temperate forest communities to shrublands or grasslands (or at the very least a shift to unstable states). While there is a potential for a shift across ecological states within the temperate forest ecosystem, that much remains unknown about whether the current woodlands have crossed a threshold to an alternate state (Fairman et al. 2016). While planned burning is applied to the landscape in an attempt to ameliorate the impacts of bushfire, the result may be an increase in the combined area burnt either deliberately or through bushfire.

The four weather systems in south-east Australia: the PDO, IOD, ENSO and SAM, are crucial drivers of fire in the south-east Australian landscape. Research has highlighted that positive IOD events precondition the landscape for bushfires in south-eastern Australia (such as Black Saturday in 2009) (Cai et al., 2009) as do positive SAM events (Mariani and Fletcher, 2016), both of which exacerbate dry conditions and increase fire fuel load across the landscape. Whilst an increase in rainfall will result in an increase in fire-fuel in dry regions, drought will dry out fuel in forests. Furthermore, in attempting to identify the specific drivers behind a changing fire regime, the process becomes complex when so many different factors are combined (DellaSala et al., 2004).

1.5.2 Fire management processes to include fauna

Considerable work has been carried out to determine appropriate protocols for prescribed burning, implemented in attempts to reduce the impacts of bushfire severity. Efforts have been centred on the need to increase the effectiveness of prescribed burning in terms of biodiversity (Penman et al., 2011) and, more specifically, to review the process of patch mosaic burning (Parr and Andersen, 2006). However, what is clear is that, for prescribed burning to be effective as an applied management tool, better informed and more clearly defined ecological values are required, in terms of meaningful and measurable objectives (Penman et al., 2011). This is best as part of an adaptive management approach (Walters and Hilborn, 1978), incorporating faunal ecology into management processes (MacHunter et al., 2009). However, there is limited information available on community assemblage responses to long-term fire planning processes (Wittkuhn et al., 2011). Adding to this, much bird conservation activity is focused on the classification and management of species at risk of extinction (Wilson et al., 2011), which excludes other species within the assemblage and does not consider overall community conservation. Spatial and temporal variations in bird abundances may suggest an increase in numbers in one location which is, in fact, masking an environmental issue in another. An example of this is illustrated by the spatial and temporal abundance differences across the range of the superb parrot (*Polytelis swainsonii*) (Manning et al., 2007). While this research highlighted a strong positive relationship between plant productivity and bird abundances, the responses by the Superb Parrot to year to year seasonal and climate variability differed across regions.

Adding yet another level of population assessment complexity, stochastic events such as climatic conditions at the time of sampling can make it difficult to make sound inferences about species population growth from short term studies (McNamara and Harding, 2004). Therefore, research needs to focus not only on goals for individual species but on trends across broad regional and national scales (Collen et al., 2009, Ford, 2011). This is essential when State-wide policies are required in management processes, as they are for prescribed burning. Therefore, to determine evidence of bird responses, studies based on larger scales are required; necessarily referencing guilds and assemblages as opposed to a focus on just single species. Studies based on whole communities may then include the results of species' interactions. Highlighted as imperative, rather than a species by species approach, research objectives need also to plan for the conservation of whole avifaunal communities (Recher, 1999).

Underpinning our knowledge on the effects of applying intensive or frequent prescribed burns to a location, is the understanding that fire has spatial and temporal complexities. There is a fundamental need to incorporate ecological values into prescribed burning protocols (Penman et al., 2011). Fire management in Australia has, to date, worked on the assumption that meeting the needs of plants will automatically meet the needs of associated fauna (Clarke, 2008) i.e. have plants – animals will come. There is a further assumption that ‘pyrodiversity begets biodiversity’ (Parr and Andersen, 2006). Neither of these assumptions give due consideration to the fact that increasing fire frequency in a location may, in fact, alter the vegetation structure and species composition which, in turn, may alter the available nesting sites and food availability for birds (Bradstock et al., 2010).

1.5.3 Bird responses to fire frequency

Prescribed burns are applied to the landscape, against a backdrop of an increase in bushfires. It is therefore prudent for a shift from research focused on only single fire events to a whole-of-landscape approach, specifically one that considers frequent fires (e.g. Leonard et al., 2016, Kelly et al., 2017). Even though bushfire has long been recognized for its importance in the Australian landscape, the impacts from increasing fire frequency in the Heathy Dry Forests of Victoria are not known. However, research has been carried out in Eucalypt Foothills Forests on effects of fire frequency on vegetation (Haslem et al., 2016) and bird responses (Leonard et al., 2016, Kelly et al., 2017).

While research investigating faunal responses to the increase in fire frequency is flagged as important (Woinarski and Recher, 1997) because the effects of fire frequency are less well understood (Leonard et al., 2016), current work has offered valuable insights into bird responses to frequent fire. Fire frequency was flagged as the most influential component of the fire regime (Kelly et al., 2017), for frequent fires can remove coarse woody debris (Aponte et al., 2014) which removes essential resources for foragers such as those that feed on damp ground (Leonard et al., 2016). It is the damp ground foraging guild birds that are impacted by areas burning three times within 40 years in Eucalypt Foothills Forests (Leonard et al., 2016).

Morrison et al. (1995) highlighted that the increase in variability of the length of the inter-fire intervals was associated with an increase in the species richness of both the fire-sensitive and fire-tolerant vegetation. Therefore, it is the variation in inter-fire interval that results in a wide variety of plants. Furthermore, there is a global shift to more frequent fires and knowledge gaps exist relating to the development and persistence of alternative vegetation states driven by changes in inter-fire intervals (Fairman et al., 2016). Further, repeated, low intensity prescribed burns have been found to reduce coarse woody debris in temperate eucalypt forests (Aponte et al., 2014), an important structural component of many forest ecosystems (Carmona et al., 2002). Therefore, examining responses by birds to repeated fires will determine whether any of these vegetation changes are impacting their abundance and persistence.

Research explicitly states that the need to increase quantitative data on faunal studies combining bushfire and prescribed burns is imperative (Lindenmayer, 2007; Adams and Attiwill, 2011). Bradstock et al. (2010) argue that while effective use of fire can promote woodland regeneration, it can also lead to the eventual decline of woodlands. This is because changing fire intervals may impact the survival of vegetation species. So, while time-since-fire will change the immediate structure, a change in inter-fire interval may result in a change of the stable state of the vegetation, or a complete removal of a plant species from a landscape (Morrison et al., 1995, Fairman et al., 2016). As birds are responding the structure of the vegetation, it was deemed important in this research to investigate whether birds in the Heathy Dry Forests are responding to different fire frequencies in the landscape. Therefore, increasing data on bird responses to fire frequency is paramount.

The research by Kelly et al. (2017) is crucial in not only highlighting the value and importance of a fire regime approach, but in highlighting bird responses, specifically to the interval between fires. They state that their findings of the inter-fire interval reflecting a greater response than time-since-fire (for some species), was in fact unexpected, and suggest that recurrent fires may produce a different set of ecological conditions, such as altered species composition and vegetation structure. Differences in bird responses to single fires compared with responses to repeated fires has also been found in the United States by Fontaine et al. (2009). They investigated bird responses to single and repeated high severity fires in the conifer forests in a region of Oregon and showed that birds responded differently to repeat fires than they did to single fires. Their results showed that repeated high-severity fire did not reduce species richness, and that bird densities were greater in repeat burns than in once-burned habitats. These results highlight the complexities that arise when attempting to unravel the factors driving changes to bird assemblages in dynamic environments.

The value of a review of responses to fire frequency, as opposed to reviewing responses to only a single fire event, can be highlighted in research by Loyn et al. (2003). In this investigation on the effects of repeated low intensity fires on birds in Eucalypt Foothills Forests, it was highlighted that no factor impacting birds operates in isolation. Burn sites were small and surrounded by forest with a mosaic of fire histories and the patchy forest around the sites may have been important for the dispersal of birds during a fire event (Loyn et al., 2003). Further adding, it was not known how species would respond if broad-scale burning was large enough to eliminate surrounding patches of unburnt vegetation. However, Loyn (1997), in assessing an East Gippsland avian population post-fire, suggested that nearby unburnt forest has a limited capacity to absorb displaced birds, as the increase observed in nearby forest, in that project, was only 10%. While the 2003 study highlighted that effects of fuel reduction burning were milder than those of severe bushfire, it was noteworthy that the most common species in this forest type were the canopy dwellers, and these were removed from the effects of small scale, groundcover reduction burns. These studies suggest that issues for birds in these areas do not arise from a one-off, low intensity fire event. Some birds forage on open, cleared ground immediately post-fire, and others forage in the epicormic growth in the next growth stage of the vegetation within the first three years. Some, such as those birds living in the canopy layers, may not be adversely affected at all by low intensity, ground burns. The question to consider is whether there is an appropriate, or required, fire frequency regime for the continued survival and existence of all the forest bird species.

1.5.4 Variations in how fire frequency is measured

Determining the fire history in vegetation is handled differently in Australia when compared with the USA. In the USA the process of dendrochronology is widely used and is extremely useful in reconstructing fire history. There is, however, a persistent belief in Australia that most eucalypt species are unsuitable for dendrochronological analyses and therefore research interest has been limited (Brookhouse, 2006). Instead, fire history is reconstructed in Australian environments by 1. sedimentary charcoal studies, and 2. reviewing the vital attributes of vegetation species present (Noble and Slatyer, 1980). Charcoal studies are rarely of sufficient resolution either temporally or spatially and so, for ecological assessment, vegetation attributes are more informative.

In Victoria, Australia, the vegetation classification system by Cheal (2010) not only ages vegetation in terms of time-since-fire, but also suggests a tolerable fire interval for each vegetation grouping, the minimum time required for vegetation in the community to persist, before fire once again occurs in the landscape. The tolerable fire interval is deemed to be different for the time required after a bushfire, as opposed to the time required after a prescribed burn. This difference is based on the assumption that prescribed burns are less intense and will remove a lesser amount of the vegetation from the environment. To date, the tolerable fire intervals have not been measured in terms of their usefulness as measures for faunal communities.

Research varies in how the inter-fire interval, or fire frequency, is accounted for in modelling. Kelly et al. (2017) used, for an inter-fire interval, a mean measure of the years between each fire. Leonard et al. (2016) used a count; the number of fires on a site in the 40 years prior. Loyn et al., (2003) reviewed the effects of repeated, low intensity fires on the avian community in Victoria, Australia. Five fire treatments were considered: frequently burned (spring), frequently burned (autumn), infrequent fires (spring), infrequent fires (autumn), unburnt. Frequent fires were defined as occurring approximately every three years, and infrequent, approximately every 10 years. On this basis, the more recent work by Cheal (2010) suggests that the minimum fire interval required for vegetation to recover, post-fire, in the vegetation class investigated by Loyn, is 10 years post low intensity fire and 15 years post high intensity fire. The question then arises as to whether the frequent and infrequent groupings should, in fact, be one ‘frequent’ grouping. While the discussion in Loyn’s research suggested there being little variation between the different treatments, the outcome may have been different if the grouping of frequency classes had been in line with the Cheal classification. It is, therefore, prudent that the tolerable fire intervals of Cheal be reviewed in terms of faunal responses i.e. determine if birds are responding in time frames reflecting the tolerable fire intervals Cheal has outlined for vegetation growth post-fire. This is essential if management is to invoke decisions impacting fauna based on the tolerable fire interval guidelines.

There is an extensive volume of research completed on predicting future fire activity in regions around the globe (Carmona-Moreno et al., 2005, Mouillot and Field, 2005, van der Werf et al., 2006, Schultz et al., 2008, Tansey et al., 2008, Flannigan et al., 2009), including predictions for a high likelihood of significant increases in fire risk across Australia (Pitman et al., 2007). Indeed, there is a mounting body of evidence on the impacts from different aspects of fire on avian species. Despite this, for the Heathy Dry Forests of south-east Australia, forests within a region flagged as being a crisis zone for bird biodiversity, there is a paucity of research on the responses by birds to multiple fires.

1.6 Response by birds to the impacts of prescribed burn severity

The prescribed burn process in Victoria, Australia follows a management protocol that incorporates an adaptive management approach to fuel assessments (Hines et al., 2010) with burning processes underpinned by available knowledge on fire behaviour, fire weather and topography, all impacting prescribed burning decisions (Tolhurst and Cheney, 1999).

There is literature available on the extent to which prescribed burns can be used to minimize biodiversity loss (Penman et al., 2011), and studies have been published on bird responses to prescribed burns (Loyn et al., 2003, Loyn and McNabb, 2015, Sitters et al., 2015). However, what is evident is that an amount of research and subsequent literature assumes that prescribed burns are reducing, not removing, all vegetation (hence fire fuel) in the environment (Moreira et al., 2003, Penman et al., 2007, Pons and Clavero, 2010, Sitters et al., 2015, Kelly et al., 2017), or have a measure of intensity as opposed to severity (Brockett et al., 2001). Fire intensity and severity differ, in that intensity is measured in terms of the output generated by fire and severity is measured in terms related to the impact from the fire on the surrounding landscape. Implicit in this may be the expectation that most prescribed burns are of low severity and controlled, with no acknowledgement that fire managers sometimes err and some prescribed burns are, indeed, severe. With the increased emphasis on prescribed burning application, the assumption is made that the outcome is positive for the environment without considering that the prescribed burn process itself may be detrimental (Fernandes et al., 2013).

One study in Portugal pine forests determined that there were no significant differences in bird abundance and richness from plots burnt by prescribed burns compared with controls (Moreira et al., 2003). This research was then referenced as indicating that the extensive use of prescribed burning to reduce understory has shown to have limited effects on birds in pine habitats (Pons and Clavero, 2010). Yet, neither paper refer to the severity of the prescribed burns being examined in Portugal; they may in fact have all been quite moderate in severity. The original research was clear in its explanation that burns were mostly 'extremely small' and that vegetation structure returned to similar levels as that on control sites five years post-burn (Moreira et al., 2003). The Portugal pine landscape recovered rapidly and may have not been impacted by prescribed burn severity however, in landscapes with longer recovery intervals between vegetation post fire age classes, or where tree hollows are a requirement for faunal species (Lindenmayer et al., 2013), severity may be an important factor.

Bushfires are commonly high severity burns and much research into the impacts from fire of this level of severity are undertaken *post-hoc* (Loyn, 1997, Robinson et al., 2014). The assumption may be that large fires can and do comprehensively burn the landscape. While this may be the case, not all bushfires burn homogeneously, not all fires remove all resources essential to fauna (Bradstock, 2008). In these cases, the bulk of biota are found to be reasonably resilient due to *in situ* landscape factors (Bradstock, 2008). In fact, the importance of unburnt vegetation remaining within bushfire zones has been highlighted (Robinson et al., 2014).

Where vegetation changes dramatically post-fire (in terms of vegetation species composition), so too will the bird community change. For example, Leavesley et al. (2010) reviewed bird assemblages post-fire in mulga woodland in the Central Australian arid zone. In this instance, the early seral stage of the woodland community was effectively one of grasslands, therefore the bird community was characterised by granivores, whereas the older, long-unburnt vegetation was characterised by foliar insectivores. It may be expected that after severe bushfires, community assemblage patterns will change, a result of the extreme change to vegetation structure. However, as research has shown, these changes may not be consistent across a landscape. Brotons et al. (2005) studied avian populations post bushfire across a regional area of the north-eastern section of the Iberian Peninsula and found that bird assemblages on sites immediately post-fire varied across a regional area. They suggest that the significant differences in post-fire bird assemblages were a response to regional scale dispersal factors impacting the colonization of habitats post bushfire. The challenge is in determining which factor is resulting in a bird response – fire severity or variations across the landscape. To model fire severity responses by birds, such that the severity factor is independently tested, is an important contribution to our knowledge of bird responses to fire.

In the Victorian Heathy Dry Forests, bushfires are invariably of greater size than managed prescribed burns. This is largely a consequence of the fact that the region is heavily fragmented, with Heathy Dry Forests existing as patches across the landscape, as opposed to continuous expanses of vegetation. Furthermore, prescribed burns are controlled. Therefore, it is feasible to consider that the severity of the burn will only impact fauna to the degree that required resources are removed.

1.7 A case for following the trends of common birds

1.7.1 Monitoring birds to aid detection of drivers of decline

Bird abundance trends directly reflect landscape change. Furthermore, most bird species are easily recognized and monitored and there is abundant available data from not only professional ornithologists and ecologists, but from a huge volunteer base of reliable observers e.g. eBird, Birdlife Australia. As such, the monitoring of birds yields trends that may serve as environmental indicators – declines in abundances are clearly highlighting an environmental threat and may be signalling potential extinction threats.

1.7.2 The monitoring of common birds as a tool for determining land health

Conserving and maintaining the landscape is necessary; to ensure that ecosystem functions are maintained, such as the protection of watersheds and soil, and the preservation of breeding grounds (Brundtland, 1987). As such, determining the health of the landscape is fundamental in this pursuit.

Birds are present in all regions of the globe and have long been recognized as efficient indicators of environmental health (Birdlife International, 2008, Drever et al., 2008, Larsen et al., 2010, Szabo et al., 2012, Birdlife Australia, 2015). In light of this, Birdlife International has produced and updated comprehensive bird assessments, generating the volume *State of The World's Birds* (Birdlife International, 2008).

Recognizing that, with frequent land-cover modifications, the generalist avian species are favoured (Wiens, 1989), further studies have added efforts to the fight against halting biodiversity decline by illustrating how common bird abundance measures are useful surrogates for determining environmental health across European countries (Gregory et al., 2005). Birdlife International utilises this data, the Pan-European Common Bird Monitoring Scheme (PECBMS) (European Bird Census Council, 2017), within their *State of The World's Birds* reporting (Birdlife International, 2008). The total of 167 species forming the PECBMS is split into common forest or common farmland groupings, and then divided across regions, such that each grouping of common taxa for a region has approximately 27 species (European Bird Census Council, 2017). Count data are then used to create indices and indicators, which are made available to not only Birdlife International, but to policy makers, scientists and the public (European Bird Census Council, 2017). The indices produced are the key outputs for various policy purposes; to effectively measure land-use and development impacts across the European Union. Two of these policies (but not limited to) are: the *Streamlining European Biodiversity Indicators*; and the *Indicators of Sustainable Development of the EU*. The indices are also biodiversity indicators for the *EU's Structural Indicator*; and form part of *The Regulation in EU's Rural Development Plans*. However, whilst the common bird indices may be useful as tools in management decisions, the challenge is in determining which factors have contributed to trends, as changes to climate precipitate changes to both the fire regime in an area and the regional climate.

1.7.3 The common species as a component of community networks

Common species represent a crucial component of community networks in terms of food web structure (Tylianakis et al., 2010). The generalist species will have the greatest measurable connectivity within the food web and a removal of any of these may lead to cascading, secondary extinctions (Dunne et al., 2002). Understanding the population trends of the generalist species may, therefore, guide us to more effectively managing the entire community, especially when considering management processes such as prescribed burn protocols (Bascompte and Stouffer, 2009). When a habitat becomes simplified, as it may post-fire, the specialists may be lost, or only return sporadically. However, a focus on the generalists will allow for the gathering of a statistically significant data set in order to understand a ‘baseline’ for fire-plant-bird interaction analysis. Owing to this role, where a community may be resilient to the random extinction of specialist species, it may be vulnerable if generalist species come under threat of extinction (Bascompte and Stouffer, 2009). In a fire-prone landscape experiencing changes to fire frequency, a potential exists for changes to occur to the structure of the network of mutualistic (e.g. plant-frugivore) and antagonistic (e.g. plant-herbivore) interactions between species; beyond changes to considering only the biodiversity measure of species richness within that assemblage. Variability in the numbers of rare birds may result in their absence, whereas the common species can be measured in terms of range contraction.

1.7.4 The value of common species in rapid assessments

As much ecological research is based on data collected over short time frames (e.g. 2-3 years) and over limited spatial scales, extending this to a whole-of-landscape approach is challenging in terms of both time and cost. One possibility to overcome this is to focus research on selected species from within bird assemblages anticipating that they act as surrogates for the entire community. The use of generalist species may be an option to overcome limits to sampling time, as they are the most abundant species and therefore easiest to monitor. Furthermore, faunal data collection is often inadequate when projects are undertaken at a small scale and over short duration (Clarke, 2008). While well-funded, long term studies can explore all species; shorter studies may benefit from a focus on more abundant taxa, important in community structure, to gain a greater understanding on dynamic changes to assemblage patterns as a result of an increase in fire disturbances.

The Atlas of Australian Birds (Barrett et al., 2003) indicates that distributions of many woodland birds had contracted in historical times, including since the previous field Atlas project (Blakers et al., 1984). However, even with over half the Australian woodland bird studies identifying a reduction in birds, there is still a paucity of data on their population trends and the variation in the populations between these two large surveys (Rayner et al., 2014). If limited funds for research were directed to a focus on the more abundant, generalist species, these common species can be used as a proxy for assessing pressures on all species. This is crucial to achieving effective whole community conservation strategies (Rayner et al., 2014).

1.7.5 Common species in adaptive management processes

Efforts to increase our level of understanding on avian species and to therefore, ensure the effective implementation of measures to maintain biodiversity, continue (Russell et al., 2009). A further tool in effective adaptive management is to predict the trends of our common woodland birds, as surrogates for entire bird communities. However, as common species may be more resilient to disturbances such as fire (Koch et al., 2011), implementing an adaptive management approach is useful. Continued monitoring and resultant modifications to management decisions will ultimately result in ongoing monitoring gradually building an understanding of rarer species. The importance here is to be clear on whether the common species are representative of the community.

As ecological systems are inherently complex there will always be a level of uncertainty in decisions made about their management. Hence, adaptive management processes are useful in underpinning resource management decisions (Allen et al., 2011). This places common woodland birds as useful tools, where the focus is in understanding ecosystem processes rather than for measures to rescue individual species. Information on bird community responses to both time since last fire and fire history of the forest landscape is important in contributing to our current knowledge base, and will aid effective management decisions, and be helpful toward determining levels of resistance, and indeed resilience, within the forest avian community in maintaining populations in a flammable landscape (Wittkuhn et al., 2011). Research in 2005 has shown that common farmland birds in Europe had been in decline for two decades (at that point in time), and the authors argued that farmland bird trends were a useful surrogate for trends in other elements of biodiversity within that habitat type (Gregory et al., 2005). Even though they were unable to find evidence to show that the woodland bird indicators were useful surrogates for other taxa in woodlands, they argued that overall, the European common bird indicators are powerful tools and enable valuable information to be communicated to policy makers to inform biodiversity measures and targets applied across Europe (Gregory et al., 2005).

Research has highlighted the extensive knowledge gaps in our understanding of the responses by plants and animals to different fire regimes (Clarke, 2008, Bradstock, 2010, Driscoll et al., 2010, Bowman et al., 2012). Clarke (2008) argued (prior to the events of Black Saturday in 2009), that management of fire regimes needed to move past floristics alone and that faunal species responses to different fire regimes were poorly understood. It is therefore imperative to extend the suite of evidence and test the current assumptions on the expected effects on avifauna from fire in the landscape.

1.8 A changing climate resulting in ‘super wicked problems’

The term ‘wicked problem’ was borne out of the inability to combine definable problems in natural science with those undefinable in social studies (Rittel and Webber, 1973). The term ‘wicked’ is applied to denote a resistance to resolution. The term has been used to describe environmental issues, where a resolution of a situation or conflict does not occur, no matter how much reasoned effort and application of science is provided (Kuchinke, 2000). Underpinning the issue will often be a social or moral conflict, where there is a philosophical objection to a proposed process that may be ill-defined and relying on political judgement for resolution (Rittel and Webber, 1973). Conversely, another example is where ‘action groups’ form, in attempts to create a voice, when policy is perceived to be not representative of either science or the public wish. In these situations, alternative methods or plans need to be implemented to pave a way forward (Kuchinke, 2000).

Following its initial use to describe conflict, there has been a further expansion on the application of the term ‘wicked problem’ with ‘super wicked problem’ (Levin et al., 2007). This type of problem is defined as being one that has four climate change issues underpinning it, namely; time is running out; the authority addressing them is weak or non-existent; that those who cause the problem are also seeking to address it and lastly; the responses are pushed into the future when it is immediate action that is required (Levin et al., 2007).

A changing climate is resulting in an increase in the frequency of bushfires, which in turn leads to a management focus on increasing prescribed burns in the landscape (e.g. Ellis et al., 2004, Teague, 2010). Both these factors combined result in more area of the landscape burnt each year. While the process of applying fire to the landscape is a response by society to manage a perceived increase in risk to property, it could be described as a ‘super wicked problem’. This highlights the imperative of investigating and developing a clear understanding of bird responses to fire.

1.9 Research themes

This thesis explores two inter-related themes (Fig 1.4).

1. What are the common, moderately common and uncommon species within the forest bird assemblage?

2. What are the effects of fire interval (in terms of time since fire), fire frequency and fire severity on the forest birds?

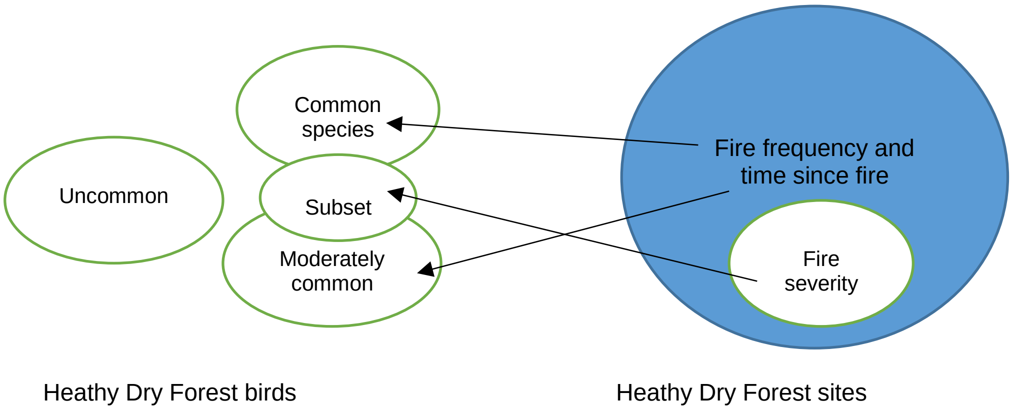


Figure 1.4 Research themes

Research themes of 1) species classification into groupings of common, less common and uncommon, and 2) factors of the fire regime impacting forest birds.

Forest birds are declining in both range and abundance in the south-east region of Australia (Birdlife Australia, 2015). Furthermore, evidence suggests that generalist species are better suited to landscapes impacted by continual change (Watson et al., 2014). Managing for all impacting processes to maintain bird biodiversity is complex. Therefore, determining which are the common species within the forest bird assemblage and reviewing their responses to fire, compared with the responses of the less common species, may determine whether the common species could be further used as indicators of land health in the region.

The second theme explores impacts of the fire regime on the forest birds. The evidence outlined in this chapter highlights the imperative of understanding forest bird responses to fire regimes (Driscoll et al., 2010), in a landscape impacted by numerous, inter-related, changing factors. Investigations on this theme focus on:

i. The common and moderately common bird responses to fire interval in terms of time-since-fire and fire frequency;

ii. The responses by a subset of the complete data set of forest birds to prescribed burns of different severities, from a subset of sites.

1.10 Research objectives

The research reported in this thesis has three general objectives:

1. To quantify forest bird responses to time since fire and fire frequency in terms of individual species and foraging guilds;

2. To quantify forest bird responses to prescribed burn severity in terms of the bird community and individual species' responses;

3. To quantify the responses of common birds compared with those of moderately common birds and therefore determine whether the common birds are representative of the entire assemblage, in terms of individual species and foraging guilds.

Specific predictions are outlined below and described in more detail in the overview and discussions of each results chapter (Chapters 3 and 4).

1.11 Thesis structure

The study area and project methodology are outlined in Chapter 2. This section discusses the region around the monitoring sites, the project design, explanations pertaining to data collection, and the identification of both fire frequency and fire severity rankings. Statistical methods are described for each results chapter.

In chapter 3 the response of birds to time since fire and fire frequency are examined. Discussion focuses on broad community responses. Following this, the common forest species' responses are compared with those species less common, to determine whether common bird responses are representative of the entire assemblage. There are three broad predictions: 1. that early post-fire vegetation in Heathy Dry Forest is characterised by distinct structural elements in the vegetation and, as such, will support distinct bird assemblages, 2. that there will be distinct individual species' responses to time since fire, a result of the immediate impact of fire on vegetation structure, and 3. that there will be little evidence of changes to assemblages based on fire frequency.

In Chapter 4 the response of birds to prescribed burns of different severities is examined using a Before-After Control-Impact project design. This was measured for the bird community in terms of alpha and beta diversity. Further, a group of individual species' responses to prescribed burn severity was modelled. Two general predictions were made regarding birds' responses to prescribed burn severity: 1. that an increase in severity would result in a short-term decrease in species richness; and 2. that there may be an increase in species turnover in the first year post-fire.

Chapter 5 is a synthesis of the main findings and reviews the results in terms of the contribution this research makes towards conserving bird biodiversity in forests of south-eastern Australia.

Chapter 6 is a discussion on the possible future for birds of the south-east region of Australia.

Plate 2 The most common bird of Heathy Dry Forests, Buff-rumped Thornbill, *Acanthiza reguloides.*

Source: Dean Ingwersen, Birdlife Australia

2 Study area and project methodology

2.1 Study area

2.1.1 Site locations

The study area, in this project, was located in the Central Victorian Uplands bioregion in central southern Victoria, Australia. The study area lies south of the township of Creswick (37.42^{o}S, 143.89^{o} E), east of Linton (37.67^{o}S, 143.52^{o}E) and includes, along its south-east edge, the Brisbane Ranges National Park (37.88^{o}S, 144.26^{o}E), (Fig 2.1). The region has a temperate climate with a winter mean monthly temperature range of 3.2^{o}C (daily minimum) to 13.8^{o}C (daily maximum). The equivalent mean monthly summer temperature range is 8.8^{o}C to 27.8^{o}C (Bureau of Meteorology, 2017). Rainfall recorded at Durham Lead (central in the study area) is representative of the region with an annual mean of 830.5 mm from 2009-2012 (Bureau of Meteorology, 2017).

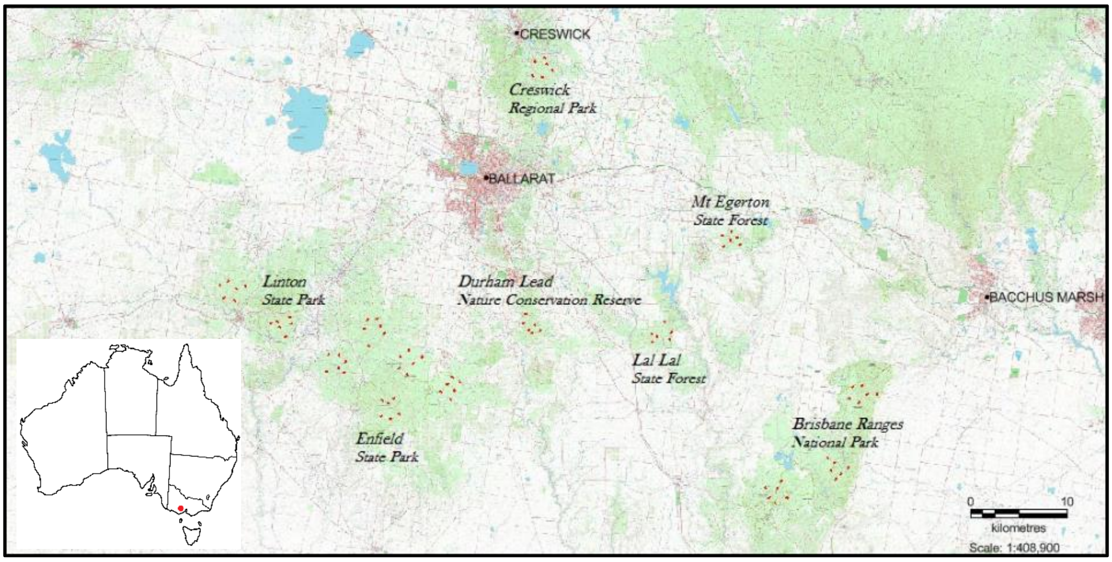


Figure 2.1 Site locations in Victoria, Australia showing 84 sites in 14 groups of six.

2.1.2 Regional climate

Victoria has been on a drying trend since the 1970's (Fig 2.2).

The Millenium Drought (1996-2010) affected most of southern Australia. Central and western Victoria experienced low rainfall throughout this period. Whilst the drought broke in 2010, both the central and western Victoria regions then continued on a drying trend throughout 2012 – 2014 (Bureau of Meteorology, 2017). Bird observations were conducted throughout this drying phase (2012-2014) (Chapter 3). Further bird observations were included for pre and post-burn analyses (Chapter 4), and those observations were conducted immediately after the Millenium Drought broke (2010).

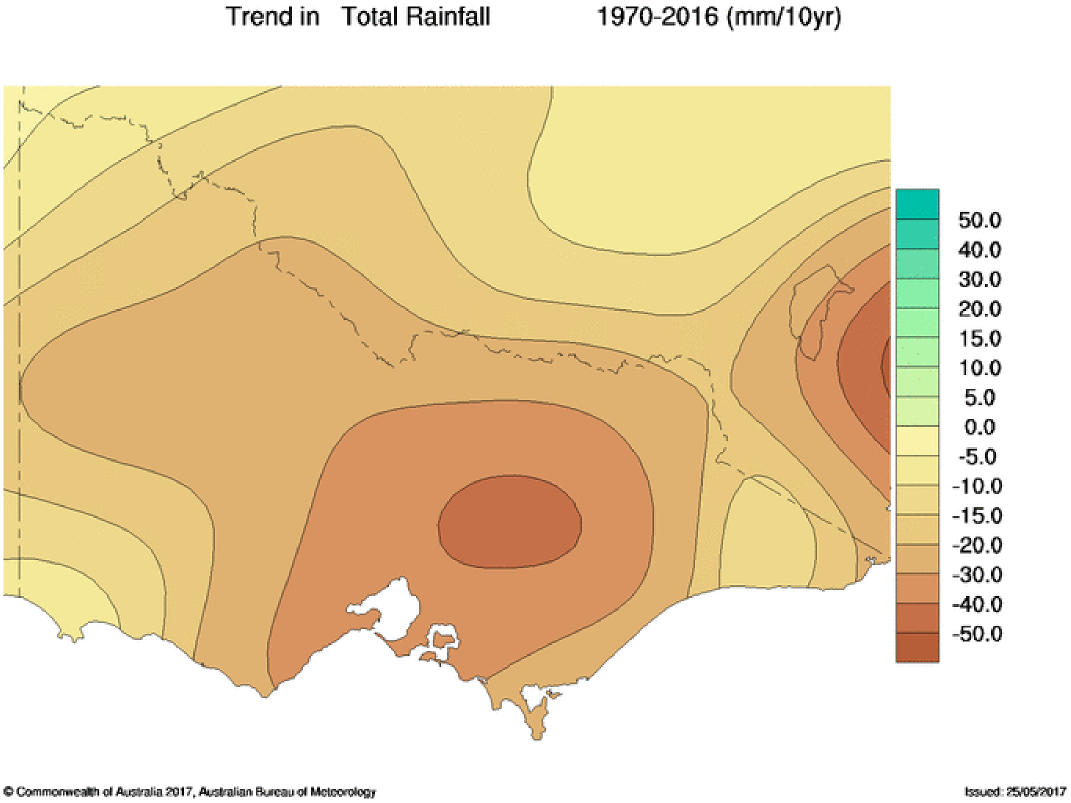


Figure 2.2 Drying trend across Victoria, Australia.

Source: Bureau of Meteorology (2017).

2.1.3 Sites for time-since-fire (TSF) and fire frequency analyses

Sites selected for this research form part of the project “Fire in Temperate Forest Landscapes”, designed as a whole-of-landscape study in dry eucalypt forests (in the midlands region), incorporating mammals, bats and birds (Palmer, 2015). This initiative forms a component of a Victorian Government program known as ‘HawkEye’, ongoing research incorporating projects in areas such as the Mallee, Otway Ranges and Gippsland Foothills. This program was implemented as a direct result of the Royal Commission into the Black Saturday fires (Teague, 2010). Recommendation 58 in the Royal Commission findings was a requirement for data collection, to monitor and model the effects of prescribed burning programs (Teague, 2010).

The site selection was based on the principle of land mosaics (Bennett et al., 2006) by selecting groups of six sites, from the same Ecological Vegetation Division (EVD) of Grassy Heathy Dry Forests (see Table 2.1), all within an area of three km diameter, to form a ‘fire-age mosaic’ (Fig 2.1). There was a total of 84 sites forming fourteen mosaics, with approximately 80 kilometres between the sites furthest west, and those furthest east. Sites were two hectares in size, a minimum of 600 metres apart and selected to be at least 50 metres distant from roads and burn edges, and at least 100 metres from drainage lines and major clearings. Site selection within the project design targeted large, relatively intact areas of native vegetation embedded within substantial areas of forested public land (Palmer, 2015).

Mosaics were selected to represent a gradient in ‘fire diversity’, from ‘low fire diversity’ where sites within the mosaic had experienced no recent fire or only one fire event, to ‘high fire diversity’ where sites within the mosaic had a diverse fire history (i.e. four to six fire events represented in a mosaic) (Palmer, 2015). It was anticipated that this design would therefore capture variations in bird abundances that may have been a result of the topographical variations across the landscape. So, where some mosaics had experienced very little or no fire in the recent past (last 34 years), others had experienced fire, either as a single fire event or as multiple fires over time (Palmer, 2015).

2.1.4 Subset of sites for fire and prescribed burn severity analyses

The project investigating bird responses to prescribed burn severity incorporated a subset of the initial 84 sites that formed part of the fire frequency project. From the initial 84, 30 sites were selected from an area with Linton to the west (37.67^{o}S, 143.52^{o}E), Durham Lead to the east (37.69^{o}S, 143.88^{o} E) and across Enfield State Park to the south (37.75^{o}S, 143.75^{o} E). Of those 30 sites, 14 were given prescribed burns during a two-week window spanning March to April in 2012, allowing for an opportunistic study. The remaining 16 sites were used as control sites. The prescribed burns were part of a Department of Environment, Land, Water and Planning (DELWP) autumn seasonal burn plan and each of the 14 sites impacted were entirely burnt.

2.2 Vegetation

The Victorian Government Department with authority over the environment (currently DELWP but previously DSE) had developed spatially explicit native vegetation data sets based on vegetation characteristics and physiographic variables (Cheal, 2010). This classification is known as Ecological Vegetation Classes (EVCs). Due to the wide range and large number of EVC's within Victoria (300), it became prudent to group EVCs into EVDs; EVCs were grouped into 32 EVDs based on ecological similarities (Cheal, 2010).

Dry, temperate eucalypt forests comprise the major natural vegetation that remains in the study area, as it does over much of southern and eastern Victoria. The major blocks of forests that remain near Linton, Enfield, Lal Lal, Creswick and the Brisbane Ranges support Grassy/Heathy Dry Forests Ecological Vegetation Division (EVD), which mostly comprises Heathy Dry Forest Ecological Vegetation Class (EVC) occurring on foothills and ridges. This EVC supports a low, relatively open understorey dominated by tussock grasses (Poaceae), with low to medium height sclerophyllous shrubs (e.g. Epacridacaeae, Fabaceae) common. The eucalypt tree canopy includes stringybark (e.g. Messmate *Eucalyptus obliqua*, Brown Stringybark *E. baxteri*) and peppermint species (Narrow-leaved Peppermint *E. radiata* and Broad-leaved Peppermint *E. dives*), as well as scattered gum-barked species such as Manna Gum (*E. viminalis*).

All sites selected for study in this project were within vegetation classified as Grassy/Heathy Dry Forests EVD and all were within the Heathy Dry Forest EVC.

2.3 Recent fire history and prescribed burn protocols

The study area has experienced both bushfires and prescribed burns over recent decades. Major bushfires in the study area in the past two decades include events centred on Enfield State Park (1995), Creswick Regional Park (1997), Linton State Park (1998), the Brisbane Ranges National Park (2006) and in Scotsburn (2015). Prescribed burns have been applied broadly across the landscape to achieve fuel reduction goals and operational objectives. The objectives for these burns are: to meet fuel reduction targets, for regeneration burns, or for targets set, following the guidelines prescribed for ecological burns (Department of Sustainability and Environment, 2004). Further to this, burns will be carried out by following management procedures which outline the processes required for creating burns of variable intensities, designed to achieve specific reductions in fire fuel. These targets may range from achieving below 50% of ground cover reduction, to near complete fuel reduction of the site (i.e. 80% of fuels removed) (Department of Sustainability and Environment, 2004).

2.4 Methodology

2.4.1 Categorical vegetation age classes based on TSF

As vegetation age was a factor in the analyses, age classes were determined for analyses based on the duration between the last fire and the dates of the bird surveys. An adaptation of an already established ageing protocol intended for Victorian Grassy/Heathy Dry Forests was utilised for determining TSF vegetation age classes with respect to the maturity and phenology of the vegetation (Table 2.1, Plates 4-8). Two descriptors are given in Table 2.1, distinguishing the regrowth vegetation of the 6 month to 2.5 year age class (dominated by epicormic regrowth on tree trunks with ground mostly bare), from the vegetation in the 2.5 – 10 year age class. In Heathy Dry Forests, the latter age class is predominantly new growth vegetation (shrubs, sedges and young trees), such that bare ground is less common.

Table 2-1 Post-fire age classes for Grassy/Heathy Dry Forests vegetation.

Source: Cheal (2010).

Code		Fire-age range	Descriptor in thesis
TSF1	Most of the ground bare, bradysporous species releasing seed, soil-stored seed germinating, resprouting species have buds activating, eucalypt coppice evident at end of stage, species not flowering, little or no litter	0 – 6 months	
TSF2	Most of the ground is bare, fire ephemerals common, bradysporous species germinating, tree seedlings evident, resprouting species vigorously growing, herbaceous fire ephemerals and annuals with first seeds set, little to no litter	6 months – 2.5 years	regrowth
TSF3	Bare soil less common, fire ephemerals in decline with longer lived fully reproductive, earlier germinants and resprouting shrubs and sedges vigorously growing, some litter accumulation	2.5 – 10 years	new growth
TSF4	Canopy cover at maximum, all fire ephemerals retreated to soil seed store, no further germination of bradysporous species, resprouting species growing, sedges common but not vigorous, canopy eucalypts flowering, herbaceous species appearing, litter accumulating and lichen establishing	10 – 35 years	
TSF5	Canopy opening, canopy eucalypts flowering, annual species more common, no further germination of bradysporous species, resprouting shrubs growing but growth rate decreasing, litter cover re-established with lichen and bryophyte cover well established	35+ years	

Typical vegetation in Victoria’s Heathy Dry Forests (EVC), post-fire age classes.

Plate 3 TSF1, 0 to 6 months, Linton.

Plate 4 TSF2, 6 months to 2.5 years, Linton.

Plate 5 TSF3, 2.5 to 10 years, Brisbane Ranges.

Plate 6 TSF4, 10 to 35 years, Creswick.

Plate 7 TSF5, 35+ years, Brisbane Ranges.

2.4.2 Categorical fire frequency rankings

Cheal (2010) estimated that the minimum tolerable fire intervals for a Grassy/Heathy Dry Forest are: 10 years post prescribed burn, and 15 years post bushfire. These classes are selected to ensure those species with the longest duration to reproductive maturity are accommodated to retain the ability to recruit; when enough flowers have matured to provide a sufficient viable seed source to replace a population. This principal is based on the incapacity of species replacing themselves when fire occurs before adult stages are reached. Further to this, Cheal (2010) suggested that the greatest vegetation response to fire frequency will appear after the second fire in this EVD, when intervals are of less duration than the suggested minimum. Understanding that the minimum tolerable fire intervals are a bench-mark measure for forest vegetation regeneration, one objective was to investigate whether in fact they are also a suitable measure for forest bird responses. With these factors in mind, sites were grouped into five ‘strength-of-impact’ categories (Table 2.2), in order to model the effects of fire frequency.

Table 2-2 Strength of fire impact on EVD3 Grassy/Heathy Dry Forests vegetation, combining fire frequency and tolerable fire intervals (TFI).

Code	**Fire frequency**	**Fire interval**	**Number of sites**
ff1	Unburnt or burnt over 34 years ago		18
ff2	One fire within last 20 years; or one fire within last 20 years plus one fire over 45 years ago		25
ff3	2-4 fires recorded since 1939	not breaching TFI	15
ff4	0-2 fires in the last 19 years + 1-3 fires > 20 years ago	TFI breached >22 years ago	14
ff5	2-3 fires within the last 19 years	TFI breached within last 19 years	12
			84

Table 2.3 shows the number of sites within each TSF and fire frequency category for this project. As some sites changed TSF classification across the life of the project, there were 115, not 84 combinations. The representation of sites across the TSF and FF groupings was unbalanced owing to the nature of the fire landscape studied, with low replication within some categories. Logically some combinations were never going to occur e.g. TSF5 (old vegetation) and ‘frequently burnt’ (ff5). The decision to maintain the number of TSF categories, rather than combining some to create a more balanced design, was made on a biological basis. TSF categories were based on an accepted classification of

seven divisions modified down to five, to include all vegetation over 35 years in the one age class. This was because structural changes in Grassy/Heathy Dry Forests after 35 years are minimal (Cheal, 2010). The remaining five TSF divisions act to separate the Grassy/Heathy Dry Forests into age classes with distinctly different vegetation structure e.g. the regrowth phase of TSF3. As a result of this issue, TSF analyses were considered independently to fire frequency analyses.

Table 2-3 Eighty-four sites in terms of TSF and fire frequency groupings.

14 sites burnt just prior to and during the project, requiring them to be split between two age classes. TSF age classes: **tsf1** 0-6 months, **tsf2** 6mths to 2.5 years, **tsf3** 2.5 to 10 years, **tsf4** 10 to 35 years, **tsf5** 35+ years. Fire frequency categories: **ff1** unburnt or burnt over 34 years ago; **ff2** one burn within last 20 years, or, one fire within last 20 years plus one fire over 45 years ago; **ff3** 2-4 fires recorded since 1939 and not breaching tolerable fire interval (TFI) of 10 years post prescribed burn or 15 years post bushfire; **ff4** 0-2 fires in the last 19 years + 1-3 fires > 20 years ago with TFI breached >22 years ago **ff5** 2-3 fires within the last 19 years with TFI breached within last 19 years.

	ff1	ff2	ff3	ff4	ff5
tsf1	0	8	4	8	0
tsf2	0	10	2	7	3
tsf3	0	12	10	2	11
tsf4	7	5	5	7	0
tsf5	13	1	0	0	0
					total 115

2.4.3 Prescribed burn severity rankings

The vegetation components of canopy scorch, bark burn and ground cover were assessed within three months of fire at all burnt sites. The extent to which these three parts of the vegetation were burnt, resulted in a percentage-based impact score. These data were pooled and a percentage for burn severity created for each burn site. Once sites were given a score of 1-10 based on their percentage burn, it was clear that each site fell within distinct groupings of either low, medium or high severity burns. Examples of ground cover, bark and canopy scorch for the three levels of prescribed burn: low, medium and high severity, are depicted in Plates 8 - 16.

Plate 8 Low severity burn, ground cover. Durham Lead.

Plate 9 Low severity burn, bark scorch. Enfield.

Plate 10 Low severity burn, canopy scorch. Enfield.

Plate 11 Medium severity burn, ground cover. Enfield.

Plate 12 Medium severity burn, bark scorch. Linton.

Plate 13 Medium severity burn, canopy scorch. Linton.

Plate 14 High severity burn, ground cover. Durham Lead.

Plate 15 High severity burn, bark scorch. Enfield.

Plate 16 High severity burn, canopy scorch. Linton.

2.5 Study designs

2.5.1 An analysis of birds' responses to TSF and fire frequency

An inventory of birds was created by censusing bird species individuals on these sites in order to explore bird-fire history relations through space-for-time substitution (Letnic et al., 2004, Muir et al., 2015) to compare assemblages within the mosaics of contrasting/different fire histories.

The fire history for each site was determined by reference to the DELWP Firemap GIS overlays from the DELWP online system. The records included fires that occurred since 1939. The 169 individual site fire events recorded were then each verified against original hard copy maps from DELWP archives and checked against the recollections of the fire and land managers. Fires occurring during the 1970's were missing from the online stored data. As some classifications for TSF categories (Table 2.1) and fire frequency categories (Table 2.2) were made incorporating a 30 to 40 year window, efforts were made to ensure that data collected back to the 1970's were accurate. Information on each of these fires was taken from original hard copy maps and included in the dataset. Of the 169 fire events recorded on individual sites, 40 were prescribed burns and there were 13 bushfire season events in total. Bushfires such as those of 1966 had ignition points in multiple locations (Table 2.4).

A table was created for 672 monitoring assessments, the overall number of surveys taken at sites, grouping each site into one of five post-fire vegetation age classes: 0-6 months (TSF1), 6 months – 2.5 years (TSF2), 2.5 years to 10 years (TSF3), 10 to 35 years (TSF4) and 35+ years (TSF5) (combined with 'last burn not recorded’). Cheal (2010) proposed six post fire ages in his classification. For the purposes of this project five were used, with the 35-45 years and 45+ years classes, along with sites with no record of burning, combined on account of the limited detail for pre-1970 fires.

Fourteen sites were given prescribed burn treatments two months prior to the first monitoring round and prescribed burns were implemented at six sites over the duration of this research, after the commencement of bird surveys.

2.5.2 Bird responses to fire and prescribed burns of varying severities

Of the 30 sites selected for this project, 14 were burnt in prescribed burns during a two-week window at the end of March through into April 2012. Of the 16 control sites, one was removed from the database as it had burnt in 2008, just prior to the scheduled pre-burn monitoring of 2010 and its vegetation was therefore much younger than other control sites. Two further sites were removed, as they were treated with prescribed burns in 2011; burnt in the time between pre and post burn monitoring. Further to this, three sites were used as controls for only part of the project due to prescribed burn treatments in 2013. Of a total of 156 monitoring sessions on sites, 84 were on burnt sites and 72 on control sites.

A Before-After Control-Impact (BACI) design was used to investigate the influence of prescribed burn severity on the avifauna in the area, which involved bird surveys before and after prescribed fire and concurrent monitoring of both burn and control sites (Stewart-Oaten et al., 1986).

Table 2-4 Burn history on sites from 1939 to 2014.

🟨 Prescribed burns 🟥 Bushfire information obtained through online DELWP fire history maps.

Included is the Forest Commission Victoria Fire Prevention Plan Burn History, revised 1981. >50% burnt, (dark green), <50% burnt, (pale green).

✖ Marks where the tolerable fire interval of 10 years post prescribed burn or 15 years post bushfire have been breached (Cheal, 2010).

Location	*Identifier*	*Recorded fires* *1939*	*1966*	*1972-1991*	*1992-2001*	*2002-2014*	*Vegetation Age Class*	*Fire frequency classification*	*Sites in severity project*
Brisbane Ranges	BR141					🟥 2006 Jan	TSF3	FF2	
	BR142				🟨 1994		TSF4	FF2	
	BR143					🟥 2006 Jan	TSF3	FF2	
	BR144					🟥 2006 Jan	TSF3	FF2	
	BR145					🟨 2013 Oct	TSF5,1	FF2	
	BR146						TSF5	FF1	
	BR151					🟥 2006 Jan	TSF3	FF2	
	BR152			🟨 1987		🟥 2006 Jan	TSF3	FF3	
	BR153			🟨 1990		🟥 2006 Jan	TSF3	FF3	
	BR154					🟥 2006 Jan	TSF3	FF2	
	BR155					🟥 2006 Jan	TSF3	FF2	
	BR156					🟨 2004 ✖ 🟥 2006 Jan	TSF3	FF5	
	BR161					🟨 2005 Apr	TSF3	FF2	
	BR162						TSF5	FF1	
	BR163					🟨 2005 Apr	TSF3	FF2	
	BR164						TSF5	FF1	
	BR165					🟨 2005 Apr	TSF3	FF2	
	BR166						TSF5	FF1	
Creswick	CR091			🟥 1977	🟥 1997 Jan	🟨 2013 spring	TSF4,1	FF3	
	CR092			🟥 1977			TSF4,5	FF1	
	CR093			🟥 1977		🟨 2003 May	TSF3,4	FF3	
	CR094			🟥 1977			TSF4,5	FF1	
	CR095			🟥 1977 ✖ 🟨 1991	✖ 🟥 1997 Jan	✖ 🟨 2010 Nov	TSF2,3	FF5	
	CR096			🟥 1977	🟥 1997 Jan	✖ 🟨 2010 Nov	TSF2,3	FF5	
Durham Lead	DU081			🟥 1972 ✖ 🟥 1983			TSF4	FF4	🟩 control

	DU082		1972			2008 Oct		TSF3	FF3	
	DU083					2012 Mar		TSF1,2	FF2	burn
	DU084		1972			2012 Mar		TSF1,2	FF3	burn
	DU085		1972			2004 Apr		TSF3	FF3	control
	DU086					2012 Mar		TSF1,2	FF2	burn
Enfield	EN031	1966				2011 Mar		TSF2,3	FF2	
	EN032	1966						TSF5	FF1	control
	EN033	1966						TSF5	FF1	control
	EN034	1966						TSF5	FF1	control
	EN035	1966				2012 Apr		TSF1,2	FF2	burn
	EN036	1966						TSF5	FF1	control
	EN041		1976 ✖	1985	1995			TSF4	FF4	
	EN042				1995			TSF4	FF2	
	EN043		1976 ✖	1985	1995	2010 Mar		TSF2,3	FF4	
	EN044				1995			TSF4	FF2	
	EN045		1976		1995			TSF4	FF3	
	EN046	1966			1995	✖ 2008 April		TSF3	FF5	
	EN051	1966 ✖	78/79		1995			TSF4	FF4	
	EN052	1966 ✖	79/80		1995	✖ 2007 Apr	✖ 2008 Feb	TSF3	FF5	
	EN053	1966 ✖	79/80		1995	✖ 2009 Apr		TSF3	FF5	
	EN054	1966 ✖	79/80		1995	✖ 2009 Apr		TSF3	FF5	
	EN055		79/80		1995	✖ 2004 Mar		TSF3	FF5	
	EN056	1966 ✖	1973 ✖	78/79	1995			TSF4	FF4	
	EN061		1973 ✖	1976	1995	2012 Apr		TSF1,2	FF4	burn
	EN062	1966 ✖	1973,✖76 ✖	1990 ✖	1995	✖ 2004 Apr		TSF3	FF5	control
	EN063	1966			1995			TSF4	FF2	control
	EN064		1973,✖76 ✖	1990 ✖	1995	✖ 2004 Apr		TSF3	FF5	control
	EN065		1973,✖76		1995	2012 Apr		TSF1,2	FF4	burn
	EN066	1966			1995	2012 Apr		TSF1,2	FF3	burn
	EN071		1972	✖79/80		2013 spring		TSF4,1	FF4	control
	EN072		1972	✖79/80		2012 Apr		TSF1,2	FF4	burn
	EN073		1972	✖79/80		2013 spring		TSF4,1	FF4	control
	EN074		1972	✖79/80		2013 spring		TSF4,1	FF4	control

	EN075			1972	✖ 79/80			2012 Apr		TSF1,2	FF4	burn
	EN076			1972	✖ 79/80			2012 Apr		TSF1,2	FF4	burn
Lal lal	LA111		1966	✖ 79/80				2011 Feb		TSF2,3	FF4	
	LA112									TSF5	FF1	
	LA113			79/80						TSF4	FF1	
	LA114			79/80				2003 Nov		TSF3,4	FF3	
	LA115			78/79						TSF4	FF1	
	LA116			79/80				2007 Nov		TSF3	FF3	
Linton	LI011		1966			1992	✖ 1998 Dec	✖ 2009 April		TSF3	FF5	
	LI012		1966			1998 Dec				TSF4	FF2	
	LI013		1966			1992		2004 Oct	✖ 2011 Dec	TSF2	FF5	
	LI014							2010 Oct		TSF2,3	FF2	
	LI015		1966							TSF5	FF1	
	LI016					1998 Dec		2013 spring		TSF4,1	FF3	
	LI021	1939						2011 Mar		TSF2,3	FF2	
	LI022	1939								TSF5	FF1	control
	LI023	1939						2012 Apr		TSF1,2	FF2	burn
	LI024							2012 Apr		TSF1,2	FF2	burn
	LI025							2012 Apr		TSF1,2	FF2	burn
	LI026	1939						2012 Apr		TSF1,2	FF2	burn
Mt Egerton	WH131			78/79				2005		TSF3	FF3	
	WH132			78/79						TSF4	FF1	
	WH133			78/79						TSF4	FF1	
	WH134			78/79				2008 Nov		TSF3	FF3	
	WH135			78/79				2005		TSF3	FF3	
	WH136			78/79						TSF4	FF1	

2.6 Data collection of bird surveys and foraging guild information

2.6.1 Bird surveys for determining fire frequency responses

Birds were monitored on all 84 sites in eight survey rounds incorporating two rounds for each of two spring/summer breeding seasons and two in winter, taken from June 2012 to February 2014.

The 20 minute search method (Loyn, 1986) was used to survey birds. This approach is accepted as a standard bird survey method, and it formed the basis of the procedure used in the Birds Australia Atlas (Barrett et al., 2003). Bird monitoring method comparisons have shown that the 20 minute area search is an effective monitoring method with advantages over three other methods tested (transect, stationary count or rolling bird survey) (Hewish and Loyn, 1989).

Sites were traversed for 20 minutes and all birds seen and heard were identified to species level. Birds flying overhead, or heard and considered to be outside the two-hectare perimeter of the plot, were not recorded. Two surveys on each site were conducted in the winter months of 2012 with two survey rounds undertaken the following spring/summer season. This was repeated in 2013 with the final summer round completed during February 2014. A total of 672 surveys was undertaken. Equal numbers of morning and afternoon monitoring sessions for both winter and spring/summer were carried out for each site and the bird survey data were pooled for each season. All surveys were carried out in mild weather conditions with no monitoring undertaken on days of extreme heat (>30°C) or when wind was greater than 20 km/hr. In extreme heat birds are less active making detection difficult, and in high winds birds cannot be seen or heard in the canopy as easily.

2.6.2 Bird surveys for determining fire severity responses

Birds were surveyed on all sites in six survey rounds incorporating two rounds before prescribed burning occurred and four rounds post prescribed burns. Pre-burn monitoring occurred in the spring/summer of 2010 and prescribed burns occurred in the autumn months of March/April 2012. The first two rounds of post-fire monitoring occurred in the spring/summer immediately after the prescribed burns in 2012/2013 with the final two sessions the following spring/summer in 2013/2014.

2.6.3 Foraging guild classifications

Bird species observed were classified into one of twelve foraging guilds (Table 3.5) prior to data analyses.

Table 2-5 Foraging guilds.

Source: Loyn et al. (2007).

B	Bark	takes invertebrates from bark on trunks and branches
C	Canopy	takes invertebrates from foliage of eucalypts and other large trees
DG	Damp ground	takes invertebrates from damp ground below shrubs, among dense understorey or among damp litter
F	Frugivore	takes soft fruit along with other food such as nectar, invertebrates or seeds
N	Nectarivores	takes nectar along with other food such as seeds (parrots) and fruit or invertebrates
OG	Open ground	takes invertebrates from open ground, quite often far from tree or shrub cover
OT	Open trees	takes invertebrates from open ground among trees or scattered tall shrubs, but not
		from damp ground below dense cover, and does not usually venture far from woody vegetation
SG	Seeds ground	takes seeds from ground or low plants such as grasses, herbs and saltmarsh

ST	Seeds trees	takes seeds from trees and shrubs or wide range of strata, or other food such as gall insects
		or insect larvae extracted from wood
TS	Tall shrubs	takes invertebrates from foliage of tall shrubs, which may form middle storey of eucalypt forests
V	Carnivore	carnivore, taking vertebrates as an important part of diet, often along with large invertebrates
		and other food such as fruit (passerines)
W	Water	water bird inhabiting inland waters

In this exercise, in the classification of the forest bird species, the Grey Shrike-thrush (refer to Table 3.2 for scientific names), often known as a generalist, was classified as a carnivore. This was done to maintain consistency with classifying species into distinct feeding methods. All other species were classified into distinct foraging guilds as per Loyn et al. (2007).

2.7 Statistical analyses

Two distinctly different methods were implemented to investigate and model TSF, fire frequency and fire severity responses. The first method was one of combining investigations and assessments on the effects of TSF and fire frequency in the form of both ordinations and statistical modelling (Chapter 3). The second method was a Before-After-Control-Impact study (Chapter 4). The opportunity to create this design arose when a number of prescribed burns were carried out concurrently on sites within the study area. Pre-burn data were available, and for this subsection of data, bird responses to fire (as either control or burns) and fire severity were modelled.

In both the TSF with fire frequency and fire severity projects, modelling of community, foraging guild and individual species responses were carried out in the R statistical environment (R Development Core Team, 2008), using the packages lme4 (Bates et al., 2016) for fitting generalised linear mixed effects models (GLMMs), and MuMIN (Barton, 2016), a multi-model inference package for model selection. GLMMs were chosen as they allow for nested data structure and possible correlation between datasets (Zuur et al., 2009).

2.7.1 Graphing outputs: a comparison between modelling techniques using vegetation age classes versus continuous age data

A comparison between modelling with continuous age class data as opposed to categorical data, was carried out in the R statistical environment (R Development Core Team, 2008). Avian abundance data collected during eight observation sessions, across 84 sites, were added together, with species grouped into foraging guilds (Table 2-5). Ten foraging guilds were modelled, using vegetation age as both continuous data in terms of 'years since fire' and as an age class category (Table 2-1). There were two foraging guilds represented in the observed data that were not modelled, as individuals observed were too few (frugivores with two individuals and water birds with six). The random effects were tested with options of mosaic and sites nested within a mosaic. An Information Theoretic Approach was then used ranking candidate models for different random effects by comparing AICc (AIC with a correction for finite sample sizes) to select the most parsimonious model (Burnham and Anderson, 2002).

All Poisson distribution models were tested for overdispersion (Zuur et al., 2015). If a model was overdispersed (considerably >1), the individual level random effect (ILRE) factor was applied to the model, transforming the Poisson model into a Poisson-lognormal model, more appropriate for overdispersed data. If the model was still overdispersed after an ILRE was applied, a Negative binomial distribution was applied (Zuur et al., 2009). In cases where the sample was underdispersed (<0.7), a Binomial distribution was applied to the model.

Comparing a model for continuous data with one of categorical data was done by running both with gamm4 (Wood and Scheipl, 2017). AICc was then used to rank the two candidate models, to determine which was the most parsimonious.

2.7.2 Analysis of community composition response to TSF and fire frequency: a multivariate approach

The influence of time since fire and fire frequency on assemblage composition, in terms of individual species presence and species grouped into foraging guilds, were both investigated using statistical analyses including non-metric multidimensional scaling (nMDS) (Clarke, 1993), permutational multivariate analysis of variance (PERMANOVA) (Anderson et al., 2008) and similarity percentage contribution of individual species or foraging guilds (SIMPER) (Clarke, 1993). These analyses were all run in PRIMER-E, version 6.1.16 (Clarke and Gorley, 2006) with PERMANOVA+ version 1.0.6 (Clarke and Warwick, 2001). Due to their partial dependence, the influence of both TSF and fire frequency on birds were both investigated separately in all analyses. Ordinations were run on log(x+1) transformed bird count data to reduce the influence of several highly abundant species. Initial investigations incorporated datasheets with breeding and non-breeding seasons set as two factors. This was subsequently changed to data being split into separate datasheets for each of the two seasons, to view seasonal ordinations separately. The transformed data were subjected to nMDS using the Bray-Curtis distance measure to quantify the level of dissimilarity in composition between samples (Bray and Curtis, 1957). Six records of a woodland duck species (Australian Wood Duck *Chenonetta jubata*) and two of Mistletoebird (*Dicaeum hiruninaceum*) were removed from the bird dataset in the ordination process. The ordination plots revealed these two species to be outliers, with all other taxa closely clustered, relative to the ordination distance between the cluster and these taxa. The two species were included in all other data analyses. In all instances three dimensional (3D) ordinations were viewed along with two dimensional (2D). In each case stress levels were <0.2 for 3D but >0.2 for 2D. As the major patterns in the 3D representations were also evident in the 2D representations, 2D were presented in the results as they provided a clearer visual presentation. PERMANOVA+ model testing was used to determine the statistical significance of the differences in TSF and fire frequency effects. This form of modelling is considered more robust than parametric statistics as it does not require the data to follow a particular distribution, acting only on the ranks of dissimilarity (Anderson et al., 2008). PERMANOVA+ also allows a nested sampling design (Anderson et al., 2008).

As dispersion levels for the random factors in a nested design should be homogenous, Permdisp was run as a prelude to the Permanova analyses, to determine if there were any significant deviations from the centroid (Anderson et al., 2008). The PERMANOVA+ models incorporated fixed factors of TSF or fire frequency, combined with season e.g. TSF*season, FF*season. Models were created to deal with spatial and temporal nesting; i.e. sites nested within mosaics where sites in one mosaic are independent from sites in a second mosaic

(Anderson et al., 2008). Both sites and mosaics were considered to be random samples from a larger pool, so were treated as random factors. Therefore, along with the fixed factors, the design also incorporated ‘site’ as nested within any mosaic. The datasets used for the model were the same as those used for species and foraging guilds in the ordinations. In two analyses TSF data would not conform to one of the age classes and the models would not run. In these instances, the analyses were simplified by removing mosaics from the model. The initial analysis was followed by pairwise testing to inform which pairs of TSF and fire frequency categories had a significant effect on composition. To determine the species that drove the multivariate patterns detected by PERMANOVA+, the category pairs were then matched within the program SIMPER using Bray-Curtis percentages to determine the species contributing most to total dissimilarity. This process was then repeated for foraging guilds.

2.7.3 Community responses to TSF and fire frequency: a univariate approach

The modelling of the avian community, in terms of foraging guild and individual species responses to both TSF and fire frequency, was carried out in the R statistical environment (R Development Core Team, 2008).

Models were run testing continuous age with categorical age, therefore data sheets used in analyses maintained 672 data lines of bird observation numbers, representing 84 sites monitored eight times. For foraging guild models, data for each species were added together to form the foraging guilds listed (Table 2.5). Data for both predictor variables of TSF and fire frequency were categorical (Tables 2.1, 2.2). The random effects were tested with options of mosaic and sites nested within a mosaic. An Information Theoretic Approach was then used ranking candidate models for different random effects by comparing AICc (AIC with a correction for finite sample sizes) to select the most parsimonious model (Burnham and Anderson, 2002).

All Poisson distribution models were tested for overdispersion (Zuur et al., 2015) (The overdispersion code is available at https://bbolker.github.io/mixedmodels-misc/glmmFAQ.html). If the sample was overdispersed (considerably >1), the individual level random effect (ILRE) factor was applied to the model, transforming the Poisson model into a Poisson-lognormal model, more appropriate for overdispersed data. In cases where the sample was underdispersed (<0.7), a binomial distribution was applied to the model. While an underdispersion factor was determined as the trigger, an underdispersion result indicated that the dataset had a large number of zeroes and/or the non-zero values were small. In these instances, raw data were converted to either presence or absence values and run in a model with a binomial distribution.

Necessary checks were undertaken for all GLMM models. Candidate models selected as most parsimonious with different random effects were checked for dispersion levels. For Poisson and Negative Binomial distributions, a Pearson residuals vs fitted values plot was run, to assess models by determining if there were any obvious patterns in the residuals (Zuur et al., 2009). No unusual patterns in residuals were identified. For Binomial distributions, model validation cannot be done in this manner as responses are either one or zero and therefore plotting Pearson residuals will give two bands (Zuur et al., 2015).

Ggplot2 was used for graphing (Wickham and Chang, 2016).

2.7.4 The response of birds to prescribed fire and varying prescribed burn severities – a Before-After Control-Impact design approach

Modelling of the avian community in terms of alpha and beta diversity responses, along with individual species responses, to both fire and fire severity, was carried out in the R statistical environment (R Development Core Team, 2008).

Two predictor variables of fire (burnt and unburnt) and fire severity (1-10) were modelled to investigate the response of birds to prescribed burns and prescribed burn severity, using GLMMs. Fire severity was modelled as continuous data. Ggplot2 was used for graphing the output data (Wickham and Chang, 2016).

There were no control sites with vegetation burnt within the last eight years. Therefore, models incorporating fire were all run with this variable as either a control or a burn; there was no need to specify the vegetation age on the control sites. Time was divided into two separate breeding seasons post-fire.

A BACI model calculates the mean before-after difference at control sites and the mean before-after difference at treatment sites, and then calculates the difference between these differences to determine the interaction effect. So, even if some of the before-after difference is due to an observer effect, we can assume it will be consistent across control and impact sites.

For community-level analyses, alpha diversity (species richness) and beta diversity (species turnover) were both used as response variables, representing the degree to which the fire changed community composition. Beta diversity was modelled for pre and post burn variation using presence-absence data, with the package betapart (Baselga and Orme, 2012). Betapart created a Simpson dissimilarity index (beta.SIM) between pre fire bird observation data and post fire bird data. This was calculated separately for two post fire time periods: pre fire in association with first year post fire and then pre-fire with second year post fire data. The index ranged from zero to one, zero indicating complete similarity and one being total dissimilarity (Baselga and Orme, 2012). The index was then used to run models testing for both fire and fire severity responses on community turnover, comparing changes in the two post fire periods.

Of the total of 51 species recorded, 10 were observed frequently enough to generate species response models (Wintle et al., 2005). The individual species models were all run using presence-absence data, to remove the impact of any outlier from individual sites having unusually greater abundances.

Selected models were run using the random effect options (mosaic, site, or site within mosaic) for biodiversity measures or for the individual species, with fixed effects (fire or fire severity), as interactive and then additive with time (Table 2.6). An Information Theoretic Approach was then used, ranking candidate models for different random effects, measuring goodness of fit and model complexity (Zuur et al., 2009). Running the package MuMIn (Barton, 2016), candidate models were ranked, comparing AICc (AIC with a correction for finite sample sizes) (Akaike, 1973) to give the most parsimonious model (Burnham and Anderson, 2002).

Fire (as control or burn) and fire severity (continuous data) were modelled separately. Models were selected with the lowest AICc for the interaction of fire x time (indicating that fire response occurred over time) or fire + time (indicating a fire response that is not time dependent). With the severity models, some investigations resulted in the most parsimonious option not requiring time as a factor i.e. there was no change over time. These models were run with severity as the only fixed effect.

Table 2-6 Full model design carried out in the R statistical environment.

Response variable	Predictor variables	Full fixed model	Full random model
Species richness	Time (before, after1, after2) Fire (burnt, unburnt)	Time*fire	Mosaic, site
Species richness	Time (before, after1, after2) Severity (1-10)	Time*severity	Mosaic, site
Species turnover (beta.SIM)	Time (before, after1, after2) Fire (burnt, unburnt)	Time*fire	Mosaic, site
Species turnover (beta.SIM)	Time (after1, after2) Severity (1-10)	Time*severity	Mosaic, site
10 individual species (presence/absence)	Time (before, after1, after2) Fire (burnt, unburnt)	Time*fire	Mosaic, site
10 individual species (presence/absence)	Time (before, after1, after2) Severity (1-10)	Time*severity	Mosaic, site

Percent variance explained (R^2) was evaluated using the method of Nakagawa and Schielzeth (2013). GLMMs were evaluated for model fit by running the packages piecewiseSEM (Lefcheck, 2016) or MuMIN (Barton, 2016), to determine marginal R^2 and conditional R^2: marginal R^2 being the variance explained by fixed factors and conditional R^2 being the variance explained by both fixed and random factors.

Plate 17 The Laughing Kookaburra, *Dacelo novaeguineae,* is less abundant in dense new-growth vegetation.

Source: Lisa Rachiele, Victorian Birders

3 The influence of time since fire and fire frequency on forest bird species

3.1 Chapter overview

Bushfires and prescribed burns are prevalent in the Australian landscape (Bradstock, 2010, Teague, 2010, Holdgate et al., 2014) and their frequency is increasing (Clarke et al., 2011, Attiwill and Adams, 2013, Fairman et al., 2016). Many of the avian species, long recognised as efficient indicators of environmental health (Birdlife International, 2008, Drever et al., 2008, Larsen et al., 2010, Szabo et al., 2012, Birdlife Australia, 2015), are decreasing in abundance across Australia (Recher, 1999, MacNally et al., 2009, Bennett and Watson, 2011, Ford, 2011, Watson, 2011, Birdlife Australia, 2015, Ceballos et al., 2017).

The increase in the use of prescribed burns in the landscape, whilst implemented in attempts to ameliorate the impacts of bushfire, may ultimately result in more of the landscape being burnt each year. While the objectives of a prescribed burn are first, to protect assets and second, to target biodiversity management generally (Department of Sustainability and Environment, 2012), ecological fire management in Australia has largely, to date, had as its focus the goal of meeting flora requirements as a surrogate for fauna diversity (Clarke, 2008). The use of vital attributes of plants has been key to planning ecological burning practices, with vegetation species determined as most vulnerable to frequent fire or fire exclusion noted as *key fire response species* (Tolhurst and Cheney, 1999). As planning for plants does not necessarily cater for the needs of animals, the planning of ecological burning should consider the tolerance regimes of fauna and encompass faunal population trends to ensure appropriate timing of prescribed burns (Clarke, 2008), to maintain between-fire intervals within which all faunal species will persist.

Bird communities have been shown to become increasingly dissimilar over time with increases in severity and occurrence of fire (Barlow and Peres, 2004). So, while it is important to define vegetation communities in terms of fire interval limits for plant species, this may not produce the appropriate conservation strategy for bird species or other fauna (Driscoll et al., 2010, Loyn, 2012). To avoid a perverse outcome for avifaunal management, it is important to add to the current vegetation knowledge, data directly related to birds. This is because there may be species or communities more influenced by the cumulative effects of fire and its frequency (Spies et al., 2012).

One technique to deal with a limited database, due to restrictions both temporally and spatially, is to focus research on selected species from within bird assemblages. The use of data on more abundant birds may be an option to overcome limits to sampling time. The common birds represent a crucial component of community networks in terms of food web structure (Tylianakis et al., 2010), and there is evidence that some of the most common species have geographic ranges that are shrinking, which requires immediate investigation (Ceballos, 2017). Understanding the population trends of the common species may therefore enable more effective management of the entire community, especially when considering management processes such as prescribed burn protocols (Bascompte and Stouffer, 2009). When a habitat becomes simplified, as it may post-fire, the specialist taxa may not remain. Further, where a community may be resilient to the random extinction of specialist species, it may be vulnerable if generalist species come under threat of extinction (Bascompte and Stouffer, 2009).

The over-arching theme in this chapter is bird responses to time since fire and fire frequency. In concert, common and moderately common species' responses to both time since fire and fire frequency were analysed. Bird responses were then examined as a part of the foraging guild each bird represented, to assess their usefulness as bioindicators, to act as a guide in management processes.

Investigations into avian responses to both time since fire and fire frequency occurred in four broad stages. The first investigation considered the most appropriate method of model design for this research, while also considering one of the project objectives – determining the value of the Cheal (2010) vegetation age classification as a management tool for predicting avian responses. Vegetation age classes determined by Cheal (2010) note categories that incorporate six monthly increments, for stages where vegetation changes may be important for the development of crucial avian resources e.g. new growth vegetation from approximately 2.5 years post fire creating dense ground cover in a vegetation type that typically has bare open ground (Table 2.1). The 'years since fire' method commonly chosen to present data in a model, using a continuous age dataset, may not illustrate the distinction between critical elements of regrowth and new growth in a Heathy Dry Forest, unless age increments were small enough in the model. A further consideration in model designs for this project is that vegetation age needs to also be presented as 70+ years. Therefore, the first investigation in this research was a modelled comparison between two methods of presenting data: a categorical versus a continuous age class. To capture most of the avian community in the comparison, foraging guilds were modelled. The only birds therefore, not in this analysis were those in the frugivore foraging guild (as only two individuals were observed) and the water birds (only six individuals were observed). The second investigation reviewed the response of the entire bird community, modelled in terms of species richness and then again with species classified into foraging guilds. Third, ordinations were run (nMDS), with statistical support (Permanova), to determine species' and guild responses and patterns across sites in response to time since fire and the frequency of fires. The species which contributed most to the assemblage patterns in the ordinations were identified as common species. Finally, the remaining group of species was divided into moderately common and uncommon classifications. Models of both time since fire and fire frequency responses were then run for the moderately common species and their responses compared with those from the most common species.

There are three broad predictions for this research. First, notwithstanding that eucalypt forest regeneration post-fire is rapid, the first prediction is that early post-fire vegetation in Heathy Dry Forest is characterised by distinct structural elements in the vegetation and, as such, will support distinct bird assemblages (Bennett et al., 1994, Gilmore, 1985, Loyn et al., 2007, Reis et al., 2016). A second prediction is that there will be distinct individual species' responses to time since the last fire, a result of the immediate impact of fire on vegetation structure (Loyn and McNabb, 2015). Bird diversity has been found to increase over time with the application of patchy, low severity fire (Sitters et al., 2015), and an increase in biodiversity occurs with species preferences for particular successional states (Bradstock et al., 2005, Parr and Andersen, 2006). It is reasonable to expect therefore that avian assemblage patterns may change over time post-fire, and that these changes will be clearly reflected in the abundance of individual common species. The third prediction is that there will be little evidence of changes to assemblages based on fire frequency, as research suggests that both small planned burns on sites with a history of disturbance, along with intense bushfires, have short-term impacts on forest avian communities, and more subtle longer-term effects (Loyn and McNabb, 2015, Muir et al., 2015, Leonard et al., 2016).

3.2 Results

3.2.1 Graphing outputs: a comparison between modelling techniques using vegetation age classes versus continuous age data

Avian foraging guild responses to time since fire were modelled three times. The first two were using vegetation aged on a continuous scale of ‘years since fire’, with raw data and then without. The model was run with both, as graphs without the raw data outliers gave an improved visualisation of results. The continuous graphs were then compared with models using ‘vegetation age class categories’ (Table 2.1). The random effect of ‘site within mosaic’ was the most parsimonious model, in all instances. Outputs show that models for five foraging guilds were most parsimonious with a ‘years since fire’ age continuum, and five were best with a categorical age classification (Table 3.1).

Table 3-1 Comparing methods of modelling vegetation age: categorical versus continuous. Generalized linear mixed models.

Model selection based on random effect structure and ranked using Akaike's Information Criteria (AICc). Most parsimonious model is shaded. Distribution curves as per Watson (2012).

Foraging guild	Vegetation age class	Distribution parameter	Distribution curve	AICc
bark	categorical	Poisson	bell	2123.6
	continuous	poisson	null	2135.9
canopy	continuous	negative binomial	irruptive	3477.3
	categorical	negative binomial	bell	3479.7
carnivores	categorical	poisson	bell	2286.6
	continuous	poisson	null	2301.0
damp ground	continuous	poisson	decline	582.5
	categorical	negative binomial	irruptive	625.0
nectarivore	categorical	negative binomial	irruptive	1351.8
	continuous	negative binomial	irruptive	1446.5
open ground	continuous	negative binomial	decline	824.0
	categorical	negative binomial	irruptive	868.8
open trees	continuous	negative binomial	irruptive	3605.4
	categorical	poisson	bell	3658.3
seeds on ground	continuous	negative binomial	irruptive	623.7
	categorical	poisson	irruptive	669.2
seeds on trees	categorical	negative binomial	irruptive	1991.9
	continuous	negative binomial	delayed	2003.0
tall shrubs	categorical	negative binomial	irruptive	1161.7
	continuous	negative binomial	decline	1190.1

Statistically, either method of presenting data is acceptable, as AICc outputs for each pair showed little difference between candidate models. However, graphed outputs present differently. The complete set of graphed outputs for the ten foraging guilds is provided in Appendix A1.

The categorical and continuous graphs reveal similar information. The abundances for birds that forage on seeds in trees illustrates this; all graphs show the greatest and maximum abundances <5 years post fire, decreasing in abundance to around 30 years and then increasing over time in vegetation older than 35 years (Fig 3.1).

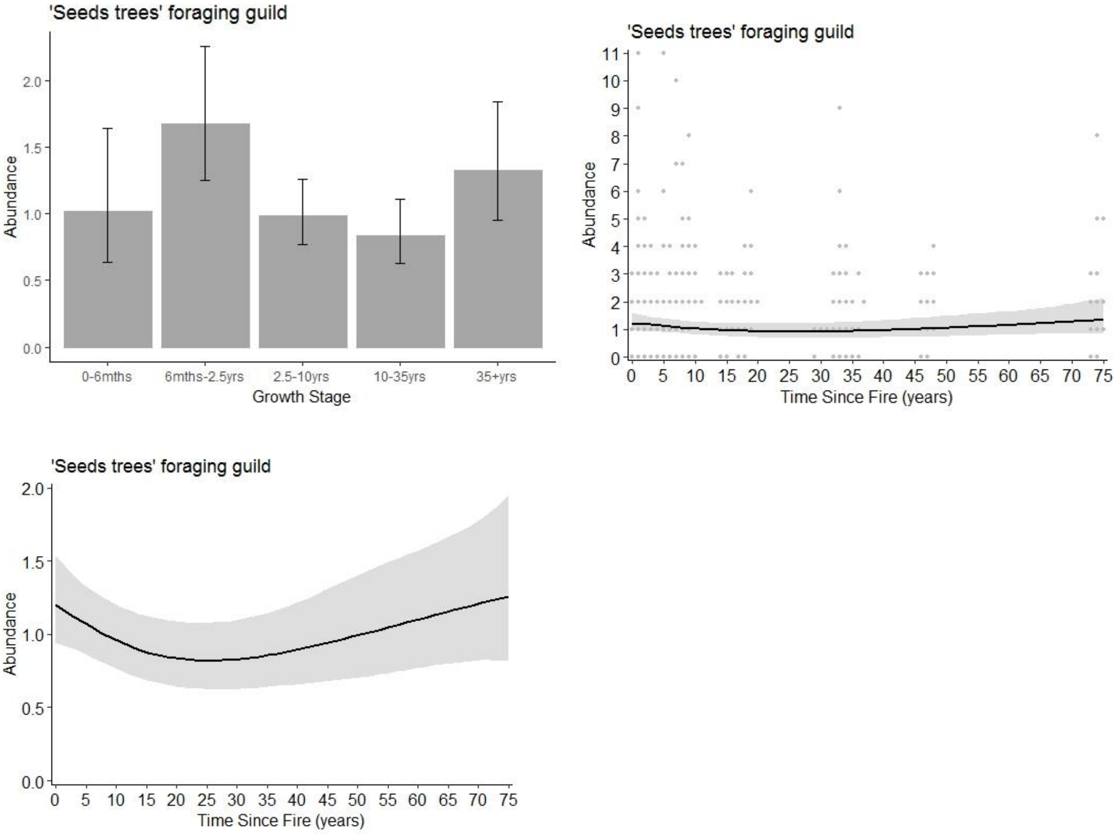


Figure 3.1 Models of ‘Seeds in tree’ forager data with vegetation as continuous age, with and without raw data and with a categorical age class for vegetation.

Generalized linear mixed models. Model selection based on random effect structure (site within mosaic) and ranked using Akaike’s Information Criteria (AICc).

It is the detail within each graph that varies across each comparison. Inherent in the design of a continuous timescale of yearly intervals, that needs to extend for 75 years, the six-month increment, relevant to regrowth vegetation appearing six months post fire (as epicormic growth in eucalypt species), will be masked (Fig 3.2).

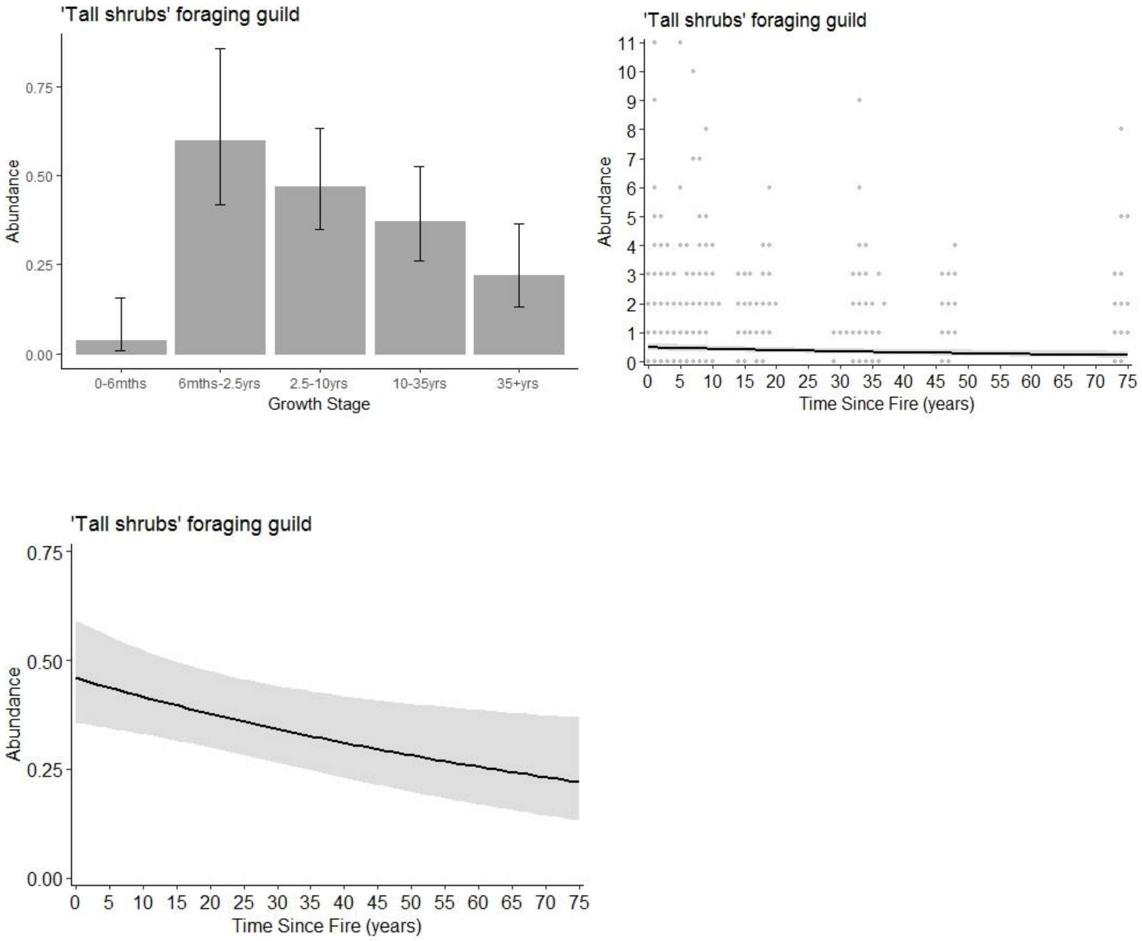


Figure 3.2 Models of Tall shrub forager data with vegetation as continuous age, with and without raw data and with a categorical age class for vegetation.

Generalized linear mixed models. Model selection based on random effect structure (site within mosaic) and ranked using Akaike's Information Criteria (AICc).

A further example of differences in detail, is the distinction between stages under ten years post-fire. The 'new growth' stage of heathy dry forest vegetation post-fire is the time during which ground cover is at its greatest extent, as new shoots develop (Table 2.1). Based on a categorical measure, this period is the 2.5-10 year age class. The comparative graphs for nectarivores all show an increase in abundance in the first ten years (Fig 3.3). The categorical output shows an increase in abundance in the 2.5-10 year age class and the continuous graph shows the greatest abundances within the first ten years. However, more information is revealed in the categorical example, as the response is shown to be greater in ‘new growth’ as opposed to the slightly younger ‘regrowth’ vegetation.

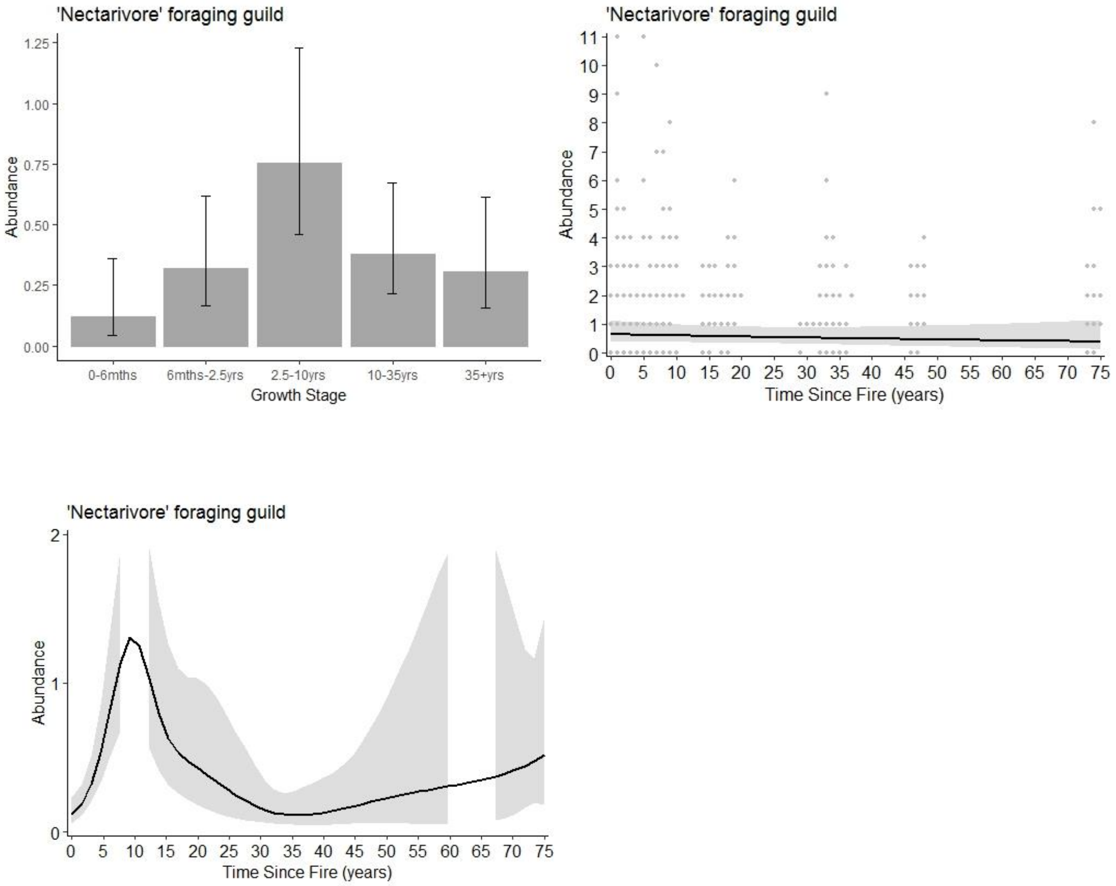


Figure 3.3 Models of Nectarivore data with vegetation as continuous age, with and without raw data and with a categorical age class for vegetation.

Generalized linear mixed models. Model selection based on random effect structure (site within mosaic) and ranked using Akaike’s Information Criteria (AICc).

Bird responses displayed similarly in each comparison across foraging guilds (Appendix A1). However, variations exist in the shape of the distribution response curves. Guild responses showed in the continuous graphs as: ‘delayed’ (Fig 3.1), ‘decline’ (Fig 3.2) and ‘irruptive’ (Fig 3.3). Further to these, the ‘bark foragers' continuous age graph presented as 'null' (Appendix A1, Table 3.1), (Watson et al., 2012). The categorical graphs highlight that these response curves are triggered by the 2.5-10 year old age class e.g. nectarivore curve is bell shaped, possibly irruptive, triggered by the 2.5-10 year old age class (Fig 3.3). Considering the importance of this age class, all analyses in this research were completed using time since fire age classifications.

3.2.2 Broad community responses to TSF and fire frequency

Overall community responses

Between winter 2012 and summer 2014, a total of 10,296 birds was recorded across a total of 56 species within 12 foraging guilds (Table 3.2). For a site comparison of TSF categories, a significant difference between age classes was only found between the newly burnt vegetation and the other classes (Fig 3.4, Table 3.2). Sites newly burnt had distinct structural changes and sites that were more than 6 months old had extensive epicormic regrowth and therefore, possibly higher numbers of lerp, which supported a higher bird abundance across the TSF2 and 3 age classes. A review of bird responses to fire frequency showed that there was little difference across fire frequency classes (Fig 3.5).

Generalized linear mixed modelling of fire frequency yielded no significant levels of difference between any of the frequency classes. So, as a community, birds do not appear to be responding to fire frequency in the terms of this method of classification. Furthermore, the tolerable fire intervals measure used for determining the minimum time required for vegetation to regenerate post-fire, appears not to be a useful measure for bird abundance differences.

Table 3-2 Species and foraging guilds of Victoria's Heathy Dry Forest.

Data are pooled abundances cross 672 monitoring sessions, on 84 sites in Central Victoria. Grouped into abundance classes: U - uncommon, M - moderately common, C - common. Common species classified as those contributing to approximately 80+% of community assemblage patterns and displaying significant differences in TSF vegetation age classes. Moderately common were birds from the remainder of the species, observed on more than 10% of sites. 'Sites' shows the number of sites, each species was observed at.

	Foraging guild		**Common name**	**Scientific name**	**Total**	**Class**	**Sites**
B	Bark forager	takes invertebrates from bark on trunks and branches	Varied Sittella	*Daphoenositta chrysoptera*	82	M	15
			White-throated Treecreeper	*Corombates leucophaea*	884	C	84
C	Canopy forager	takes invertebrates from foliage of eucalypts and other large trees	Black-faced Cuckoo-shrike	*Coracina novaehollandiae*	9	U	6
			Satin Flycatcher	*Myiagra cyanoleuca*	2	U	2
			Shining Bronze Cuckoo	*Chalcites lucidus*	12	U	8
			Weebill	*Smicrornis brevirostris*	7	U	1
			Rufous Whistler	*Pachycephala rufiventris*	145	M	47
			Striated Pardalote	*Pardalotus striatus*	24	M	17
			Grey Fantail	*Rhipidura albiscapa*	560	C	77
			Spotted Pardalote	*Pardalotus punctatus*	422	C	79
			Striated Thornbill	*Acanthiza lineata*	1809	C	82
DG	Damp ground insectivores	takes invertebrates from damp ground below shrubs, among dense understorey or among damp litter	Common Blackbird	*Turdus merula*	1	U	1
			Eastern Yellow Robin	*Eopsaltria australis*	65	M	31
			White-browed Scrubwren	*Sericornis frontalis*	41	C	21
F	Frugivore	takes soft fruit along with other food such as nectar, invertebrates or seeds	Mistletoebird	*Dicaeum hirundinaceum*	2	U	2
N	Nectarivores	takes nectar along with other food such as seeds (parrots) and fruit or invertebrates	Little Lorikeet	*Glossopsitta pusilla*	2	U	1
			Little Wattlebird	*Anthochaera chrysoptera*	2	U	2
			New Holland Honeyeater	*Phylidonyris novaehollandiae*	17	U	5
			Brown-headed Honeyeater	*Melithreptus brevirostris*	58	M	28
			Eastern Spinebill	*Acanthorhynchus tenuirostris*	35	M	22
			Red Wattlebird	*Anthochaera carunculata*	123	M	25
			White-naped Honeyeater	*Melithreptus lunatus*	39	M	17
			Yellow-faced Honeyeater	*Lichenostomus chrysops*	67	M	24
			White-eared Honeyeater	*Lichenostomus leucotis*	207	C	51
OG	Open ground	takes invertebrates from open ground, quite often far from tree or shrub cover	Southern Whiteface	*Aphelocephala leucopsis*	4	U	1
			Yellow-rumped Thornbill	*Acanthiza chrysorrhoa*	9	U	4
			Australian Magpie	*Cracticus tibicen*	233	C	45

OT	Open trees	takes invertebrates from open ground among trees or scattered tall shrubs, but not from damp ground below dense cover, and does not usually venture far from woody vegetation	Flame Robin	*Petroica phoenicea*	4	U	3
			Hooded Robin	*Melanodryas cucullata*	2	U	1
			Pallid Cuckoo	*Cacomantis pallidus*	11	M	11
			Spotted Quail-thrush	*Cinclosoma punctatum*	17	M	10
			Buff-rumped Thornbill	*Acanthiza reguloides*	2339	C	83
			Scarlet Robin	*Petroica boodang*	361	C	81
			Superb Fairy-wren	*Malurus cyaneus*	196	C	47
			White-winged Chough	*Corcorax melanorhamphos*	379	C	23
SG	Seeds ground	takes seeds from ground or low plants such as grasses, herbs and saltmarsh	Blue-winged Parrot	*Neophema chrysostoma*	16	U	6
			Common Bronzewing	*Phaps chalcoptera*	37	M	22
			Diamond Firetail	*Stagonopleura guttata*	4	U	2
			Galah	*Eolophus roseicapillus*	19	U	7
			Red-browed Finch	*Neochmia temporalis*	7	U	3
			Sulphur-crested Cockatoo	*Cacatua galerita*	35	M	17
ST	Seeds trees	seeds from trees/shrubs or wide range of strata, or other food such as gall insects or insect larvae extracted from wood	Crimson Rosella	*Platycercus elegans*	797	C	83
TS	Tall shrubs	takes invertebrates from foliage of tall shrubs, which may form middle storey of eucalypt forests	Yellow Thornbill	*Acanthiza nana*	1	U	1
			Fan-tailed Cuckoo	*Cacomantis flabelliformis*	72	C	46
			Golden Whistler	*Pachycephala pectoralis*	21	M	10
			Brown Thornbill	*Acanthiza pusilla*	203	C	68
V	Carnivore	carnivore, taking vertebrates as an important part of diet, often along with large invertebrates and other food such as fruit	Brown Goshawk	*Accipiter fasciatus*	1	U	1
			Sacred Kingfisher	*Todiramphus sanctus*	2	U	2
			Southern Boobook	*Ninox novaeseelandiae*	2	U	2
			Wedge-tailed Eagle	*Aquila audax*	1	U	1
			Australian Raven	*Corvus coronoides*	83	M	23
			Pied Currawong	*Strepera graculina*	23	M	15
			Grey Currawong	*Strepera versicolor*	212	C	63
			Grey Shrike-thrush	*Colluricincla harmonica*	413	C	79
			Laughing Kookaburra	*Dacelo novaeguineae*	171	C	45
W	Water	water bird inhabiting inland waters	Australian Wood Duck	*Chenonetta jubata*	6	U	1

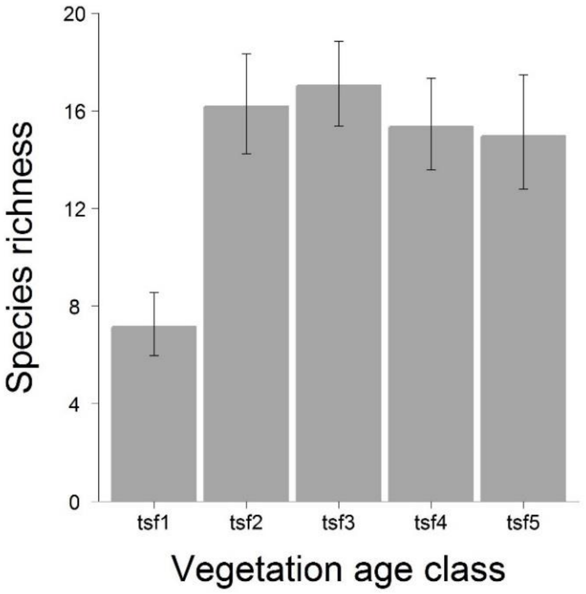


Figure 3.4 Bird community responses to five TSF vegetation age classes.

Generalized linear mixed models. Model selection based on random effect structure (mosaic) and ranked using Akaike's Information Criteria (AICc). TSF categories: tsf1 0-6 months, tsf2 6 months – 2.5 years, tsf3 2.5 years to 10 years, tsf4 10 to 35 years, tsf5 35+ years, combined with last burn not recorded.

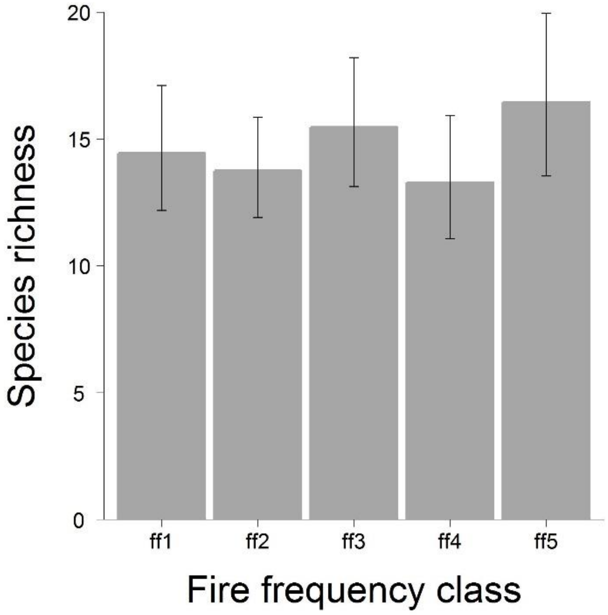


Figure 3.5 Bird community responses to five fire frequency classes.

Generalized linear mixed models. Model selection based on random effect structure (mosaic) and ranked using Akaike's Information Criteria (AICc). Classifications: **ff1** unburnt or burnt over 34 years ago; **ff2** one burn within last 20 years, or, one fire within last 20 years plus one fire over 45 years ago; **ff3** 2-4 fires recorded since 1939 and not breaching tolerable fire interval (TFI) of 10 years post prescribed burn or 15 years post bushfire; **ff4** 0-2 fires in the last 19 years + 1-3 fires > 20 years ago with TFI breached >22 years ago **ff5** 2-3 fires within the last 19 years with TFI breached within last 19 years.

Avian responses in terms of foraging guilds

Generalized linear mixed models were developed to investigate broad foraging guild responses across the TSF age classes. Two foraging guilds, water birds and frugivores, were not modelled, as monitoring returned numbers of only six and two individuals respectively. The random effect of ‘site within mosaic’ was the most parsimonious model, in all instances.

As ten guilds were modelled and graphed and each tested with five reference levels to determine significant differences between TSF and fire frequency categories, only the significant results are tabled for TSF and fire frequency (Table 3.3, 3.4). However, graphed outputs, for each guild in terms of TSF and fire frequency, are included in Appendix A2.

Among the remaining ten guilds, all showed significant differences in assemblages surveyed in newly burnt vegetation compared with other age classes (Table 3.3). It was clear when monitoring sites that a number of the burns, occurring on sites that categorized the TSF1 age class, had resulted in extensive ground vegetation removal, bark scorch and canopy burn. All feeding groups would have been greatly impacted in terms of food availability, protection and nest site availability in the first six months post fire.

Responses by the foraging guilds fell into three broad groups. The first group was those that were less abundant in post fire vegetation, with no apparent differences in other vegetation age classes. This group comprised insectivores that forage from bark and canopy, and on open ground (may be far from cover) and open ground among trees (Table 3.3).

The second broad group comprised those that not only responded with a significant drop in abundance in newly burnt vegetation, but further, showed significant increases in the regrowth vegetation as opposed to the older, less dense vegetation. This group comprised insectivores that feed from tall shrubs (Fig 3.6*i*), insectivores that feed from damp ground, seed-eaters that feed close to the ground, and nectarivores (Table 3.3, Fig 3.6*ii*). Nectarivores were seen to be most common in regrowth of 2.5 to 10 years, observed on sites as having the densest vegetation, where eucalypts were flowering at the time of surveys.

Table 3-3   Bird community and foraging guild responses to five TSF vegetation age classes.

Generalized linear mixed models. Model selection based on random effect structure (‘mosaic’ for community and ‘site within mosaic’ for foraging guilds) and ranked using Akaike’s Information Criteria (AICc). Reference is the TSF age class used as the model reference; positive estimates indicate reference mean is the lowest. Only variations between age classes giving a 95% and greater confidence level are reported. TSF categories: (1) 0-6 months, (2) 6 months – 2.5 years, (3) 2.5 years to 10 years, (4) 10 to 35 years, (5) 35+ years, combined with last burn not recorded. Refer to Table 4.1 for scientific names.

				Abundances per TSF age classes				
Community	Foraging Guild	Akaike weight	Ref	TSF Low	TSF High	Estimate	SE	P
Forest birds		0.74	1	1	2	0.82	0.10	3.86E-16
			1	1	3	0.87	0.10	<2E-16
			1	1	4	0.76	0.10	1.42E-13
			1	1	5	0.74	0.12	1.81E-10
	Bark	1.00	1	1	2	0.59	0.16	0.02E-2
			1	1	3	0.47	0.18	0.023
	Canopy	1.00	2	1	2	-0.41	0.21	0.040
			2	3	2	-0.40	0.15	0.009
			2	4	2	-0.48	0.16	0.003
			2	5	2	-0.65	0.18	0.04E-2
	Damp ground	1.00	3	1	3	-2.44	0.81	0.003
			3	2	3	-0.91	0.42	0.030
			3	4	3	-0.96	0.45	0.036
			3	5	3	-1.76	0.67	0.009
			1	1	2	1.53	0.76	0.042
	Nectarivores	1.00	3	1	3	-1.82	0.03E-1	<2E-16
			3	2	3	-0.85	0.03E-1	<2E-16

			3	4	3	-0.68	0.03E-1	<2E-16
			3	5	3	-0.89	0.03E-1	<2E-16
			1	1	2	0.96	0.003	<2E-16
			1	1	4	1.13	0.003	<2E-16
			1	1	5	0.91	0.003	<2E-16
	Open tree	1.00	1	1	2	0.74	0.23	0.001
			1	1	3	0.56	0.25	0.023
	Seeds on ground	1.00	1	1	2	2.35	1.03	0.022
			1	1	3	2.40	1.07	0.025
			2	4	2	-1.45	0.53	0.005
			3	4	3	-1.50	0.49	0.002
			5	4	5	-1.22	0.58	0.040
	Seeds trees	1.00	2	3	2	0.53	0.19	0.004
			4	4	2	0.70	0.19	0.02E-2
			4	4	5	0.46	0.21	0.025
	Tall shrubs	0.94	1	1	2	2.78	0.73	0.01E-2
			1	1	3	2.54	0.74	0.05E-2
			1	1	4	2.30	0.74	0.002
			1	1	5	1.77	0.77	0.020
			2	5	2	1.01	0.29	0.5E-3
			3	5	3	0.77	0.28	0.5E-2
	Carnivore	1.00	1	1	2	0.60	0.17	<0.2E-3
			2	4	2	-0.47	0.16	0.003

One of the common species in the guild of insectivores that forage from damp ground, the White-browed Scrubwren, was mainly found amongst dense regrowth stands of bracken *(Pteridium esculentum).* Both flowering regrowth and bracken were less prevalent in the older, more open vegetation than in regrowth of intermediate age.

The third broad group of TSF foraging guild responses involved two guilds (seed-eaters that feed in trees and the carnivores) that were scarce in recently burnt stands and showed complex patterns in older regrowth. Both guilds showed a preference for young regrowth and old vegetation stands (TSF2 and TSF5) rather than intermediate age classes (Fig 3.6 *iii*, *iv*, Table 3.3), perhaps reflecting their need for open stands within regrowth.

The fire frequency models were designed to investigate a fire frequency response and further, to determine whether the TFI parameter applied to vegetation age classes is an appropriate classifier for bird responses (Table 3.4). Modelling the entire community against fire frequency showed little variation between classes suggesting no apparent frequency or TFI responses (Fig 3.5). A similar lack of response to frequent fires was shown by a number of foraging guilds: the seed-eaters and foragers that feed on open ground, as well as those that feed on seeds in trees, along with the nectarivores.

Table 3-4 Bird community and foraging guild responses to five fire frequency classes.

Generalized linear mixed models. Model selection based on random effect structure (‘mosaic’ for community and ‘site within mosaic’ for foraging guilds) and ranked using Akaike’s Information Criteria (AICc). Reference is the fire frequency age class used as the model reference; positive estimates indicate reference mean is the lowest. Only variations between age classes giving a 95% and greater confidence level are reported. Fire frequency categories**:** **ff1** unburnt or burnt over 34 years ago; **ff2** one burn within last 20 years, or, one fire within last 20 years plus one fire over 45 years ago; **ff3** 2-4 fires recorded since 1939 and not breaching tolerable fire interval (TFI) of 10 years post prescribed burn or 15 years post bushfire; **ff4** 0-2 fires in the last 19 years + 1-3 fires > 20 years ago with TFI breached >22 years ago **ff5** 2-3 fires within the last 19 years with TFI breached within last 19 years.
Refer to Table 4.1 for scientific names.

			Abundances per fire frequency class				
Foraging Guild	Akaike weight	Ref	TSF Low	TSF High	Estimate	SE	P
Bark	0.77	5	2	5	-0.47	0.22	0.028
		5	4	5	-0.48	0.23	0.043
Nectarivore	1.00	5	1	5	-0.24	0.01	2.00E-16
		5	2	5	0.19	0.01	2.00E-16
		5	3	5	0.51	0.01	2.00E-16
		5	4	5	0.02	0.01	7.3E-06
Seeds ground	1.00	5	1	5	-0.74	0.01	2.00E-16
		5	2	5	-0.35	0.01	2.00E-16
		5	3	5	-0.86	0.01	2.00E-16
		5	4	5	-0.56	0.01	2.00E-16
Seeds trees	1.00	1	2	1	-0.44	0.22	0.049
Carnivore	1.00	1	2	1	-0.43	0.21	0.036
		1	4	1	-0.51	0.24	0.032

Whilst few guilds showed significant differences between fire frequency categories, six of the ten guilds revealed increased abundances in ff5 (frequently burnt) sites: bark, canopy, damp ground, open tree, seed on ground, tall shrubs and carnivorous feeders (Appendix 2). This response ties in with the responses by each of these guilds in that they all had abundance increases in regrowth vegetation (tsf2) and/or new growth vegetation (tsf3) (Table 3.3). While there was a clear preference for sites burnt frequently, there was no distinction between those with TFI's and those where TFI's were not maintained (Fig 3.7*i, ii, iii*).

The model for the guild of carnivorous birds resulted in a significant increase in abundance on unburnt sites over sites with one burn in the last 20 years (ff2) as well as sites with 0-2 burns (with any TFI breach over 20 years prior) (ff4), (Fig 3.7*iv*). This result aligns with this guild's response to TSF in that there was a preference for sites with open ground – tsf2, tsf5 (Fig 3.6*iv*).

Bark foragers showed an increase in sites with multiple burns (TFI's broken) compared with other fire frequency classes (Fig 3.7*iii*). Further, there was little distinction between unburnt (ff1) and frequently burnt sites (ff5). This guild also showed little difference in abundances across TSF age classes other than clearly lower abundances on newly burnt sites (Table 3.3). The reduced abundances in bark foragers across sites with infrequent burning (ff2, ff3 ff4) suggests that habitat for bark foragers can be produced either by frequent burning or by leaving sites unburnt for long periods, but not by intermediate strategies of infrequent burning.

Overall, there was little evidence, at the broad community scale, of response to fire frequency and no evidence that vegetation TFI's are an appropriate predictor of bird abundances.

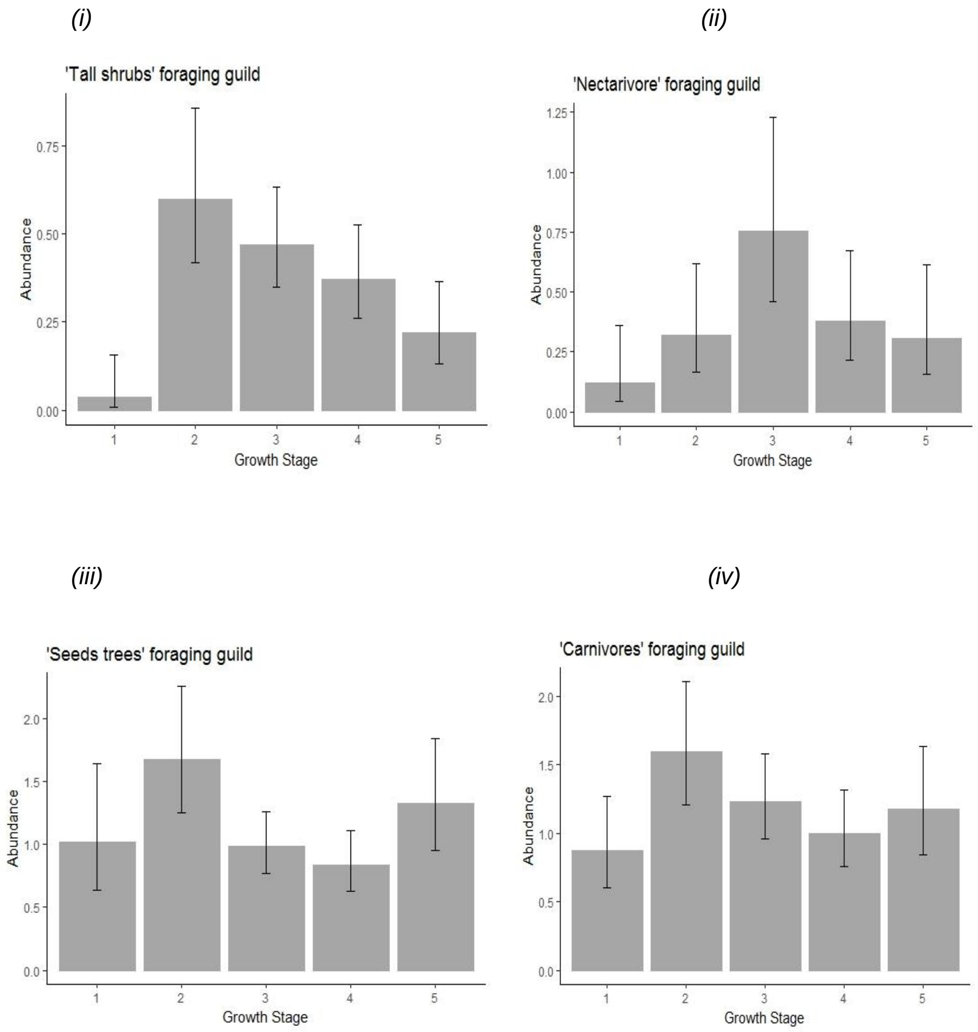


Figure 3.6 Foraging guild responses to TSF vegetation age classes.

Generalized linear mixed models. Model selection based on random effect structure (site within mosaic) and ranked using Akaike's Information Criteria (AICc). TSF categories: **1** 0-6 months, **2** 6 months – 2.5 years, **3** 2.5 years to 10 years, **4** 10 to 35 years, **5** 35+ years, combined with last burn not recorded.

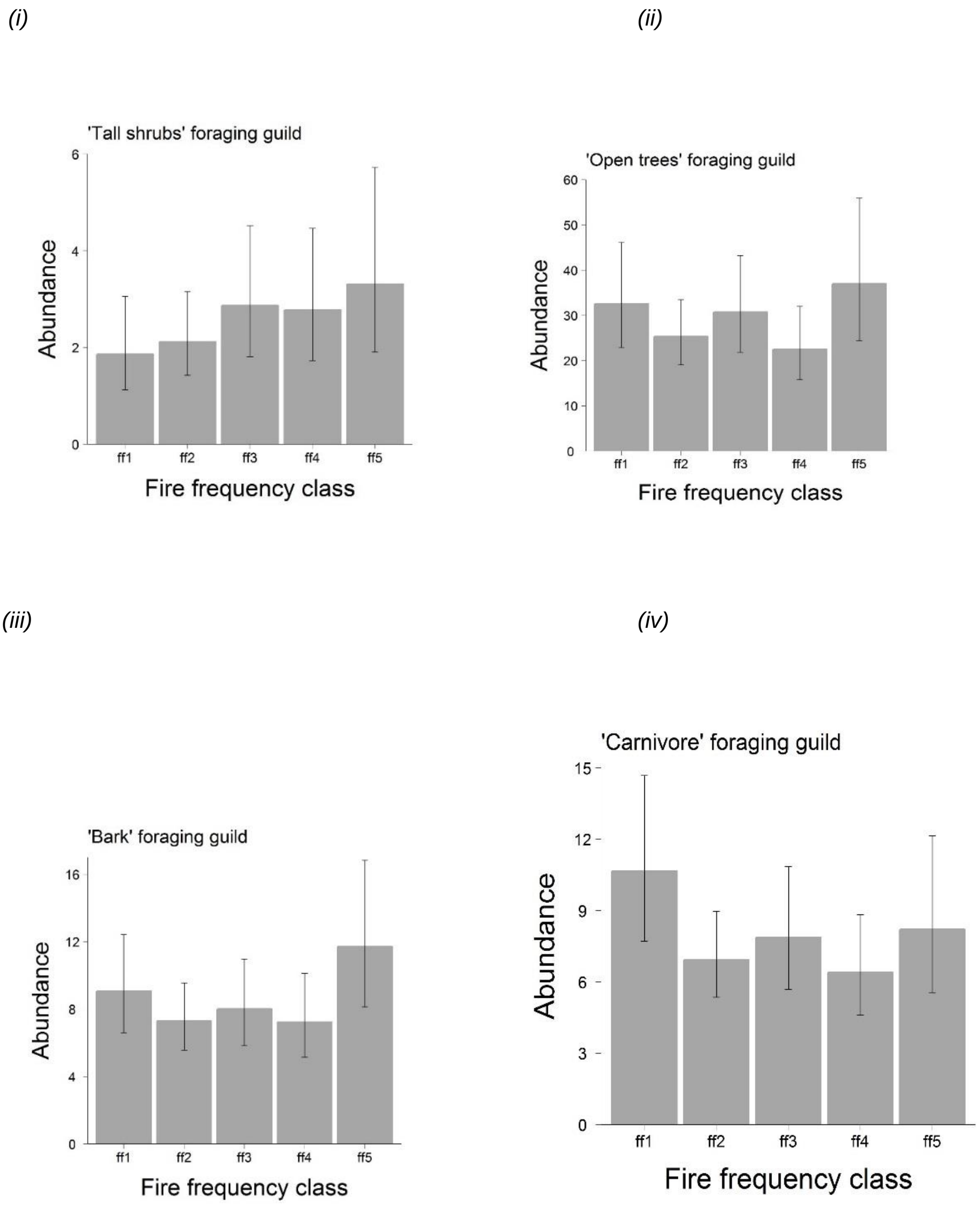


Figure 3.7 Foraging guild responses to fire frequency classes.

Generalized linear mixed models. Model selection based on random effect structure (site within mosaic) and ranked using Akaike's Information Criteria (AICc). Fire frequency categories: **ff1** unburnt or burnt over 34 years ago; **ff2** one burn within last 20 years, or, one fire within last 20 years plus one fire over 45 years ago; **ff3** 2-4 fires recorded since 1939 and not breaching tolerable fire interval (TFI) of 10 years post prescribed burn or 15 years post bushfire; **ff4** 0-2 fires in the last 19 years + 1-3 fires > 20 years ago with TFI breached >22 years ago **ff5** 2-3 fires within the last 19 years with TFI breached within last 19 years.

3.2.3 *Community responses to TSF ordinations illustrating patterns over time*

Species' responses - time since fire

The nMDS ordination of the similarity between species assemblages on each site, with TSF categories as a factor, indicated compositional differences between several TSF age classes (Fig 3.8).

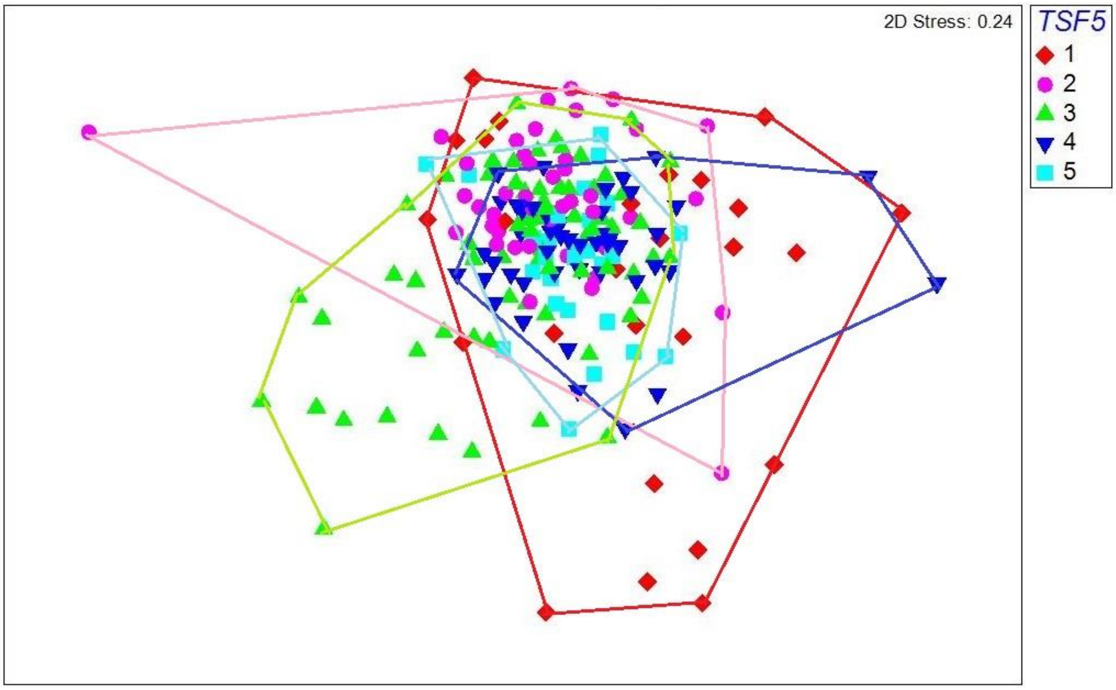


Figure 3.8 Similarity in species assemblages on each site, with factor of TSF vegetation age classes.

nMDS ordination of species on sites with TSF. TSF categories: (1) 0-6 months, (2) 6 months – 2.5 years, (3) 2.5 years to 10 years, (4) 10 to 35 years, (5) 35+ years, combined with last burn not recorded. 3D equivalent ordination had 0.18 stress level.

As uncommon species tend to plot as outliers they tend to skew the plot e.g. TSF2; the ordination shows a central cluster of points, where some site assemblages were common to all TSF ages.

The clustering of individual sites into unique TSF age class groupings, e.g. TSF3, indicated unique assemblage compositions as a property of TSF (Fig 3.8). This ordination suggests some temporal changes to assemblage patterns, an example being the newly burnt (hence open), clear ground (TSF1) group as distinct (on some sites) to the 2.5-10 year since fire (ysf) dense regrowth vegetation (TSF3). The ordination illustrates how the oldest vegetation (TSF5) had the least variation in assemblage patterns, sharing species with all stages. The lack of assemblage variation among the sites with TSF4 and TSF5 age classes is clear with all TSF4 and TSF5 sites grouped within the central cluster of sites of other age classes. This showed that the greatest differences in assemblage patterns existed between the newly burnt (0 – 6 month) and the dense regrowth (2.5 – 10 year) vegetation. Amongst the regrowth vegetation, the White-eared Honeyeater was observed most consistently across all sites of this age class, with a greater abundance observed in the winter months. Further, the regrowth vegetation sites were not geographically clustered, as every broad location (Linton, Enfield, Creswick, Durham Lead, Lal Lal, Mt Egerton and the Brisbane Ranges) had representative sites of this age class.

This result was supported by the outcomes of the Permanova+ analysis which demonstrated a significant effect for TSF (pseudo-F=1.666, P=0.0096) and also for season (pseudo-F=8.7026, P=0.0002).

Pairwise testing was performed to identify the age classes which provided the greatest contribution to dissimilarity between assemblages. The greatest contrasts were found between TSF1 (newly burnt age class) and TSF3 (2.5-10 ysf), (t=2.6826, P=0.001). There were also clear dissimilarities in assemblage structure between TSF1 and TSF5 (35+ysf), (t=1.6455, P=0.007) and between TSF1 and TSF2 (.5-2.5 ysf), (t=1.6988, P=0.008).

Fifteen species contributed to the 79% cut-off used to measure dissimilarity in bird assemblages across sites using a SIMPER one-way analysis, comparing the newly burnt vegetation (TSF1) and the other three age classes (TSF3, 5 and 2) (Table 3.5). Of these, twelve species were present in all comparisons. This group comprised the Australian Magpie, Buff-rumped Thornbill, Crimson Rosella, Grey Fantail, Grey Currawong, Grey Shrike-thrush, Laughing Kookaburra, Scarlet Robin, Spotted Pardalote, Striated Thornbill, White-throated Treecreeper, and the White-winged Chough. Further to these, the Superb Fairy-wren and the White-eared Honeyeater contributed to the dissimilarity between TSF1 (newly burnt) and TSF3 (2.5-10 ysf), and the Brown Thornbill to dissimilarity between TSF1 and TSF3 along with TSF1 compared with TSF2 (.5-2.5 ysf) (Table 3.5). Of the total White-eared Honeyeaters and Superb Fairy-wrens observed (Table 3.2), the greatest abundances for both these species appeared in the flowering vegetation of TSF3 (Fig 3.17). The Brown Thornbill had a preference for vegetation classes TSF2 and TSF3 (Fig 3.15) and was mainly observed on sites feeding in foraging flocks on post-fire insects and lerps.

Table 3-5 SIMPER results of Bray-Curtis similarity percentages giving species contributions to sites with different TSF vegetation age classes.

One-way analysis with data transformed log(x+1). Bird acronyms: BFTH Buff-rumped Thornbill, STTH Striated Thornbill, CRRO Crimson Rosella, GYFA Grey Fantail, WTTR White-throated Treecreeper, GYST Grey Shrike-thrush, WEHO White-eared Honeyeater, SPPA Spotted Pardalote, SCRO Scarlet Robin, WWCH White-winged Chough, SUFW Superb Fairy-wren, GYCU Grey Currawong, AUMA Australian Magpie, LAKO Laughing Kookaburra, BRTH Brown Thornbill.

Groups 3 & 1
Average dissimilarity = 64.58

Species	Gp 3 Av.Abun	Gp 1 Av.Abun	Av.Diss	Diss/SD	Cont%	Cum.%
BFTH	1.40	0.74	7.81	1.30	12.10	12.10
STTH	1.03	1.16	6.76	1.27	10.46	22.56
CRRO	0.61	0.63	4.47	1.29	6.93	29.49
GYFA	0.46	0.43	4.12	1.01	6.37	35.86
WTTR	0.72	0.68	3.63	1.28	5.62	41.48
GYST	0.47	0.16	3.40	1.22	5.27	46.75
WEHO	0.32	0.07	2.84	0.72	4.40	51.16
SPPA	0.38	0.14	2.78	1.12	4.31	55.47
SCRO	0.38	0.33	2.69	1.22	4.16	59.63
WWCH	0.10	0.34	2.60	0.50	4.03	63.66
SUFW	0.27	0.15	2.17	0.88	3.36	67.03
GYCU	0.19	0.23	2.12	0.90	3.29	70.31
AUMA	0.14	0.22	2.09	0.63	3.24	73.55
LAKO	0.10	0.24	1.95	0.64	3.02	76.57
BRTH	0.26	0.04	1.90	0.93	2.94	79.51

Groups 1 & 2
Average dissimilarity = 62.05

Species	Gp 1 Av.Abun	Gp 2 Av.Abun	Av.Diss	Diss/SD	Cont%	Cum.%
BFTH	0.74	1.49	7.61	1.27	12.27	12.27
STTH	1.16	1.30	5.91	1.17	9.53	21.79
CRRO	0.63	0.89	4.88	1.29	7.87	29.66
SPPA	0.14	0.70	4.47	1.24	7.20	36.86
GYFA	0.43	0.47	3.82	0.97	6.16	43.02
WTTR	0.68	0.91	3.57	1.22	5.76	48.78
WWCH	0.34	0.32	3.34	0.60	5.38	54.16
GYST	0.16	0.53	3.24	1.21	5.21	59.37
AUMA	0.22	0.31	2.66	0.74	4.29	63.66
SCRO	0.33	0.48	2.64	1.18	4.26	67.92
LAKO	0.24	0.22	2.32	0.83	3.74	71.65
GYCU	0.23	0.31	2.31	1.03	3.73	75.38
BRTH	0.04	0.34	2.21	1.05	3.56	78.94

Groups 4 & 1
Average dissimilarity = 59.96

Species	Gp 4 Av.Abun	Gp 1 Av.Abun	Av.Diss	Diss/SD	Cont%	Cum.%
BFTH	1.31	0.74	8.22	1.21	13.71	13.71
STTH	1.25	1.16	6.97	1.19	11.62	25.33
CRRO	0.56	0.63	4.83	1.16	8.06	33.38
GYFA	0.39	0.43	3.95	1.05	6.58	39.96
WTTR	0.81	0.68	3.72	1.08	6.20	46.16
GYST	0.42	0.16	3.17	1.25	5.29	51.45
SPPA	0.34	0.14	2.85	1.11	4.75	56.20
SCRO	0.33	0.33	2.79	1.17	4.66	60.86
WWCH	0.12	0.34	2.79	0.50	4.65	65.51
LAKO	0.22	0.24	2.74	0.82	4.56	70.07
AUMA	0.20	0.22	2.55	0.69	4.25	74.32
GYCU	0.20	0.23	2.36	0.89	3.94	78.26

Groups 5 & 1
Average dissimilarity = 59.24

Species	Gp 5 Av.Abun	Gp 1 Av.Abun	Av.Diss	Diss/SD	Cont%	Cum.%
BFTH	1.23	0.74	7.31	1.29	12.34	12.34
STTH	1.04	1.16	6.3	1.35	10.63	22.98
CRRO	0.77	0.63	4.98	1.38	8.40	31.38
GYFA	0.37	0.43	3.94	1.06	6.66	38.03
WWCH	0.33	0.34	3.82	0.64	6.44	44.47
WTTR	0.82	0.68	3.36	1.07	5.67	50.14
SPPA	0.37	0.14	3.05	1.25	5.14	55.28
SCRO	0.45	0.33	2.96	1.31	5.00	60.29
LAKO	0.25	0.24	2.85	0.91	4.81	65.10
GYST	0.34	0.16	2.74	1.22	4.63	69.72
GYCU	0.25	0.23	2.55	0.97	4.30	74.02
AUMA	0.13	0.22	2.14	0.63	3.61	77.63

Species' responses – fire frequency

Similar analyses of investigative ordinations with statistical support, were run with fire frequency factors. Permanova+ results for the whole community suggested that while there was a seasonal variation across mosaics (Pseudo f=1.61, P=0.001), there was no fire frequency effect evident with the complete dataset (Pseudo f=1.20, P=0.192). Results from pairwise testing indicated significant dissimilarities between unburnt sites (ff1) and frequently burnt sites (ff5) (t=1.06, P=0.041), in winter months. This suggests that species are responding to vegetation changes in winter months; a response not apparent in the summer months. It may be that food resources are scarce in the winter months, driving birds to areas of winter flowering. In summer months, after spring vegetation flowering and regrowth, food resources are more readily available across the landscape. This aligns with the result whereby the greatest differences in assemblage patterns existed between the newly burnt and regrowth vegetation. White-eared Honeyeaters were common in the regrowth vegetation, and were observed more frequently in winter months.

Results from a SIMPER analysis indicated that there were 17 species contributing to 85% of the dissimilarity between ff1 and ff2 in the winter months. These comprised the fifteen species listed as important in the TSF analyses as well as the Fan-tailed Cuckoo and the White-browed Scrubwren. As the Fan-tailed Cuckoo is a seasonal migrant, this result may be independent of seasonal vegetation structure.

3.2.4 Classification of species into three abundance classes

Influential species contributing to 80% of assemblage patterns

Seventeen species were identified as contributing approximately 80+% of assemblage patterns based on the SIMPER analysis. Not surprisingly, many in this group were species whose abundances across all site observations were the greatest (Table 3.2). However, some of this cohort were not in the highest abundances, but were instrumental in assemblage patterns: e.g. Australian Magpie, Superb Fairy-wren, Laughing Kookaburra and the White-eared Honeyeater. These four species each contributed only 2% of the total community abundance. This is in contrast to abundance levels of a number of the species observed much more frequently: the Buff-rumped Thornbill (22% of observations), Striated Thornbill (17%) and White-throated Treecreeper and Crimson Rosella (each 8%) (Table 3.2). Considering this, models were run to check for significant levels of difference between TSF age classes for the common species.

Species grouped as ‘common’, ‘moderately common’ and ‘uncommon’

Once the most influential species had been classified, the remaining species were separated into two groups: ‘moderately common’ being those that had been observed on >10% of the sites, leaving the remainder as ‘uncommon’ (Table 3.2). The ‘common’ and the ‘moderately common’ species were all modelled as GLMM’s using TSF and then fire frequency categories in the modelling. The exception to this were the Spotted Quail-thrush for fire frequency and the Golden Whistler for TSF. The data were too few for these species, when modelled with these factors. While only significant results are reported in tables, all graphs are included (Appendix A3). Subsequent discussions include references to responses by species that may not have statistical significance but are relevant to the discussion.

3.2.5 A review of responses to TSF and fire frequency by the 'common' and 'moderately common' forest bird species

Of the 17 species listed as common (Table 3.2), 15 species not only contributed to the majority of the variations in assemblage patterns, but also displayed significant TSF age class differences (Table 3.6). Of the remaining species observed, 16 were classified as 'moderately common', as these were observed on over 10% of the 84 sites (Table 3.2). To determine if generalizations could be made about foraging guild responses to TSF and fire frequency, and if in fact common species responses could be indicative of entire assemblage patterns, common and moderately common species were grouped into their foraging guilds and all were modelled against TSF and fire frequency (Table 3.6, 3.7).

The results for the fire frequency modelling showed fewer species responding significantly, using this categorical method. However, some species did display fire frequency responses and these are referred to in relevant sections.

Table 3-6 Common and moderately common species with lowest and highest abundance TSF vegetation age class and the statistical significance between the age classes.

Generalized linear mixed models of species. Model selection based on random effect structure (RE) and ranked using Akaike's Information Criteria (AICc). Reference (Ref) is the TSF age class used as the model reference; positive estimates indicate reference mean is the lowest. Only variations between TSF age classes giving a 95% and greater confidence level are reported. TSF categories: (1) 0-6 months, (2) 6 months – 2.5 years, (3) 2.5 years to 10 years, (4) 10 to 35 years, (5) 35+ years, combined with last burn not recorded. Refer to Table 3.2 for scientific names.

Common species **Abundances per TSF age class**

Foraging Guild	Common name	Akaike weight	Ref	TSF Low	TSF High	Estimate	SE	P
Open Trees	Buff-rumped Thornbill	1.00	1	1	2	2.10	0.14	<2E-16
	(RE mosaic/site)		1	1	3	2.13	0.16	<2E-16
			1	1	4	1.78	0.16	<2E-16
			1	1	5	2.10	0.22	<2E-16
	Scarlet Robin	0.97	1	1	2	1.30	0.25	2.26E-07
	(RE mosaic)		1	1	3	1.29	0.26	8.34E-07
			1	1	4	1.14	0.27	2.70E-05
			1	1	5	1.59	0.29	4.99E-08
	Superb Fairy-wren	0.76	1	1	3	2.99	0.97	2.05E-03
	(RE mosaic)		1	1	4	1.79	0.91	4.83E-02
			1	1	5	2.46	1.05	1.90E-01
	White-winged Chough	1.00	3	3	1	1.69	0.58	3.50E-03
	(RE mosaic/site)		3	3	2	2.45	0.55	7.23E-06
			3	3	4	1.42	0.66	3.06E-02
Seeds in trees	Crimson Rosella	1.00	2	1	2	-1.54	0.16	<2E-16
	(RE mosaic/site)		2	3	2	-0.36	0.14	1.09E-02
			2	4	2	-0.63	0.17	2.00E-04
			5	1	5	-1.61	0.22	8.66E-13
			5	3	5	-0.43	0.18	1.84E-02
			5	4	5	-0.70	0.20	3.93E-04
Canopy	Grey Fantail	1.00	2	1	2	-1.57	0.21	1.42E-13
	(RE mosaic/site)		2	4	2	-0.74	0.25	2.90E-03
	Spotted Pardalote	1.00	2	1	2	-2.88	0.36	1.30E-15
	(RE mosaic/site)		2	3	2	-0.55	0.19	3.20E-03
			2	4	2	-0.77	0.22	5.00E-04
			2	5	2	-0.62	0.25	1.44E-02
	Striated Thornbill							
	(RE mosaic/site)	1.00	5	5	1	-0.84	0.20	2.79E-05
Nectarivore	White-eared Honeyeater	1.00	3	1	3	-2.77	0.69	5.55E-05
	(RE mosaic/site)		3	2	3	-0.93	0.46	4.31E-02
Carnivores	Grey Currawong	1.00	1	1	2	1.45	0.33	8.26E-06
	(RE mosaic/site)		1	1	3	1.18	0.37	1.39E-03
			1	1	4	1.00	0.38	9.58E-03
			1	1	5	1.44	0.43	9.10E-04
	Grey Shrike-thrush	0.97	1	1	2	2.22	0.32	2.77E-12
	(RE mosaic/site)		1	1	3	2.24	0.33	1.38E-11
			1	1	4	2.12	0.33	1.69E-10
			1	1	5	1.89	0.37	2.67E-07
	Laughing Kookaburra	1.00	1	1	2	20.37	4.04	4.74E-07

	(RE mosaic/site)		1	1	4	20.11	3.92	2.90E-07
			1	1	5	22.13	5.11	1.52E-05
			3	3	2	20.71	3.75	3.23E-08
			3	3	4	20.45	3.57	1.02E-08
			3	3	5	22.47	4.86	3.72E-06
Bark forager	White-throated Treecreeper	1.00	1	1	2	1.38	0.17	<2E-16
	(RE mosaic/site)		1	1	3	1.30	0.18	7.13E-13
			1	1	4	1.32	0.18	3.29E-13
			1	1	5	1.52	0.17	5.73E-06
Open ground	Australian Magpie	1.00	2	1	2	-1.83	0.30	6.84E-10
	(RE mosaic/site)		2	4	2	-0.96	0.43	2.43E-02
Tall shrubs	Brown Thornbill	0.51	1	1	2	3.32	0.72	3.45E-06
	(RE mosaic)		1	1	3	3.40	0.72	2.51E-06
			1	1	4	2.99	0.72	3.63E-05
			1	1	5	2.86	0.75	1.00E-04

Moderately common species

Nectarivore	Eastern Spinebill	0.76	3	1	3	-1.9218	0.7939	1.55E-02
	(RE mosaic)		3	2	3	-2.5693	1.0507	1.45E-02
			3	4	3	-2.9717	1.0526	4.80E-03
Seeds on ground	Sulphur-crested Cockatoo (RE mosaic)	0.76	2	4	2	-2.8104	1.3562	3.82E-02
	Common Bronzewing	0.76	3	1	3	-2.6923	-2.236	2.53E-03
	(RE mosaic)		3	2	3	-1.9034	-1.984	4.73E-02
			3	4	3	-1.7981	-1.968	4.91E-02
			3	5	3	-2.0919	-1.953	5.09E-02
Damp ground	Eastern Yellow Robin (RE mosaic)	0.76	3	1	3	-1.9706	0.8866	2.62E-02

Table 3-7 Common and moderately common species with lowest and highest abundance fire frequency classes and the statistical significance between classes.

Generalized linear mixed models of species responses. Model selection based on random effect structure (RE) and ranked using Akaike's Information Criteria (AICc). Reference (Ref) is the fire frequency class used as the model reference; positive estimates indicate reference mean is the lowest. Only variations between fire frequencies giving a 95% or greater confidence level are reported.
Fire frequency categories: (**1**) unburnt or burnt over 34 years ago; (**2**) one burn within last 20 years, or, one fire within last 20 years plus one fire over 45 years ago; (**3**) 2-4 fires recorded since 1939 and not breaching tolerable fire interval (TFI) of 10 years post prescribed burn or 15 years post bushfire; (**4**) 0-2 fires in the last 19 years + 1-3 fires > 20 years ago with TFI breached >22 years ago (**5**) 2-3 fires within the last 19 years with TFI breached within last 19 years. Refer to Table 3.2 for scientific names.

Common species **Abundances per fire frequency class**

Foraging Guild	Common name	Akaike weight	Ref	TSF Low	TSF High	Estimate	SE	P
Open Trees	Scarlet Robin	0.95	5	2	5	-0.44	0.22	0.05
	(RE mosaic/site)		5	3	5	-0.53	0.25	0.03
			5	4	5	-0.50	0.25	0.04
Carnivores	Grey Currawong	0.98	5	1	5	0.10	0.001	2.00E-16
	(RE mosaic/site)		5	2	5	-0.59	0.001	2.00E-16
			5	3	5	-0.45	0.001	2.00E-16
			5	4	5	-1.03	0.001	2.00E-16
	Laughing Kookaburra	1.00	1	2	1	-1.30	0.46	5.00E-03
	(RE mosaic/site)		1	5	1	-1.62	0.62	0.01
Tall shrubs	Brown Thornbill	0.50	1	1	3	0.58	0.29	0.04
	(RE mosaic/site)		1	1	5	0.88	0.36	0.01
Moderately Common Species								
Carnivores	Pied Currawong (RE mosaic)	0.78	2	2	5	1.92	0.94	0.04
Nectarivore	Brown-headed Honeyeater	0.78	2	2	1	1.46	0.71	0.04
	(RE mosaic)		2	2	3	1.59	0.70	0.02

Carnivores

This group comprised three common species: Laughing Kookaburra, Grey Shrike-thrush and Grey Currawong. The moderately common species were the Pied Currawong and Australian Raven.

All individual species' models for this guild showed that abundances were lower in newly burnt vegetation (Appendix A3), reflecting the response of the carnivore group as a guild (Fig 3.6*iv*) and the avian community more generally (Fig 3.4).

The Laughing Kookaburra models show a lower abundance in new growth (2.5-10 year old) vegetation (TSF3) (Fig 3.9 *i, iii*, Table 3.6). The lower abundance of Kookaburras in this age class is not representative of the carnivore foraging guild but does reflects its preference for open stands of vegetation, suitable for pounce feeders, as opposed to denser ground cover in the form of new growth vegetation, that may impede their sight for feeding. Further to a preference for open stands of older vegetation, they showed a preference for unburnt sites over sites burnt at all (Fig 3.9 *ii, iv*, Table 3.7). TSF and fire frequency are confounded (old sites are not frequently burnt) so it is impossible to tell if it is TSF or fire frequency, or both, that drives the observed trends in Laughing Kookaburras.

The Laughing Kookaburra was unique in its response to TSF and fire frequency. No other carnivore showed any major abundance differences in vegetation older than six months (Appendix A3). The only other fire frequency response for birds of this guild that were modelled, was that of the Pied Currawong, where modelling suggests a significant increase on sites frequently burnt (Table 3.7).

Of interest is the possible relationship between the Laughing Kookaburra and the Grey Currawong. The results for these two species were more closely examined. Results are presented for both these species as abundance data (Fig 3.9 and 3.10 *i* and *ii*), and further, as presence/absence data (Fig 3.9 and 3.10 *iii* and *iv*), to remove the significance of one site where kookaburras were observed in greater abundance. The binomial graph for the kookaburra shows it to be almost absent on sites of both newly burnt (TSF1) and new growth (TSF3) vegetation (Fig 3.9 *iii*). The confidence intervals on this graph span from 0-1, reflecting a near complete presence or absence on every site within the particular age class.

The abundance graph suggests a reduction, but still a presence in newly burnt and new growth (Fig 3.9 *i*). Indeed, examination of raw data reveals a drop in occurrence in the new growth vegetation. Of the total number of sites Laughing Kookaburras were observed, 12% of presence observations were in newly burnt vegetation, 16% were in new growth, while the remaining 3 classes each had 24% of presence observations. In contrast, the Grey Currawong abundance and occurrence across age classes older than six months, varies little (Fig 3.10 *i, iii*). For the kookaburra, both the binomial and the abundance graph show a preference by the species for unburnt vegetation (Fig 3.9 *ii, iv)*. The Grey Currawong exhibited no fire frequency preferences (Fig 3.10 *ii, iv*).

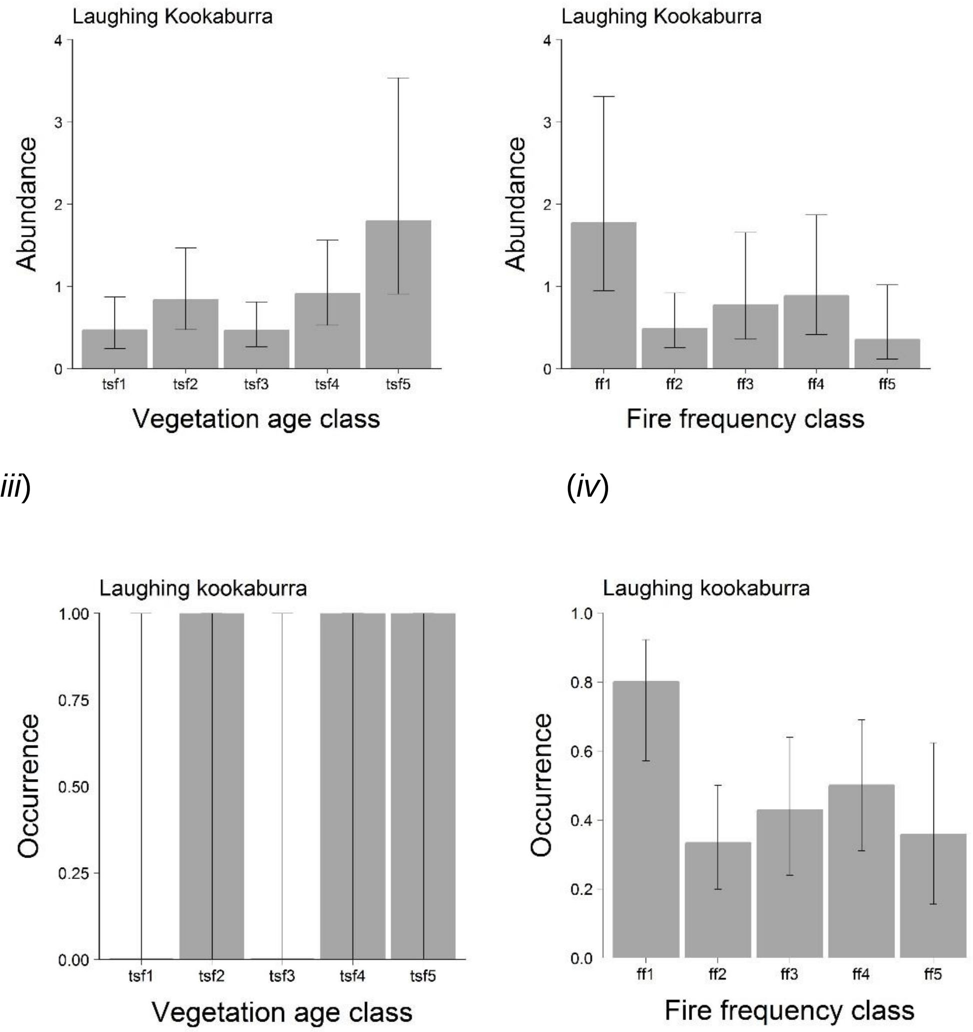
A further comparison was made by reviewing changes in presence observations for these two species, together and independent of one another, across the vegetation age classes (Table 3.8).

Table 3-8 Comparison of Laughing Kookaburra and Grey Currawong presence data, together as species and independent of one another.

Comparison of presence data with presences for each species split per vegetation age classes when observed individually and together, as a percent.

Vegetation age class	Species alone		Species together	Vegetation feature
	Laughing Kookaburra	Grey Currawong		
TSF1 (0-6 mths)	10	8	29	ground bare
TSF2 (6 mths-2.5 yrs)	21	25	21	ground bare
TSF3 (2.5-10 yrs)	18	29	4	dense new growth
TSF4 (10-35 yrs)	29	20	25	open vegetation
TSF5 (35+ yrs)	23	18	21	open vegetation

The results from the review of presence data for the two species highlight a decrease in the presence of Laughing Kookaburras in new growth vegetation, when not with Grey Currawongs. For Grey Currawongs, the new growth vegetation may be their preferred age class. However, the occurrence of both species together in this vegetation, was rare (Table 3.8).



(*i*) (*ii*)

(*iii*) (*iv*)

Figure 3.9 Laughing Kookaburra responses to TSF and fire frequency.

Modelling run as a Poisson distribution (*i, ii)* and then as Binomial presence/absence (*iii, iv*).

Generalized linear mixed models. Model selection based on random effect structure (site within mosaic) and ranked using Akaike's Information Criteria (AICc). TSF categories: **tsf1** 0-6 months, **tsf2** 6 months – 2.5 years, **tsf3** 2.5 years to 10 years, **tsf4** 10 to 35 years, **tsf5** 35+ years, combined with last burn not recorded. Frequency categories: **ff1** unburnt or burnt over 34 years ago; **ff2** one burn within last 20 years, or, one fire within last 20 years plus one fire over 45 years ago; **ff3** 2-4 fires recorded since 1939 and not breaching tolerable fire interval (TFI) of 10 years post prescribed burn or 15 years post bushfire; **ff4** 0-2 fires in the last 19 years + 1-3 fires > 20 years ago with TFI breached >22 years ago **ff5** 2-3 fires within the last 19 years with TFI breached within last 19 years.

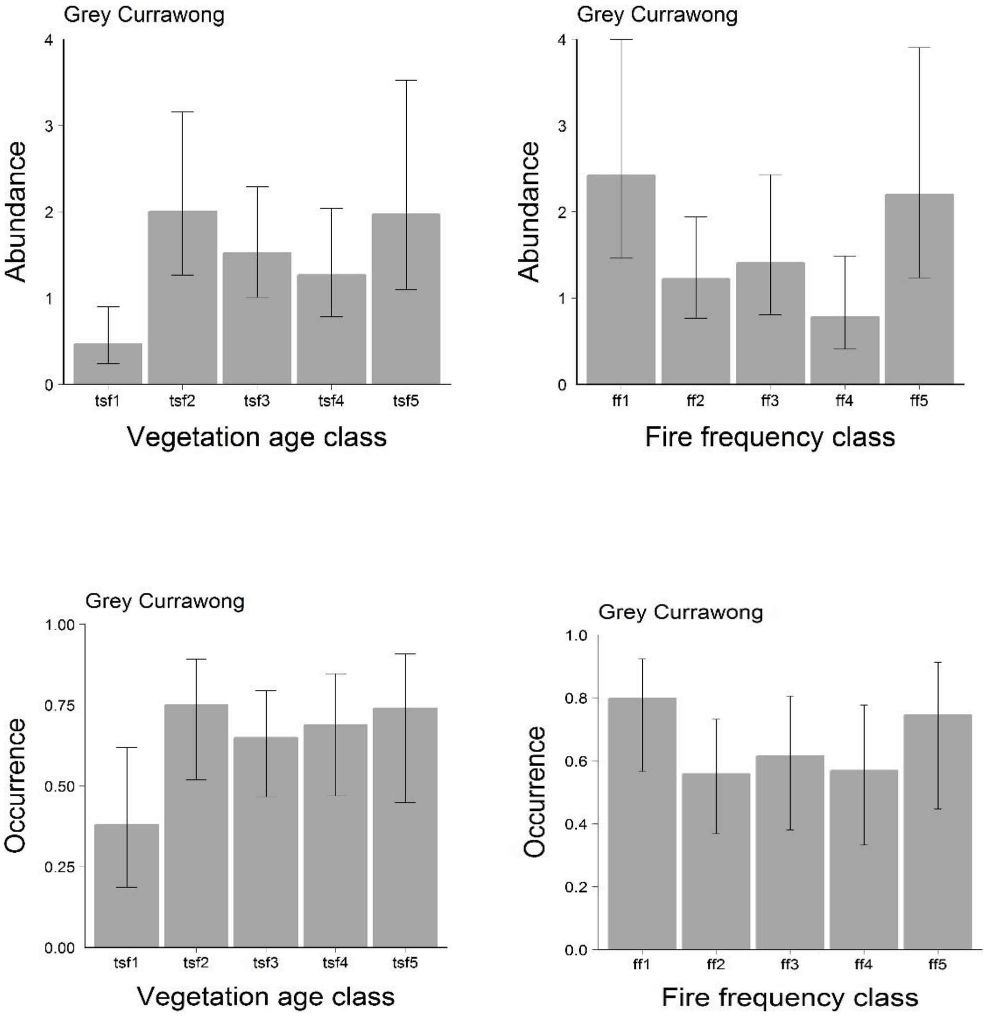


Figure 3.10 Grey Currawong responses to TSF and fire frequency. Modelling run as a Poisson distribution and then as Binomial presence/absence.

Generalized linear mixed models. Model selection based on random effect structure (site within mosaic) and ranked using Akaike's Information Criteria (AICc). TSF categories: **tsf1** 0-6 months, **tsf2** 6 months – 2.5 years, **tsf3** 2.5 years to 10 years, **tsf4** 10 to 35 years, **tsf5** 35+ years, combined with last burn not recorded. Frequency categories: **ff1** unburnt or burnt over 34 years ago; **ff2** one burn within last 20 years, or, one fire within last 20 years plus one fire over 45 years ago; **ff3** 2-4 fires recorded since 1939 and not breaching tolerable fire interval (TFI) of 10 years post prescribed burn or 15 years post bushfire; **ff4** 0-2 fires in the last 19 years + 1-3 fires > 20 years ago with TFI breached >22 years ago **ff5** 2-3 fires within the last 19 years with TFI breached within last 19 years.

Canopy-foraging insectivores

The group of canopy feeders comprised three common species: Grey Fantail, Spotted Pardalote and Striated Thornbill. The moderately common species were the Striated Pardalote and Rufous Whistler.

The Grey Fantail graphs present outputs that broadly reflect the responses of the individual species modelled for this guild (Fig 3.11, Appendix A3). Grey Fantails showed a significant increase in abundance in regrowth and new growth vegetation (TSF2, 3). This result aligns with the Grey Fantail response in the fire frequency model. Even-though results were not significant, the graph highlights an increase in abundance on sites frequently burnt (TFI's broken), (Fig 3.11, Table 3.7).

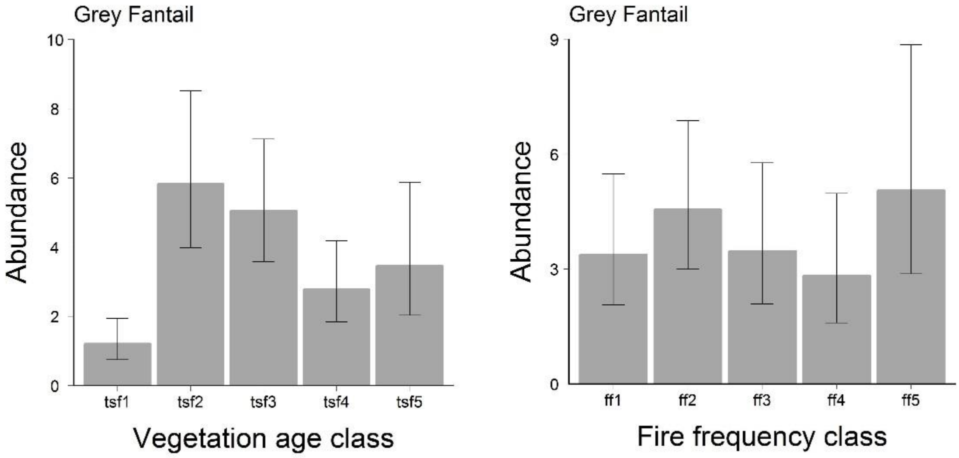


Figure 3.11 Grey Fantail responses to TSF and fire frequency.

Generalized linear mixed models. Model selection based on random effect structure (site within mosaic) and ranked using Akaike's Information Criteria (AICc). TSF categories: **tsf1** 0-6 months, **tsf2** 6 months – 2.5 years, **tsf3** 2.5 years to 10 years, **tsf4** 10 to 35 years, **tsf5** 35+ years, combined with last burn not recorded. Frequency categories: **ff1** unburnt or burnt over 34 years ago; **ff2** one burn within last 20 years, or, one fire within last 20 years plus one fire over 45 years ago; **ff3** 2-4 fires recorded since 1939 and not breaching tolerable fire interval (TFI) of 10 years post prescribed burn or 15 years post bushfire; **ff4** 0-2 fires in the last 19 years + 1-3 fires > 20 years ago with TFI breached >22 years ago **ff5** 2-3 fires within the last 19 years with TFI breached within last 19 years.

Bark-foraging insectivores

There were two species reviewed from the bark foraging guild: the White-throated Treecreeper and Varied Sittella. White-throated Treecreeper numbers were lower in newly burnt vegetation and the Varied Sittella was absent (Appendix A3). While the modelling result was not significant, the graph for the Varied Sittella highlights an increase in occurrence in regrowth vegetation (Appendix 3A).

In terms of fire frequency, there are no notable results for the White-throated Treecreeper. However, the graph outputs for the Varied Sittella model show an increase in occurrence on sites that had been burnt more frequently than only once (Appendix 3A). This reflects that Varied Sittellas are responding to fire through a greater abundance in frequently burnt sites, and that this may be on account of a greater abundance of their insect food source which may be stimulated by regrowth vegetation.

Nectarivores

The abundance of only one nectarivore, the White-eared Honeyeater, was influential in the assemblage patterns by being almost exclusively detected in new growth vegetation (TSF3), (Fig 3.12, Table 3.6). This species was observed feeding on honeydew amongst the new growth and amongst tall shrubs. There were five moderately common nectarivores in this study: Brown-headed Honeyeater, White-naped Honeyeater, Yellow-faced Honeyeater, Red Wattlebird and Eastern Spinebill. The Eastern Spinebill and Red Wattlebird both showed a similar response to the White-eared Honeyeater, in being more abundant in new growth vegetation (Table 3.6, Appendix A3). While no other nectarivore exhibited any significant responses across any TSF age classes, there was a further response besides an increase in abundance or occurrence in new growth. Graphs for both the White-naped and Yellow-faced Honeyeaters reveal a preference for older vegetation (TSF4), further supported by the graph of Brown-headed Honeyeater showing a greater occurrence in vegetation older than 35 years (Appendix A3).

Modelling for fire frequency showed that no nectarivores responded significantly to the fire frequency classes.

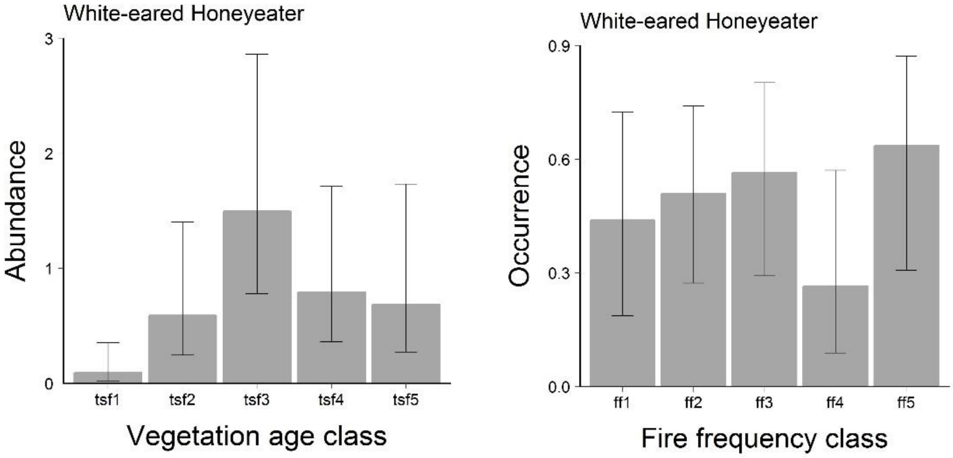


Figure 3.12 White-eared Honeyeater responses to TSF.

Generalized linear mixed models. Model selection based on random effect structure (site within mosaic for tsf and mosaic for ff) and ranked using Akaike's Information Criteria (AICc). TSF categories: **tsf1** 0-6 months, **tsf2** 6 months – 2.5 years, **tsf3** 2.5 years to 10 years, **tsf4** 10 to 35 years, **tsf5** 35+ years, combined with last burn not recorded. Frequency categories: **ff1** unburnt or burnt over 34 years ago; **ff2** one burn within last 20 years, or, one fire within last 20 years plus one fire over 45 years ago; **ff3** 2-4 fires recorded since 1939 and not breaching tolerable fire interval (TFI) of 10 years post prescribed burn or 15 years post bushfire; **ff4** 0-2 fires in the last 19 years + 1-3 fires > 20 years ago with TFI breached >22 years ago **ff5** 2-3 fires within the last 19 years with TFI breached within last 19 years.

Damp ground foraging insectivores

There were two moderately common species reviewed in this foraging guild: the White-browed Scrubwren and the Eastern Yellow Robin.

The models for both species show an increase in regrowth vegetation (TSF2), after a reduced presence in the six months post-fire vegetation for the Eastern Yellow Robin and absence in this age class for the White-browed Scrubwren (Appendix 3A). Both species showed a preference for new growth vegetation (TSF3) above all age classes, this being significant for the Eastern Yellow Robin (Table 3.6). Neither species displayed a fire frequency response, although both frequency graphs do reveal the lowest occurrence for these species occurred on unburnt sites (Appendix 3A).

Graphing the frequency data for both species did not result in any significant outputs. However, both the Eastern Yellow Robin (Appendix A3) as well as the Damp ground guild (Appendix A2) showed them to be rare in sites unburnt or of only one burn, with an increase in occurrence across sites that both follow and breach the tolerable fire interval for Heathy Dry Forest vegetation. This result demonstrates that the tolerable fire interval measure for plants is not an efficient measure for damp ground foragers.

Insectivores that feed from open ground among trees or shrubs, the 'Open Tree' foraging guild

This group comprised four influential species: the Buff-rumped Thornbill, Scarlet Robin, Superb Fairy-wren and the White-winged Chough, as well as the moderately common species, the Pallid Cuckoo and Spotted Quail-thrush.

The Scarlet Robin results broadly reflect that of the group, with a significantly lower occurrence in newly burnt vegetation, rapidly increasing in abundance, six months post burn (Fig 3.13 *i,* Appendix A3). While the Superb Fairywren showed a preference for new growth vegetation (TSF3), (Appendix A3), the increase six months after fire was consistent across species, with no significant differences in the increased abundances and occurrences modelled. However, the White-winged Chough graph shows a significantly lower occurrence in new growth vegetation (Table 3.6, Appendix A3).

The Scarlet Robin model for frequency showed a significant increase on sites burnt frequently with the tolerable fire intervals breached (Fig 3.13*ii,* Table 3.7). This result is supported in this guild by the graphed output for the Buff-rumped Thornbill (Appendix A3), the species observed in greatest abundance (Table 3.2). The fire frequency binomial graph of the Buff-rumped Thornbill shows a confidence interval spanning 0-1 which reflects a presence on nearly every site in this fire frequency category (Appendix A3).

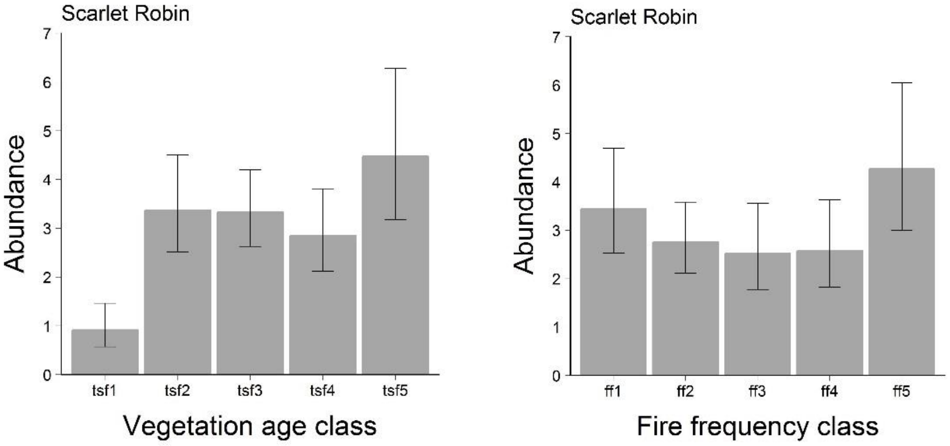


Figure 3.13 Scarlet Robin responses to TSF and fire frequency.

Generalized linear mixed models. Model selection based on random effect structure (site within mosaic) and ranked using Akaike's Information Criteria (AICc). TSF categories: **tsf1** 0-6 months, **tsf2** 6 months – 2.5 years, **tsf3** 2.5 years to 10 years, **tsf4** 10 to 35 years, **tsf5** 35+ years, combined with last burn not recorded. Frequency categories: **ff1** unburnt or burnt over 34 years ago; **ff2** one burn within last 20 years, or, one fire within last 20 years plus one fire over 45 years ago; **ff3** 2-4 fires recorded since 1939 and not breaching tolerable fire interval (TFI) of 10 years post prescribed burn or 15 years post bushfire; **ff4** 0-2 fires in the last 19 years + 1-3 fires > 20 years ago with TFI breached >22 years ago **ff5** 2-3 fires within the last 19 years with TFI breached within last 19 years.

Responses by species in this guild to time since fire, highlight the complexities associated with generalised assessments being made when reviewing guild responses. Five of the six species reviewed were most frequently observed in new growth vegetation. However, the White-winged Chough was observed less amongst new growth (Appendix A3).

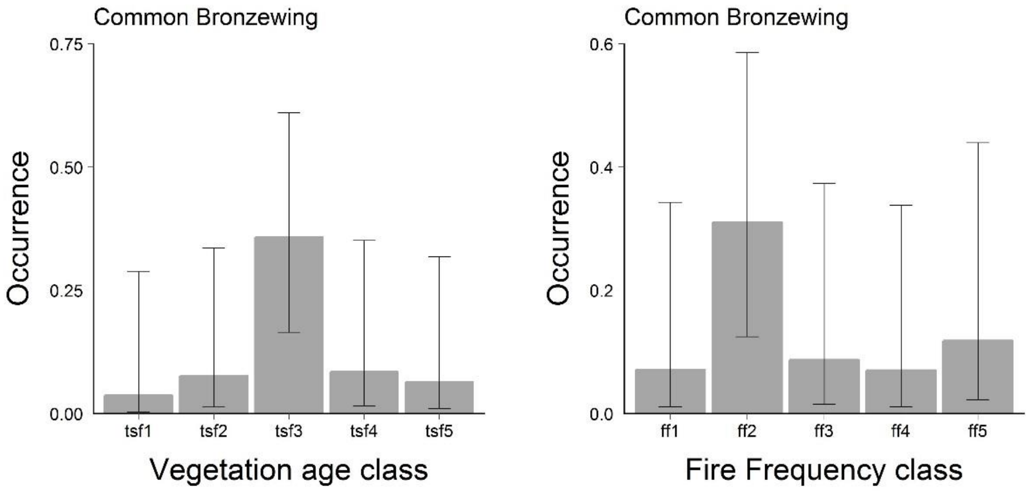
Open ground foragers

There was one species reviewed in this foraging guild: the Australian Magpie. Whilst significantly more abundant in regrowth vegetation (Table 3.6), it did not respond significantly to fire frequency.

Seed-eaters feeding from ground or low vegetation

There were two moderately common species in the guild that feed on seeds on the ground: Common Bronzewing and the Sulphur-crested Cockatoo.

The Common Bronzewing displayed a distinct TSF response with a significantly higher abundance in new growth vegetation compared to all other age classes (Fig 3.14*i*, Table 3.6). Its greater abundance on sites with only one burn may be an artefact of the project design as many of the burn sites were of recent burns (Fig 3.14*ii*).


Common Bronzewing

Common Bronzewing

Figure 3.14 Common Bronzewing responses to TSF and fire frequency.

Generalized linear mixed models. Model selection based on random effect structure (mosaic) and ranked using Akaike's Information Criteria (AICc). TSF categories: **tsf1** 0-6 months, **tsf2** 6 months – 2.5 years, **tsf3** 2.5 years to 10 years, **ff1** unburnt or burnt over 34 years ago; **ff2** one burn within last 20 years, or, one fire within last 20 years plus one fire over 45 years ago; **ff3** 2-4 fires recorded since 1939 and not breaching tolerable fire interval (TFI) of 10 years post prescribed burn or 15 years post bushfire; **ff4** 0-2 fires in the last 19 years + 1-3 fires > 20 years ago with TFI breached >22 years ago **ff5** 2-3 fires within the last 19 years with TFI breached within last 19 years.

The Sulphur-crested Cockatoo was absent from newly burnt sites and results reveal a significant preference for young vegetation (TSF2) compared with the older open stands of TSF4 (Table 3.6). However, it did not appear to respond to fire frequency.

Insectivores that forage from tall shrubs

Two common species, the Brown Thornbill and Fan-tailed Cuckoo, and one moderately common species, the Golden Whistler, were reviewed in the tall shrub foraging guild.

The Fan-tailed Cuckoo did not respond to either TSF or fire frequency. The Golden Whistler numbers were too few to model for TSF. However, due to a lack of presence on frequently burnt sites such that observations were restricted to four frequency categories, modelling for frequency was run for this species. However, results reveal little. The Brown Thornbill responded to TSF with a significantly lower abundance in the newly burnt sites, in keeping with the community response (Fig 3.15*i,* Table 3.6). Further, the Brown Thornbill responded to fire frequency with significantly higher numbers in sites with multiple burns (TFI's unbroken) as well as in sites subject to multiple burns (TFI's unbroken), compared to unburnt sites (Fig 3.15*ii,* Table 3.7). The Brown Thornbill appears to be responding to fire frequency but is not responding to a TFI measure.

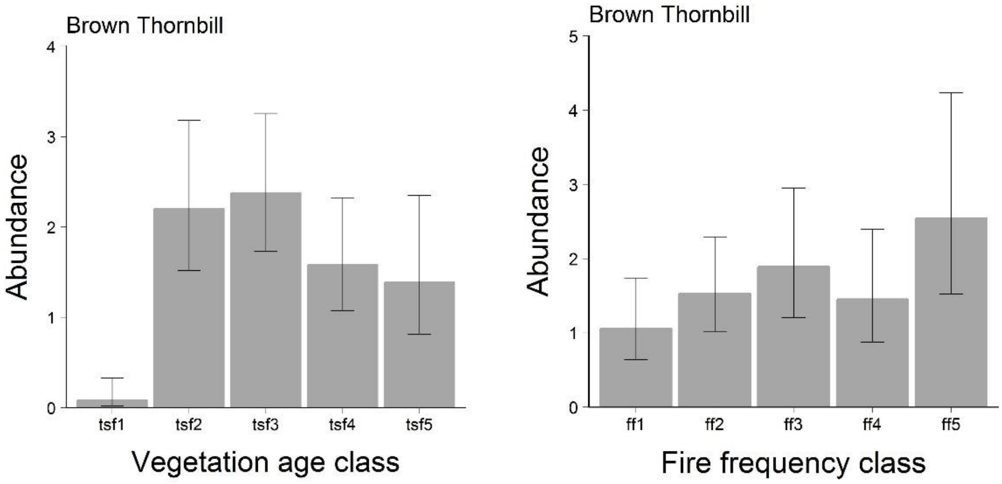


Figure 3.15 Brown Thornbill responses to TSF and fire frequency.

Generalized linear mixed models. Model selection based on random effect structure (mosaic for tsf and site within mosaic for ff) and ranked using Akaike's Information Criteria (AICc). TSF categories: **tsf1** 0-6 months, **tsf2** 6 months – 2.5 years, **tsf3** 2.5 years to 10 years, **tsf4** 10 to 35 years, **tsf5** 35+ years, combined with last burn not recorded. Frequency categories: **ff1** unburnt or burnt over 34 years ago; **ff2** one burn within last 20 years, or, one fire within last 20 years plus one fire over 45 years ago; **ff3** 2-4 fires recorded since 1939 and not breaching tolerable fire interval (TFI) of 10 years post prescribed burn or 15 years post bushfire; **ff4** 0-2 fires in the last 19 years + 1-3 fires > 20 years ago with TFI breached >22 years ago **ff5** 2-3 fires within the last 19 years with TFI breached within last 19 years.

The responses by birds foraging amongst tall shrubs are similar to responses by those foraging on damp ground, in that both guilds support the notion that the measure of tolerable fire interval is not useful for predicting fire frequency responses by birds.

Seeds-eaters that forage at all levels including trees

There was one species representing the foragers of seeds in trees, the Crimson Rosella. It did not exhibit a fire frequency response however it did reflect two distinct responses to TSF: a significantly higher abundance in both regrowth (TSF2) and long unburnt vegetation (TSF5) (Table 3.6, 3.7).

3.3 Discussion

Fire regimes in the forests of south-eastern Australia are changing relative to regimes in recent decades, not only with an alteration in fire interval patterns (Gill, 1975), a result of a changing climate (Cary et al., 2012), but also with the increase in the frequency of both bushfires and prescribed burns (Pitman et al., 2007, Fairman et al., 2016, Steffen et al, 2017). This may place some avifaunal communities in a vulnerable position; with fire regimes becoming more complex (Driscoll et al., 2010, Bradstock, 2012). In an era where Government protocol is to manage the landscape for asset protection by burning (Department of Sustainability and Environment, 2012), we need to be clear on what impacts, if any, this vegetation management process (combined with bushfire events) has on forest birds (Driscoll et al., 2010). Limitations exist on funding that often impact on the continuation of monitoring programs, and species' responses may lag fires, and so trends may not be fully recognized under short term research programs (Clarke, 2008). As such, determining responses with a space-for-time approach is useful in providing information on bird responses in successional post-fire vegetation stages.

3.3.1 What are the bird responses to the most recent fire?

This research used Cheal's (2010) measure of vegetation age classes (time since fire) as the basis for examining bird abundance trends in Heathy Dry Forests. As birds are responding to the vegetation structure (Loyn, 2012), the older classes of Cheal's classification were grouped into one, such that any sites investigated that were over 35 years in age were all grouped together. Differences in eucalypt Heathy Dry Forest older than 35 years are difficult to distinguish by all but a skilled botanist, and the structural components required by birds change little in areas older than 35 years. The time since fire categories used were logical in terms of observed vegetation changes in each class, reflecting differences in vegetation structure to which the birds would have been responding. Indeed, highlighted in the modelling comparison between the use of continuous age versus categorical age data, the time since fire categories revealed information that may have been masked in modelling with a continuous classification that did not pick up six-month increments, over 70 plus years.

The first prediction for this research was that early post-fire vegetation in Heathy Dry Forest is characterised by distinct structural elements in the vegetation and, as such, will support distinct bird assemblages. In Heathy Dry Forests there was limited evidence of this occurring. In the first six months, abundances of all species dropped, but no individual species or group was specific to this age class only. Over time, abundances increased, with the ordinations illustrating that changes over time were not different assemblages but rather, changes in species abundances of very similar assemblages.

There was one exception to the notion that species abundances increased rapidly post-fire, evident with nectarivores. They showed a distinct preference for the densest, new growth vegetation in the 2.5 to 10-year-old vegetation stands. These birds were near absent in all but new growth vegetation; increases in abundances largely attributable to the numbers of White-eared Honeyeaters. Whilst sites of the new growth age class were not clustered (sites with new growth vegetation were monitored across the project region), the project design did have more 2.5 to 10-year-old sites than other age classes.

On site, vegetation differences were evident between the two regrowth age classes (TSF2, TSF3), hence the decision was made to maintain the class distinction. The younger regrowth stands (6 months to 2.5 years) had eucalypts with thick and quite short epicormic growth covering entire blackened tree trunks. On these sites, ground cover was apparent but minimal. The older regrowth and new growth (2.5 to 10 years) had re-sprouting grasses and dense shrub-like vegetation covering the entire ground. The younger regrowth vegetation had an abundance of birds feeding within the epicormic growth, however the older regrowth and new growth had birds feeding within canopy and ground levels. The distinction between these two vegetation age classes proved important as the Laughing Kookaburra was abundant amongst regrowth vegetation, but absent from the denser new growth age class.

The increase in bird abundances in the regrowth vegetation leads to two points of discussion. Firstly, ‘mega-fires’ across Victoria are occurring more frequently. In recent times, major areas of the landscape have been burnt by bushfire twice inside 11 years (Fairman et al., 2016). Whilst research into three bushfires in Spain has already highlighted that not all vegetation species survive fire in all situations (Rodrigo et al., 2004), Fairman et al. (2016) argue that what is unknown and of concern is how the increase in bushfire frequency will impact the fire sensitive (obligate seeders) and the fire tolerant (re-sprouting) trees in the landscape. Rodrigo et al. (2004) highlighted that re-sprouters displayed the greatest resilience to frequent fire. However, they contrasted these results with predictions for plants with few seeds, potentially failing to regenerate, as fire frequency increases. Further research has highlighted that plants take several years to establish pre-fire seed stocks, therefore subsequent fires occurring in the juvenile stages of obligate-seeders can threaten species (Bowman et al., 2014). Just as Victoria's Heathy Dry Forest has a tolerable fire interval requirement of 15 years post bushfire, it was highlighted by Rodrigo et al., (2004) that, with species in their models, fire intervals of less than 10 to 15 years may lead to the disappearance or reduction of plant species. As the climate in south-east Australia is drying, further investigations into the tolerable limits of fire tolerant vegetation species are required. What is already known is that these fire tolerant species have temperature tolerances that will likely result in site-location shifts as temperatures increase and rainfall decreases (Mok et al., 2012, Enright et al., 2015). The drying landscape and the increase in the frequency of large scale fires, both potentially impacting the persistence of fire tolerant vegetation species, may have important consequences for the forest birds. In Heathy Dry Forest, the abundance of birds was highest in re-sprouting trees. The birds observed may have been responding to the re-sprouting epicormic growth of the eucalypts for both insects as their food resource and predator protection. A threat to this vegetation is a direct threat to bird biodiversity in the region.

A second important consideration is that of the timing of prescribed burns. It has been determined that the most critical period for development of young birds is whilst they are being fed in the nest (Lack, 1950). Furthermore, research describes that temperate region species breed in spring (James and Shugart, 1974), as this is the season that yields the greatest volume of food (Lack, 1950). Until recent times, prescribed burns were carried out across Victorian forests in autumn and through winter months. In fact, this research used sites burnt in the prescribed burn process in autumn months only. Now however, prescribed burning is practised all year. Dangerous fire weather can occur in any month today, and so, in attempts to meet prescribed burning targets, burns occur when opportunity arises, irrespective of season. A further point is that autumn prescribed burns are more intense and severe than spring burns, due to vegetation having dried extensively in the summer months (Tolhurst and Cheney, 1999). However, while spring prescribed burns, occurring after winter rains, might be considered safer to manage, they are also considered more unpredictable (Tolhurst and Cheney, 1999). While fuel moisture may be high, changes to weather patterns in Victoria in the spring months can result in rain events ceasing and fires burning for some time (Tolhurst and Cheney, 1999). More recently, down the east coast of Australia, in the south-west Australian regions and even in Queensland, bushfires are occurring in spring months. This change in the timing of prescribed burns and bushfires may have major implications for avifauna by reducing food availability and killing nestlings during the breeding season.

If the declining trend in avian biodiversity in this region is to be arrested, research needs to continue into the identification of species' responses to a changing fire regime. An adaptive management approach is required, to continually update bird abundance trends, if management decisions are to be in the best interest of avian biodiversity. Penman et al. (2011) highlighted the need for including more extensive ecological values into prescribed burning programmes. Further to this, Kelly et al. (2017), by combining different factors of the fire regime and creating species distribution models, highlight that many studies review individual fire events, but that it is now imperative that there is a shift to studies that incorporate data across multiple fires into modelling. In Victoria, the State Government operates on a premise of burning for biodiversity (Department of Sustainability and Environment, 2012). However, what is clear in this research is that, whilst many species increased in abundance in the post-fire regrowth and new growth vegetation, the Laughing Kookaburra was found to decline in abundance and occurrence due to this denser vegetation. A ‘pyrodiversity begets biodiversity’ premise (Taylor et al., 2012) will not benefit the Laughing Kookaburra if the ground cover is dense with new growth and it cannot detect prey. Understanding and managing for all species’ requirements is an imperative, if the declining trend in avian biodiversity is to be reversed. Recognising that broad community or guild responses may not highlight the responses of all individual species is fundamental to ensuring, and managing for, the survival of all bird species.

3.3.2 Can we describe foraging guild responses to fire, in broad terms, for each guild?

Even though species may exploit the same environmental resources in a similar way to others within guilds (Root, 1967), as this study and others have demonstrated (Thiollay, 1992, Lindenmayer et al., 2000), not all members of a guild may respond similarly to disturbance. It may not be possible to predict responses between closely related species (Lindenmayer et al., 1999, Lindenmayer et al., 2000) and factors that influence population density may vary between closely related species. Koch et al. (2011) suggested that foraging guild responses are better predictors of environmental change than species’ commonness, therefore this research combined responses by guilds, with responses by individual common species.

Some foraging guilds can be described in broad response terms for all the species examined within that guild. For example, the nectarivores and the canopy foragers were all most abundant in the denser regrowth age classes. However, general descriptors to describe a foraging guild are not always possible and complexities within guilds were highlighted.

Responses by species within the carnivore foraging guild highlight the difference in responses by species within guilds. The Laughing Kookaburra was greatly reduced in abundance in dense new growth vegetation, whereas the Grey Currawong showed a preference for this vegetation type. While the notion that ‘competitive exclusion’ may be a factor impacting Laughing Kookaburra abundance, these two species have been found to coexist (Recher and Davis, 1998). Indeed, results in this research highlight that not only do these species co-exist in Heathy Dry Forests, but they drop in occurrence together, when the vegetation is at its densest, during the 2.5 – 10 year age class. The Laughing Kookaburra and Grey Currawong forage for prey from different levels in the vegetation. The Laughing Kookaburra is a pounce predator and it is possible that the ground to mid-level regrowth may have been impacting prey visibility, whereas the Grey Currawong was mostly seen feeding while walking across damp ground. However, the Grey Currawong has also been found to forage at height (Recher, 2016), therefore this versatility in sourcing prey from different levels in the vegetation (i.e. more generalist) may mean that the Grey Currawong may have a competitive advantage. An added factor impacting kookaburra abundance in dense vegetation is that new growth ground cover may not be suitable for a broad range of reptiles; open canopy environments being essential for ectothermic animals for basking (Michael et al., 2011). As lizards are a major component of a Laughing Kookaburra's diet, not only does the denser ground cover potentially impede vision of prey, but a component of the kookaburra's diet may be entirely absent from the area, and what is left must be competed for with the Grey Currawongs.

Another foraging guild displaying a complex set of responses was the insectivores that feed on the open ground. The Buff-rumped Thornbill, Scarlet Robin and Superb Fairy-wren all displayed an aversion to newly burnt vegetation with no clear response differences across other vegetation age classes. The White-winged Chough on the other hand, avoided dense new growth. As an open-ground forager the White-winged Chough would have difficulty feeding in closed, low vegetation and preferred a more open landscape. However, in the same guild the Spotted Quail-thrush and the Pallid Cuckoo preferred denser new growth. This is an important distinction, as the Spotted Quail-thrush is noted as being in decline (Barrett et al., 2003), and listed as endangered in some regions. To maintain species biodiversity, the responses of those species noted as being in decline need to be clearly understood and considered in concert with common species responses.

3.3.3 Do we see distinctly different bird assemblages in the vegetation of different post-fire age classes?

In this research, there was evidence confirming the second prediction, that there would be distinct individual species' responses to time since fire. Vegetation affords birds with nesting sites, protection and food. Therefore, each successional stage in post-burn vegetation will vary in which of these crucial components is available. If we consider a hot fire within a Heathy Dry Forest (one that was not patchy and removed the entire ground cover, tree bark and canopy levels) then the successional stages post-fire could be expected to have distinct differences, as seen by the birds.

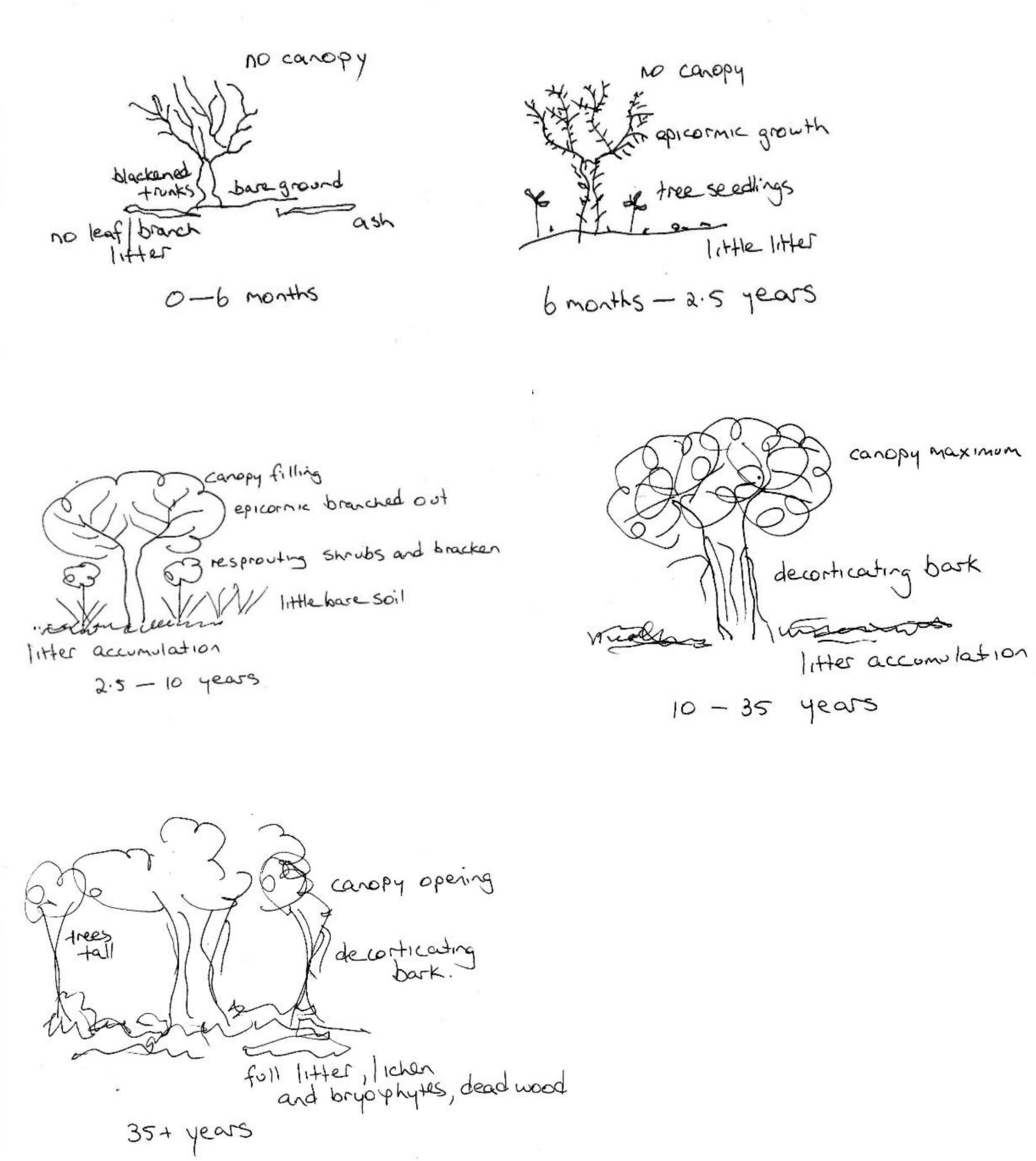


Figure 3.16 Diagrammatic Heathy Dry Forest vegetation successional stages post fire.

Fire will trigger immediate assemblage variations as birds respond to the changes in resource availability (Reis et al., 2016). From the diagrammatic representation of the post-fire successional stages in Heathy Dry Forest (Fig 3.17), and the bird abundances recorded within each of these age classes, it is possible birds were responding to changes in the canopy density, presence (or absence) of decorticating bark, leaf and dead wood litter, and the ‘openness’ of the vegetation.

In Heathy Dry Forest the newly burnt vegetation, with canopy, bark and ground cover all removed, was missing the open-tree feeders such as the Buff-rumped Thornbills, Scarlet Robins and Superb Fairy-wrens. These sites did not have any nectarivores. The White-throated Treecreeper, as a bark forager, was low in abundance and the Brown Thornbill, as a tall shrub forager, was also absent. Six months later, and the epicormic growth was extensive. The Crimson Rosella was abundant, as it fed on seeds in the trees, as were the canopy feeders such as the Grey Fantail and Spotted Pardalote. Along with them, the Sulphur-crested Cockatoo fed on the ample seeds on the ground. Three to ten years post fire and regrowth had moved through to its maximum phase. Branches from the short epicormic growth, shrubs and bracken were all dense. There was little to no bare ground. The nectarivores, such as the White-eared Honeyeater and the Eastern Spinebill, were at their greatest abundances, the Common Bronzewing fed on seeds on the ground, and the Eastern Yellow Robin numbers were high as they fed on damp ground. The White-winged Chough and the Laughing Kookaburra were scarce, these species preferred an open feeding ground. By the time this vegetation was 35 years old the trees had all grown to their tallest, the decorticating bark was extensive and the vegetation was open, with limited shrubs. The Crimson Rosella dominated bird numbers, feeding on the seeds in the trees and carnivores, such as the Laughing Kookaburra and Grey Currawong, were once again obvious in the landscape.

As a community, there were distinct differences with assemblage patterns in each vegetation age class. The most striking were the community patterns immediately post-fire and again with the dense regrowth vegetation. Immediate post-fire vegetation supported an assemblage with lower abundances across all foraging guilds. On sites that had been extensively burnt (100% severity), it appeared that maybe birds had little protection, limited nesting sites and a paucity of food resources when bark, canopy and ground cover were all removed. Fire disturbance has been shown to affect the short-term response of insects (Elia et al., 2011). In the regrowth vegetation on the other hand, the abundances increased for all those species, such as the nectarivores, that thrived on and within dense flowering vegetation. The age class with the densest post-fire vegetation supported nearly all the nectarivores recorded. The greatest similarities between assemblages were observed in the older vegetation classes; which supported all those foraging guilds preferring open vegetation stands, fuller canopies and decorticating bark, along with complete ground cover of rotting logs, bracken and leaf litter.

Other studies, for example those in mixed-conifer forests, indicate that vegetation stands post fire are suitable for specific foraging guilds and fire specialist species (Hutto, 1995, Nappi and Drapeau, 2009) and will therefore result in a temporary increase in abundance of early stage species, along with those species that appear on burn sites only (Smucker et al., 2005). The bird communities in these conifer forests, immediately post fire, are recognised to differ greatly in composition to later vegetation successional stage bird communities. The Heathy Dry Forest results are not this distinct. Whilst displaying a unique assemblage immediately post fire, there were no species or foraging guild abundance increases in newly burnt vegetation and there were no species unique to a burnt environment. The abundances of all species decreased in recently burned vegetation.

Unique assemblage patterns have been shown to exist in mixed-species eucalypt forests in Victoria, in age classes other than newly burnt. Other studies have shown that dense regrowth, approximately 3 to 10 years after logging or fire, will support high densities of birds that favour shrub layers, including a suite of nectarivores dominated by White-eared Honeyeaters, whereas mature stands support a different suite of nectarivores (Loyn, 1980, Loyn, 1985, Muir et al., 2015, Leonard et al., 2016, Kelly et al., 2017).

3.3.4 How quickly are species abundances bouncing back post fire?

Community and individual species' responses, immediately post fire, were clear; abundances dropped within the first six months post burn. However, low abundances postburn were usually followed with high abundances in regrowth and then new growth vegetation.

In Heathy Dry Forest, vegetation recovery is swift. After six months, the epicormic regrowth on trunks was dense, serotinous seedlings had sprouted, and grasses and brackens were regenerating. For the ten years following fire, in

Heathy Dry Forest, the regrowth and subsequent new growth is at its densest. Of those species that had low abundances in the newly burnt vegetation, many had corresponding high abundances in the regrowth and new growth age classes. This is illustrated in the species from different foraging guilds feeding at different levels within the vegetation, for example, the Spotted Pardalote as a canopy feeder, the Scarlet Robin and Superb Fairy-wren as those feeding on open ground among trees and shrubs, and the White-throated Treecreeper as a bark forager. The Spotted Pardalote was likely responding to an increase in the abundance of psyllid insects within developing epicormic growth (Loyn and McNabb, 2015). With a home range of approximately ten hectares (Debus, 2006), Scarlet Robins are sedentary and select nest sites in specific *Eucalyptus* species (Robinson, 1990). As such, it could be expected that they are vulnerable to fire, as fire would render trees unsuitable for nesting. What is possible in these results is that they may have been responding to resource availability in the young regrowth vegetation. The Superb Fairy-wren prefers low shrubs and tall herbs interspersed with bare ground (Emison et al., 1987), both features of regrowth and new growth vegetation. While these results showed significant abundances in dense new growth, aged 2.5 to 10 years, their abundance in the younger regrowth of 6 months to 2.5 years was not significant. This was partly in keeping with other studies that showed they avoid dense regrowth after bushfire (Loyn, 1997). Superb Fairy-wrens are known to tolerate urban settings (Harrisson et al., 2013) and their persistence throughout landscapes altered by humans indicates a resilience to landscape changes and an inherent ability to exploit changing conditions (Trollope et al., 2009). However, as they are territorial, sedentary and weak fliers, Superb Fairy-wrens may be reluctant to move to nearby locations (Harrisson et al., 2013) and so are likely vulnerable to fire. As Superb Fairy-wrens are currently in decline across north-central Victoria, despite their resilience across landscape types (Harrisson et al., 2013), further monitoring of their population trends is necessary to ensure population declines are detected and managed. The White-throated Treecreeper's response broadly aligns with results stating that they feed mainly from rough-barked eucalypts and that they have an aversion for smooth bark trees (Loyn et al., 2007), therefore they might be expected to avoid bark-scorched vegetation. Following this, for their abundance to be greater in young regrowth vegetation suggests they were responding to epicormic regrowth providing an abundant supply of invertebrate food (Loyn et al., 2007). The response by the White-throated Treecreeper is consistent with previous results where they were found to prefer rough-barked trees from which they gleaned insects (Noske, 1985).

As birds have been shown to display strong site fidelity, a rapid regrowth response by forest vegetation is a crucial factor for bird abundances to recover rapidly (Lindenmayer et al., 2014).

3.3.5 Are birds responding to fire frequency?

The third prediction in this research was that there will be little evidence of changes to assemblages based on fire frequency. This study confirmed this prediction, with little evidence for a community fire frequency response.

There were a limited number of individual species' responses to fire frequency. Two species that showed a preference for unburnt sites were the Laughing Kookaburra and the Australian Raven. The canopy feeding Grey Fantail showed a preference for sites burnt multiple times, corresponding with their high abundances on regrowth vegetation. However, the abundances of most species in unburnt sites were similar to those in sites burnt multiple times.

The response of one species, the tall-shrub foraging Brown Thornbill, challenged the utility of the tolerable fire interval measure, (as a vegetation measure), as a predictor of bird abundances. This species was abundant on sites burnt multiple times – with both tolerable fire intervals breached, as well as in those sites where the interval was not breached. Other studies have shown that the abundance of Brown Thornbills dropped in recently burnt sites (Loyn and McNabb, 2015), suggesting that a negative relationship with fire frequency may be likely. However, this result highlights that there is a distinction between the age of vegetation, as per a recent burn, and a location being burnt multiple times. Whilst Brown Thornbills may not be abundant on newly burnt sites, if resources are available, they will persist in a location that has been burnt a number of times in recent decades.

While studies have determined that an increase in fire frequency in some vegetation types may result in a simplification of the vegetation structure with a reduction in the understorey (Albanesi et al., 2012), the structure of the Heathy Dry Forest tends to become simplified through time, naturally. As the vegetation matures, the structure of the understorey greatly reduces. By 35 years, the Heathy Dry Forest has a vertical profile that has simplified to one consisting largely of ground cover, bark and a canopy. The very nature of this vegetation type may therefore be cause for concern. Research in woodlands has found that habitats of low complexity supported a reduced abundance in species that are common in the Heathy Dry Forests: the Buff-rumped Thornbill, Spotted Pardalote, Grey Shrike-thrush, Scarlet Robin and White-winged Chough (Watson et al., 2003).

Other research into fire frequency within different vegetation types has also yielded a similar result, of little response by birds. Repeated burning in tropical forests of Kalimantan did not lead to an impoverishment of avifauna; species richness stayed the same, although abundances dropped (Slik and Balen, 2005). Similarly, repeated fire in Oregon, USA, did not reduce bird species richness in the mixed evergreen forests (Fontaine et al., 2009).

3.3.6 Fire frequency as a categorical classification – was this method a useful measure?

Fire frequency was measured, in this project, in terms of the number of times the site had been burnt over time. This was further segregated based on Cheal's (2010) 'tolerable fire intervals' which stipulated that 15 years is required for Heathy Dry Forest to persist post bushfire and 10 years post prescribed burn. The intention in this study was to not only assess avian responses to burn frequency, but to also determine the appropriateness of the tolerable fire interval measure, being used as a guide for birds. Species were not seen to be responding to the interval in that there were no discernible differences in abundances on sites with TFI broken versus TFI unbroken. This was evident with the guilds of damp ground and tall shrub foragers both having significant increases in abundance on sites burnt – regardless of fire interval.

Responses by birds to the fire frequency measure used in this project were minimal. There was one exception, with the Laughing Kookaburra showing a preference for unburnt sites. However, as this project design meant that time since fire and fire frequency could not be modelled together due to the configuration of sites – it is not clear on modelling alone whether the Laughing Kookaburra's reduced absence on frequently burnt sites and reduced absence on sites with dense regrowth vegetation was a response to the most recent fire, or the frequency of burning, or both. A biologically-based judgement suggests that, at the very least, the response is to time since the last fire, in that dense regrowth obscures prey, or prey is greatly reduced. Further, a fire event may render a nest hollow unusable. Therefore, fire events may shape a kookaburra's territory.

Models for the Laughing Kookaburra were initially run on abundance data, highlighting significant results for both time since fire and fire frequency. The outcome from the more conservative presence-absence data, with the impact of one site with higher abundance minimized, was the same. Therefore, even though Kelly et al. (2017) highlight that bird responses can sometimes vary (importantly) with the use of abundance data versus presence-absence data, the Laughing Kookaburra responses are clear. This species shows are clear aversion to regrowth vegetation as well as burnt sites.

The lack of Laughing Kookaburras on newly burnt sites was unexpected. As a pounce predator, it could be expected that they would be present on the open sites for easy access to prey. However, as many of the recently burnt sites were comprehensively burnt, with no ground cover, bark or canopy remaining, most prey may have perished.

This fire frequency research was designed in such a way as to answer specific questions. As such, the outcomes need to be considered in line with the fire frequency classification used – an actual number of burns per site and a tolerable fire interval, creating classification divisions. Research by others has shown that the changes to habitat structure from the most recent fire strongly influence faunal responses (Haslem et al., 2011). Furthermore, that repeated fires, especially severe fires, may in fact be altering the vegetation species composition and structure (Fairman et al., 2016, Kelly et al., 2017). A more informative method of analysing bird abundance data may be the use of inter-fire intervals (the mean of years between fires), modelled with vegetation data, to make judgements on changes to vegetation that may be impacting birds *per se* (Kelly et al., 2017).

Kelly et al., (2017) found the species they examined, in Victorian foothills, to show resilience to the effects of fire regimes. However, attention also needs to focus on the potential for changes to vegetation (Fairman et al., 2016) that may occur as the frequency of bushfires and prescribed burns increase. Part of the judgement on bird impacts from fire lies not only with changes to fire frequency, but also with the severity of each fire event, as those fires most severe may substantially alter the structure of the vegetation.

3.3.7 How effective are the common species from each foraging guild, in illustrating possible fire responses for the entire foraging guild they represent?

The benefits gained from monitoring common species are broad-reaching, including explanations as simple as: answering questions related to the most apparent species will substantially explain the world around us (Gaston, 2008); to those more complex, such as that common species can be used to reflect community changes (Koch et al., 2011). Further, common species may be more responsive to anthropogenic activities and, therefore, drive community assemblage patterns (La Sorte and Boecklen, 2005). Or conversely, common birds may be less responsive and more resistant as they can shift resource base. All these points add weight to the benefits of placing emphasis on researching their trends.

The abundance responses of common bird species to fire in Heathy Dry Forest have been shown to largely represent that of their respective foraging guilds, based on time since fire and the frequency of fires. Even those foraging guilds with varied individual species' responses (such as that shown by the carnivores, where pounce feeders preferred open vegetation, and ground feeders increased in abundance in regrowth vegetation), had common birds whose responses largely reflected that of the guild. The ongoing monitoring of common species' abundances will, therefore, give an overview reflecting many of the species in a Heathy Dry Forest.

It is recognised however, that a focus on common species does not necessarily capture the status of those species endangered, or those under direct threat of extinction. In fact, uncommon species were not considered in this study. However, as monitoring continues, and the database for all taxa grows, trends for rarer species become clearer and management can be applied and modified accordingly. While attention does need to be given to managing rare or threatened species when considering the processes behind a changing biodiversity (Atauri and de Lucio, 2001), much can be gained from a focus on the common species (Gaston, 2008) as they provide insights into ecosystem processes.

3.3.8 Common species trends as a database – an adaptive management approach

In Australia, there is a need for management processes to incorporate, not just vegetation ecology, but to include targeted research incorporating faunal ecology (Haslem et al., 2011). Further to this, there is an emerging emphasis on adaptive management (MacHunter et al., 2009, Cheal, 2010); an integrated approach combining ecology with fire management practises. The basic tenet of an adaptive management process for prescribed burning is iterative; to monitor and then modify management accordingly.

Common birds, with their ease of detection, are already used as indicators of environmental change (Drever et al., 2008, O'Connell et al., 2000, Järvinen and Väisänen, 1979). Therefore, determining the population dynamics of common species may offer substantial information on their relationships with other birds and their environment (Gaston, 2008, Gregory et al., 2005). It is necessary to be clear in focus and direction; that different species have different sensitivities, and hence value, in environmental assessment. It is possible therefore, to take this current research further and direct some focus onto common forest birds that may be key indicator species with specific fire responses (Lindenmayer et al., 2000). In monitoring the trends of these species, an attempt is made to capture responses to environmental changes that might not be detected through ecological monitoring of other less abundant fauna or vegetation species. Investigation into the behavioural responses and resource needs of each of the common forest species needs to be undertaken to determine their position and value as indicators in a forest fire management protocol. Nonetheless, an adaptive management approach of continued monitoring to determine responses of the common birds will give valuable insight into ecosystem scale changes occurring around them. This is important, as it represents a readily accessible database that can be analysed, while collecting data on the less abundant, specialist or endangered species.

Research discussion has noted the paucity of long-term data (Ford et al., 2001, Bennett and Watson, 2011). However, in recent decades *The Atlas of Australian Birds* (Blakers et al., 1984) was created, which was subsequently followed with an update (Barrett et al., 2003). Even though a lack of more long-term data is highlighted in literature (e.g. Bennett and Watson, 2011), there is scope for combining trends for common species with information that already exists, and grouping the common birds to landscape types, similar to that in the Pan-European Monitoring Bird Scheme (PEMBS) in Europe. The PEMBS (created by the European Bird Census Council) has taken bird data analysis a step further (than anything in Australia) by utilizing common bird abundances (http://www.ebcc.info/pecbm.html). The PEMBS members monitor the abundance data of common birds, in groups of forest and farmland birds, and measure trends across Europe. Key to this process are two points: the first, that birds are good indicators for biodiversity across all taxa, and second, that farmland species abundance trends can be directly aligned with changes in agricultural landscapes. Based on these fundamental points, the PEMBS has created the European common bird index and identified indicator species.

This research highlighted that 17 species are the main contributors to assemblage patterns in Heathy Dry Forest. A similar system to the PEMBS, making online data available, could be implemented in Australia. This research can be duplicated to determine the common bird species across all landscapes in Australia. Taking this even one step further, citizen science campaigns could be created to continually add to the data bank of common bird abundances.

Recent research by Simmonds et al. (2017, personal communication) at The University of Queensland has taken a similar approach in that their research has a focus on capturing broad-scale bird trends across Australia. In their current investigations, continental-scale spatial vegetation datasets have been created, and they have linked a database of Australian bird habitat associations to these spatial data. They have therefore, been able to produce extirpation maps for 463 terrestrial bird species. From these maps Simmonds et al. have designed metrics of lost habitat for land birds, potentially an important tool for investigating species decline.

So, while there is a constant effort to increase our understanding of our bird communities (Russell et al., 2009), in order to effectively manage them, a further tool in effective adaptive management is to predict the trends of our common forest birds, as they may act as surrogates for entire bird communities. As common species may be resilient to disturbances such as fire (Koch et al., 2011), implementing an adaptive management approach is useful in that this process incorporates continued monitoring and resultant modifications to management decisions, with ongoing monitoring gradually building an understanding of the rarer species.

3.3.9 The Laughing Kookaburra as a fire indicator species at risk

The Laughing Kookaburra is a sedentary species that maintains complex nuclear social groups of a monogamous breeding pair with up to six helpers (Legge and Cockburn, 2000). The flock is territorial and the size of that territory depends on the flock size and their ability to defend space (Parry, 1973). As the home range of territorial species is confined, this suggests that they may be vulnerable to the potential impacts from repeated burning of most or all of their territory. While it has been shown that we may expect the impact of prescribed burns undertaken every three to ten years to have minimal impact on common birds (Loyn and McNabb, 2015), the effects on the Laughing Kookaburra may, in fact, be substantial.

In this study, the Laughing Kookaburra was the only species that showed significant responses to both time since fire and fire frequency. It was greatly reduced in abundance from newly burnt sites; simply explained by suggesting that the extreme fires had burnt any available prey. The Laughing Kookaburra's presence in the new growth and regrowth vegetation however, was a little more complex. It was present in the younger 6 month – 2.5 year regrowth, but reduced in abundance in the denser 2.5 – 10-year-old regrowth and new growth. This suggested that there was a threshold in vegetation density for the Laughing Kookaburra; in the denser new growth vegetation it could not detect prey. Further, the Laughing Kookaburra showed a significant increase in abundance in unburnt vegetation, as opposed to sites burnt at all, no matter what the frequency. These responses have implications for prescribed burn management for if forests frequently burn, resulting in a persistent new growth vegetation state, the Laughing Kookaburra may remain absent.

In 2004 the Laughing Kookaburra was reported as being widespread (Legge, 2004) and yet, in 2015, this species was noted to be currently in decline along the entire east coast of Australia (Birdlife Australia, 2015). This observation, when seen in combination with these data and the present burning policy, raises concern for the fate of this iconic, common species. The extent to which this observation applies widely needs to be substantiated while there remains an opportunity to intervene, before the population becomes critically low. While the results in this thesis may be finding a risk to the Laughing Kookaburra populations in forest reserves, its mainstay may be rural and agricultural settings with vegetation open enough for basking prey and pouncing; but old enough to have ample tree hollows.

3.3.10 Validating the Cheal (2010) classification system for predicting bird responses to fire

Creating models using either continuous age data and then, modelling with Cheal classifications, highlighted the importance of considering the timescale being reviewed and the context of the enquiry. While both methods result in graphs with similar response curves (Watson, et al., 2012), the categorical graphs show more detailed information in the first ten years post-fire. This may be important when determining the timing of prescribed burns.

Plate 18 The White-throated Treecreeper*, Corombates leucophaea*, is impacted by high severity prescribed burns.

Source: Dean Ingwersen, Birdlife Australia

4 Forest bird responses to prescribed fire and varying prescribed burn severities

4.1 Overview

In recent decades ‘mega-fires’ in the Victorian landscape have increased in frequency (Fairman et al., 2016) and this trend is expected to continue (Pitman et al., 2007). This has resulted in an increase in efforts to ameliorate bushfire impacts on both human life and assets (Teague, 2010). While prescribed burning is used as a means of managing fire fuel loads across different landscapes (Brockett et al., 2001, Artman et al., 2005, Brudvig et al., 2007, Fernandes et al., 2013), in Victoria, prescribed burns are also applied to the landscape with the objectives of maintaining and improving the resilience of natural ecosystems (Department of Sustainability and Environment, 2012). So, as efforts extend to minimise bushfire impacts as well as to promote ecological sustainability, so there will be an increase in the extent of the landscape burnt (Teague, 2010).

For fire to be used effectively for conservation we need to be able to predict species responses. But this is challenging, as species have spatially and temporally varying responses to fire (Bilney et al., 2011, Lindenmayer et al., 2014, Nimmo et al., 2014, Kelly et al., 2017). The challenge is that species respond to vegetation change, and the way fire affects vegetation depends on numerous factors (Monamy and Fox, 2000, Driscoll et al., 2012).

Fire severity is an aspect of the fire regime that has a major influence on post-fire vegetation dynamics, and hence species’ responses (Smucker et al., 2005, Kotliar et al., 2007, Fontaine and Kennedy, 2012, Lindenmayer et al., 2014). While the response of fauna to fire severity gradients has been studied in some Australian regions, many of the Victorian studies to date have been conducted on taller/wetter/denser forests (Lindenmayer et al., 2014, Robinson et al., 2014, Loyn and McNabb, 2015, Haslem et al., 2016). There is a paucity of research on fire severity impacts on fauna in Heathy Dry Forests.

Much of the surveying and subsequent modelling of bird responses to the impacts of fire – either bushfire or prescribed burns – occurs without prior knowledge of an impending fire. Necessarily then, such research is usually only based on post-fire bird data. The opportunity presented itself for a sub-project to be developed that incorporated BACI design principles to investigate in detail bird responses to specific prescribed burns. A BACI design is beneficial as it enables an assessment of the possible impacts of the prescribed burns, independent of (i.e. while controlling for) other natural phenomena.

A group of sites were burnt as part of a prescribed burn process, all within a fortnight (March, April 2012). Even though field work for this research did not begin until June 2012, bird data had been collected on these sites in the spring/summer season of 2010 (refer Preface). Furthermore, the severity of the burns ranged from moderate burns where much vegetation remained unburnt, to intense fires, where close to 100% of the vegetation was burnt.

To investigate the impacts of prescribed burn severity on birds, alpha and beta diversity responses were analysed, as well as the occurrence of individual species. The sites were all in Heathy Dry Forest EVC's across three locations of Central Victoria, Australia.

Two general predictions were made regarding the responses of birds to prescribed burn severity. First, any high severity prescribed burn could be expected to alter the structure of the vegetation, such as the ground cover and bark layers, as well as the canopy, reducing its density. However, studies have shown that, due to the speed of recovery of vegetation post-fire in eucalypt forests and woodlands, few early successional avian species are detected (Lindenmayer et al., 2008, Loyn and McNabb, 2015, Sitters et al., 2014a, Sitters et al., 2015). Therefore, it could be expected that after severe burns there would be a decrease in the species richness of birds, along with low individual species' occurrence, followed by a rapid recovery. This rapid vegetation response may influence the bird assemblages, so a second prediction was formulated, that there may be an increase in species turnover in the first year post-fire, representing a unique assemblage of birds responding to a burnt environment.

4.2 Site Locations

Thirty sites were selected as part of the study of impacts of prescribed burn severity on birds. There were 14 burn sites and initially 16 control sites (refer Methods 3.1.4). Control sites, unburnt sites near to the sites affected by a prescribed burn, were reduced to a total of 13 (refer Methods 2.1.4). The position of the sites relative to the area of each of the 2012 prescribed burns near them are provided on maps (Fig 4.1).

In most cases the burn edges were at roads (Figure 4.1). As a number of the project sites were selected to be a mimimum of 50 metres from road edges, this then placed some project sites within 100 metres of prescribed burn edges.

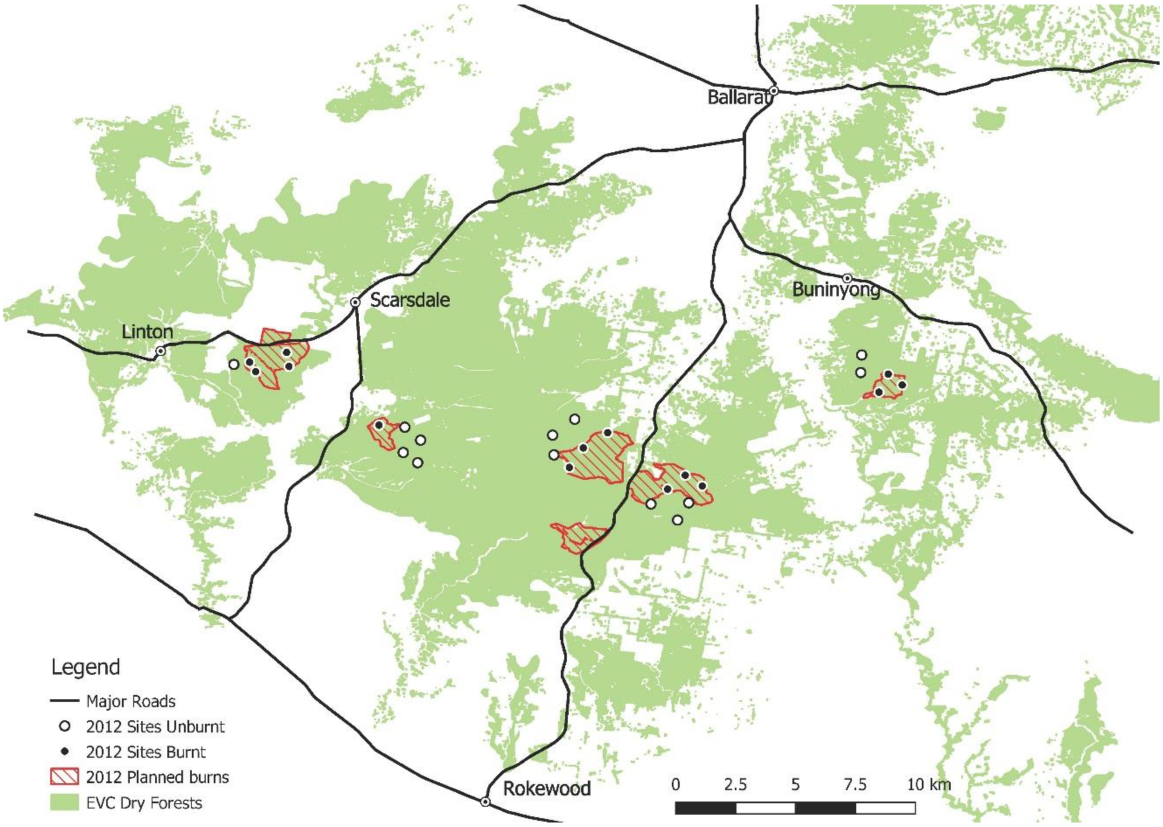


Figure 4.1 Site locations in Victoria, Australia relative to prescribed burns of 2012.

2012 prescribed burn areas illustrated with cross-hatch markings.

4.3 Results

Fifty-one species were detected in the surveys conducted during three spring/summer breeding seasons (2010, 2012-13, 2013-14), across 27 sites - a total of 2825 birds counted.

There were no distinct differences in either alpha or beta diversity between sites and further, there was little response to severity in any of the species modelled. The one exception was the response of the White-throated Treecreeper, which was less abundant in the second year after severe fires.

4.3.1 Alpha and beta diversity responses to fire and the severity of prescribed burns

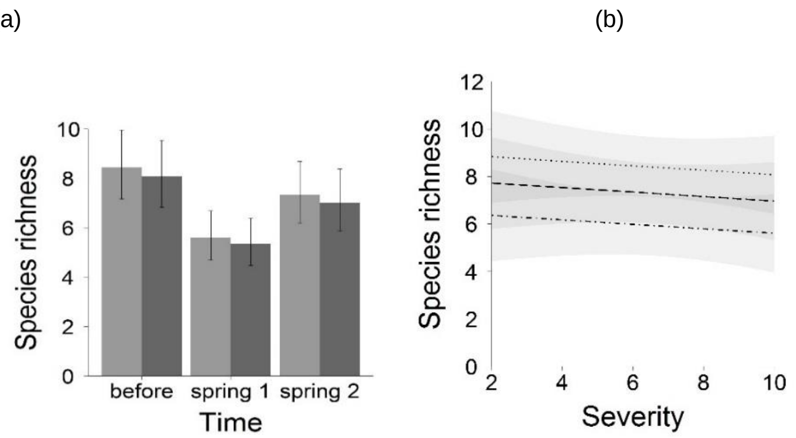
Bird species richness and species turnover were not influenced by prescribed fire, or fire severity, at burnt sites (Fig 4.2(a) – (d), Tables 4.1, 4.2). Whilst the data showed no response by species richness to fire severity, but a change in abundance over time, the species richness graph for fire illustrated how impacts in the first and second year post fire were on both control and burn sites. Therefore, it shows that the responses were not fire related (Fig 4.2 (a)).

There was no evidence of a species turnover response to fire severity. Birds showed a limited response to community assemblage patterns post fire and further, there were limited changes to their community assemblage with an increase in prescribed burn severity (Fig 4.2 (c-d)). The large difference between marginal and conditional R^2 for both fire and severity indicated that any variations in turnover may be a result of small spatial scale geographical variations (Tables 4.1, 4.2).

Species richness

(a) Fire response: control/burnt

(b) Severity response



Species turnover

(c) Fire response: control/burnt

(d) Severity response

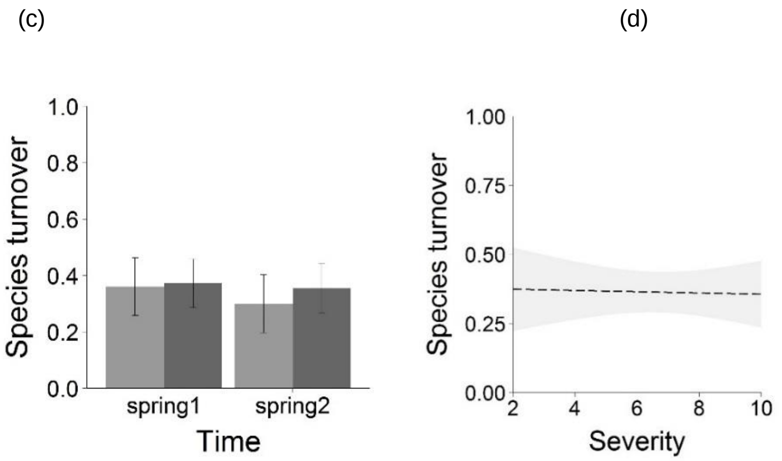


Figure 4.2 Avian responses to fire and the severity of prescribed burns in terms of biodiversity measures of species richness and turnover.

The community response of species richness (a,b) and species turnover (c,d) to prescribed fire and the fire severity at burnt sites. All fire graphs have pairs of control sites (light grey) with burn sites (dark grey) for time periods before the autumn prescibed burn (before) followed by the first observation year (spring 1) and the second observation year (spring 2). The species turnover graphs are a Simpson dissimilarity measure of species turnover between before and spring 1 compared with before and spring 2. Predictions are from generalized linear mixed models and grey shading represents 95% confidence limits.

4.3.2 Individual species' responses to prescribed fire and the severity of prescribed burns

The occurrence of nine of the ten species analysed was not influenced by prescribed fire, or fire severity at burnt sites.

Species with a response to both prescribed fire and prescribed burn severity

Models showing significant relationships were developed for one species: the White-throated Treecreeper (*Cormobates leucophaeus*). The Akaike weight of 0.57 for the model of fire changes over time (selected from two candidate models) suggests that there is approximately equal evidence supporting either interactive or additive models. Therefore, it is unclear if prescribed burning is having an influence on the occurrence of this species. The fire response graph illustrates a decrease in occurrence on burn sites in the second year post-fire (Fig 4.3 (a)). Severity modelling outputs for this species gave a high Akaike weight for model selection (0.99) (Table 4.2). The severity graph suggested a second year post-fire decrease in occurrence with high severity burns (Fig 4.3 (b)).

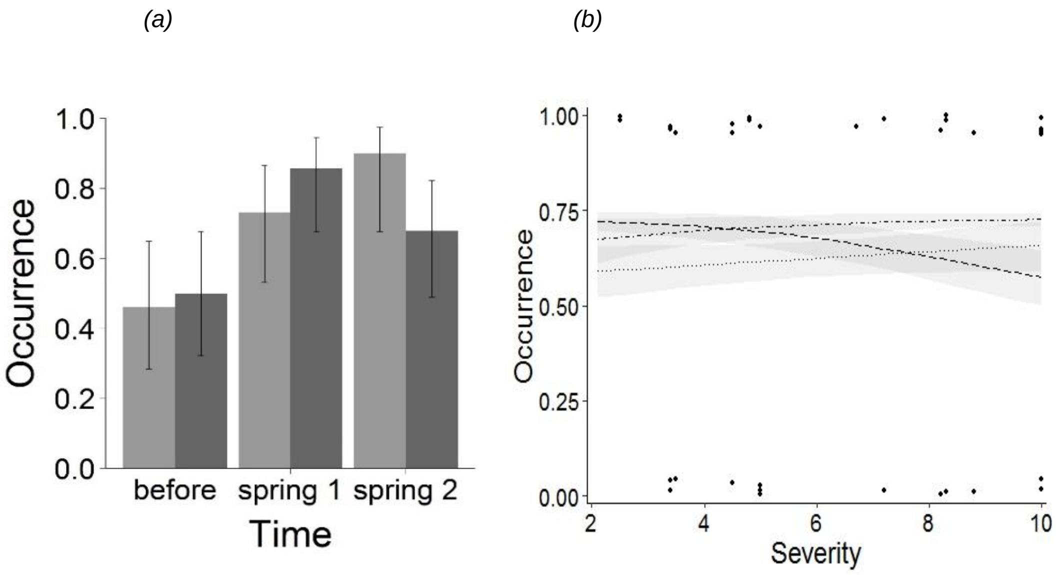


Figure 4.3 White-throated Treecreeper (*Corombates leucophaea*) responses to fire and the severity of prescribed burns.

Fire graph (a) has pairs of control sites (light grey) with burn sites (dark grey) for time periods before the autumn prescibed burn (before) followed by the first observation year (spring 1) and the second observation year (spring 2). Severity responses are modelled (b) with continuous data, showing raw data spread across the severity scale. The fixed effect of time is llustrated with a baseline for before (dotted line) on the burnt sites with comparisons for the two years post fire: spring 1 (dot and dash) and spring 2 (long dash). Predictions are from generalized linear mixed models and errors are 95% confidence limits.

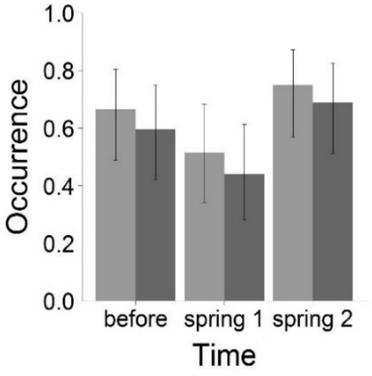
Species with a limited fire and prescribed burn severity response

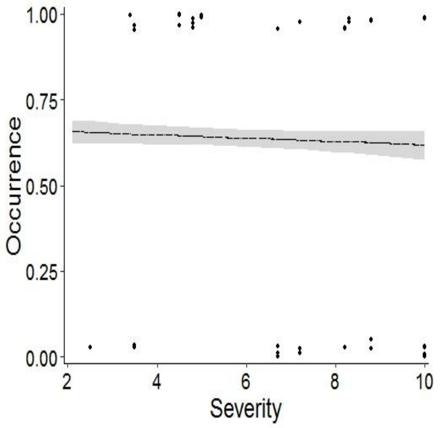
Models of occurrence for the remaining nine abundant species showed no detectible fire or severity responses by any of these species. Top ranked models for five species excluded the factor of time; there were no variations in severity responses by those species changing over time (Buff-rumped Thornbill, Crimson Rosella, Grey Currawong, Spotted Pardalote and the Striated Thornbill, Figs 4.4 (b), (d), (f), (h), (j)). Further to this, any changes in occurrence for these five species in graphs modelling fire were reflected equally between both burn and control sites (Fig 4.4 (a), (c), (e), (g), (i)).

(a) (b)

Buff-rumped Thornbill

Acanthiza reguloides

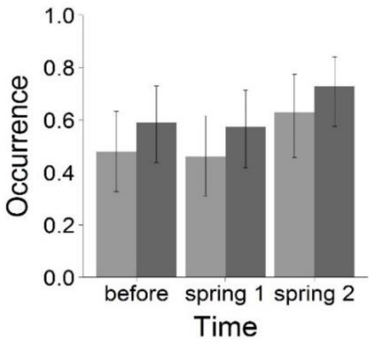


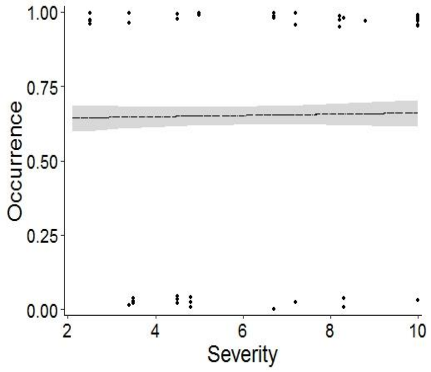


(c) (d)

Crimson Rosella

Platycercus elegans

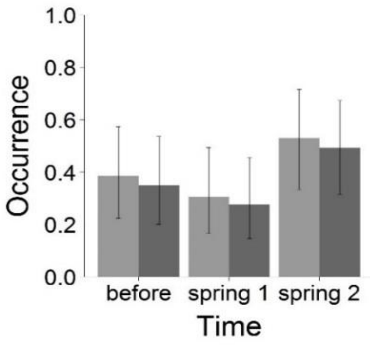


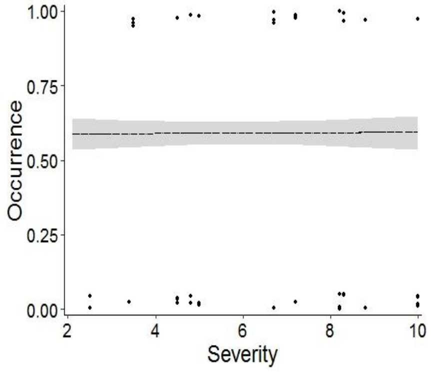


(e) (f)

Grey Currawong

Strepera versicolor

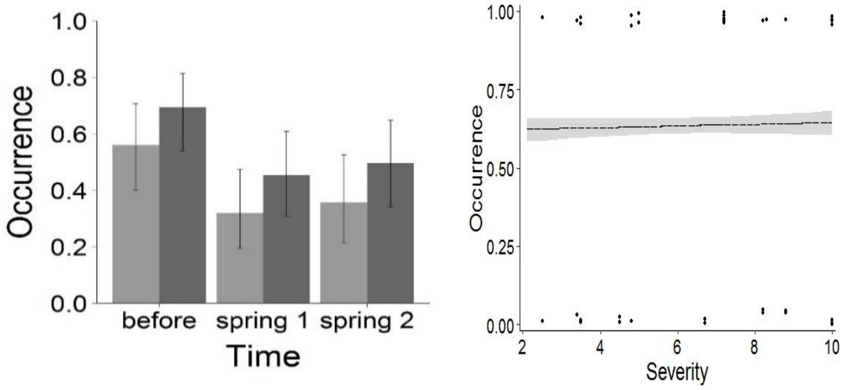




(g) *(h)*

Spotted Pardalote

Pardalotus punctatus



(i) *(j)*

Striated Thornbill

Acanthiza lineata

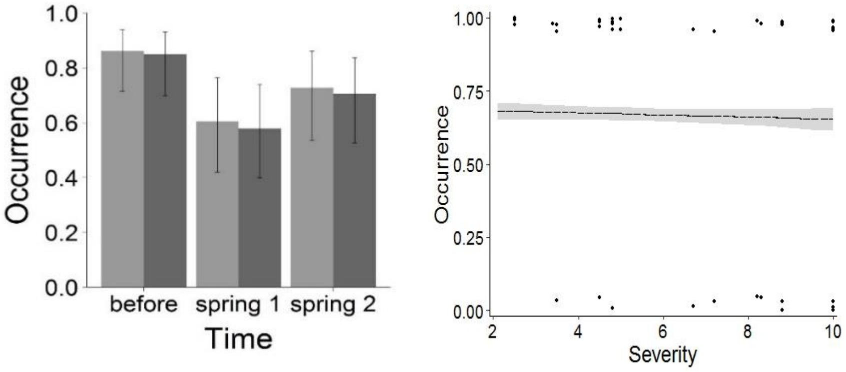


Figure 4.4 Species responses to fire and the severity of prescribed burns (excluding time).

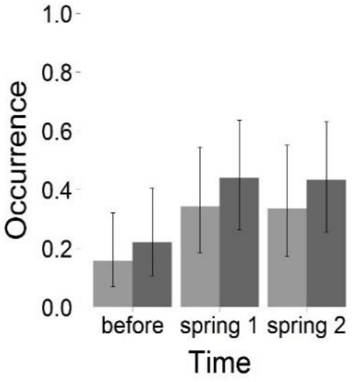
Fire graphs (a, c, e, g, i) have pairs of control sites (light grey) with burn sites (dark grey) for time periods before the autumn prescibed burn (before) followed by the first observation year (spring 1) and the second observation year (spring 2). Severity responses are modelled (b, d, f, h, j) with continuous data, showing raw data spread across the severity scale. Predictions are from generalized linear mixed models and errors are 95% confidence limits.

The occurrence of the remaining four species (Grey Fantail, Grey Shrike-thrush, Rufous Whistler and Scarlet Robin) changed over time, but temporal changes were similar at burnt and unburnt sites (Fig 4.5 (a), (c), (d), (f)). At burnt sites, fire severity had no detectable effect on the occurrence of these species (Fig 4.5 (b), (d), (f), (h)).

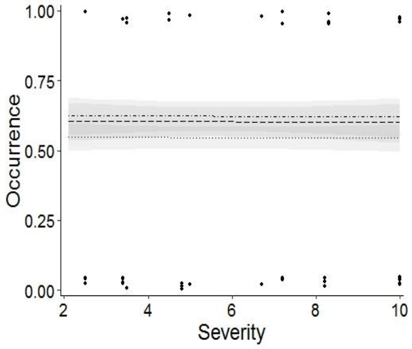
Grey Fantail

Rhipidura albiscapa

(a)



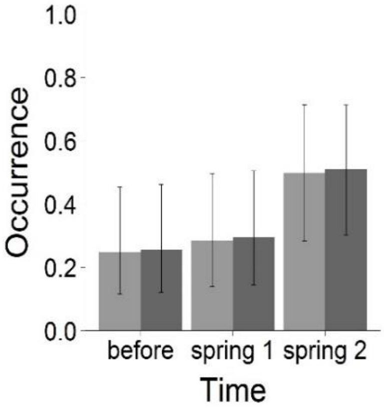
(b)



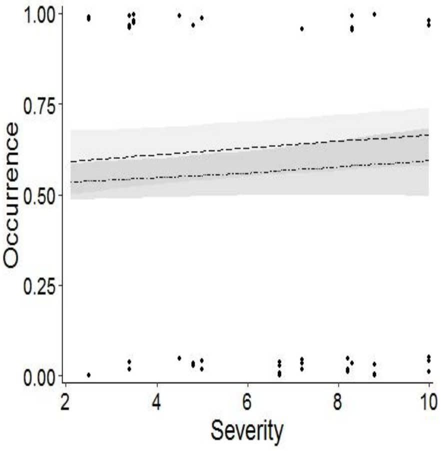
Grey Shrike-thrush

Colluricincla harmonica

(c)



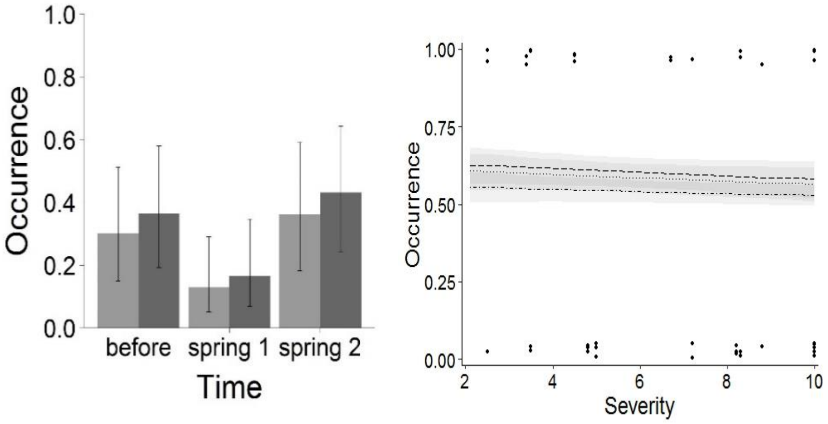
(d)



(e) *(f)*

Rufous Whistler

Pachycephala rufiventris



(g) *(h)*

Scarlet Robin

Petroica boodang

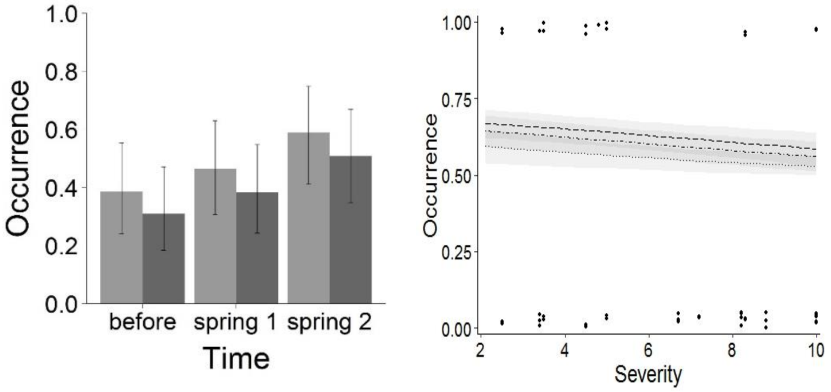


Figure 4.5 Species responses to fire and the severity of prescribed burns (including time).

Fire graphs (a, c, e, g) have pairs of control sites (light grey) with burn sites (dark grey) for time periods before the autumn prescibed burn (before) followed by the first observation year (spring 1) and the second observation year (spring 2). Severity responses are modelled (b, d, f, h) with continuous data, showing raw data spread across the severity scale. The fixed effect of time is llustrated with a baseline for before (dotted line) on the burnt sites with comparisons for the two years post fire: spring 1 (dot and dash) and spring 2 (long dash). Predictions are from generalized linear mixed models and errors are 95% confidence limits.

Table 4-1 Avian responses to prescribed burns in terms of biodiversity measures of species richness and species turnover and the likelihood of occurrence of 10 species.

Two stage model selection based on: the interaction of the random effects (RE) and the interaction of fixed effects of burnt sites with time. Models ranked using Akaike's Information Criteria (AICc). Top ranked models are reported as Akaike weights from two potential candidates, interactive or additive combinations. Fixed effect of time is reported as two measures: first year post fire (a1) and second year post fire (a2) with parameter estimates and standard errors (SE), along with each statistical significance (P). Positive estimates indicate an incease above the intercept mean. The intercept is the reference level in the model. In species models 'before' data was run as the reference. R^2marginal and R^2conditional give the model fit with R^2(m) representing fixed effects and R^2(c) representing fixed plus random effects. Predictions are from generalized linear mixed models. Outputs for turnover model gave a t-value rather than a p-value as the measure of the difference relative to the variation in the sample data.

Response variable	Model	Time	Akaike weight	Estimates	SE	P	R^2(m)	R^2(c)
species richness	burnt + time (RE mosaic)		0.84	-0.04	0.06	0.48	0.17	0.27
		a1		-0.41	0.07	2.72 e-08		
		a2		-0.14	0.07	0.05		
species turnover	burnt + time (RE site)		0.74	0.03	0.06	0.58	0.02	0.64
		a1		0.04	0.03	1.32		
		a2		-0.04	0.03	-1.32		
Buff-rumped Thornbill	burnt + time (RE mosaic)		0.89	-0.30	0.35	0.40	0.06	0.09
Acanthiza reguloides		a1		-0.63	0.40	0.12		
		a2		0.41	0.43	0.34		
Crimson Rosella	burnt + time (RE mosaic)		0.87	0.45	0.33	0.17	0.04	0.04
Platycercus elegans		a1		-0.08	0.39	0.85		
		a2		0.62	0.42	0.14		
Grey Currawong	burnt + time (RE mosaic)		0.88	-0.15	0.37	0.69	0.04	0.10
Strepera versicolor		a1		-0.35	0.42	0.40		
		a2		0.59	0.42	0.16		

Grey Fantail	burnt + time (RE site)		0.82	0.42	0.51	0.41	0.07	0.24
Rhipidura fuliginosa		a1		1.02	0.46	0.03		
		a2		0.99	0.48	0.04		
Grey Shrike-thrush	burnt + time (RE mosaic)		0.88	0.04	0.38	0.91	0.06	0.16
Colluricincla harmonica		a1		0.19	0.44	0.66		
		a2		1.10	0.44	0.01		
Rufous Whistler	burnt + time (RE mosaic)		0.90	0.29	0.40	0.47	0.09	0.19
Pachycephala rufiventris		a1		-1.07	0.48	0.03		
		a2		0.28	0.43	0.52		
Scarlet Robin	burnt + time (RE site)		0.80	-0.33	0.37	0.37	0.04	0.09
Petroica multicolor		a1		0.33	0.41	0.42		
		a2		0.84	0.42	0.05		
Spotted Pardalote	burnt + time (RE mosaic)		0.59	0.57	0.33	0.09	0.08	0.08
Pardalotus punctatus		a1		-1.00	0.40	0.01		
		a2		-0.84	0.41	0.04		
Striated Thornbill	burnt + time (RE mosaic)		0.72	-0.11	0.39	0.78	0.09	0.13
Acanthiza lineata		a1		-1.41	0.48	3.00E-03		
		a2		-0.85	0.50	0.09		
White-throated Treecreeper	burnt x time (RE site)		0.57	0.15	0.55	0.78	0.18	0.18
Cormobates leucophaeus		a1		0.64	0.89	0.47		
		a2		-1.60	1.01	0.11		

Table 4-2 Avian responses to severity of prescribed burns in terms of biodiversity measures of species richness and species turnover and the likelihood of occurrence for 10 species.

Two stage model selection based on: the interaction of the random effects (RE) and the interaction of fixed effects of burnt sites with time. Models ranked using Akaike's Information Criteria (AICc). Top ranked models are displayed with Akaike weights from three potential candidates: interactive or additive combinations, or excluding time as a factor. Fixed effect of time is reported as two measures: first year post fire (a1) and second year post fire (a2) with parameter estimates and standard errors (SE), along with each statistical significance (P). Positive estimates indicate an increase above the intercept mean. The intercept is the reference level in the model. In all cases 'before' data was run as the reference. R²marginal and R²conditional give the model fit with R²(m) representing fixed effects and R²(c) representing fixed plus random effects. Predictions are from generalized linear mixed models. Outputs for turnover model gave a t-value rather than a p-value as the measure of the difference relative to the variation in the sample data.

Response variable	**Model**	**Time**	**Akaike weight**	**Estimates**	**SE**	**P**	**R²(m)**	**R²(c)**
species richness	severity + time (RE site)		0.75	-0.01	0.02	0.45	0.12	0.28
		a1		-0.35	0.24	3.80E-03		
		a2		-0.14	0.23	0.22		
species turnover								
	severity (RE site)		0.69	-2.00E-03	0.01	-0.170	0.00	0.79
Buff-rumped Thornbill								
Acanthiza reguloides	severity (RE mosaic)		0.69	-0.11	0.08	0.22	0.02	0.02
Crimson Rosella								
Platycercus elegans	severity (RE mosaic)		0.63	0.05	0.11	0.63	0.01	0.06
Grey Currawong								
Strepera versicolor	severity (RE mosaic)		0.53	0.01	0.15	0.92	0.00	0.08
Grey Fantail	severity + time (RE site)		0.62	-0.01	0.13	0.91	0.09	0.25
Rhipidura fuliginosa		a1		1.49	0.65	0.02		
		a2		1.15	0.65	0.07		

Grey Shrike-thrush	severity + time (RE mosaic)		0.58	0.20	0.19	0.29	0.12	0.38
Colluricincla harmonica		a1		0.00	0.63	1.00		
		a2		1.25	0.62	0.04		
Rufous Whistler	severity + time (RE mosaic)		0.49	-0.13	0.11	0.25	0.11	0.14
Pachycephala rufiventris	RE mosaic	a1		-0.98	0.65	0.13		
		a2		0.32	0.56	0.57		
Scarlet Robin	severity + time (RE site)		0.53	-0.24	0.10	0.02	0.17	0.19
Petroica multicolor		a1		0.89	0.61	0.15		
		a2		1.36	0.62	0.03		
Spotted Pardalote								
Pardalotus punctatus	severity (RE mosaic)		0.83	0.06	0.09	0.49	0.01	0.01
Striated Thornbill								
Acanthiza lineata	severity (RE mosaic)		0.59	-0.09	0.09	0.32	0.02	0.02
White-throated Treecreeper	severity x time (RE site)		0.99	0.19	0.16	0.24	0.38	0.41
Cormobates leucophaeus		a1		0.24	0.32	0.46		
		a2		-0.79	0.30	0.01		

4.4 Discussion

4.4.1 Can we see any changes to species richness after high severity prescribed burns?

High severity prescribed burns had the effect of removing 100% of the ground cover, tree bark and canopy layers. It might be predicted therefore, that the dramatic structural changes would result in a short term drop in species richness. However, there was no evidence of this occurring. Other studies have found that many species decline in abundance soon after severe bushfires e.g., in East Gippsland, Victoria (Loyn, 1997) and in the Victorian Central Highlands (Loyn and McNabb, 2015). However, in this study, most of the subsequent drops in the occurrence of individual species modelled on severely burnt sites were reflected in both burn and unburnt control sites. The result of little change in individual species occurrence extended across the community, with no clear evidence of a change to species richness.

4.4.2 Is there a change to species turnover levels after a high severity prescribed burn?

Given the widely recognised impact of fire on many birds, it was predicted that there would be an increase in species turnover immediately post-fire, with turnover returning to pre-burn levels by the second year. Again, however, there was little evidence in this study of this occurring. This result aligns with those from other south-east Australian studies that indicate a scarcity of early-successional stage species in eucalypt forests and woodlands (Lindenmayer et al., 2008, Lindenmayer et al., 2014, Sitters et al., 2014a, Loyn and McNabb, 2015, Sitters et al., 2015). So, there may be a paucity of species that may enter the woodland soon after a fire to drive the expected increase in turnover.

The low species turnover in the Heathy Dry Forest may be also due to the rapid vegetation regrowth that has been noted to occur in the south-east Australian forests (Lindenmayer et al., 2008). The sites in this project were burnt in autumn, and rapidly developed dense epicormic growth on the severely burnt sites by spring. It was evident that this regrowth provided protection and food resources for birds. Other studies have yielded similar results; where species abundances' can be directly related to vegetation density (Monamy and Fox, 2000).

In contrast to this outcome of low turnover are results from elsewhere that show distinct post-fire assemblages. Fire in the mulga woodland in the central Australian arid zone results in a post-fire grassland vegetation, with a distinct bird community, dominated by granivores (Leavesley et al., 2010). In the United States, the North American conifer-dominated forests also have a post-fire landscape with a distinct group of early successional stage bird species (Kotliar et al., 2007, Smucker et al., 2005). This may be explained in that, whilst recovery is rapid in Heathy Dry Forest vegetation, the post-fire regrowth of conifer forests occurs at a much slower rate (Franklin et al., 2002).

A further explanation for a lack of turnover with species may have been the closeness in proximity of burn sites to the edge of burns. A number of the burn sites were within 100 metres of the burn edge, which, in many cases, was a narrow dirt road. Furthermore, burn areas were relatively small and contained within the landscape. Therefore, birds from monitored sites had little distance to fly to escape the fire and readily returned, to vegetation that had rapidly regrown.

A further aspect to consider in this research is the impact of edge effects on birds across this study area. Considerable research has been applied to edge effects (Cadenasso et al., 2003a, Cadenasso et al., 2003b, Ries et al., 2004) and what is evident is that, although edge communities are complex, edge responses can be predictable and consistent (Ries et al., 2004). Abrupt anthropogenic edges, such as those generated by severe prescribed burns, have been shown to result in a decline for some birds (Comfort et al., 2016). Furthermore, bird responses at edges vary relative to vegetation density (McWethy et al., 2009). So, as surveys may have been observing birds attracted to burn edges, associations with the nature of the burn need to be made with care. This is particularly so given that three of the ten species individually assessed (White-throated Treecreeper, Grey Shrike-thrush and Grey Fantail) have been found to be more abundant in edge sites (Berry, 2001).

A final aspect to consider is based on the process of selecting species to model, combined with site position. Individual species modelled were selected based on their high abundance relative to others. Furthermore, many of the sites were positioned next (within 100 m) to edges of roads and fire. Therefore, inherent in this design, some of the interior species, which are typically specialists requiring larger woodland areas (Ambuel and Temple, 1983) and species more sensitive than generalist species (Canaday, 1996), were not considered in individual species' assessments.

4.4.3 The White-throated Treecreeper and its response to fire severity

The one species unique in its response to fire severity was the White-throated Treecreeper. Displaying site fidelity, the White-throated Treecreeper would have remained in the area post fire. This was also evident in the study of Loyn (1997), where the White-throated Treecreeper was also found to maintain abundance immediately post bushfires (Loyn, 1997).

While the White-throated Treecreeper normally has a preference for feeding from rough-barked trees (Noske, 1985), in this project it would have found ample invertebrates in the rapidly growing and dense epicormic foliage, that was evident on the severely burnt sites in the first spring post-fire. This was even after its preferred bark habitat was burnt in the high severity fires. This interpretation aligns with another study that illustrated that the White-throated Treecreeper responded with a reduced abundance post bushfire, however it was to a much lesser extent than other species (Loyn and McNabb, 2015). The response in the Heathy Dry Forest was different to that seen in other forests, for example, the Canadian boreal forests. In these vegetation biomes, fire severity is crucial. Burned boreal forests are habitats for many deadwood-dependent species (Hutto, 1995, Nappi and Drapeau, 2009), and so the severity of a fire determines the responses of the deadwood-associated insects and birds. In these environments, in order to supply the post-fire specialist birds with food, deadwood-dependent invertebrate species must survive severe fires on unburnt snags or lightly burnt stands (Nappi et al., 2010). In this project, the burns of high severity left little habitat other than blackened stumps and ash in the landscape for some weeks. However, the regrowth of the vegetation was rapid.

In the Heathy Dry Forest, it was in the second year after fire, on sites of high severity burns, that the treecreeper’s occurrence declined greatly. As the White-throated Treecreeper is a sedentary bird that roosts and nests in tree hollows, its requirement for a supply of hollows (Loyn et al., 2007) may not have been met. As such, its vulnerability to fire may be more a consequence of its needs for nesting, rather than for food.

A further explanation of the treecreepers’ response to severe fires can be given, based on research with ground-dwelling mammals (Recher et al., 2009). Recher et al., (2009) described how fire did not result in the extirpation of their model species however, abundances dropped in the years following the fire as drought conditions persisted (Recher et al., 2009). Similarly, in this project, the second-year decline by the treecreeper may have also been a response to a drying climate, rather than only a response to fire severity.

4.4.4 A BACI design for testing prescribed burn severity illustrating an environmental impact other than fire

No driver of change operates in isolation, and, as has been discussed, rainfall is a major factor in determining not only vegetation health but the drying of the landscape and the fuel available to fire. A strong research approach designed to identify the influence of a single factor is to examine impact and control sites both before, and after, an impact. The value of this BACI design is two-fold. The approach provides an output for a pre-post fire comparison of bird abundance and occurrence. Second, where a response is common to both control and impact sites, both before and after an impact, the influence of a further environmental factor may be revealed. In this project, both burn and control sites were changed post-fire. Broad community abundance, and the rates of occurrence of some individual species, decreased post-fire across all sites. The occurrence of some species recovered in the second year post-fire, however the species richness for the community as a whole, did not. However, this was reflected across both control and burn sites, a response that aligned with a decrease in rainfall in those years. Even though rainfall in 2010 and 2011 marked the end of the Millennium Drought across regions of Australia, the western and central regions continued on a drying trend through 2012 to 2014 (Bureau of Meteorology, 2017). In fact, the annual rainfall for Scotsburn, Victoria (representative of the study area) has been recorded as: 2010 - 948mm; 2011 - 912mm; 2012 - 815mm; 2013 - 705mm; and 2014 - 622mm (Bureau of Meteorology, 2017).

As the monitoring of birds post-fire occurred across this time-period, the drop in abundance on control and burn sites may be a drought response.

While the importance of not only the phenological development of vegetation, but also the competition and predation on species cannot be ignored, it has been shown that successful bird breeding operates within a range largely driven by two factors - temperature and precipitation (Baker, 1939, James and Shugart, 1974). Whilst this research did not have as its central focus a question directly related to changes in rainfall, what was clear were the decreases in abundance and occurrence that corresponded with substantial decreases in rainfall across 2012-2014. This illustrates that, even though the Heathy Dry Forest birds were resilient to small-scale prescribed burns of any severity, that overall, abundances may have been in decline. However, it is understood that annual fluctuations in rainfall can have dramatic and immediate effects on bird breeding (Gibbs et al., 2011). Hence, the outcomes in this project illustrate the limitations of surveys that extend even across several years. Longer-term studies, that can more clearly capture short term pulses in breeding, along with the longer-term abundance trends and fluctuations, are needed to partition the effects of fire regimes under a variable and changing climate.

4.4.5 A more robust design for fire severity research

Of importance when modelling bird data is the nature in which the data is formatted, i.e. presence-absence data as opposed to abundance data. It is important to recognize that the factors that drive the occurrence of a species may be different to those which influence the abundance of that species (Howard et al., 2014). Therefore, the sensitivity of different bird species to fire may go undetected if the inappropriate form of data is used in an analysis (Kelly et al., 2017). There are times when outputs from models based on presence-absence data are best. This may be, for example, when action is urgently needed to protect a threatened species where rarity precludes analysis based on abundance data (Guisan et al., 2013). However, as has been highlighted, extirpation rates for birds are increasing and species' ranges are shrinking (Chapter 1.1.3). So, where even relatively abundant birds are at risk, accurate predictions of ecological consequences to an environmental impact are required and here, abundance data is preferred (Howard et al., 2014) to capture demographic changes to a species' range across the landscape (Bradstock et al., 1997). For a more robust set of results, a project with more bird data (either spatially or temporally), than what was available in this project, would allow for a robust analysis base on abundance models. When species are rare, there is little more information in variations in abundance data than in occurrence data alone. For more common species, abundance data contains more information and will, therefore, reveal more about population dynamics than that based on mere presence or absence (Kelly et al., 2017).

Plate 19 Two common carnivores in Heathy Dry Forests, with Grey Currawong (*Strepera versicolor)* (top) possibly out-competing Laughing Kookaburra (*Dacelo novaeguineae)* in dense new-growth (2.5 – 10 year old) vegetation.

Source: Grey Currawong – Steve Happ, Laughing Kookaburra – Danny Gunn, Victorian Birders

5 Summary and synthesis of results

5.1 Overview

Fire is a major form of natural disturbance that shapes biome distributions worldwide, influences both the carbon cycle and climate, and helps to maintain the structure and function of fire-prone communities (Bond and Keeley, 2005, Bowman et al., 2009, Bradstock et al., 2010). The ubiquitous nature of fire across many ecosystems therefore engenders research into the impacts of fire on ecological communities, necessary because of threats to flora and fauna, as a result of changes to fire frequency and severity (Gill, 2012). Moreover, inappropriate fire regimes may threaten the persistence of species (Whelan et al., 2010), and may even result in irreversible shifts in ecosystem states (Lindenmayer et al., 2011). Understanding the different patterns between aspects of the fire regime, is imperative to maintaining both flora and fauna biodiversity.

Fire is an agent of vegetation disturbance, that can influence avian assemblage patterns, with some birds responding positively to a fire event, or conversely, fire may pose a significant threat to those birds with narrow habitat requirements (Woinarski and Recher, 1997, Brown et al., 2009). A changing fire regime has been flagged as one of the key drivers of avian biodiversity decline both globally (Birdlife International, 2008), and within Australia (Birdlife Australia, 2015). The challenge lies in maintaining a fire regime in the landscape such that all habitat requirements for avifauna are met (Clarke, 2008, Loyn and McNabb, 2015). Yet, there remain substantial gaps in knowledge on bird responses to fire.

Furthermore, research lags well behind the rate at which the fire regime is changing in levels of frequency and severity.

Mediterranean regions around the globe are impacted by changing weather patterns, resulting in complex interactions between some of the factors impacting birds, that may result in avian species decline. This thesis is a case study for examining bird responses in a Mediterranean region, where climate drivers combine to heavily impact bird responses.

The south-east region of Australia has a diverse range of vegetation types. In Victoria, vegetation is classified into 32 Vegetation Divisions (EVDs), each of which is further categorised as various Ecological Vegetation Classes (EVCs) (Cheal, 2010). Heathy Dry Forest is one of the EVCs within the Grassy/Heathy Dry Forest EVD and patches of this forest type have been fragmented, and much reduced in extent. However, areas of forest do remain, but avian species' abundance trends within these landscapes are not consistent, for many species abundances in the entire south-east region of Australia are in decline (Birdlife Australia, 2015). As fire is a pervasive component of forest and woodland regions in south-east Australia, an understanding of changes to the fire regime across vegetation types is fundamental to efforts to curb biodiversity decline. However, there is a paucity of research into Heathy Dry Forest bird responses. Therefore, this synthesis compares patterns of bird responses between the Heathy Dry Forests investigated in this thesis and a broader range of eucalypt Foothills Forests (of south-west, central and north-east regions of Victoria) (Loyn et al., 2003, Loyn and McNabb, 2015, Haslem et al., 2016, Kelly et al., 2017); semi-arid Murray Mallee vegetation (Brown et al., 2009, Taylor et al., 2012, Taylor et al., 2013, Watson et al., 2012); along with the woodlands of the Australian Capital Territory (Watson et al., 2002). Comparisons between results also draws on research that incorporates a range of vegetation communities including; Heathland, Forby, Foothills and Wet Forest in the Otway Ranges (Sitters et al., 2014a, Sitters et al., 2014b, Sitters et al., 2015), and a range of Foothills Forests in East Gippsland (Loyn, 1997). While factors such as topography and elevation variations along with logging practices are all important factors impacting vegetation density, forests in the west and central regions of Victoria have experienced less rainfall in recent decades than their eastern counterparts (Bureau of Meteorology, 2017). Research confirms that Foothills vegetation in Victoria is strongly influenced by rainfall (Haslem et al., 2016), and therefore understandably, vegetation to the west in Victoria, is less dense. The research conducted in the region most similar to that of the Heathy Dry Forest vegetation class studied in this thesis, is that of Loyn et al. (2003), who examined bird responses to low severity prescribed burns in the Wombat Forest of central Victoria.

Bushfire events in the south-east Australian landscape are increasing in frequency (Fairman et al. 2016). Furthermore, in recent years, when bushfires and prescribed burn areas are combined, the result has been more area burnt. Whilst the prescribed burn protocol follows the ‘pyrodiversity begets biodiversity’ theme, not all vegetation age classes benefit all birds. In the Otways Foothills Forests, avian species richness was found to be positively associated with vegetation age class diversity (Sitters et al., 2014a). However, in the Murray Mallee vegetation, species richness was not associated with fire-mediated heterogeneity. In the Mallee, species richness reduces in regrowth vegetation (Taylor et al., 2012). Similarly, in Heathy Dry Forests, Laughing Kookaburras are less abundant in dense new growth vegetation (2.5 to 10 years old), supporting the notion that regrowth vegetation post-fire is not necessarily favoured by all birds. Therefore, the imperative for a comprehensive understanding on all bird responses is clear.

5.2 Synthesis of results

5.2.1 Bird responses to the temporal patterns of fire

Successional dynamics, following disturbances such as fire, are represented by temporal changes in the vegetation (Clements, 1916, Gleason, 1927, Connell and Slatyer, 1977); for example, from pioneer to climax stage (Helle and Mönkkönen, 1985). Fundamental to managing fire to benefit biodiversity is knowledge of how the temporal arrangement of fires may influence biota (Driscoll et al., 2010). Therefore, much fire ecology research is based on temporal changes in species' occurrence, distribution and abundance post-fire (Fox, 1982, Keeley et al., 2005, Watson et al., 2012, Sitters et al., 2014a).

It is known that the immediate post-fire response of avifauna is substantial, occurring in response to removed or changed food resources, but that the response is often of short duration (Woinarski and Recher, 1997). The results in this thesis for Heathy Dry Forest birds exemplify the response that Woinarski and Recher describe, in that immediately post-fire, abundances dropped across all foraging guilds, reflected across most individual species analyses. This reduced bird abundance remained for only six months. After six months, vegetation resprouted and new growth developed, with bird abundances rapidly recovering. The caveat to this generalisation is that this research focused on those species driving 80% of the assemblage patterns, and further, those that totalled over 90% of the total birds observed. Whilst there was a cohort of 23 species observed in small numbers, their numbers were too few to allow for their individual responses to be effectively reviewed. Nonetheless, in Heathy Dry Forests there were no species identified as post-fire specialists. This result aligns with research on the Shrubby Dry, Foothills and Damp Forests vegetation sites of Wombat State Forest (Central Victoria), in which low intensity prescribed burns resulted in little difference being found between bird responses on both control and burnt sites (Loyn et al., 2003). In this research conducted in vegetation similar to the Heathy Dry Forests (in terms of climate, rainfall and vegetation), there was evidence for some increases in open ground foragers and seed eaters on burnt areas, but there were no distinct post-fire specialists. Results in the Wombat Forest region suggested a high degree of stability in terms of not only total bird count abundances but also of species composition. These results differ a little with that of Loyn (1997), who undertook research in East Gippsland, Victoria. Loyn’s research area (1997) was further east than that of the Heathy Dry Forests of this thesis and incorporated five vegetation communities: warm temperate rainforest, closed heath, dry, wet and damp sclerophyll forest. In that research, Loyn stated that whilst their numbers were too low to statistically analyse, there was some evidence of a small carnivore influx post bushfire. Nonetheless, most of the carnivores were resident birds and there were only two species (Nankeen Kestrel and Brown Falcon) that were not seen on other occasions. Nevertheless, his results showed that some species that forage from open ground behaved as early-stage specialists in the denser forest types such as rainforest (Loyn 1997). A further comparison can be drawn with that of research conducted immediately post bushfire-events of Black Saturday (2009) in Victoria (Lindenmayer et al., 2014). While their research highlighted complex responses by birds to landscape-level fire severity, they concluded that there is a paucity of species in the Central Highlands montane ash forests that specialize on early successional vegetation. The Heathy Dry Forest result therefore broadly reflects that of other vegetation types across Victoria, with no foraging group or individual species apparent as a post-fire specialist.

Habitat changes reflecting immediate fire impacts on bird abundances have been found to vary significantly between different vegetation types (Loyn, 1997). Loyn’s research (in East Gippsland, Victoria), described how habitats vary in their recovery rates and that post-fire total bird abundance was reduced to 60% of the initial levels, but had recovered across all vegetation types within three years. A similar result was found by Lindenmayer et al. (2008), in Booderee National Park (southern coast of New South Wales, south-east Australia). In their research on bird responses to varying degrees of vegetation complexity post-fire, they also found that bird assemblages had recovered in three years. The results in Heathy Dry Forests suggest a more rapid recovery. Bird abundances had recovered six months post-fire in the time-since-fire investigation (Chapter 3) and furthermore, alpha and beta diversity results (Chapter 4) suggested no changes to species richness or turnover between pre and post-fire bird abundances, with recovery of the bird community occurring within a year.

Broad-scale generalisations, of bird responses to fire aligned with a vegetation type, are not straight-forward. Reviewing further aspects of the fire regime are fundamental to a clear understanding of bird responses (Kelly et al., 2017). Loyn's research in East Gippsland (1997) reviewed differences between heathland and rainforest bird communities in terms of post-fire impacts on vegetation. In his research, bushfire had in fact had minimal impact in the rainforest region as the physical structure had survived the fire. Further, the heathland had above-ground parts of shrubs burnt, requiring below-ground lignotubers to sprout and these, Loyn suggested, did not offer a productive food source for birds within three years. Yet, it is expected that rainforests are less adapted to fire and may take decades to recover (Russell-Smith and Stanton, 2002), and that heathland recovery may be rapid (Fox, 1982). However, results from Heathy Dry Forest suggest this vegetation has distinct age classes in the first ten years post-fire, to which species and assemblages are responding. Whilst the broader community showed little variation across age classes older than six months, individual species and foraging guilds may have been responding to structural vegetation changes (Chapter 3). Furthermore, these structural changes were what distinguished the Heathy Dry Forest in the classification by Cheal (2010). The first five divisions in the Cheal classification system are useful in terms of applying vegetation measures to bird responses; such as epicormic growth with an increased insect food supply (6 month to 2.5 year age class), or the increased shrub cover (2.5 – 10 year age class). However, the divisions in this system for vegetation over 35 years were not tested, due to the little structural difference visible in this biome after vegetation is 35 years old post fire.

Avifaunal species are attracted to particular successional stages (Bradstock et al., 2005, Parr and Andersen, 2006), which was evident in the Heathy Dry Forest biome. In fact, the ‘habitat accommodation model’ suggests that species will reach peak abundance as changes in the vegetation fulfil the habitat requirements of each species (Fox, 1982). In this model, species do not modify local physical conditions, but rather, conditions are changing in response to other factors (Fox, 1982). Furthermore, rates of bird recolonization after fire depends on habitat development, structure and floristic aspects of the vegetation (Loyn, 2012). There is evidence for a ‘habitat accommodation’ model fit in Heathy Dry Forests. Species increased in abundance in the regrowth and new growth vegetation, from an apparent response to food resource availability. There was however, a distinction between the Heathy Dry Forest’s regrowth (6 months to 2.5 years) and new growth (2.5 to 10 year) stages. Young regrowth (6 month to 2.5 years) appeared as epicormic growth on trunks, the ground was still bare in early months. Bird species that normally fed in the canopy such as the Spotted Pardalote and Grey Fantail, were in greatest abundance in this vegetation age class. Whilst insect responses to fire events vary across species (Elia et al., 2011), there is evidence from research in Foothills Forests that suggests that in Heathy Dry Forests, the Spotted Pardalote was likely responding to an increase in the abundance of psyllid insects within developing epicormic growth (Loyn and McNabb, 2015). The older regrowth and shooting new growth (2.5 to 10 years) had serotinous seedlings sprouting, and grasses and brackens were regenerating. Other birds, such as the nectarivores, were in greatest abundance in this new growth comprising dense ground cover and flowering trees. The patterns shown by the species in new growth vegetation correspond with the eucalypt nectar production, important as the primary food source for these guilds (Ford and Paton, 1977), and also important as a resource for many invertebrates (Horskins and Turner, 1999). It is during the first ten years post fire that the greatest differences between bird assemblage patterns were found in the Heathy Dry Forest. In fact, the oldest vegetation class reviewed in this thesis showed the most similarity in bird assemblage patterns to other vegetation age classes, sharing bird species with all stages (Fig 3.8).

It is suggested that declines in occurrence may relate to an inability by a species to obtain enough resources (Fox, 1982). In Heathy Dry Forests, the Laughing Kookaburra exemplifies this notion. The Laughing Kookaburra (along with other birds in Heathy Dry Forests), had returned six months post-fire, to vegetation with dense epicormic regrowth (6 months to 2.5 years). As new growth covered the ground (2.5 to 10 years), abundances of kookaburras once again declined. Laughing Kookaburras are highly social, territorial and in fact sedentary, maintaining territories all year. Further, the social group has a dominant pair supported by up to five helpers (Legge and Cockburn, 2000). Assuming they fly from fire in Heathy Dry Forests and survive, they must be able to persist for six months in the territory of other Laughing Kookaburras, before they return. Then, they must again leave as the revegetation creates ground cover. Dense vegetation may not be suitable for a broad range of reptiles that they feed on; open canopy environments being essential for ectothermic animals for basking (Michael et al., 2011). In fact, Loyn (1980) investigated dense stands of the Foothills Forests of central Gippsland Victoria, and determined that Laughing Kookaburras were absent. Furthermore, Loyn (1997) found that Laughing

Kookaburras were most numerous in East Gippsland when there was plenty of open ground. Surviving in a landscape which, on being repeatedly burnt, creates a perpetual ground-cover growth stage in the vegetation, may be a challenging issue for a species that lives for over ten years (Legge, 2000).

Principles of competition and inhibition form a fundamental component of succession theory (Connell and Slatyer, 1977). Species may out-compete one another for resources (Gause, 1932). This is possibly the case with the carnivores in Heathy Dry Forests, with Grey Currawongs and Laughing Kookaburras both common species, and yet, there is a low number of sites with both species present (Table 3.8). Where species are less suited, or where resources are less optimum, one species may out-compete the other (Fox, 1981, Fox, 1982). This may be a contributing factor for the drop in occurrence of Laughing Kookaburras in new growth vegetation, in that, there is a drop in suitable prey and furthermore, there is competition for remaining resources with the Grey Currawongs. Grey Currawongs have been observed foraging from hanging bark at canopy height (Recher, 2016). In fact, research has shown that approximately 70% of their foraging is at ground level and the remaining 30% occurs in foliage up to five metres above ground (Recher and Davis, 1998). In Heathy Dry Forests, Grey Currawongs were mostly observed foraging on the ground. However, this versatility may afford them greater access to food than the Kookaburras – kookaburras pounce from height, onto prey that are limited in dense regrowth. Nonetheless, research has shown that Grey Currawongs and Laughing Kookaburras do in fact coinhabit the same areas. Recher and Davis (1998) found that on eight woodland sites, censused over a two-week period, Laughing Kookaburras were observed on two of eight sites; both of which also had Grey Currawongs. Similarly, in Heathy Dry Forests, both species were observed within the same site on numerous occasions (Table 3.8). However, the frequency dropped when ground cover was dense. The ability to coexist in an area means that prey must be ample. In the Heathy Dry Forest post-fire, dense, new growth, the competition for a limited food supply may be too great for the Laughing Kookaburra.

Reporting by Birdlife Australia (2015) is not explicit in suggesting reasons for the recent, rapid decline of the Laughing Kookaburra. Their report highlights that the entire avian community of the south-east region of Australia is under threat, and that the decline of the Laughing Kookaburra is of concern. Research by MacNally et al. (2009) describes the role that drought has played in reducing Laughing Kookaburra numbers. As more of the forest landscape is burnt in area each year by bushfires and prescribed burns, and areas of Heathy Dry Forest perpetuate a dense new-growth vegetation, Laughing Kookaburra abundances may decline in Heathy Dry Forest regions - regions being impacted by an increase in fire frequency and extent, and underpinned by a drying climate. It may be that the mainstay for the Laughing Kookaburra may become semi-rural, rural and agricultural regions that provide an open landscape, with nearby water holes and with trees, old enough to have hollows.

A number of challenges exist, when managing for different vegetation types in the landscape. As both research in the arid Mallee region along with research on the forests of the Otways Ranges in southern Victoria showed, rates of resource development can be highly variable (Haslem et al., 2011, Sitters et al., 2014b).

Furthermore, as Mallee research showed, further exemplified by the Laughing Kookaburra response in Heathy Dry Forests, management strategies promoting new growth and regrowth vegetation will not necessarily benefit all species (Brown et al., 2009, Taylor et al., 2013, Connell et al., 2017). Management focus must be clear that while many species will increase in abundance in regrowth (6 months to 2.5 years) and new growth (2.5 to 10 years) vegetation (Chapter 3), and indeed, these vegetation age classes are where nectarivore abundances may reach their peak (Figure 3.3); there are species whose abundances will decline in dense new growth.

Another challenge in managing the biodiversity of birds within Heathy Dry Forests may be related to scale. The ‘shifting-mosaic steady state’ suggests that if the distribution of vegetation is averaged over sufficient space or time, the proportion of the landscape at each successional stage is constant and the landscape is in equilibrium. This is despite the fact that non-equilibrium states are evident at smaller scales (Bormann and Likens, 1979). Whilst this concept does acknowledge that spatiotemporal variations are disturbance driven, it overlooks the complexities that may arise from multiple disturbances (Mori, 2011). It does not consider that some vegetation communities, such as Heathy Dry Forests, are heavily fragmented, with bushfire frequency expected to increase (Pitman et al., 2007), in a region that is on a drying trend. Other research has exemplified the resultant issues from low rainfall levels, by highlighting how rainfall is a dominant influence on vegetation and therefore, low rainfall impacts the species richness of native animals (Kelly et al., 2012, Sitters et al., 2014a).

Further exemplifying the need to review drivers of change together, research in woodlands of the Australian Capital Territory has shown how reducing both woodland patch size and vegetation complexity, impacted species not previously identified as threatened. Of the seven species this research identified (Weebill, Brown Thornbill, Buff-rumped Thornbill, Spotted Pardalote, Grey Shrike-thrush, Scarlet Robin and White-winged Chough), all were observed in the Heathy Dry Forest research and only the Weebill was not noted as a common species (Watson et al., 2003). Heathy Dry Forest investigations in this research made clear that the common species' responses are broadly representative of the guild they represent. Therefore, monitoring those species noted as common in Heathy Dry Forests, and flagged as being impacted by threatening processes by Watson et al. (2003), may be valuable, to flag potential issues for other less common species.

If the fire management goal in a region is to perpetuate natural fluctuations in vegetation structure, the abundances of species dependent on vegetation structure may fluctuate as well. This has implications for bird biodiversity, exemplified in research conducted in semi-arid Mallee vegetation (Haslem et al., 2011). In this research, authors state that fire management must explicitly acknowledge that fire affects fauna and fuel differently, over extended time-frames. In fact, researchers into Mallee fire landscapes argue that the application of a 'pyrodiversity begets biodiversity' paradigm, converting long unburned vegetation to young regrowth, is a direct threat to bird diversity (Taylor et al., 2012). Heavily impacted landscapes favour generalist species, species that are better able to utilise changes in land-cover type (Wiens, 1989, Le Viol et al., 2012, Watson et al., 2014). Indeed, this is the case in Heathy Dry Forests, with the uncommon species comprising only one percent of the total community. If fire management fails to recognize and manage for the requirements of the rarer species, localised extinctions may result.

5.2.2 Bird responses to an increase in fire frequency in the landscape

Woinarski and Recher (1997) discuss how fire is a major threat to birds of fragmented, eucalypt forests in that fire may limit the resources of birds whose distribution is already restricted. Furthermore, those with limited reproductive potential or poor dispersal abilities are significantly threatened. They highlighted then (in 1997) that of greatest significance as a threatening process to avian communities, was an increase in fire frequency. Indeed, in Victoria the increase has been realised, not only in terms of the frequency of ‘mega-fires’ (Fairman et al., 2016) but with the application of prescribed burns increasing area burnt (Teague, 2010).

Whilst studies have determined that an increase in fire frequency in some vegetation types may result in a simplification of the vegetation structure with a reduction in the understorey (Albanesi et al., 2012), the structure of the Heathy Dry Forest tends to become simplified through time, naturally. As the vegetation matures, the structure of the understorey greatly reduces. By 35 years, the Heathy Dry Forest has a vertical profile that has simplified to one consisting largely of ground cover, bark and a canopy.

Comparisons of bird responses, between Heathy Dry and Foothills Forests, may be most relevant in a review of fire frequency impacts on birds because in part, these vegetation communities are similar. The slopes and ridges of Foothills Forests support drier, more open forest dominated by eucalypts (Kelly et al., 2017), much like that of Heathy Dry Forests. Maximum vegetation diversity estimates in Foothills Forests vary from approximately 50 to 100 years (Loyn, 1980). However, the distinct vegetation stages in the first ten years post-fire that birds are responding to, that impact the structural stages in canopy, bark and ground layers, are similar between both the Heathy Dry and Foothills vegetation types (Cheal, 2010).

There was limited evidence for a fire frequency response by birds in Heathy Dry Forests. There was little discernible difference in the bird community response between sites burnt once in the last 20 years, compared with sites frequently burnt. In fact, it was clear that birds were responding to the distinct successional changes in vegetation, since the most recent fire (Chapter 3.4.3). Haslem et al. (2016) investigated multiple fires in patches of Foothills Forests across central and eastern Victoria. Their results were similar to the current results, in that they found little evidence for post-fire succession being influenced by fire history. Aponte et al. (2014) had investigated the impact of repeated prescribed burns on coarse woody debris in the Wombat State Forest (central Victoria), and found that repeated burns have the potential to significantly decrease coarse woody debris stocks. Haslem et al. (2016) stated that coarse woody debris is an important resource in foothills forests. But nonetheless, they found minimal effects from preceding fires on vegetation structure. However, Haslem et al. (2016) argue that consideration must be given to future, shorter inter-fire intervals and more severe fires, both potentially impacting important resources such as coarse-woody debris and hollow-bearing trees in Foothills Forests. Indeed, as was evident in Heathy Dry Forests with the cavity-nesting White-throated Treecreepers, removal of nest sites may have resulted in a reduction in the occurrence of treecreepers the second spring after fires (Chapter 4).

Even though community and foraging guild responses to fire frequency were minimal in Heathy Dry Forests, there were three species of carnivore that responded to the frequency of fires on sites. The Laughing Kookaburra and the Australian Raven both preferred sites that had experienced infrequent burning, with their greatest abundances apparent in open, older, unburnt vegetation. Grey Currawongs however, preferred sites that had been frequently burnt. In Heathy Dry Forests carnivores may be impacted by competitive exclusion, competing for prey that is less abundant or obscured (Chapter 3). It would be of benefit to have a larger spatial and temporal database of Heathy Dry Forest carnivore observations, to further investigate the relationships between the carnivores. Indeed, research has highlighted that carnivore numbers in avian datasets are often low, limiting analyses and subsequent discussions on this guild (Recher et al., 1985, Recher and Davis, 1998, Loyn et al., 2003).

Thomas et al. (2006) described the challenges that arise when attempting to determine the cause of range changes for species, when some of the combined factors affecting distribution are related to a changing climate. Indeed, this thesis described the complex drivers of climate that change fire weather in the south-east Australian and more specifically, Victorian regions. If the goal is to limit biodiversity decline, then ongoing research on bird responses to fire frequency may well need to be designed such that data modelling can capture responses to numerous questions related to different aspects of the fire regime. While the results in this thesis suggested a limited response by Heathy Dry Forest birds to fire frequency, this response was based on a measure of the number of times a location had been burnt in recent decades. This result aligned with work by Loyn et al. (2003), in the Wombat State Forest, in which he found a high degree of stability in total bird abundance in vegetation subject to one of five experimental low-intensity repeated burning treatments. However, more recent research by Kelly et al. (2017), in their investigations on flora and fauna in Foothills Forests in the Central Highlands, Victoria, determined that in fact, inter-fire interval was the most important variable for plants and animals when data were pooled. An example of their extensive results is that they found that the occurrence of some bird species, particularly the Eastern Yellow Robin and White-Browed Scrubwren (both species in Heathy Dry Forests), increased with longer fire intervals. This highlights the importance of project design capturing results that reflect the changes in the vegetation that the birds are responding to, combined with a fire frequency factor.

Underpinning succession theory is the concept of the ‘vital attributes’ of the vegetation species in a community. These are the life-history characteristics vital to the role of the dominant vegetation species in a replacement sequence (Noble and Slatyer, 1980). Victoria’s vegetation classification system, modified by Cheal (2010), incorporates ‘vital attributes’ for the dominant vegetation species within each vegetation group. For Heathy Dry Forests, the minimum ‘tolerable fire intervals’ are 10 years post-prescribed burn and 15 years post-bushfire, required for the dominant species to persist (Cheal, 2010). The current research determined that the measure of tolerable fire interval, is not of value in predicting bird responses in Heathy Dry Forests (Chapter 4.3.4.9). Whilst research has been undertaken to review fire frequency in terms of an inter-fire interval in Foothills Forests (Leonard et al., 2016, Kelly et al., 2017), to the best of my knowledge, minimum ‘tolerable fire interval’ limits have not previously been tested, for their efficiency as predictors of bird abundances in Heathy Dry Forests. If the aim is to investigate fire frequency impacts on avian species, then project design incorporating an inter-fire interval will result in output that gives more meaningful results, as inherent in this type of design, avian species’ tolerance thresholds will be tested.

5.2.3 Bird responses to the different severities of prescribed burns

Prescribed burns are applied to the landscape in Victoria with increasing frequency, collectively covering a large expanse of land. Whilst the application of fire is primarily to reduce bushfire impacts, prescribed burns are also carried out in efforts to increase biodiversity (Penman et al., 2011). Hobbs (2010) described how the use of fire for habitat maintenance and diversity assumes there will be some optimum frequency that will maintain vegetation diversity. Hobbs explained that this is a complex process due to a lack of detailed information on the population dynamics of many species, however he is referring to only vegetation species. Indeed, the process is complex. While the case can be made that prescribed burns in Victoria known as Ecological Burns are designed to burn at varied intensities to achieve different goals of fuel reduction, fire intensity and fire severity are not the same thing. A burn designed as low intensity may be impacted by weather conditions on the day and result in a high severity fire. Fire management needs to consider that prescribed burns of high severity may result in varying bird responses, rather than assume that all prescribed burns will be of low and controlled impact. Whilst the literature on the faunal responses to fire is increasing, some argue that there is a paucity of information on how animals respond to fire severity (Kotliar et al., 2007, Lindenmayer et al., 2013). To the best of my knowledge, this Heathy Dry Forest investigation on the impacts of different severity prescribed burns, is the first study of this kind in this vegetation type.

There are knowledge gaps on broad-scale wildlife responses to fire (Clarke, 2008) and while there is some research into prescribed burn impacts on fauna (Tolhurst, 1992), there is still insufficient data to determine if current prescribed burn practices have been successful in meeting multiple objectives (Penman et al., 2011). Furthermore, the assumption is broadly made that prescribed burns will be of controlled intensity and severity; reduce only a portion of fire fuel from the landscape. In fact, research by (Loyn et al., 2003) investigated the effects of repeated low-intensity fire on bird abundance in Foothills Forests and the project design was such that the five fire treatments investigated incorporated only low severity burns. This project formed part of a multidisciplinary project (Tolhurst, 1992), designed to address questions related to fuel reduction burning in Foothills Forests. However, as this thesis showed, not all prescribed burns result in low severity impacts to the vegetation structure to which birds are responding. The case may be argued that the burn objective may be to remove a large percentage of fuel cover, however it is doubtful that the intent would be to remove all tree bark-cover and canopy. However, this situation can and does occur (Plates 8 – 16).

For prescribed burns to be an effective form of fire management in the Victorian landscape, and one that results in a positive outcome for bird biodiversity, a broad-scale knowledge of a proposed burn area is required. Indeed, Pastro et al. (2014) argued in their extensive investigation into vertebrate alpha and beta diversity responses to bushfire and prescribed burns across both northern and southern hemisphere, that ecological theory underpinning prescribed burn process, such as patch mosaic burning, is context dependent. This argument was based on their disparate responses of vertebrate diversity (both alpha and beta) to fires. Their argument was that 'no one-size-fits-all', as they highlighted that both high and low severity burns can result in positive effects on affected taxa.

Birds in Heathy Dry Forests showed little evidence for a prescribed burn severity response. Research by Loyn and McNabb (2015) investigated short term responses to bushfire in a Damp Heathy Woodland is south-east Victoria (Bunyip State Park). They found that after severe fire, birds were 77% less numerous in the first winter post-fire but that some redistribution had occurred by the first spring. Indeed, the current research on Heathy Dry Forest bird responses may have displayed a similar trend. The burns were in autumn, and the bird observations used in analyses were collected the first and second spring, post fire. There was no investigation into the immediate decline in abundance post-fire across winter months, however by the first spring, results showed that assemblages had recovered. Therefore, results suggest a rapid recovery by most species, regardless of a prescribed burn severity.

Results from this research, when combined with results from work by others, suggest that the impacts from a severe fire may be confounded by fire size. Lindenmayer et al. (2014) reviewed bird responses to the bushfires of Black Saturday, across Victoria. In their research, pre-fire bird data enabled a comprehensive pre and post burn assessment of a day of extensive severe fires across Victoria. They found strong evidence of the effects of landscape-level fire severity and fire extent on bird species richness. Whilst there are clearly many differences between a whole-of-landscape severe bushfire and a controlled prescribed burn (eg fuel loads, fire weather and climate on the day), the severe fires of the Heathy Dry Forest prescribed burns were quite different in that extent was reduced, compared to the extensive and comprehensive fires of Black Saturday. The Heathy Dry Forest prescribed burns were not whole of landscape. Prescribed burns were small and close to unburnt vegetation stands (Chapter 5).

Some research into fire severity impacts emphasises the degree of canopy burn as being a major determinant of bird responses to fire severity. Haslem et al. (2016), in their research across the Foothills Forests of central and eastern Victoria, focused questions on frequent fires. However, their findings were relevant to fire severity. They argue that the primary way fire affects vegetation structure is via attributes such as fire severity. They found that the severity of the most recent fire influenced post-fire canopy regeneration, that low severity fires in Foothills Forests rarely affect canopy strata. Further to this, Lindenmayer et al. (2014) found in their research into the Black Saturday bushfire impacts on birds, that most species persist when the overstorey remains intact. However, in the Heathy Dry Forest, both alpha and beta diversity comparisons, between pre and post-fire bird abundances, were unchanged, regardless of fire severity. This speedy recovery of bird assemblages was evident on-ground, as birds were numerous in the extensive regrowth vegetation. In fact, birds from the canopy-feeding guild were abundant in the epicormic regrowth.

Broadly, the Heathy Dry Forest bird community was not impacted by severe prescribed fire. However, there was one species impacted by severe burns. The White-throated Treecreeper reduced in occurrence in the second breeding season post-fire. Haslem et al. (2016) discuss how the severity of the most recent fire will strongly influence key resources. As the White-throated Treecreeper maintains site fidelity, it is possible their nesting and roosting sites were reduced. Haslem et al. (2016) argue that fires in Foothills Forests are not stand-replacing and that canopy species survive even severe individual bushfires and regenerate. However, as the Heathy Dry Forest research has exemplified, severe fires, even those of prescribed burns, may in fact remove sites for nesting and roosting. Nesting sites are critical to both birds and mammals displaying site fidelity (Lindenmayer et al., 2013), though most hollow-dependent birds only use hollows for nesting and hence only need them in breeding season.

Varying the severity of prescribed burns in the landscape will therefore impact birds by the degree with which nesting or roosting sites and hollows are removed. As many avian species display site fidelity, territorial birds and those flying short distances, will be the species most affected by severe prescribed burns. Furthermore, severe prescribed burns carried out in spring, may therefore not only kill young but also impact birds by completely removing the nest site. Lastly, fire size is important. In landscapes, such as Heathy Dry Forests, fragmented patches of forest may afford little vegetation close by for birds to move in to post-fire, while the burnt forest regenerates - regardless of the speed of vegetation recovery.

> ***“While it is reasonable to expect management agencies to have taken reasonable preparatory measures to limit the negative effects of large fires, it needs to be acknowledged that the capacity of management agencies to control widespread wild fires ignited by multiple lightning strikes in drought conditions on days of extreme fire danger is going to be similar to their capacity to control cyclones. If the public could allow agencies and their political masters to become a little less defensive, the climate might be right for a more rigorous examination and refinement of the ecological basis of fire management in this country.”***

Clarke, M. (2008), Catering for the needs of fauna in fire management: science or just wishful thinking? P390

Source: AAP

6 Discussion – the future for the birds in the forests of south-east Australia

6.1 Birds surviving in a changing fire regime

Fire regimes across Australia exhibit biogeographical variations (Bradstock, 2010), with differences in fire patterns largely determined by variations to rainfall (Russell-Smith et al., 2007), impacting both the biomass growth that supplies fire fuel and the dryness of the landscape, hence its capacity to burn (Bradstock, 2010). The climate in Australia is changing, in that global warming is resulting in a change to baseline temperatures, above long-term variability. In fact, Bureau of Meteorology figures released for winter 2017 in Australia highlighted that the mean maximum temperatures, recorded for Australia as a whole, were 1.9° C above the baseline 1961 to 1990 average (Bureau of Meteorology, 2017). It is predicted that there is a ninety percent chance that global temperatures will rise by 2.0° C to 4.9° C this century (Raftery et al., 2017). While these predictions have large errors, at least some level of global warming is expected, regardless of future carbon emission controls, due to the legacy from past emissions (Mauritsen and Pincus, 2017). Under warming scenarios of an increase of only 1.5° C, El Niño episodes are predicted to become twice as frequent (with five events per century increased to ten) (Cai et al., 2014). Such events lead to hot dry summers in south-eastern Australia, with increased risk of severe bushfires. Research analysing the impact of climate change on fire indicate an increase in fire frequency (Beer and Williams, 1995, Cary, 2002). Analyses of more recent results concluded with a suggested increase in temperature resulting in an increase in forest fires across much of Australia (Pitman et al., 2007). The predictions are for bushfire events to become more frequent and extreme. More ‘mega-fires’ are ‘on the horizon’.

A changing climate is resulting in complex interactions between the drivers of fire weather in the south-east Australian region. With a goal of reducing avian biodiversity decline, extensive research efforts need to continue, to increase and update the database on bird responses to fire frequency. This creates a set of challenges. One challenge is that the tolerable fire intervals required for vegetation to persist do not necessarily translate to the inter-fire interval required for avian species to remain within tolerable survival limits.

A second challenge it that the synergies created by complex climate drivers impacting the fire regime, may result in complex outcomes for birds, that may challenge interpretations. An example of this is in the recent discussions between authors interpreting mammal responses to factors of fire frequency and extent of fire, as authors discussed differing views on aspects of mammal responses to fire size (Griffiths et al., 2015a, Griffiths et al., 2015b, Russell-Smith et al., 2015). As numerous drivers impact bird responses, differing views on the reasons underpinning responses may ultimately impact fire management decisions.

As mega-fires impact on human life and assets, the management response may ultimately be one based on perceived best-benefit to people, rather than one that best serves biodiversity (Teague, 2010). However, prescribed burn area quotas, as opposed to strategic burning, are not in the best interests of protecting assets or for maintaining all levels of biodiversity. A recent shift in Victoria, from a quota-based burn strategy to a risk-based approach supports this view. Perpetuating dense new growth vegetation may in fact increase fuel load and furthermore, dense new growth vegetation stages are detrimental to the ongoing persistence of iconic species such as the Mallee Emu-wren and the Laughing Kookaburra. However, the challenge lies in best serving community when mega-fires do occur; pressure comes to bear on fire managers in that they need to be seen to be responding and reacting to fire events and doing all necessary to ameliorate impacts from future fires. Yet, some argue that there is little empirical basis for a patch-mosaic protocol in burn management and therefore, fire managers lack clear guidance on how to best optimise fire management strategies (Griffiths et al., 2015b). What is clear is that a ‘pyrodiversity begets biodiversity’ protocol benefits many Heathy Dry Forest birds but does not benefit all species. Fire management processes must view each region flagged for burning, independently. Not only do bird species have a varied range of requirements, but vegetation stands of similar species will differ markedly across Victoria, in response to differing levels of rainfall. This will result in spatial variations in bird responses.

Victoria’s Flora and Fauna Guarantee Act lists, as a threatening process: “Inappropriate fire regimes causing disruption to sustainable ecosystem processes and resultant loss of biodiversity” (Department of Environment Land Water and Planning, 2017). Therefore, Recommendation 56 (Teague, 2010), with its purpose of increasing area burnt in the landscape, could be viewed as a threatening process. There is a dilemma in this, in that there is a planning challenge, as more people move from city locations to rural and semi-rural properties. What will happen in the coming decades as increases in temperature result in more bushfires? The compromise will be surely to sacrifice parts of the landscape to repeated, regular burning, to keep fire fuel in these areas to a minimum. This may mean that we need to maintain large wooded areas where the fire regime is managed for the needs of the biota, quite separate from those areas of the landscape where the emphasis is on protecting private property.

6.2 Management of a Heathy Dry Forest landscape

Long-term planning is critical for ensuring a long-term conservation success for birds in Heathy Dry Forests. The challenge lies with incorporating climate changes, land-use changes and fire responses into any management framework.

Modelling bird responses to fire in Heathy Dry Forests showed that the response to the scale of a fire varied. Broad guild responses had more parsimonious relationships with site scale elements, as all models of best fit had site nested within the mosaic. Yet, individual species modelling highlighted that some species had more parsimonious modelling with mosaic elements, reflecting landscape scale responses. Responses were not consistent for all species within the one guild. For example, with nectarivores, the White-eared Honeyeater had more parsimonious relationships with site level and yet the Eastern Spinebill models were most parsimonious with mosaic. A landscape scale response by some birds aligns with research by others from both Eucalypt Foothills Forests (Leonard et al., 2016) and in the Central Highlands of Victoria (Lindenmayer et al., 2014), in that burned areas around sites may act as population sinks (Lindenmayer et al., 2014). Further, with high mobility and with many species having large home ranges, bird responses may be better reflected by landscape variables than site variables (Leonard et al., 2016). This highlights that responses are complex and while guild results are useful, individual species may each respond differently. To capture all responses and to manage for all species such that all persist in the landscape, a framework needs to consider as many variations as possible.

Research by Tulloch et al. (2016) outlines a management process for the best range of fire intervals for keystone plant species. A version of this approach could be adopted for managing the Heathy Dry Forest in Victoria.

The research by Tulloch et al (2016) incorporated the development of 'time-varying transition matrices' to determine whether their target vegetation species would, throughout all growth stages, survive different fire regime scenarios. In their research they created a framework of four scenarios: no burning, periodic management burning, random wildfire burning and current wildfire and periodic management burning. With these four scenarios, they modelled plant development and growth over time, with varying levels of vegetation-patch connectivity.

The responses by birds in the Heathy Dry Forest falls broadly into two groups: those that flourish in dense post-fire new growth, and those that require open stands of vegetation. These two requirements are in direct conflict. It is clear that a 'pyrodiversity begets biodiversity' premise does not suffice. The challenge is in creating a patchwork of Heathy Dry Forest such that all species persist. The challenge is made all the more complex when drought conditions prevail, bushfire becomes more frequent, and prescribed burns are undertaken right throughout the year to protect human life and assets. The patches of Heathy Dry Forest in Central Victoria are few – interspersed in a landscape that largely consists of agricultural fields and urban developments. The imperative is to create a framework that models outcomes of fire scenarios, to aid in effective management – to control and plan prescribed burning so that an adequate extent of aged, open patches remains amongst the new growth.

The results by Tulloch et al. (2016) illustrate that no single management strategy is effective in maintaining populations of all species. Instead, diversity in their work was maximised after they investigated combinations of environmental conditions and patch connectivity under different fire scenarios. So too, Heathy Dry Forest patches will benefit from a management process that attempts to frame connectivity, fire responses and climate into one framework.

6.3 The Laughing Kookaburra impacted by ‘synergistic wicked problems’

In an era where changes in global temperatures are impacting the frequency and intensity of drought, flood and fire events, there is a new level of complexity to the phrase ‘wicked problem’ - we now have, ‘synergistic wicked problems.’ These can be defined as multiple wicked problems, where the application of one or more ‘wicked problems’, compounding a ‘super wicked problem’, will create as yet unknown results, as one of the problems is driven by a changing climate. While research has already highlighted the complex effects that can be created by combining different drivers of native species decline (Thomas et al., 2006), it could be expected that synergies could result when a changing climate is one of the drivers. This may be the case for the Laughing Kookaburra.

The first ‘wicked problem’, that impacts the range and affects the abundance and occurrence of the Laughing Kookaburra, is that of ‘development’. This impacts the Laughing Kookaburra in two ways: extensive land clearing across Australian States removes the old hollows that Laughing Kookaburras require; and, a permit process in Victoria permits land-owners or developers to ‘control’ this species. Not restricted in its impacts on only the Laughing Kookaburra, land development and management will always be at odds with many native species, as more and more native vegetation is removed for urban spread. Further, management of native animals, by removal from either public or private lands, will continue to divide community opinion.

The second ‘wicked problem’ that affects the Laughing Kookaburra, impacts its level of protection by its exclusion from research and hence protection processes. The two Australian states of Western Australia and Tasmania do not consider it a native species. As its major decline is down the east coast of Australia, the Tasmanian status is of most concern. The Laughing Kookaburra was known to be introduced to the north and the north-west of the state in the early 1900’s (Green et al., 1988), and it is referred to in literature as an exotic species (Green et al., 1988, MacDonald and Kirkpatrick, 2003, Koch et al., 2008). Koch et al., (2008) reviewed threatening processes to hollow-using vertebrate fauna and stated that, as the Laughing Kookaburra is an introduced species, there was no need to review any potential threatening processes for their ongoing survival. Further, it has been suggested that possibly thousands of kookaburras have been killed in Tasmania by individuals hoping to conserve native fauna (Green et al., 1988). As the research by Green et al. (1988) was published some decades ago, this situation may no longer be occurring. The decline in this species on the mainland of Australia could require Tasmanians to review their conservation processes and include the Laughing Kookaburra in management processes as a native species. Just as efforts are made to preserve and protect the Tasmanian Devil, and in doing so they are moved to and raised in Australian locations other than Tasmania; it is possible that as a century has passed since their introduction into Tasmania, the Laughing Kookaburra does, in fact, need Tasmania's focus and protection. There is likely to be ongoing resistance, which will challenge moves to implement measures in the interest of the Laughing Kookaburra.

Finally, a 'super-wicked' problem impacts the occurrence and abundance of the Laughing Kookaburra. Abundances have been shown to decrease after prolonged periods of drought (MacNally, 2009). Further, this research highlights a reduction in Laughing Kookaburra occurrence and abundance in post-fire new growth vegetation. As results here showed, while Grey Currawongs and Laughing Kookaburras do co-exist, there is the possibility that competitive exclusion may be contributing to the decline of Laughing Kookaburras in regrowth vegetation. This requires further investigation.

Research by Opdam and Wascher (2004) discusses how combining the two key pressures of habitat fragmentation and climate change on biodiversity, will result in synergetic effects. This is largely due to the different spatial scales involved – the local range contraction of species combined with the dynamic climate aspects that occur at a landscape scale. They state: “Future biodiversity research and conservation strategies are facing the challenge to re-orient their focus and scope by integrating spatially and conceptually more dynamic aspects at the landscape level”. This is clearly the challenge for the conservation of the Laughing Kookaburra, as focus and scope needs to include a complex set of factors, which includes a dynamic climate.

Barrett et al., (2003) noted that there was no change in Laughing Kookaburra abundances from the 1977-81 atlas to the 1998-2002 atlas however, in the current State of Australia’s Birds Report (2015), a major decline is reported. Little more than a decade has passed, and the reality is that one of Australia’s most iconic species is threatened due to processes from ‘synergistic wicked problems’.

6.4 The future for bird biodiversity in Heathy Dry Forests

One percent of the abundance of Heathy Dry Forest birds comprises the uncommon species. Put into context, there are 23 species making up the 1% - approximately 41% of the total species (56) recorded. In the observations of over 10,000 birds, these uncommon birds were observed at the rate of approximately one every two hours, as opposed to the more common species that were recorded at an average rate of 45 per hour. This poses a challenge for ecologists, as recordings can be so few and collecting data for implementing meaningful conservation measures takes time.

We have data bases now that are larger and more robust, and so we have a capacity to monitor declines in what would be thought of as secure species. But these are facing new threats, with warming and drying climates and these threats bring management responses that may be largely directed at protecting people before biodiversity. It should not be a surprise that we will experience another wave of biodiversity decline in response to the drivers of the ‘Great Acceleration’. So, our birds are facing new and unpredictable risks in coming decades. If we use these data bases, aware that, like the iconic case of the Passenger Pigeon (*Ectopistes migratorius)*, bird populations can and do crash, we can be forewarned and act before populations contract to the point where their resilience is compromised. The next wave of endangered species may well be those we see commonly today. We will almost certainly lose some that comprise the 1%, at least locally.

The Heathy Dry Forest bird community responses to fire are complex. So too will be the plans to manage for the conservation of all birds in this forest. On the level of foraging guild, the omnipresent nectarivores in regrowth and dense new growth vegetation seemingly flag that a protocol of burning the landscape is of great benefit. For this guild, there is no doubt, for even the uncommon of the nine species of nectar feeders observed were found amongst the regrowth. In contrast, an entire guild of open ground foragers, such as the Australian Magpie, have a preference for the open ground around the base of trees, dense with epicormic regrowth. Then there are the tall-shrub insectivores, like the Brown Thornbill, that prefer the older, more open vegetation. Management of the Heathy Dry Forests must consider all guild responses, to ensure the persistence of all species.

There were two species that were highlighted in this body of research: the White-throated Treecreeper and the Laughing Kookaburra. Both are hollow-nesters.

Both are likely to be greatly impacted as fires increase in frequency, for the chance of remaining old-growth trees persisting decreases with each fire event. Neither bird will persist in a landscape devoid of old trees with hollows. Indeed, the White-throated Treecreeper is nest specific. It therefore becomes crucial that some of the landscape of old vegetation is protected as much as possible, and not subjected to prescribed burning, to ensure the persistence of hollow-nesting birds.

Data bases, and our scientific capacity to interpret them, provide us with the capacity to predict risks to our biodiversity. However, risk identification and appropriate landscape management requires a steady hand. The implementation of an eight percent prescribed burn recommendation after a major conflagration was not indicative of the steady hand we need to manage biodiversity, and risk to our property, in our landscapes into the future. Risks to biodiversity rely on the knowledge base, an educated society and the wisdom to implement measures that protect all assets, natural and cultural.

Plate 20 The White-eared Honeyeater, *Lichenostomus leucotis*, is the most abundant nectarivore in Heathy Dry Forests.

Source: Harvey Perkins

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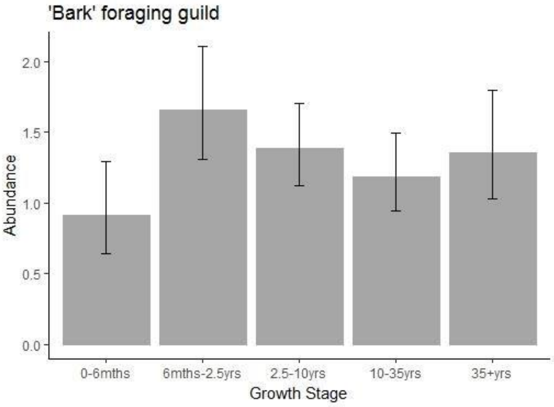
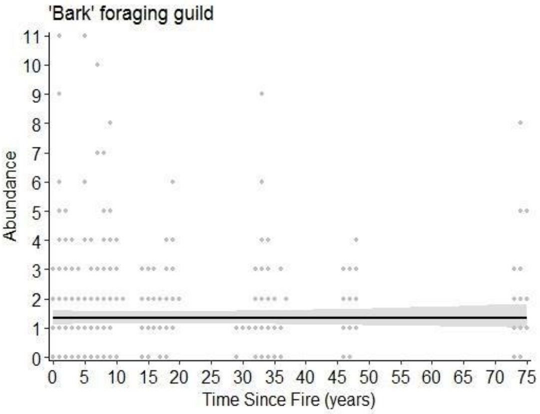
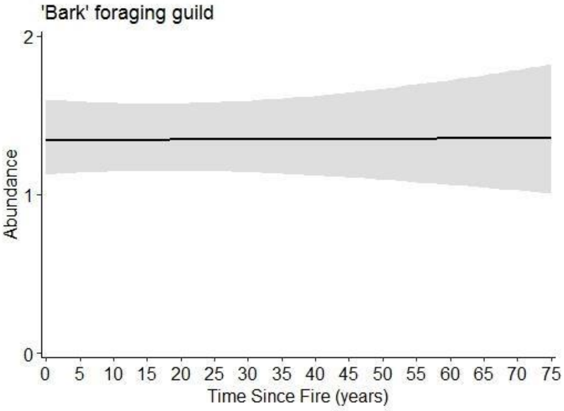
8 Appendices

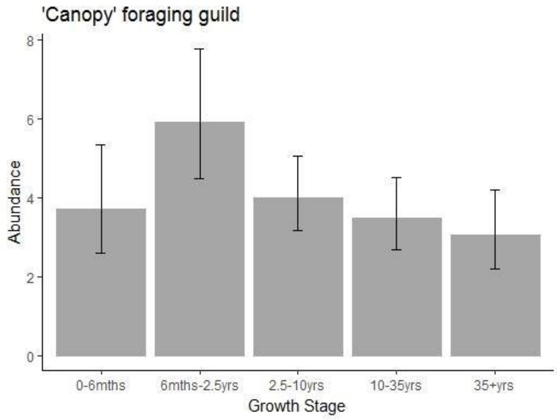
A1. Graphing outputs: a comparison between modelling techniques using vegetation age classes versus continuous age data.

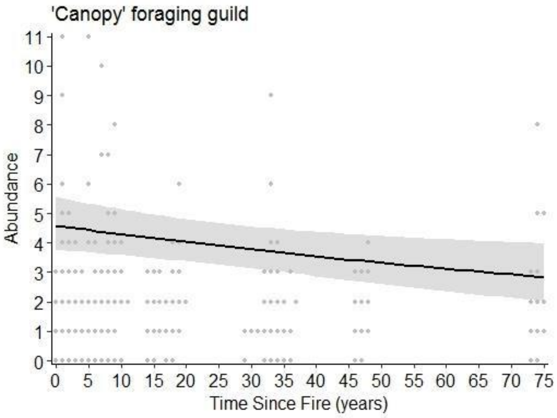
Generalized linear mixed models. Heathy Dry Forest abundance graphs for avian foraging guilds, with data presented using both continuous and categorical vegetation age measures. Continuous data is modelled twice: with and without raw data, to highlight outliers. All birds observed over eight monitoring sessions (2012-2014) on 84 sites in heathy dry forests of central Victoria are included, with the exception of two guilds: frugivores (with only two individuals observed) and water birds (with six individuals observed). Full model design carried out in the R Statistical environment (R Development Core Team, 2008). Model selection based on random effect structure (site within mosaic) and dispersion parameters and ranked using Akaike's Information Criteria (AICc). Representative models for each age measure were compared using gamm4 (Wood and Scheipl, 2014), to test for the most parsimonious model. Raw abundance data is presented on the continuous graphs.

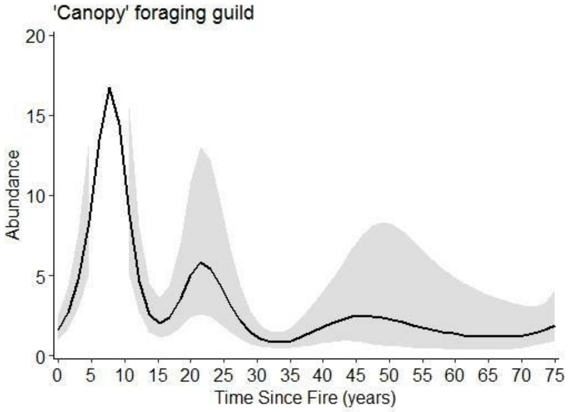


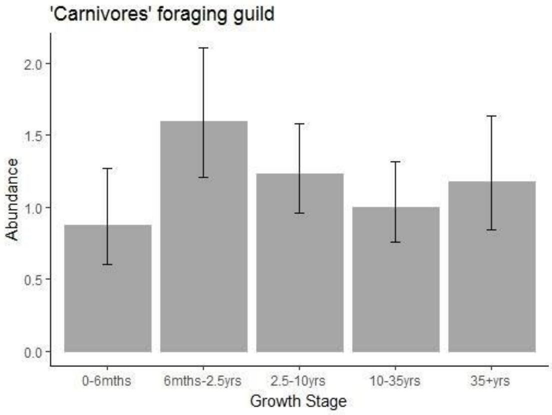


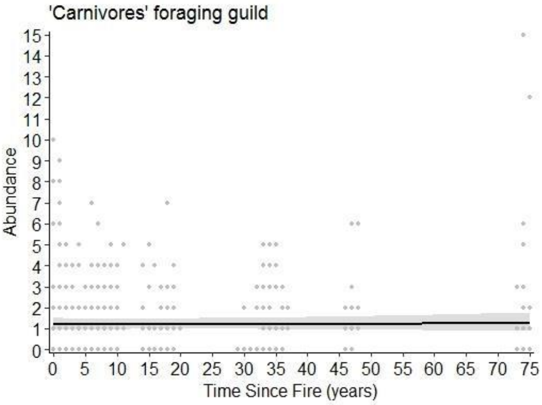


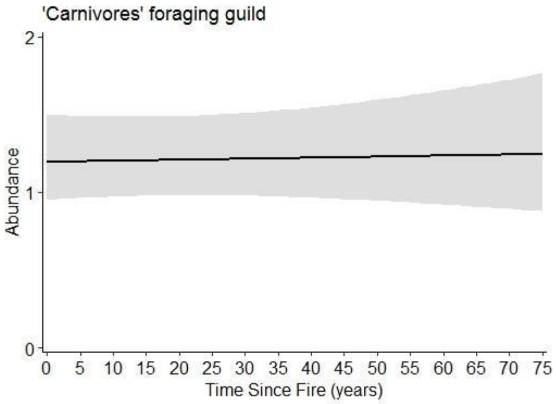

'Canopy' foraging guild
Abundance
0
2
4
6
8
0-6mths
6mths-2.5yrs
2.5-10yrs
10-35yrs
35+yrs
Growth Stage

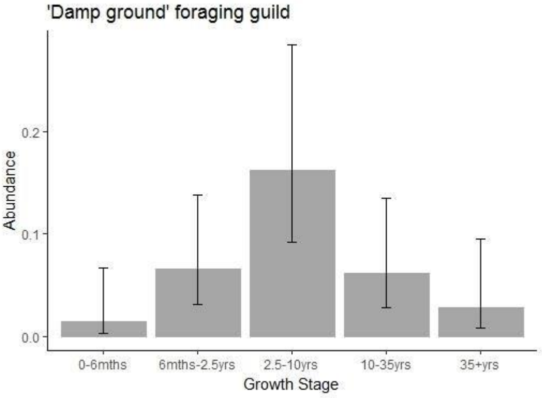

'Canopy' foraging guild
Abundance
0
1
2
3
4
5
6
7
8
9
10
11
0
5
10
15
20
25
30
35
40
45
50
55
60
65
70
75
Time Since Fire (years)

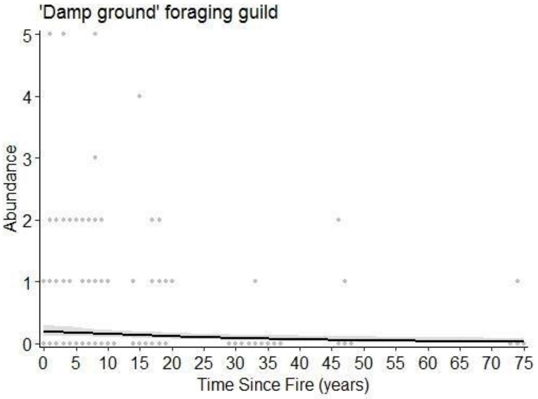

'Canopy' foraging guild
Abundance
0
5
10
15
20
0
5
10
15
20
25
30
35
40
45
50
55
60
65
70
75
Time Since Fire (years)

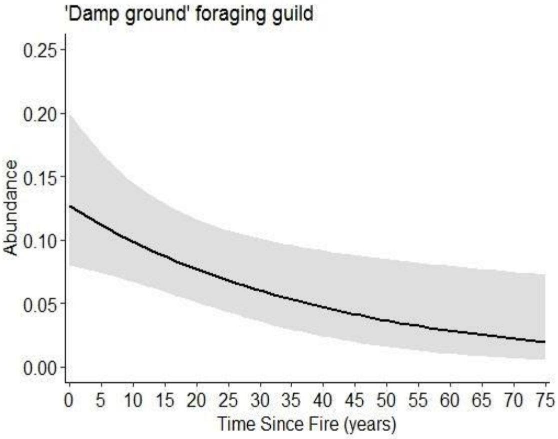

'Carnivores' foraging guild
Abundance
0.0
0.5
1.0
1.5
2.0
0-6mths
6mths-2.5yrs
2.5-10yrs
10-35yrs
35+yrs
Growth Stage

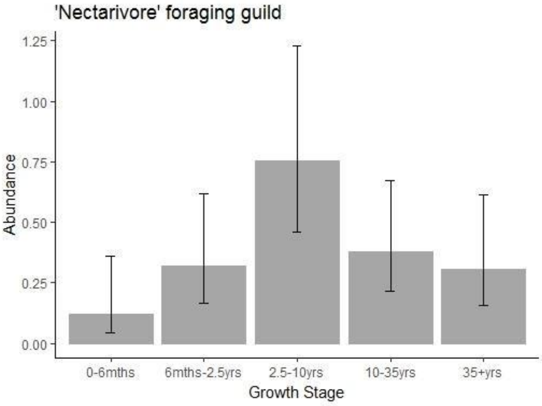

'Carnivores' foraging guild
Abundance
0
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
0
5
10
15
20
25
30
35
40
45
50
55
60
65
70
75
Time Since Fire (years)

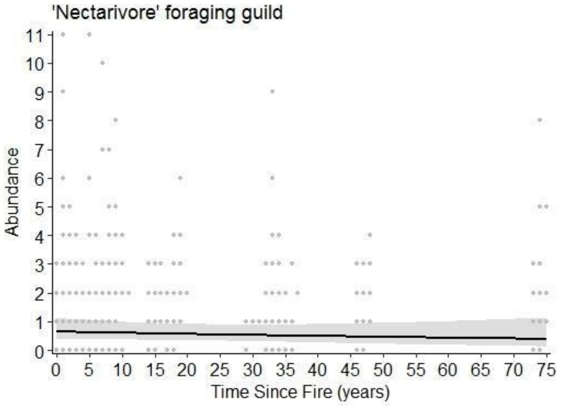

'Carnivores' foraging guild
Abundance
0
1
2
0
5
10
15
20
25
30
35
40
45
50
55
60
65
70
75
Time Since Fire (years)

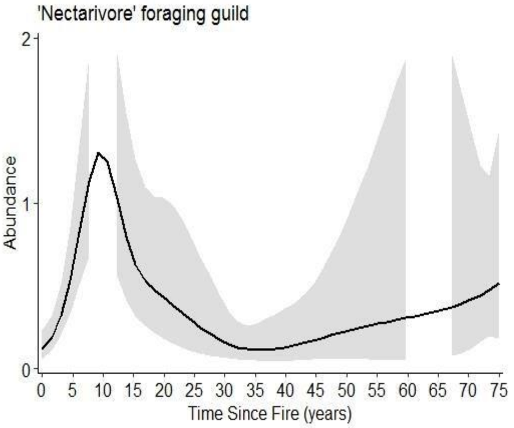

'Damp ground' foraging guild
Abundance
0.0
0.1
0.2
0-6mths
6mths-2.5yrs
2.5-10yrs
10-35yrs
35+yrs
Growth Stage

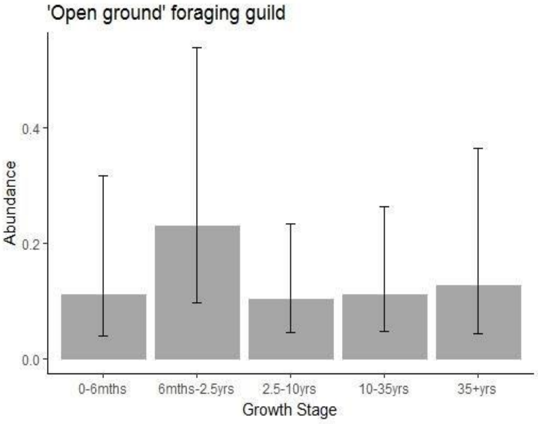

'Damp ground' foraging guild
Abundance
0
1
2
3
4
5
0 5 10 15 20 25 30 35 40 45 50 55 60 65 70 75
Time Since Fire (years)

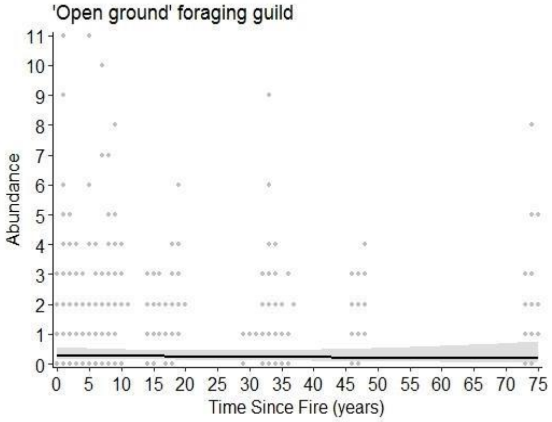

'Damp ground' foraging guild
Abundance
0.00
0.05
0.10
0.15
0.20
0.25
0 5 10 15 20 25 30 35 40 45 50 55 60 65 70 75
Time Since Fire (years)

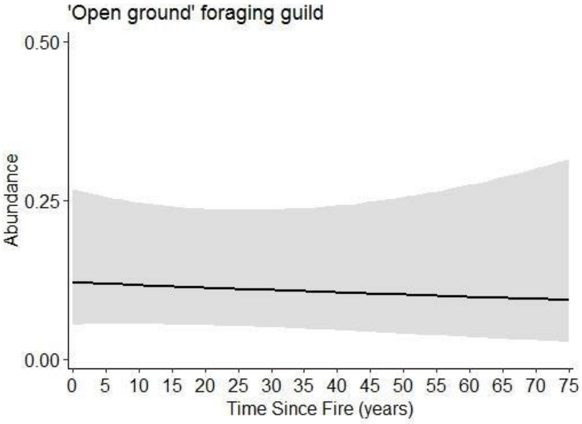

'Nectarivore' foraging guild
Abundance
0.00
0.25
0.50
0.75
1.00
1.25
0-6mths
6mths-2.5yrs
2.5-10yrs
10-35yrs
35+yrs
Growth Stage

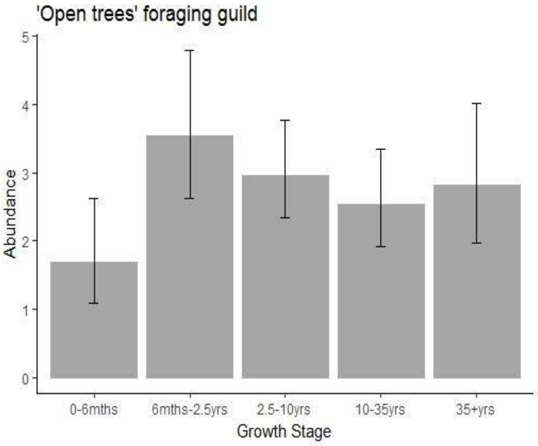

'Nectarivore' foraging guild
Abundance
0
1
2
3
4
5
6
7
8
9
10
11
0 5 10 15 20 25 30 35 40 45 50 55 60 65 70 75
Time Since Fire (years)

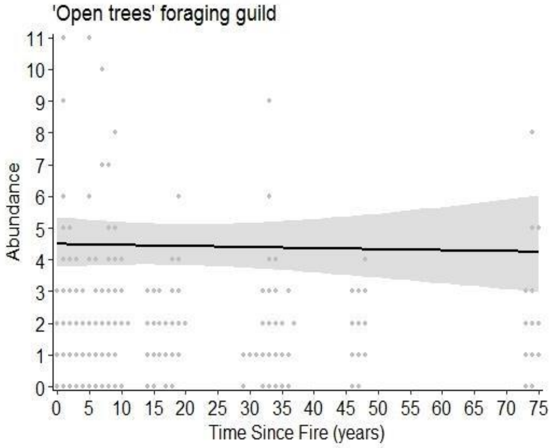

'Nectarivore' foraging guild
Abundance
0
1
2
0 5 10 15 20 25 30 35 40 45 50 55 60 65 70 75
Time Since Fire (years)

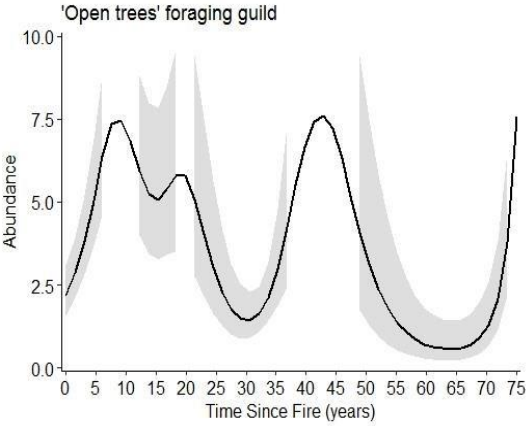

'Open ground' foraging guild

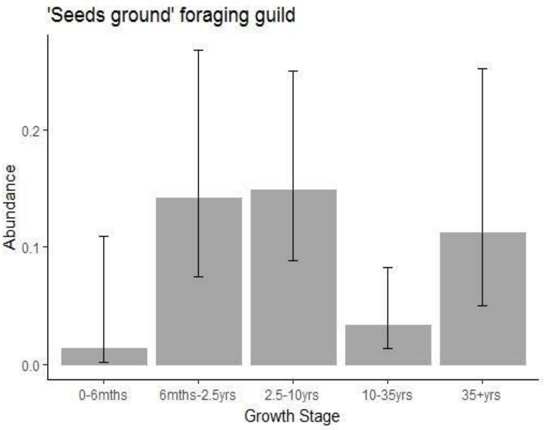

'Open ground' foraging guild


'Open ground' foraging guild

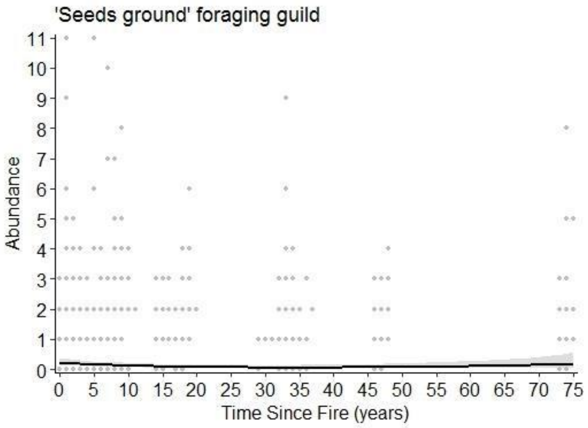

'Open trees' foraging guild


'Open trees' foraging guild

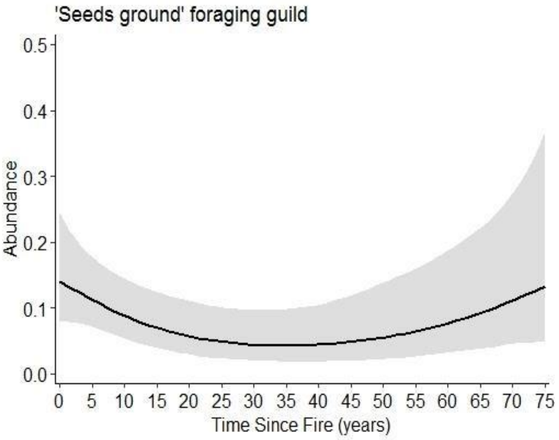

'Open trees' foraging guild



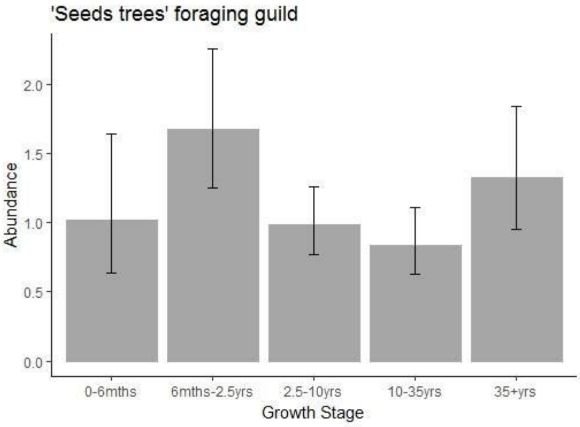
'Seeds ground' foraging guild



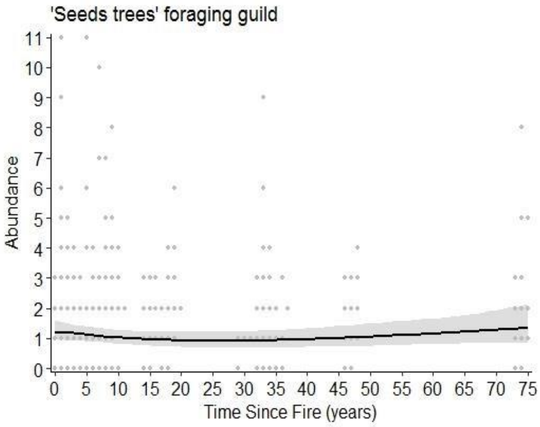
'Seeds ground' foraging guild



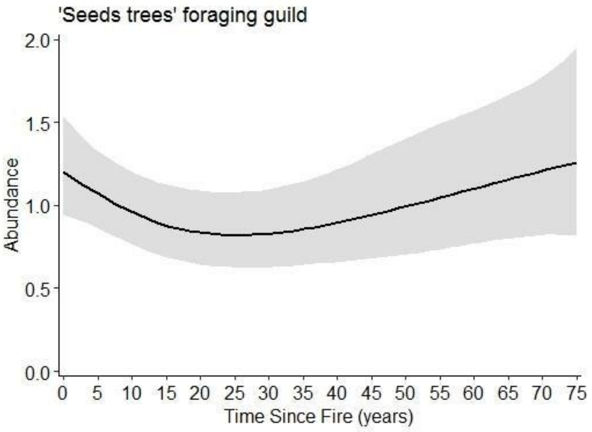
'Seeds ground' foraging guild



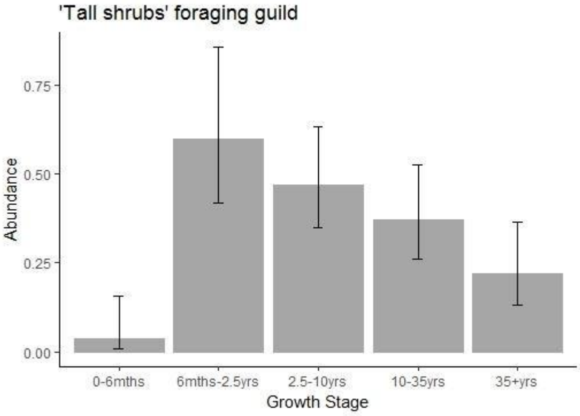
'Seeds trees' foraging guild

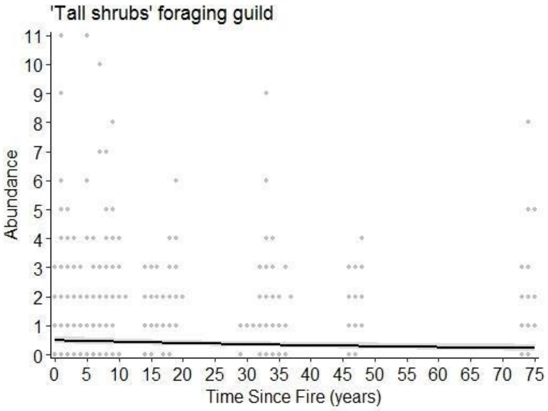


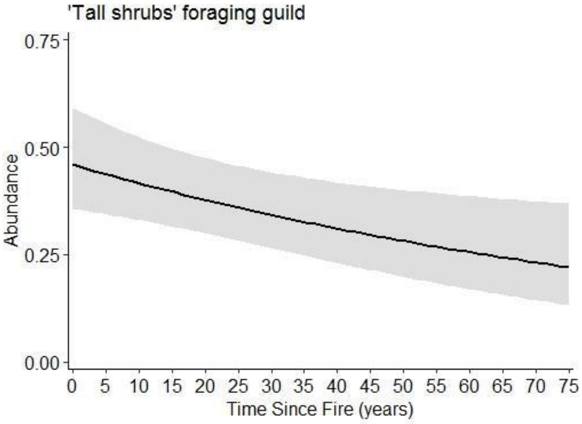
'Seeds trees' foraging guild



'Seeds trees' foraging guild


'Tall shrubs' foraging guild
Abundance
0.00
0.25
0.50
0.75
0-6mths
6mths-2.5yrs
2.5-10yrs
10-35yrs
35+yrs
Growth Stage


'Tall shrubs' foraging guild
Abundance
0
1
2
3
4
5
6
7
8
9
10
11
0 5 10 15 20 25 30 35 40 45 50 55 60 65 70 75
Time Since Fire (years)


'Tall shrubs' foraging guild
Abundance
0.00
0.25
0.50
0.75
0 5 10 15 20 25 30 35 40 45 50 55 60 65 70 75
Time Since Fire (years)

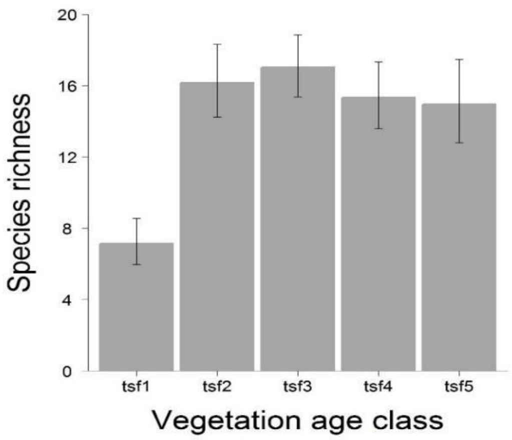
A2 Graphing outputs: community and foraging guild responses to time since fire and fire frequency.

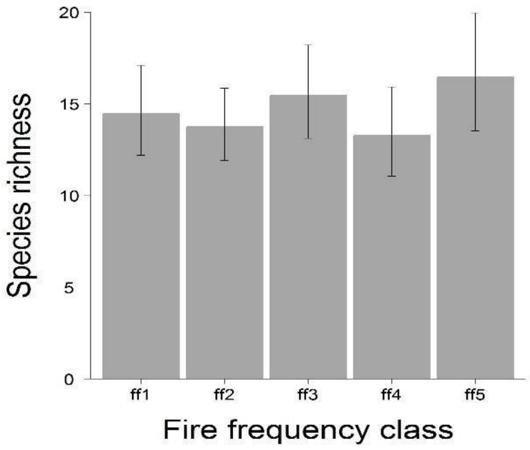
Generalized linear mixed models. Heathy dry forest abundance graphs for avian foraging guilds, with data presented using categorical vegetation age and fire frequency measures. All birds observed over eight monitoring sessions (2012-2014) on 84 sites in heathy dry forests of central Victoria are included, with the exception of two guilds: frugivores (with only two individuals observed) and water birds (with six individuals observed). Full model design carried out in the R Statistical environment (R Development Core Team, 2008). Model selection based on random effect structure of mosaic (community) or site within mosaic (guilds), and dispersion parameters, and ranked using Akaike's Information Criteria (AICc).

TSF categories: **tsf1** 0-6 months, **tsf2** 6 months – 2.5 years, **tsf3** 2.5 years to 10 years, **tsf4** 10 to 35 years, **tsf5** 35+ years, combined with last burn not recorded.

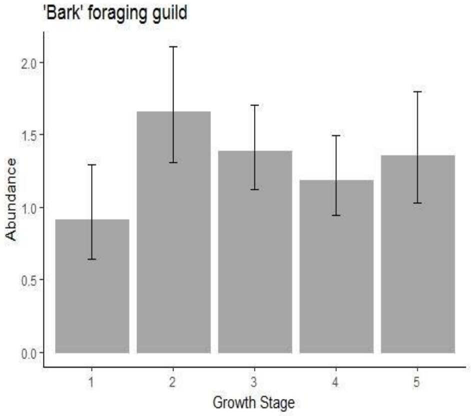
Fire frequency categories: **ff1** unburnt or burnt over 34 years ago**; ff2** one burn within last 20 years, or, one fire within last 20 years plus one fire over 45 years ago; **ff3** 2-4 fires recorded since 1939 and not breaching tolerable fire interval (TFI) of 10 years post prescribed burn or 15 years post bushfire; **ff4** 0-2 fires in the last 19 years + 1-3 fires > 20 years ago with TFI breached >22 years ago **ff5** 2-3 fires within the last 19 years with TFI breached within last 19 years**.**

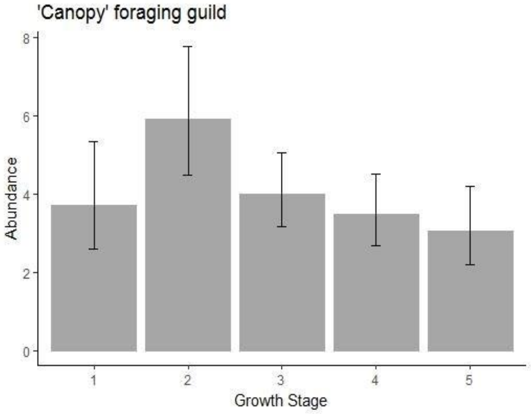
Avian community responses

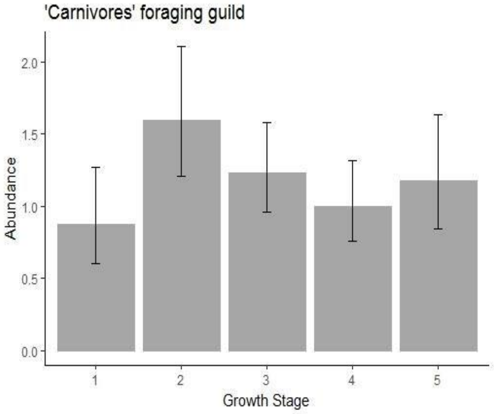


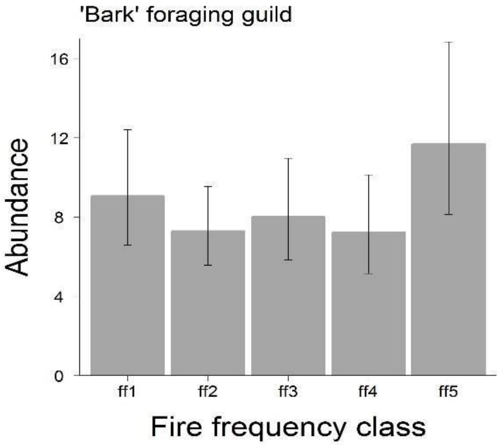


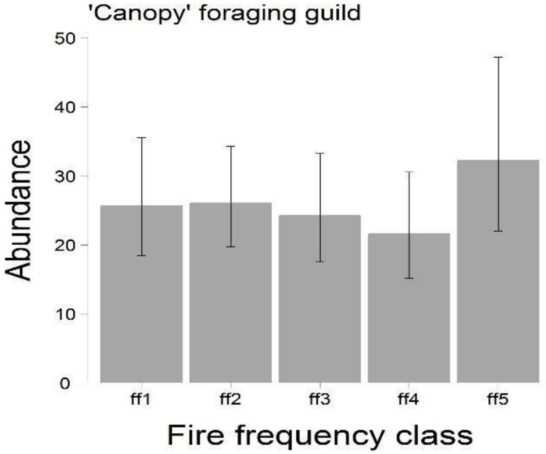
Foraging guild responses

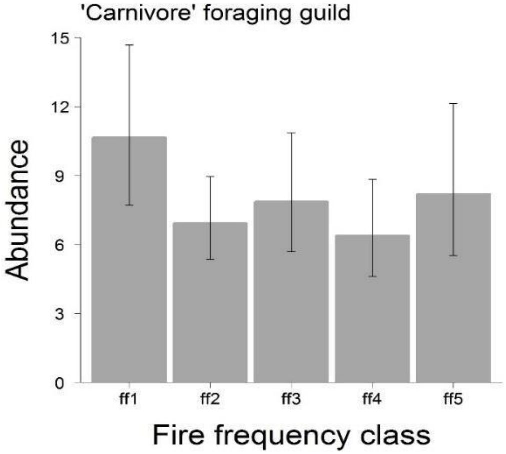

'Bark' foraging guild

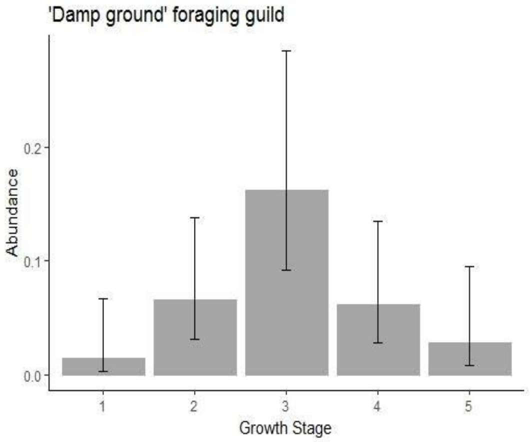

'Canopy' foraging guild


'Carnivores' foraging guild

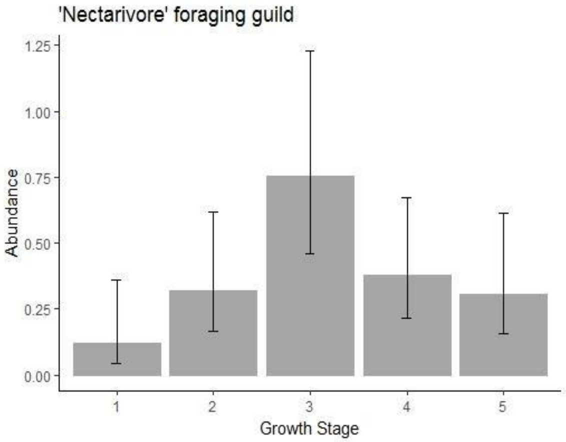

'Bark' foraging guild


'Canopy' foraging guild

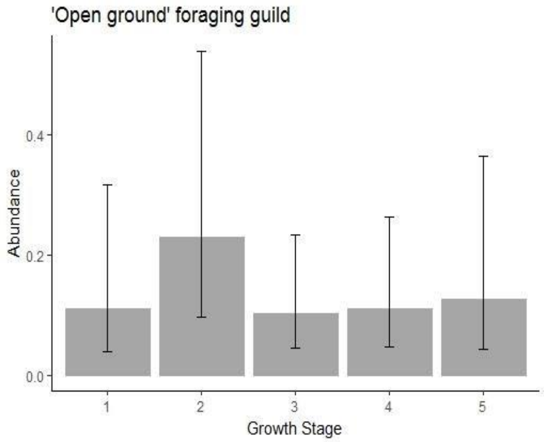

'Carnivore' foraging guild



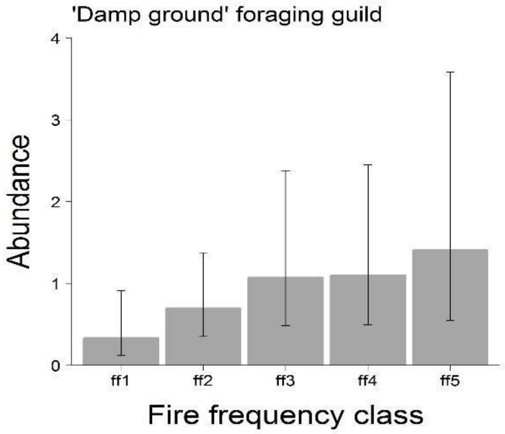
'Damp ground' foraging guild



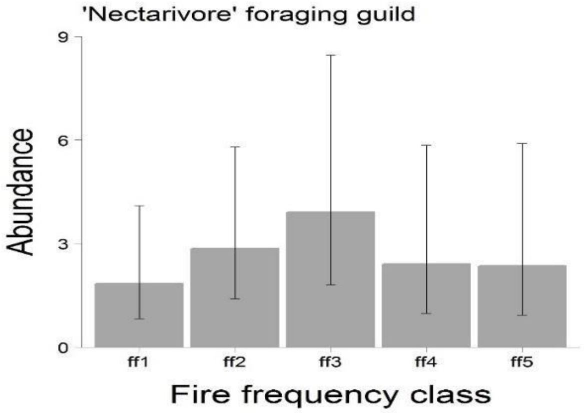
'Nectarivore' foraging guild



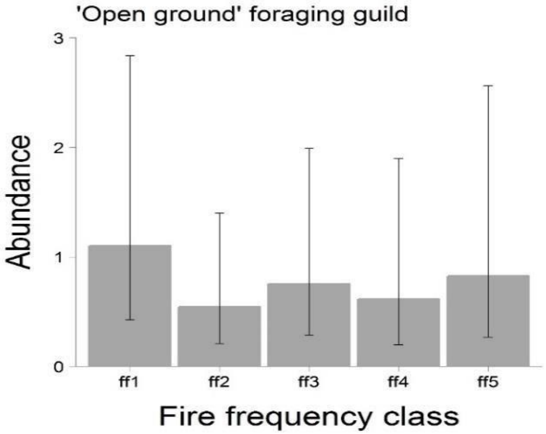
'Open ground' foraging guild



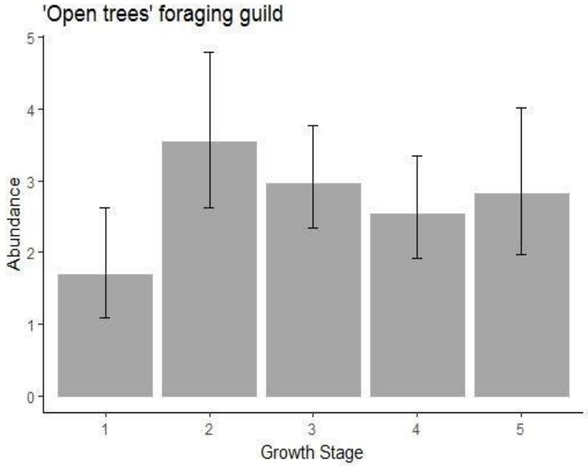
'Damp ground' foraging guild

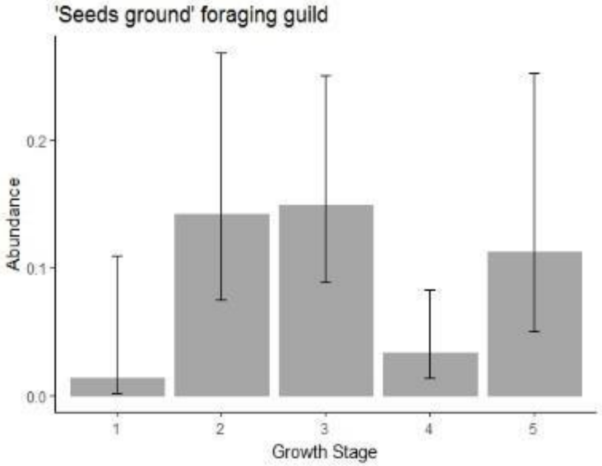


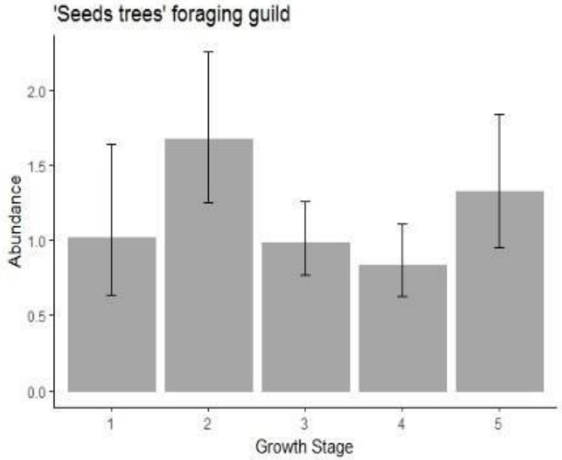
'Nectarivore' foraging guild

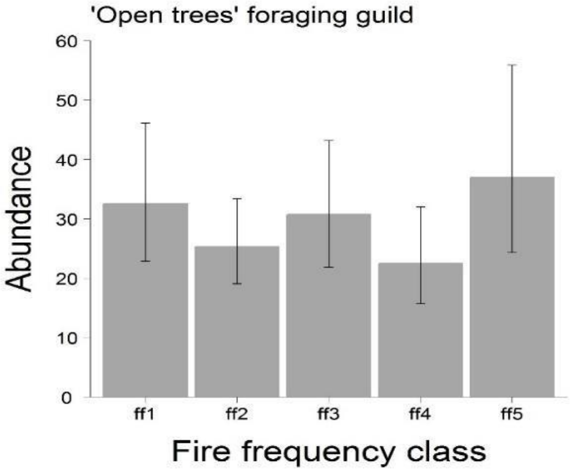


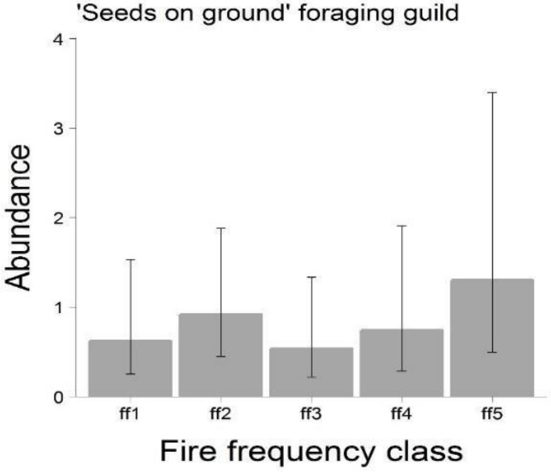
'Open ground' foraging guild

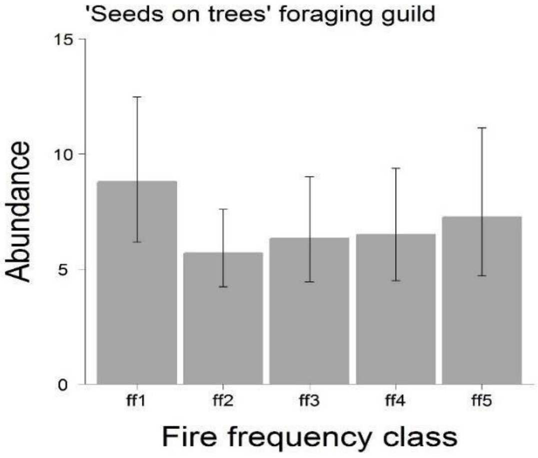

'Open trees' foraging guild
Abundance
0
1
2
3
4
5
1
2
3
4
5
Growth Stage

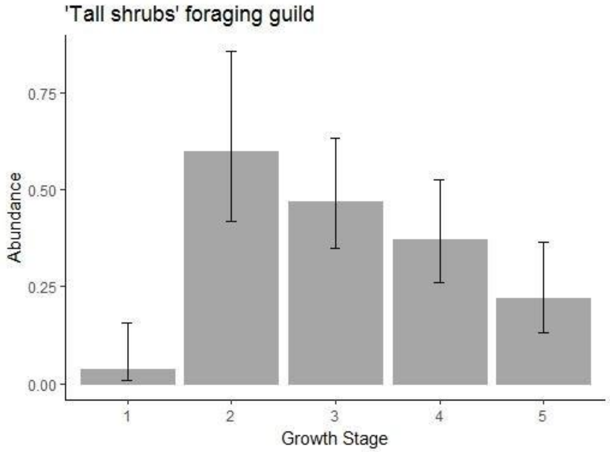

'Seeds ground' foraging guild
Abundance
0.0
0.1
0.2
1
2
3
4
5
Growth Stage

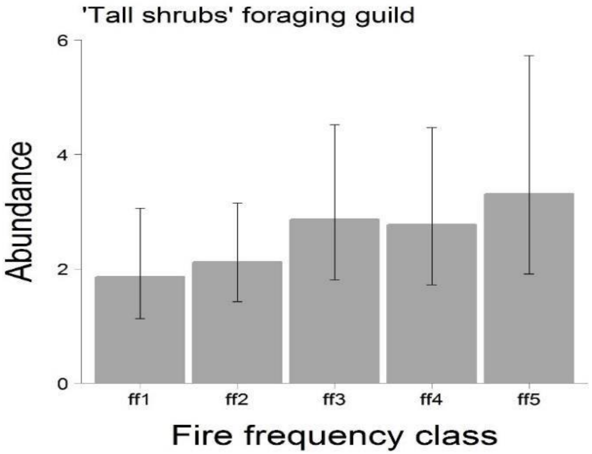

'Seeds trees' foraging guild
Abundance
0.0
0.5
1.0
1.5
2.0
1
2
3
4
5
Growth Stage


'Open trees' foraging guild
Abundance
0
10
20
30
40
50
60
ff1
ff2
ff3
ff4
ff5
Fire frequency class


'Seeds on ground' foraging guild
Abundance
0
1
2
3
4
ff1
ff2
ff3
ff4
ff5
Fire frequency class


'Seeds on trees' foraging guild
Abundance
0
5
10
15
ff1
ff2
ff3
ff4
ff5
Fire frequency class


'Tall shrubs' foraging guild
Abundance
0.75
0.50
0.25
0.00
1
2
3
4
5
Growth Stage


'Tall shrubs' foraging guild
Abundance
6
4
2
0
ff1
ff2
ff3
ff4
ff5
Fire frequency class

A3 Graphing outputs: individual species’ responses to time since fire and fire frequency.

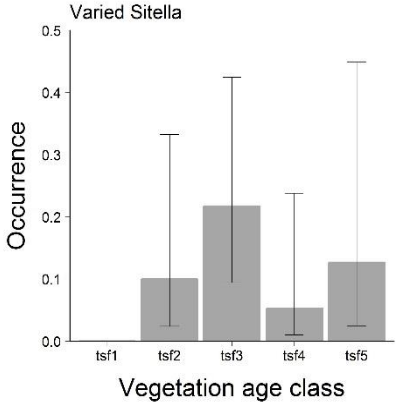
Generalized linear mixed models. Heathy Dry Forest abundance graphs for birds identified as ‘common’ (contributing to approximately 80% of assemblage patterns) and ‘moderately common’ (remaining birds in the community that were observed on >10% of all sites). Graphs are presented using categorical vegetation age of fire frequency categories. Full model design carried out in the R Statistical environment (R Development Core Team, 2008). Model selection based on random effect structure (RE) and dispersion parameters (resulting in some species presented as abundance and some as occurrence) and ranked using Akaike’s Information Criteria (AICc). Categories with no value indicate an absence of the species in that category. A confidence interval on a binomial distribution from zero to one indicates that the species was found on every site in that category.

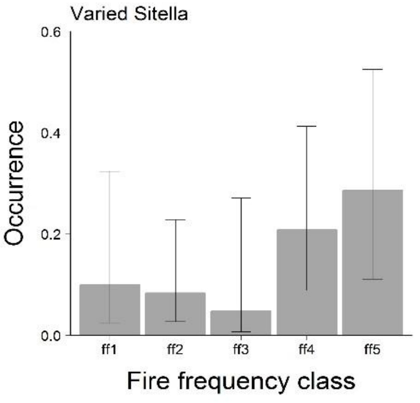
TSF categories: **tsf1** 0-6 months, **tsf2** 6 months – 2.5 years, **tsf3** 2.5 years to 10 years, **tsf4** 10 to 35 years, **tsf5** 35+ years, combined with last burn not recorded.
Fire frequency categories: **ff1** unburnt or burnt over 34 years ago**; ff2** one burn within last 20 years, or, one fire within last 20 years plus one fire over 45 years ago; **ff3** 2-4 fires recorded since 1939 and not breaching tolerable fire interval (TFI) of 10 years post prescribed burn or 15 years post bushfire; **ff4** 0-2 fires in the last 19 years + 1-3 fires > 20 years ago with TFI breached >22 years ago **ff5** 2-3 fires within the last 19 years with TFI breached within last 19 years**.**

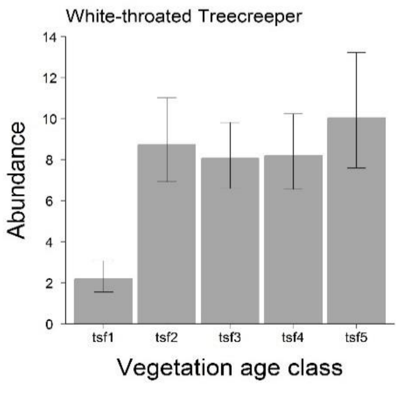
Spotted Quail-thrush for fire frequency and Golden Whistler for both TSF, were both too low in number to model.

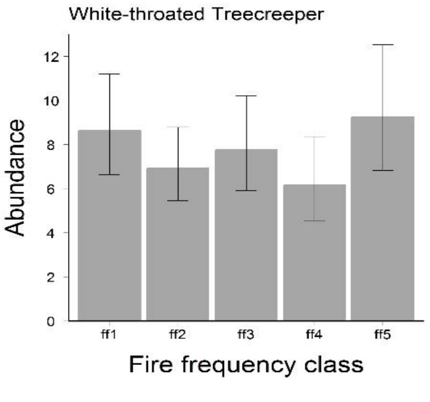
‘Bark’ foraging guild.

RE - Varied Sitella tsf: mosaic/site ff: mosaic; White-throated Treecreeper tsf and ff: mosaic/site



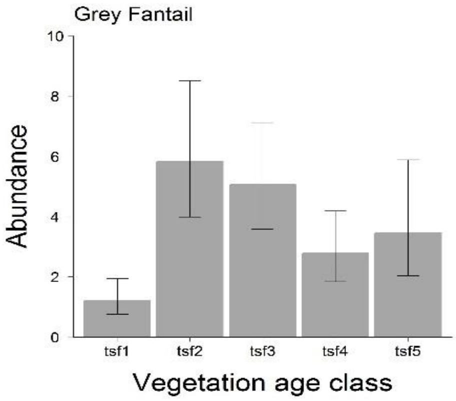


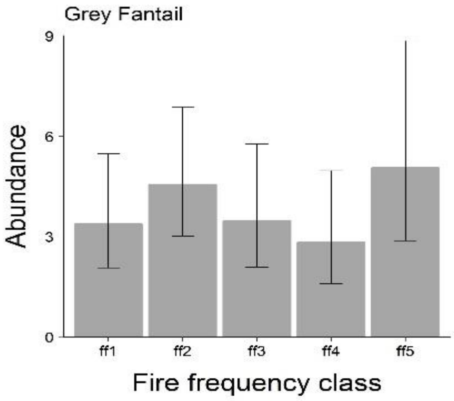


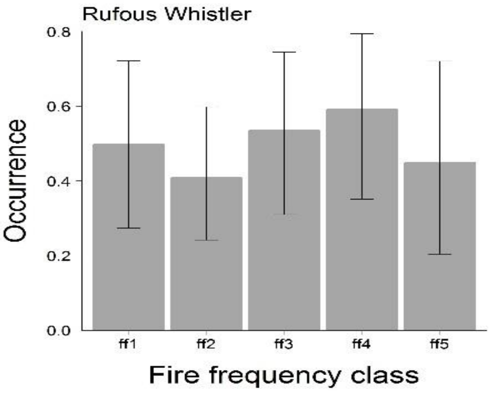


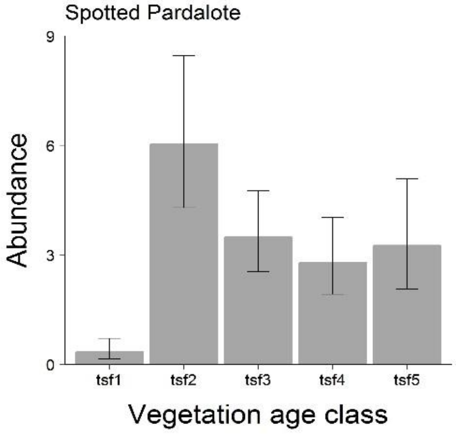
‘Canopy’ foraging guild

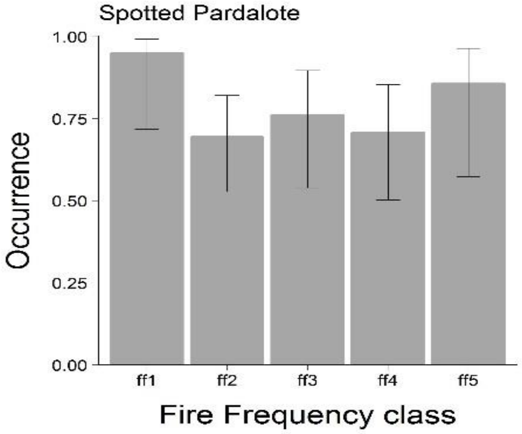
RE – Grey Fantail tsf and ff: mosaic/site; Rufous Whistler tsf and ff: mosaic; Spotted Pardalote tsf: mosaic/site ff: mosaic; Striated Pardalote tsf and ff: mosaic; Striated Thornbill tsf: mosaic/site, ff: mosaic

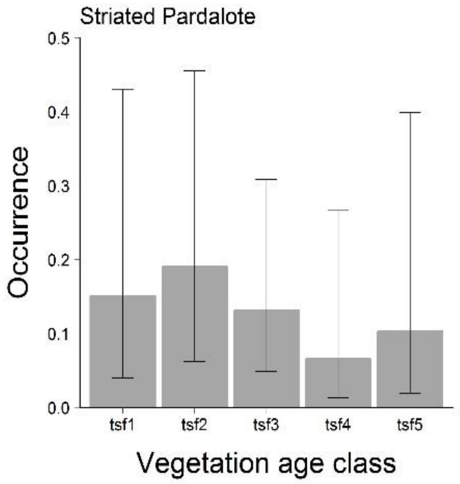


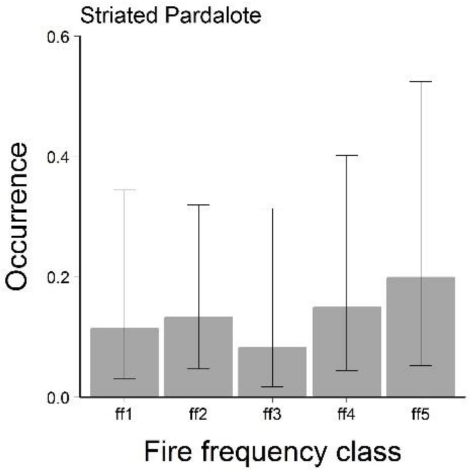


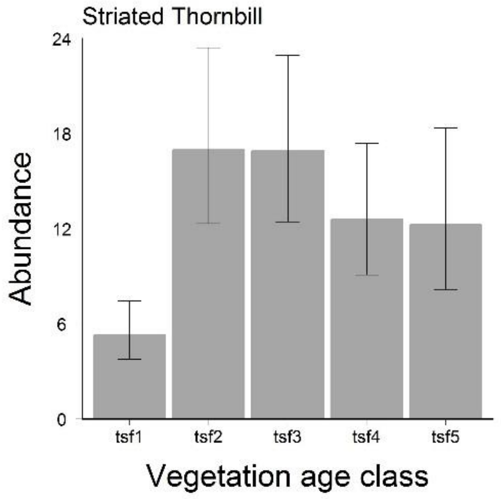


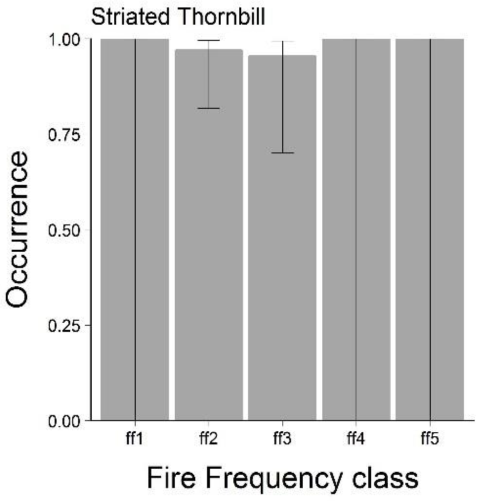






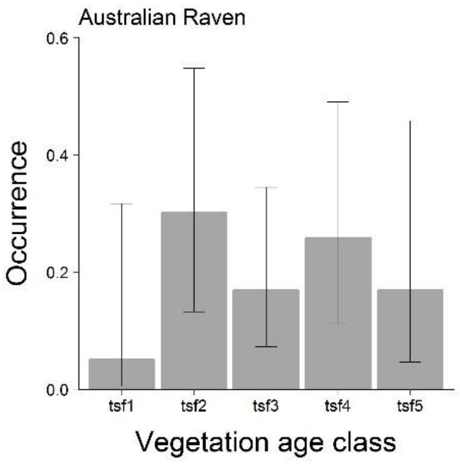


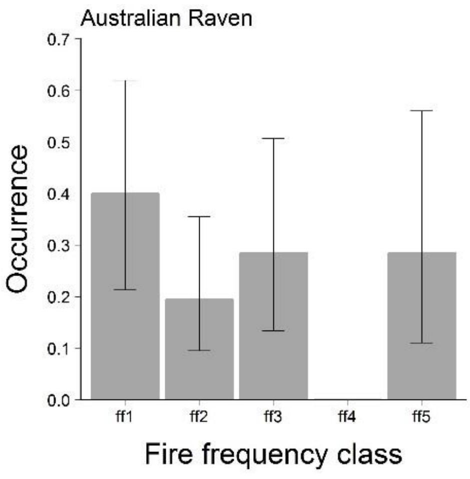


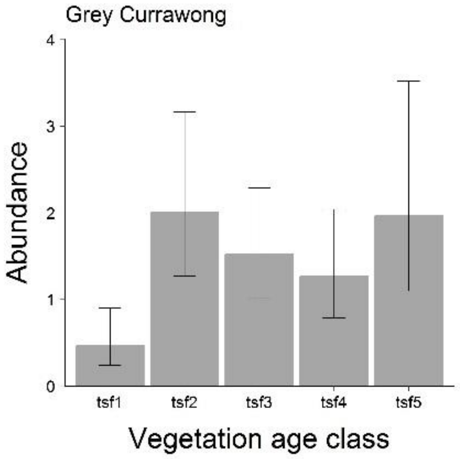


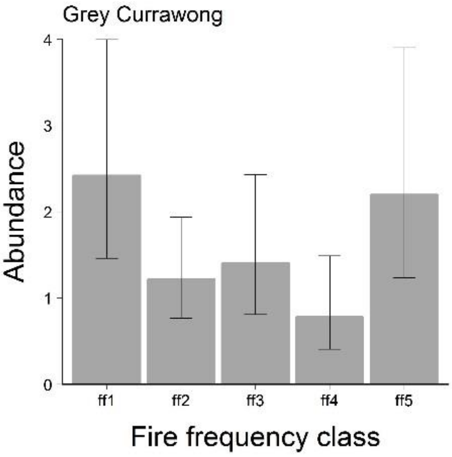
‘Carnivore’ foraging guild

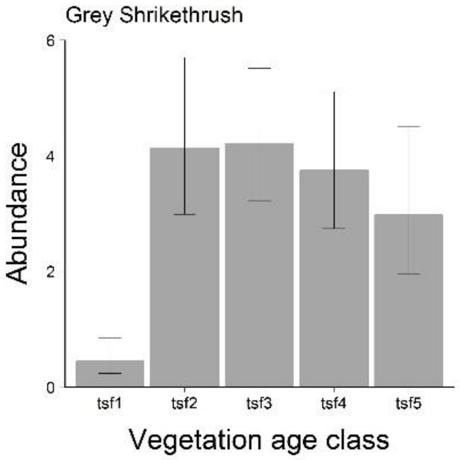
RE - Australian Raven tsf and ff: mosaic; Grey Currawong, Grey Shrikethrush and Laughing Kookaburra tsf and ff: mosaic/site; Pied Currawong tsf and ff: mosaic

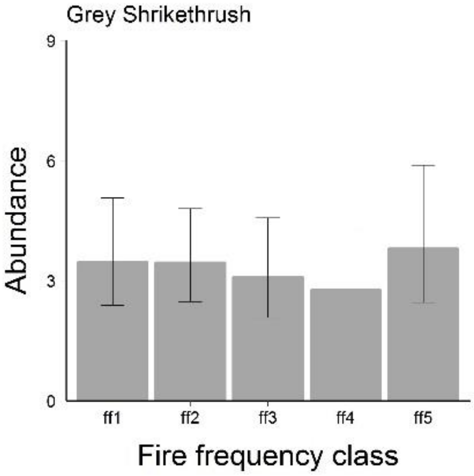


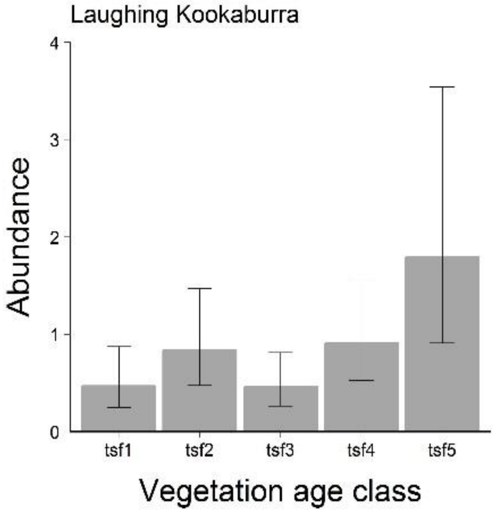


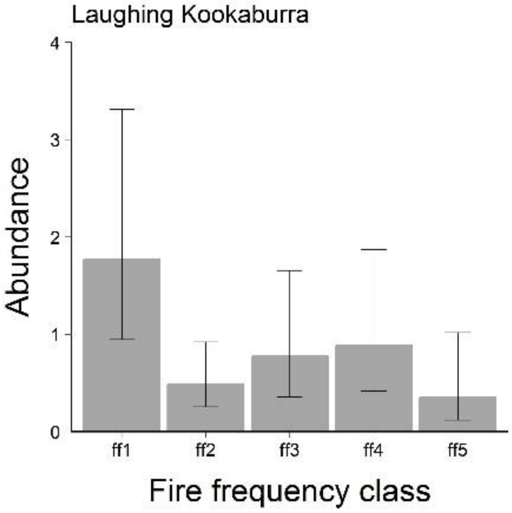

Grey Currawong
Abundance
0
1
2
3
4
tsf1
tsf2
tsf3
tsf4
tsf5
Vegetation age class

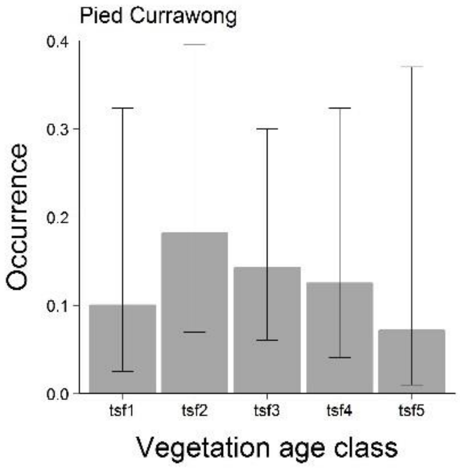

Grey Currawong
Abundance
0
1
2
3
4
ff1
ff2
ff3
ff4
ff5
Fire frequency class

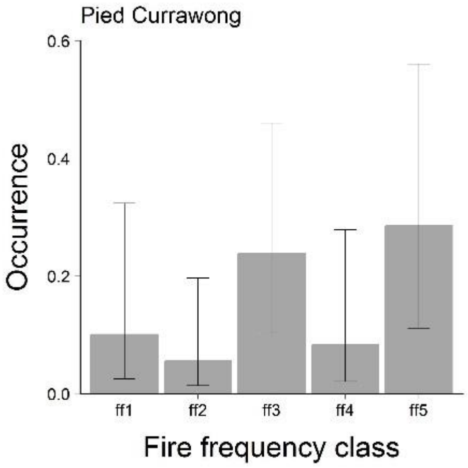

Grey Shrikethrush
Abundance
0
2
4
6
tsf1
tsf2
tsf3
tsf4
tsf5
Vegetation age class


Grey Shrikethrush
Abundance
0
3
6
9
ff1
ff2
ff3
ff4
ff5
Fire frequency class


Laughing Kookaburra
Abundance
0
1
2
3
4
tsf1
tsf2
tsf3
tsf4
tsf5
Vegetation age class

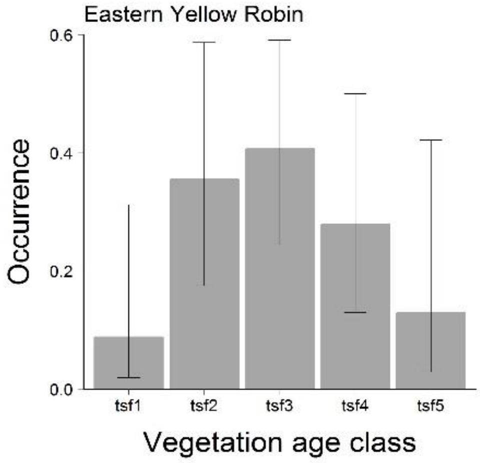

Laughing Kookaburra
Abundance
0
1
2
3
4
ff1
ff2
ff3
ff4
ff5
Fire frequency class

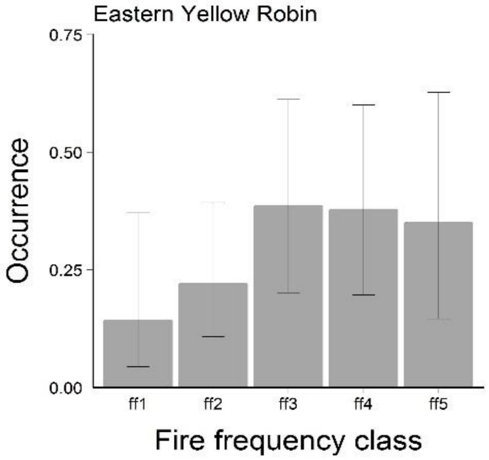


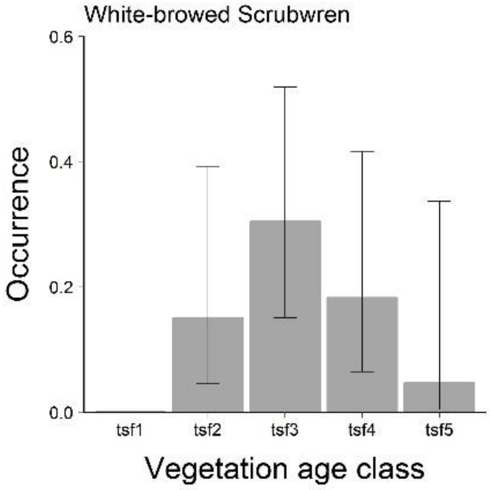


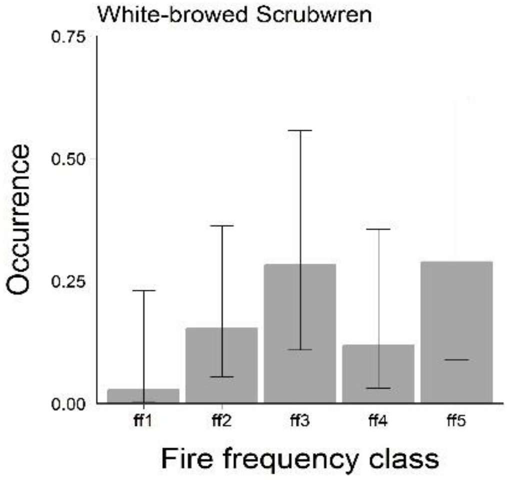
‘Damp ground’ foraging guild

RE – Eastern Yellow Robin, White-browed Scrubwren tsf and ff: mosaic



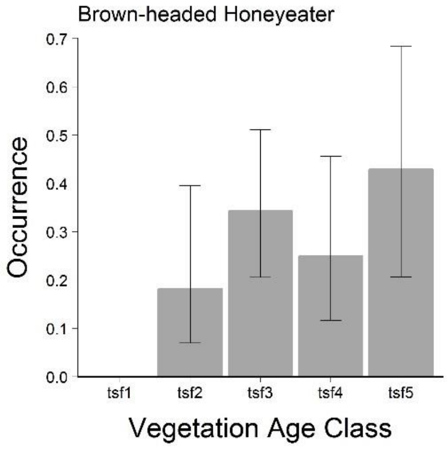


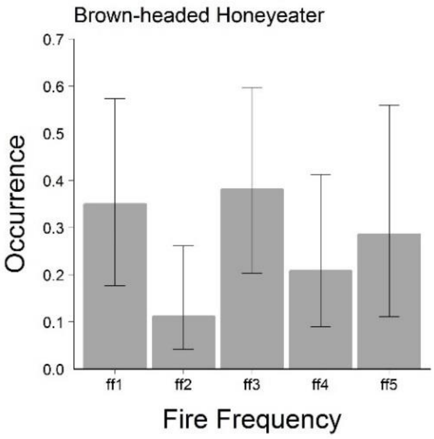


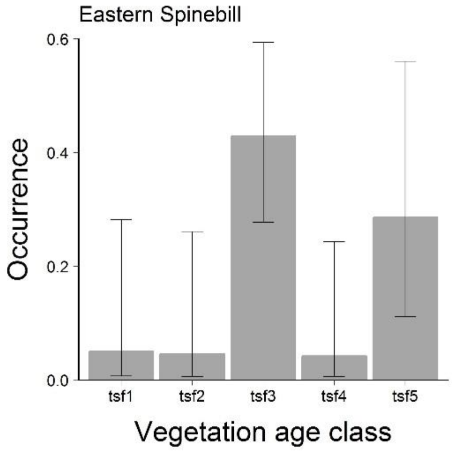


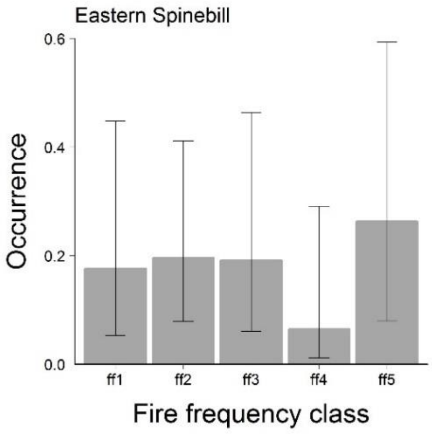
‘Nectarivore’ foraging guild

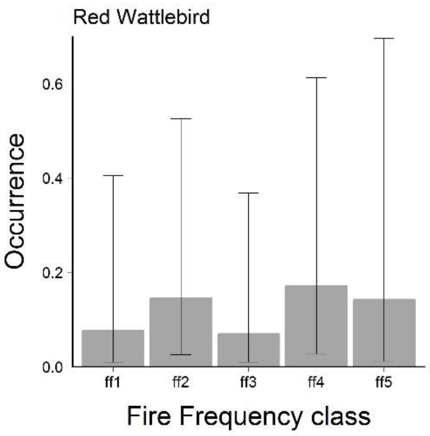
RE – Brown- headed Honeyeater, Eastern Spinebill, Red Wattlebird, White-naped Honeyeater and Yellow-faced Honeyeater tsf and ff: mosaic; White-eared Honeyeater tsf: mosaic/site ff: mosaic

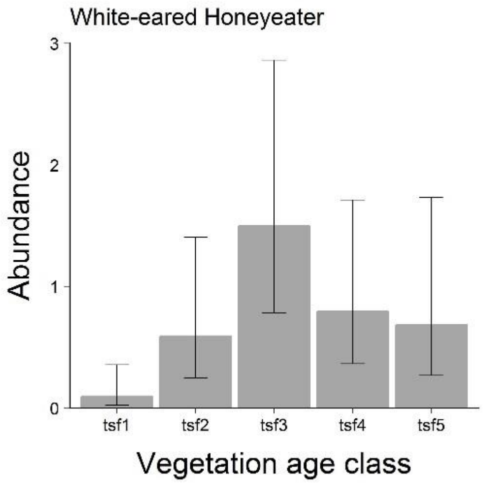


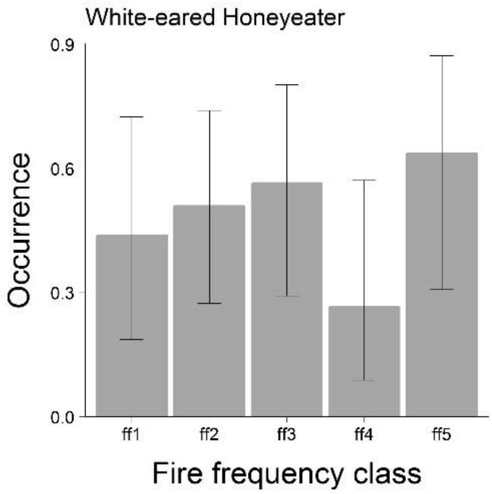


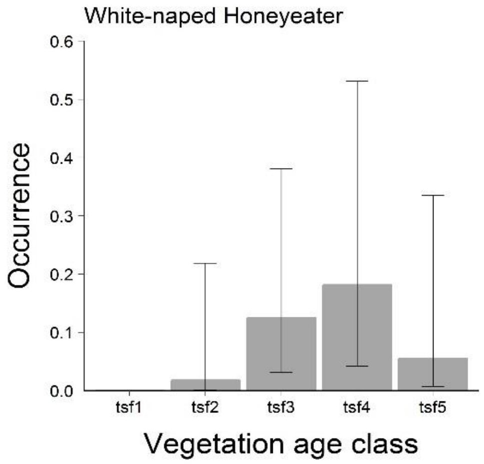


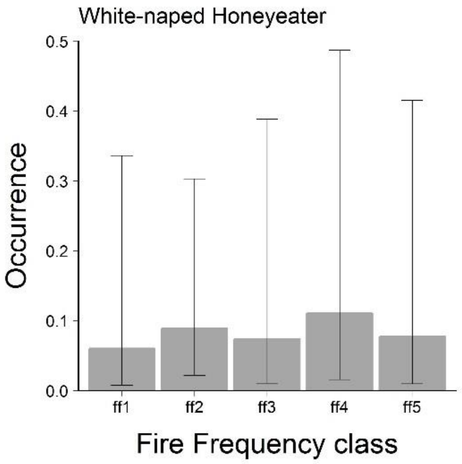


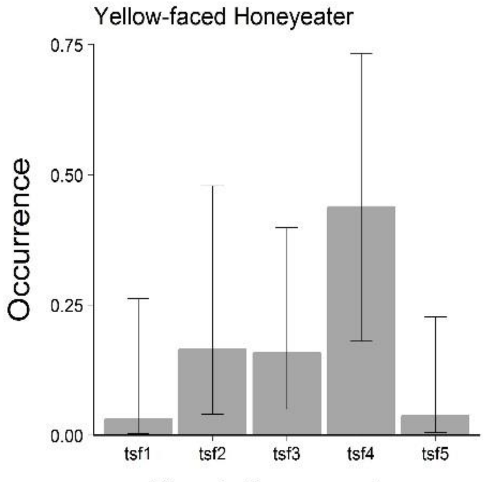


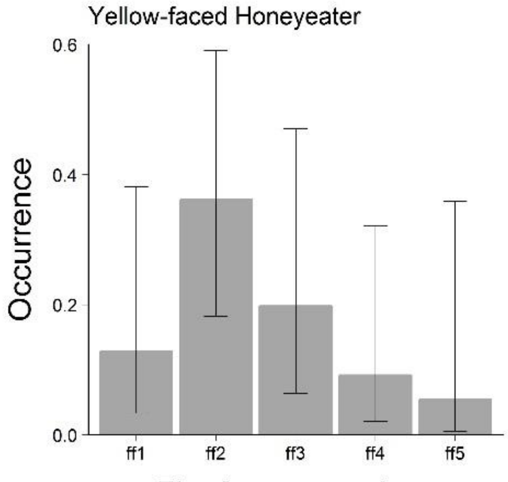

White-eared Honeyeater
Abundance
0
1
2
3
tsf1
tsf2
tsf3
tsf4
tsf5
Vegetation age class


White-eared Honeyeater
Occurrence
0.0
0.3
0.6
0.9
ff1
ff2
ff3
ff4
ff5
Fire frequency class


White-naped Honeyeater
Occurrence
0.0
0.1
0.2
0.3
0.4
0.5
0.6
tsf1
tsf2
tsf3
tsf4
tsf5
Vegetation age class

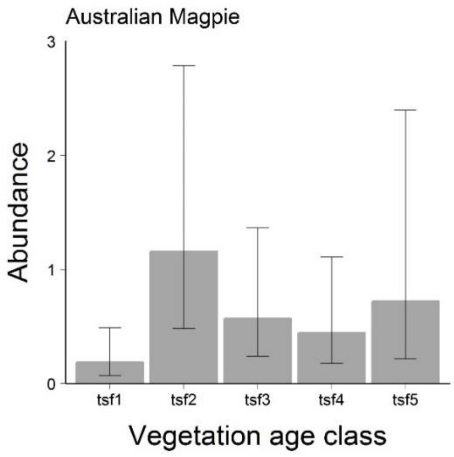

White-naped Honeyeater
Occurrence
0.0
0.1
0.2
0.3
0.4
0.5
ff1
ff2
ff3
ff4
ff5
Fire Frequency class

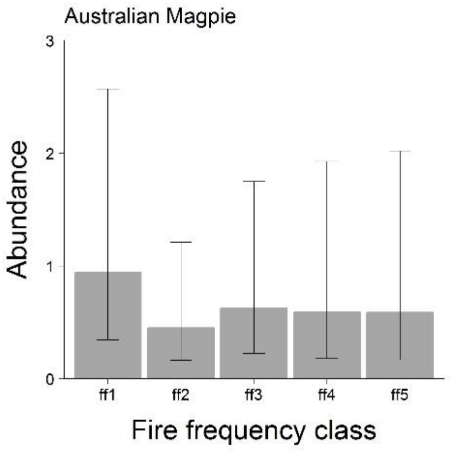

Yellow-faced Honeyeater
Occurrence
0.00
0.25
0.50
0.75
tsf1
tsf2
tsf3
tsf4
tsf5
Vegetation age class


Yellow-faced Honeyeater
Occurrence
0.0
0.2
0.4
0.6
ff1
ff2
ff3
ff4
ff5
Fire frequency class

‘Open ground’ foraging guild

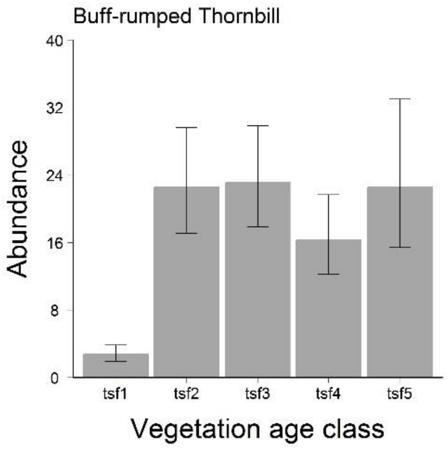
RE – Australian Magpie tsf and ff: mosaic/site

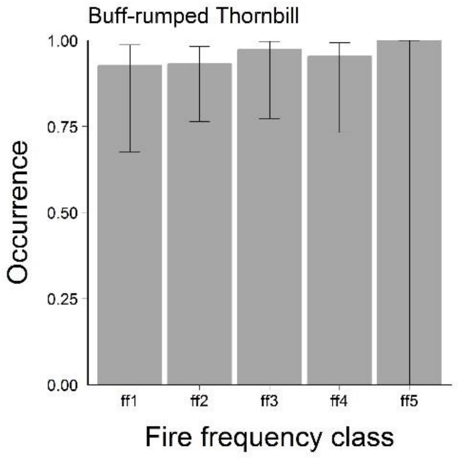


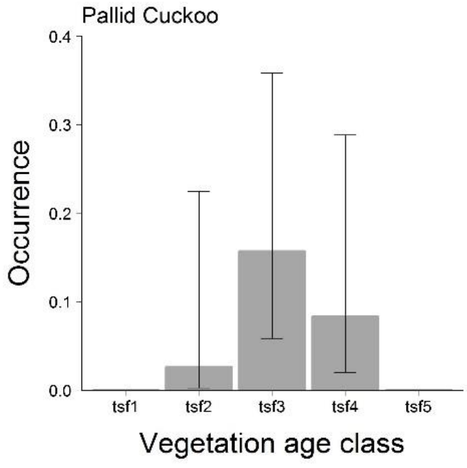


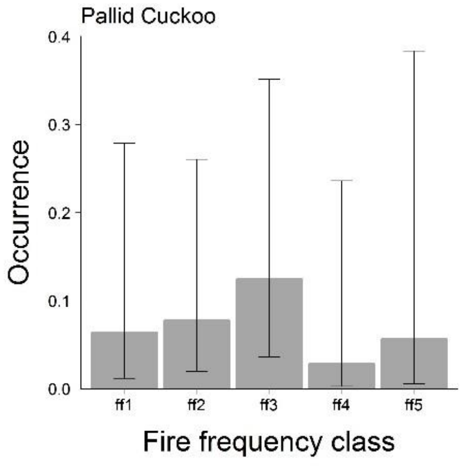
‘Open trees’ foraging guild

RE – Buff-rumped Thornbill and White-winged Chough tsf: mosaic/site ff: mosaic; Pallid Cuckoo and Superb Fairywren tsf and ff: mosaic; Scarlet Robin tsf and ff: mosaic/site; Spotted Quail-thrush tsf: mosaic

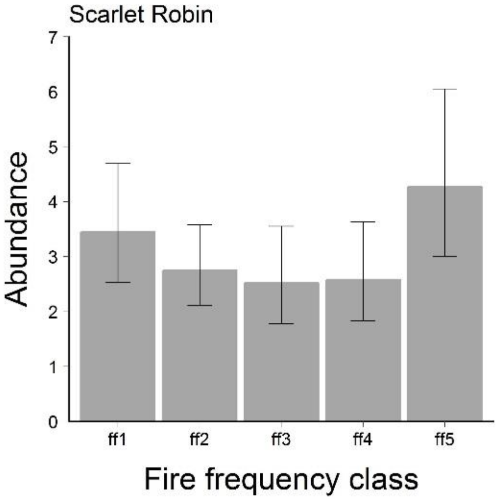


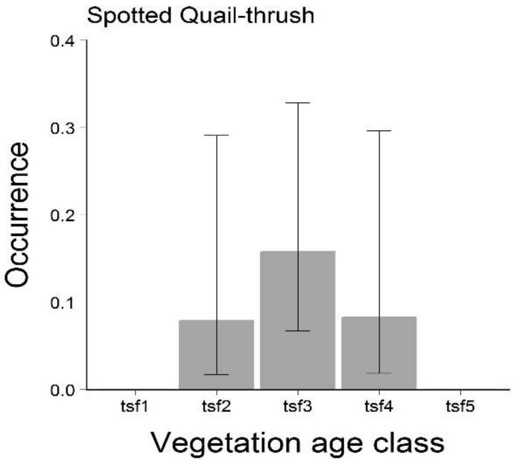


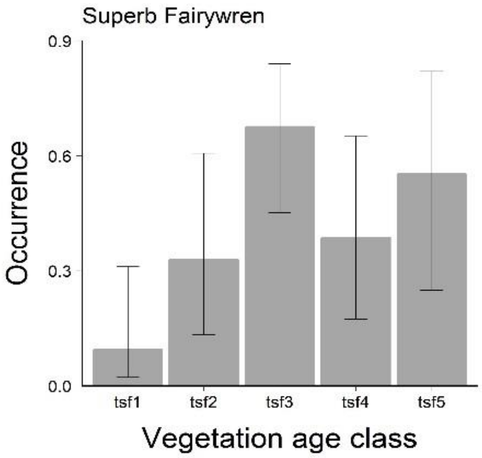


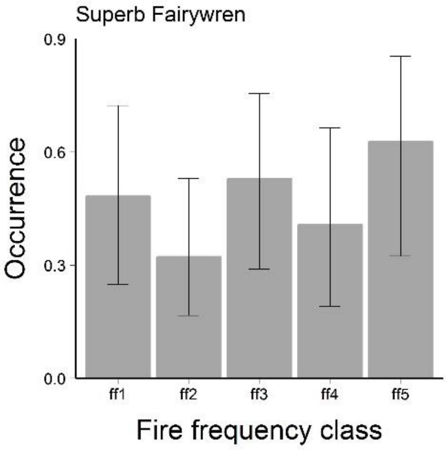


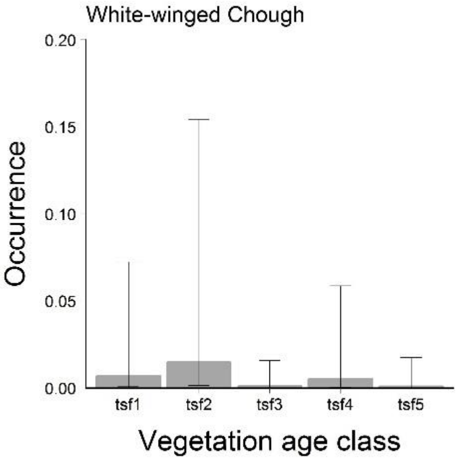

Scarlet Robin
Abundance
0
1
2
3
4
5
6
7
tsf1
tsf2
tsf3
tsf4
tsf5
Vegetation age class

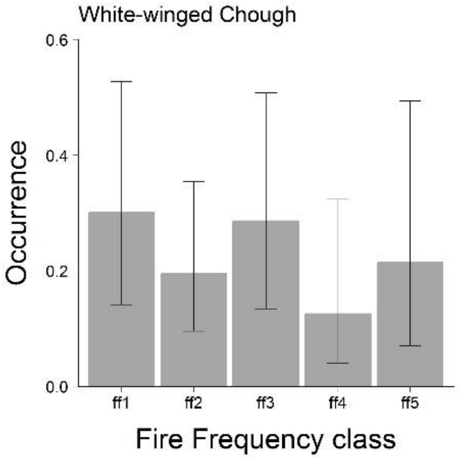

Scarlet Robin
Abundance
0
1
2
3
4
5
6
7
ff1
ff2
ff3
ff4
ff5
Fire frequency class


Spotted Quail-thrush
Occurrence
0.0
0.1
0.2
0.3
0.4
tsf1
tsf2
tsf3
tsf4
tsf5
Vegetation age class


Superb Fairywren
Occurrence
0.0
0.3
0.6
0.9
tsf1
tsf2
tsf3
tsf4
tsf5
Vegetation age class

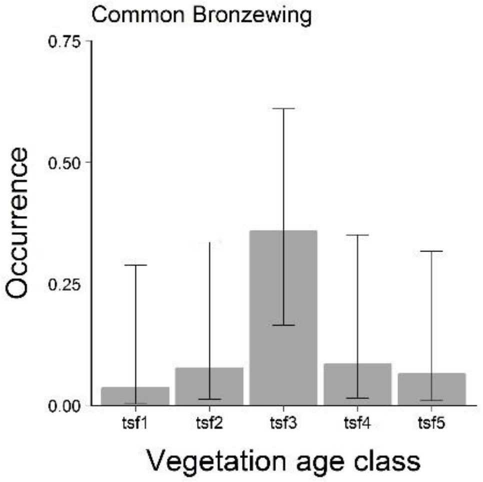

Superb Fairywren
Occurrence
0.0
0.3
0.6
0.9
ff1
ff2
ff3
ff4
ff5
Fire frequency class

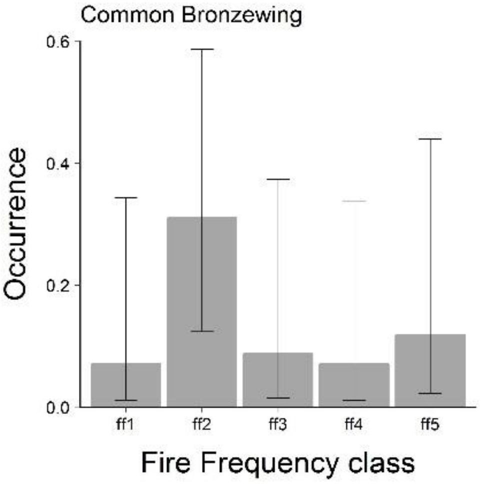


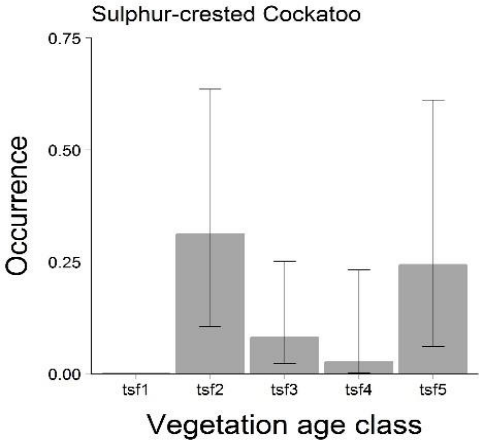


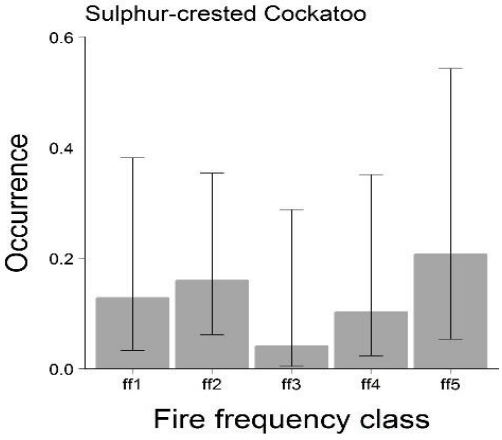
'Seeds on ground' foraging guild

RE – Common Bronzewing and Sulphur-crested Cockatoo tsf and ff: mosaic



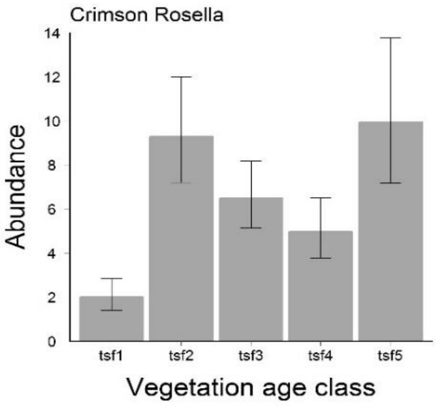


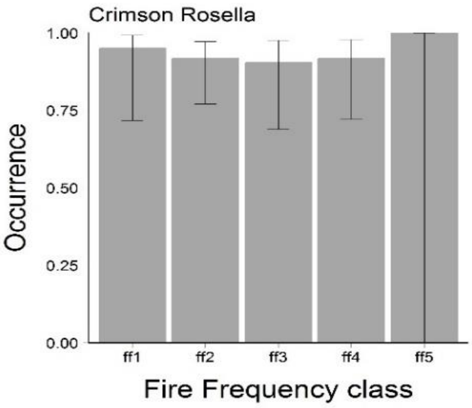




‘Seeds on trees’ foraging guild

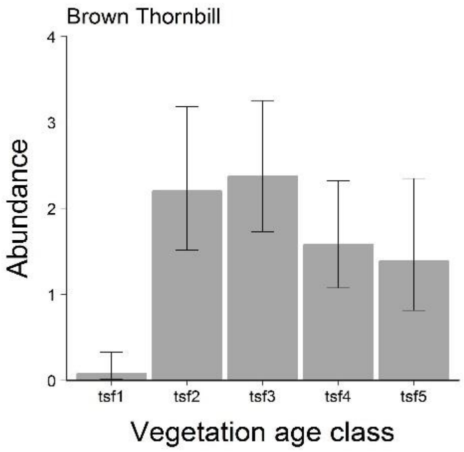
RE – Crimson Rosella tsf: mosaic/site ff: mosaic

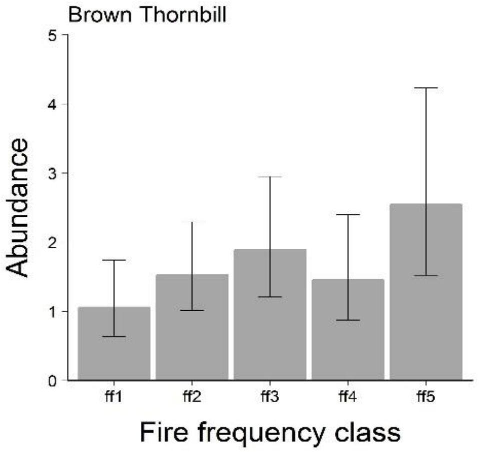


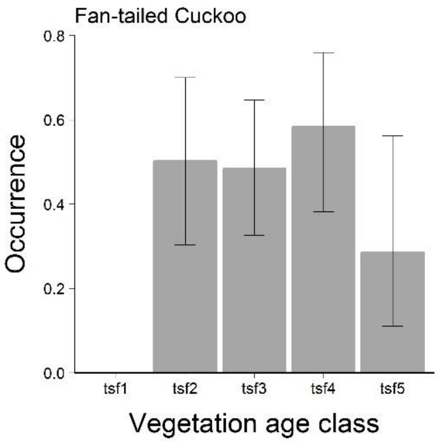


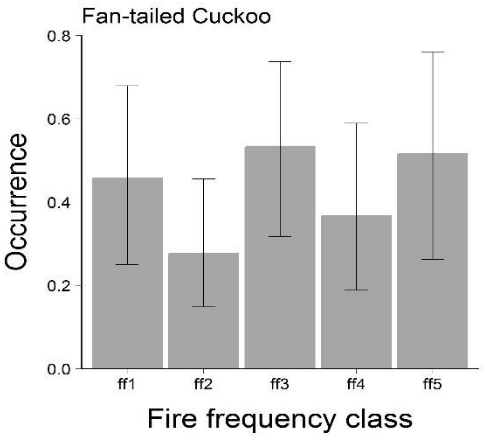
‘Tall shrubs’ foraging guild

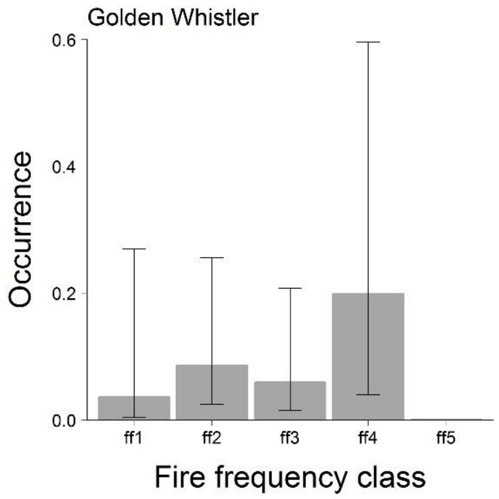
RE – Brown Thornbill tsf: mosaic ff: mosaic/site; Fan-tailed Cuckoo tsf and ff: mosaic; Golden Whistler ff: mosaic










Golden Whistler
0.6
0.4
0.2
0.0
Occurrence
ff1
ff2
ff3
ff4
ff5
Fire frequency class