

# Foothills Fire and Biota

Fire and Adaptive Management Report no. 96



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# Acknowledgements

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## Executive summary

Fire is a key driver of ecological structure and function in many ecosystems. Fires can also threaten human life and property. Planned burning is commonly used to reduce the risk of bushfire and can also be used to achieve ecological outcomes. However, in many instances the ecological effects of planned burning are poorly understood. As a result managers often lack the information required to formulate planned burning strategies that are likely to meet ecological aims. This is particularly the case for the Foothills Forest ecosystem in Victoria, where ecological complexity is a product of variation in environmental factors and fire regimes. The high flammability of the vegetation, proximity to human settlements and the role of fire as an agent of ecosystem renewal make sound fire management planning both highly complex and crucially important in this system.

In 2012, the Department of Environment, Land, Water and Planning (DELWP) commissioned a consortium of researchers from the Arthur Rylah Institute, Deakin University, La Trobe University and the University of Melbourne to investigate relationships of biota to fire and other drivers within Foothills Forest. The overarching aim of the Foothills Fire and Biota (FFB) project was to provide models of the relationships of the biota to fire history and landscape patterns that can be used to assist in predicting biodiversity outcomes of bushfire and planned burns in Foothills Forest.

'Foothills Forest' comprises what is commonly referred to as 'messmate-peppermint' or 'mixed-species' forests. Foothills Forest forms an extensive contiguous band on both the northern and southern lower-mid slopes of the Great Dividing Range and also occurs in the Otways region.

Analysis of fire history within Foothills Forest indicated that approximately 42% of the system has experienced fire since 2000, while around 10% of the area has been burnt at below the average minimum tolerable fire interval for foothills Ecological Vegetation Divisions (EVDs; approximately 14 years).

Data on the biota of Foothills Forest were drawn from existing data sets held by DELWP and the consortium institutions. Collectively, the data sets provided large sample sizes for flora, vegetation structure, birds and mammals. Combining the data sets provided greater analytical power, but also entailed some disadvantages, including increased 'noise' due to methodological variation in data collection, loss of information due to abundance data being converted to presence/absence, and the consequent need for more complex analyses. Most significantly, collating and managing the resultant large data sets proved a major task. Working with these data sets provided some lessons for any future large scale monitoring programs. A robust monitoring program requires sound survey design, which flows from carefully framed questions that address the processes of interest. Our study also highlighted: the importance of good data management and curation; careful consideration of predictor variables; and the need for targeted surveys to capture data on rare or threatened species.

We examined responses of vegetation structure, floristic composition, bird abundance (within guilds,) and mammals, to fire regime attributes and a range of environmental variables. Time since fire was the fire regime attribute most consistently related to biota, although vegetation structure, birds and some mammals also exhibited relationships with fire type (bushfire/planned burn). The majority of bird guilds were responsive to fire frequency and landscape patterns of fire regimes, specifically, showing negative relationships with increasing proportion of the landscape recently burnt by bushfire. In many instances, variables representing climate, topography and vegetation type had as much or greater influence on the biota than fire regime.

Comparison of the relative influence of fire regime variables on species grouped by taxonomic category (birds, mammals, plants) showed that time since fire and inter-fire interval were both, on average, the most influential on species occurrence. Inter-fire interval tended to be more influential on plants than animals. Fire type (bushfire versus planned burn; a surrogate for severity) generally had little influence on species occurrence.

Species relationships to time since fire were examined in relation to the currently defined growth stages for Foothills EVDs. For the growth stages for which data was available, the timing of transitions in species occurrences approximated current growth stages. We found no evidence suggesting alternative, more appropriate, growth stage definitions.

Optimisation of geometric mean abundance (GMA) was used to identify the proportional distribution of Foothills EVD growth stages that maximises biodiversity (i.e. the optimal growth stage structure; DELWP 2015). In all EVDs, the highest values of GMA were obtained with the 'landscape' divided between the youngest and oldest age classes. This pattern reflects species 'specialising' in young or old growth stages, while few species show strong association with mid-successional vegetation. The species analysed did not include a number that are of particular interest for management (e.g. rare/threatened species), due to a paucity of data on these species. The results confirm the importance of old forest for biota. Many plant species associated with early growth stages have a weedy/colonising ecology. Similarly, early successional birds are generally open country species that move into recently burnt areas. These species are likely to persist in the landscape under most feasible management regimes, and it is unlikely to be necessary to manage to produce the high levels of young age classes suggested by the optimisation results.

We examined change in GMA and the occurrence of species that were responsive to time since fire under four hypothetical fire history scenarios. The scenarios represented realistic alternative strategies for planned burning for the East Central Bushfire Risk Landscape. The scenarios did not include bushfires. There was little difference in GMA or species occurrence amongst the scenarios, nor did outcomes under the scenarios differ greatly from those predicted under the optimal growth stage distribution. These findings suggest that most Foothills Forest species studied, including those that exhibit strong responses to time since fire, are likely to persist under a range of growth stage distributions produced by planned burning activities.

Overall, the project results indicate that the Foothills Forest system is relatively resilient, in the sense that most common species are likely to persist on a landscape scale if subjected to fire regimes within the range of historical variation. This apparent resilience is likely to be due to many structural and habitat features of Foothills Forest being relatively loosely related to time since fire (compared to other ecosystems, as well as the effects of fire regimes being mediated by environmental variation).

It is important to note that we were only able to include relatively common and widespread species in our analyses. Rare or threatened species may be so because they have fire regime responses that differ from the majority of species in the system. Further study of the relationships of rare species to fire regimes would be beneficial. In addition, the data analysed within this project were mostly collected from large, contiguous blocks of Foothills Forest. However, a substantial proportion of the Foothills Forest system is fragmented and/or adjoins agricultural or urban areas. In this context, interactions of fire regimes with other drivers (e.g. exotic plants and animals, including predators) may result in different outcomes for biota than those observed in more intact forest.

Our results confirm that fire is a key driver in the Foothills Forest system, but its influence is often equalled or outweighed by that of climate, topography and biogeography. The overall resilience of the biota of the foothills system may mean that managers have more latitude around, for example, growth stage distributions than in more fire sensitive systems. This may allow ecological fire management resources to be directed towards protecting and promoting fire sensitive species and habitats that occur within, or are interspersed among, Foothills Forest. One important aim should be to preserve and enhance the extent of old vegetation, which, by definition, will take decades to replace if lost.

# 1 Background

Fire is an important driver of ecosystems globally (Bowman *et al.* 2009). Variation in fire events and fire regimes contributes to landscape heterogeneity (Schoennagel *et al.* 2008), which influences the distribution and abundance of biota (Turner *et al.* 2003). Fires can also threaten human life and assets. Gill *et al.* (2013) characterised the management of potentially negative outcomes of wildfire as the “world wildfire problem”. This problem is likely to become more pressing as global climate change promotes more frequent and intense forest fires (e.g. Flannigan *et al.* 2009).

The “wildfire problem” is manifest in Australia. In forested areas, eucalypts and other scleromorphic plants produce large amounts of fuel (Walker 1981; Keeley *et al.* 2013). Hot summers, particularly when coupled with periodic drought, result in landscape-wide availability of fuel sufficiently dry to burn. In these conditions, an ignition can quickly grow into an intense and fast moving fire (Cruz *et al.* 2012).

Planned burning is often used in eucalypt forest to reduce hazards to human life and assets, and for ecological purposes (Gill 2008). However, the ecological effects of different fire regimes, which is fundamental to planned burning, are poorly understood (Driscoll *et al.* 2010). Species differ in their response to fire regimes and the optimum fire regimes for most species are not known (Whelan *et al.* 2002; Clarke 2008). This problem is compounded when trying to determine an optimum regime for a suite of species occurring within a landscape (Bradstock *et al.* 2005; Di Stefano *et al.* 2013; Kelly *et al.* 2014).

The fire ecology of some forest ecosystems has been well studied (e.g. Mackey *et al.* 2002; Lindenmayer 2009; Wittkuhn *et al.* 2011), but most work has focused on only a single aspect of the fire regime, often at relatively small temporal and spatial scales (e.g. York 1999; Loyn 1997, 2003; Tolhurst 2003). While such studies have yielded valuable insights, knowledge is limited about many facets of forest fire ecology (Spies *et al.* 2012). Examples include the interactive effects of unplanned and planned fire, and the interactions of fire regimes with other environmental attributes (e.g. topography, climate variation). As a consequence, land managers frequently lack the information needed to predict the ecological outcomes of planned burning programs and to devise burning strategies to achieve ecological objectives (Gill 2012a).

In Victoria, recent work (e.g. MacHunter *et al.* 2009; Cheal 2010) has provided important pieces of the fire management/biodiversity puzzle, in particular in relation to defining and using ecological resilience measures (McCarthy 2012). However, further work is required to put these pieces together in such a way as to provide guidance for managers. This is particularly the case for the Foothills Forest ecosystem, where ecological complexity is a product of variation in environmental factors and fire regimes. The high flammability of the vegetation, proximity to human settlements and the role of fire as an agent of ecosystem renewal, make sound planning for fire management both highly complex and crucially important.

## 2 Project outline

This project was commissioned by DELWP<sup>1</sup> in 2012. The project was carried out by a consortium of researchers from the Arthur Rylah Institute, Deakin University, La Trobe University and the University of Melbourne. The overarching aim of the project was to provide models of the relationships of biota to fire history and landscape patterns that can be used to assist in the prediction of the biodiversity outcomes of bushfire and planned burns in Foothills Forest.

The main questions that the project addressed were:

1. How does the interplay of landscape, vegetation and fire history influence biodiversity?
2. How is biodiversity related to vegetation growth stages (a function of time since fire), and what mix of growth stages should managers aim for in order to conserve biodiversity?
3. How can managers best use fire to maintain and enhance biodiversity values within Foothills Forest?

The approach to answering these questions consisted of three components:

1. Modelling of biotic responses to fire regime and environmental variables.
2. On the basis of these models, determining the distribution of vegetation type (EVD) growth stages that maximises the geometric mean abundance (i.e. GMA optimisation) of species. This component included testing existing growth stages (Cheal 2010) against empirical relationships of species occurrences to time since fire.
3. Exploring the outcomes for biodiversity of various planned burning scenarios within a subset of the foothills system (the East Central Bushfire Risk Landscape).

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<sup>1</sup> Throughout this report 'Department of Environment, Land Water and Planning/DELWP' also refers to the Department under previous names, e.g. Department of Environment and Primary industries (DEPI), Department of Sustainability and Environment (DSE).

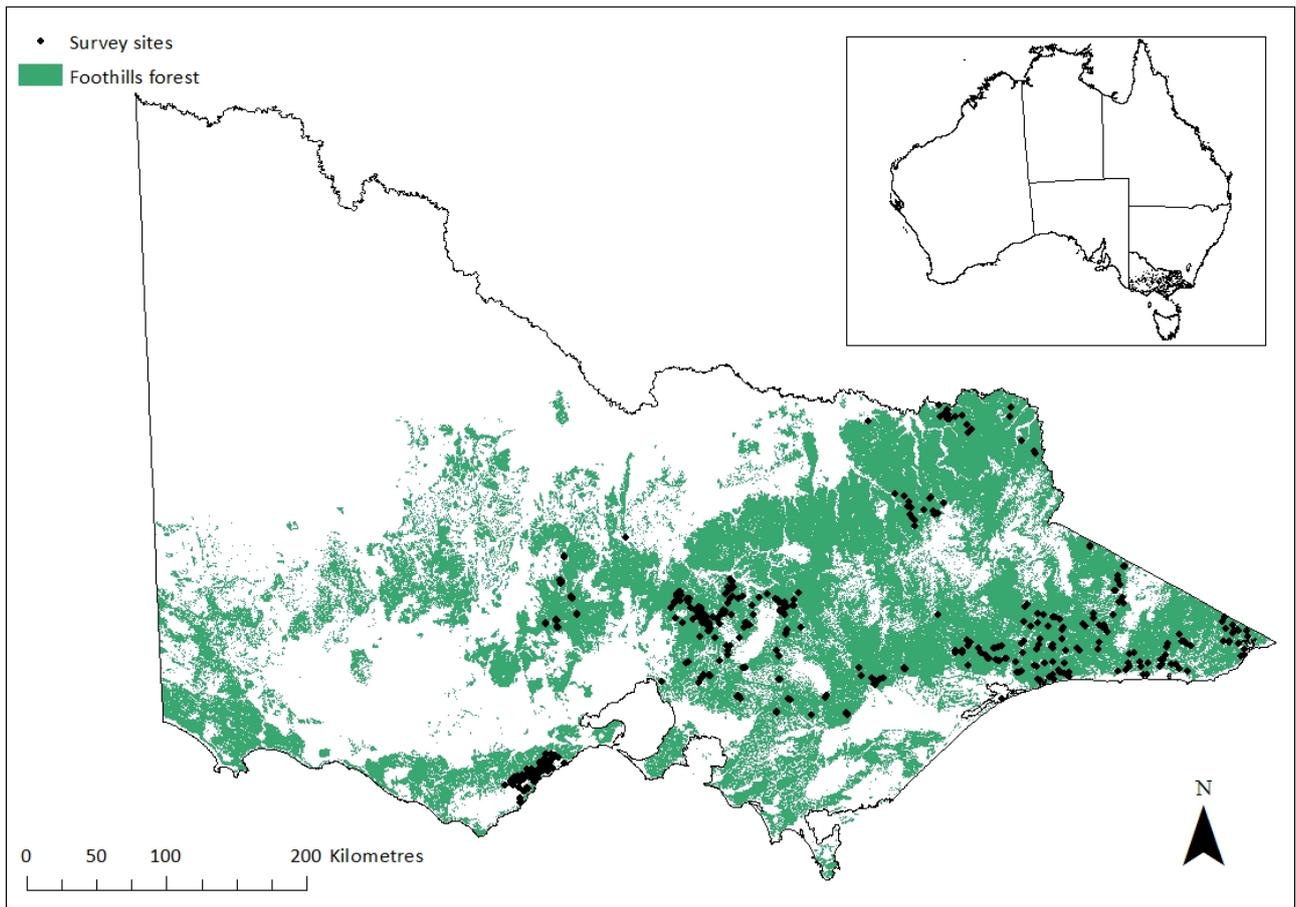
### 3 Foothills Forest: definition

'Foothills Forest' comprises what is commonly referred to as 'messmate-peppermint' or 'mixed-species' forests. It is made up of an intergrading complex of dry and 'damp' eucalypt-dominated communities that occupy the ecological space between the river red gum (*Eucalyptus camaldulensis*) and box (*E. melliodora*, *E. microcarpa*, *E. polyanthemos*) eucalypt forests and woodlands of the coastal and inland plains, and the tall wet eucalypt forests dominated by *E. regnans*, *E. nitens* and *E. delegatensis* of high rainfall, montane areas. Structurally, the Foothills Forests consist of (tall) open forest (*sensu* Specht 1981). The canopy is dominated by varying proportions of *Eucalyptus obliqua*, *E. radiata* and *E. dives*, with *E. sieberi* becoming common in the east. It is important to note that individuals of the dominant eucalypt species of Foothills Forest generally survive fire, regenerating from epicormic buds and/or basal shoots (Gill 1975; Christensen *et al.* 1981). Therefore, unlike in wet eucalypt forest dominated by *Eucalyptus regnans* or *E. delegatensis*, fires are not stand replacing in the sense that most trees are killed and a new cohort replaces them. Instead, Foothills Forest generally exhibits a multi-aged structure, although cohorts of eucalypts originating from seedlings that established following severe fire may be evident.

Sclerophyll shrubs form an understorey of variable density (Conn 1993). The ground layer is often dominated by the fern *Pteridium esculentum*, though in some instances grasses (e.g. *Poa* spp., *Microlaena stipoides*) or graminoids (e.g. *Lomandra* spp., *Lepidosperma* spp.) are prominent. In moister situations, such as along sheltered gullies, *E. obliqua*, *E. globulus* spp. *biscostata* or *E. cypellocarpa* form the canopy, with the understorey often including broad-leaved shrubs/small trees such as *Bedfordia arborescens*, *Pomaderris aspera*, *Coprosma quadrifida* and *Olearia* spp., along with tree ferns (*Dicksonia antarctica*, *Cyathea* spp.) and ground ferns. In terms of the vegetation classification system used in Victoria, our definition of Foothills Forest encompasses the Ecological Vegetation Divisions (EVDs) Tall Mixed Forest (EVD 7), Foothills Forest (EVD 8), Forby Forest (EVD 9), Moist Forest (EVD 10), Granitic Hillslopes (EVD 17) and elements of Grassy/Heathy Dry Forest (EVD 3) (Cheal 2010). Transitions amongst the communities that comprise Foothills Forest, associated with variation in soils and topography, can occur over short distances.

Foothills Forest forms an extensive contiguous band on both the northern and southern lower-mid slopes of the Great Dividing Range (600-1200 m asl), extending from central to north-east Victoria and East Gippsland (Figure 1). Other major occurrences are on the slopes of the Otway Ranges, south Gippsland and far south-western Victoria. In total, Foothills Forest covers approximately 75,000 km<sup>2</sup>. Foothills Forest predominately occurs on loamy soils derived from Silurian sedimentary rock, with granitic rock occurring in some areas. Soils may be fertile but are often highly leached and tend to be shallow in situations such as along ridges (Gibbons and Rowan 1993).

Average annual rainfall in the regions occupied by Foothills Forest ranges between approximately 700-1600 mm (BoM). Average minimum and maximum temperatures in the warmest month (February) are in the ranges 12-14°C and 22-29°C respectively, with corresponding ranges for the coldest month (July) being 2-5°C and 12-14°C. However, variability in weather and climate around these averages is likely to be ecologically significant (Gutschick and BassiriRad 2003). Particularly important is the occurrence of hot (>35°C) days and extended drought associated with El Nino climate conditions, which impose physiological stress on organisms (e.g. Seabrook *et al.* 2011) and promote intense fires (Williams and Karoly 1999).

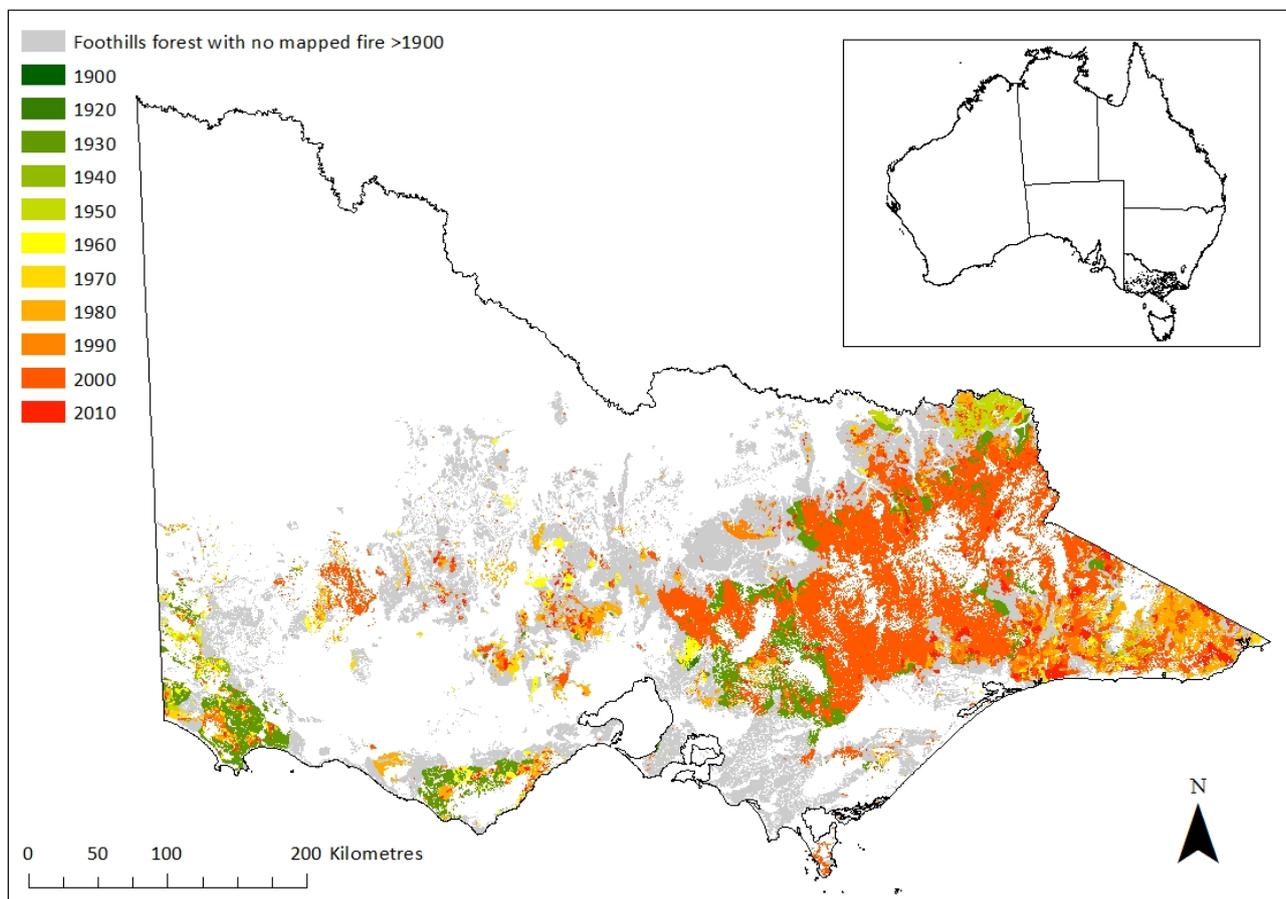


**Figure 3.1.** Extent of Foothills Forest vegetation in Victoria. Black dots indicate survey sites that were used in the project.

## 4 Fire history analysis

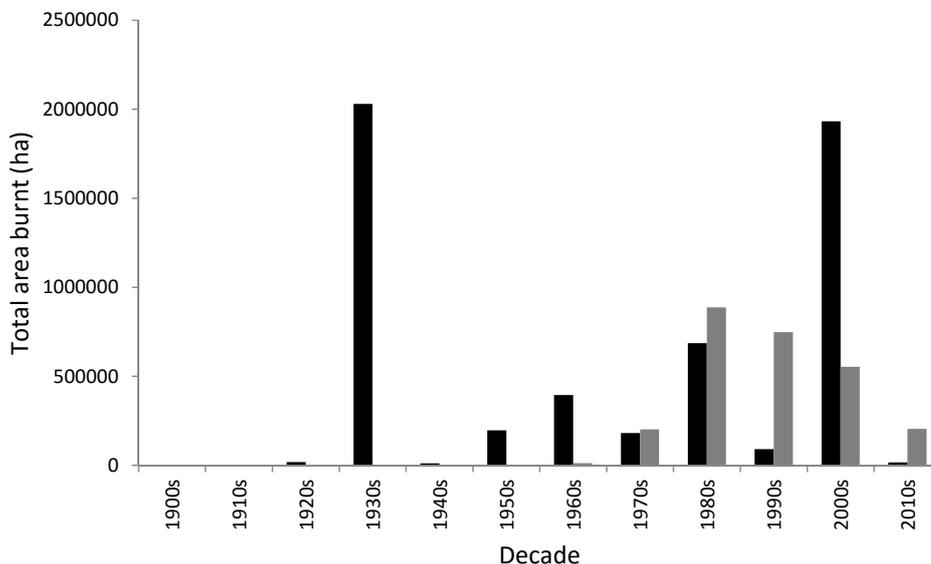
We conducted an analysis of fire history of the Foothills Forest system (excluding areas of fragmented Foothills Forest in western Victoria and South Gippsland), using spatial data provided by DELWP (FIRE 100 YEAR layer). Since 1900, around two-thirds of this area (3,788,117 of 5,556,855 ha) is mapped as having experienced fire at least once, while around 1,768,738 ha (32%) has no mapped fire (Figure 4.1). It is important to note that some of this ostensibly 'unburnt since 1900' vegetation will have burnt over the last century as not all fires have been mapped. This is particularly true for smaller fires before 1980. Similarly, fire mapping typically represents the perimeter of fires and does not account for burn patchiness within fire boundaries.

Approximately 42% of Foothills Forest is mapped as having last burnt since 2000, mostly during major bushfires that occurred in the period 2000-2010. Areas last burnt prior to 1940 are concentrated on the margins of the region along the northern slopes of the Great Dividing Range, in the higher rainfall areas of the Central Highlands and in the Otway Ranges.

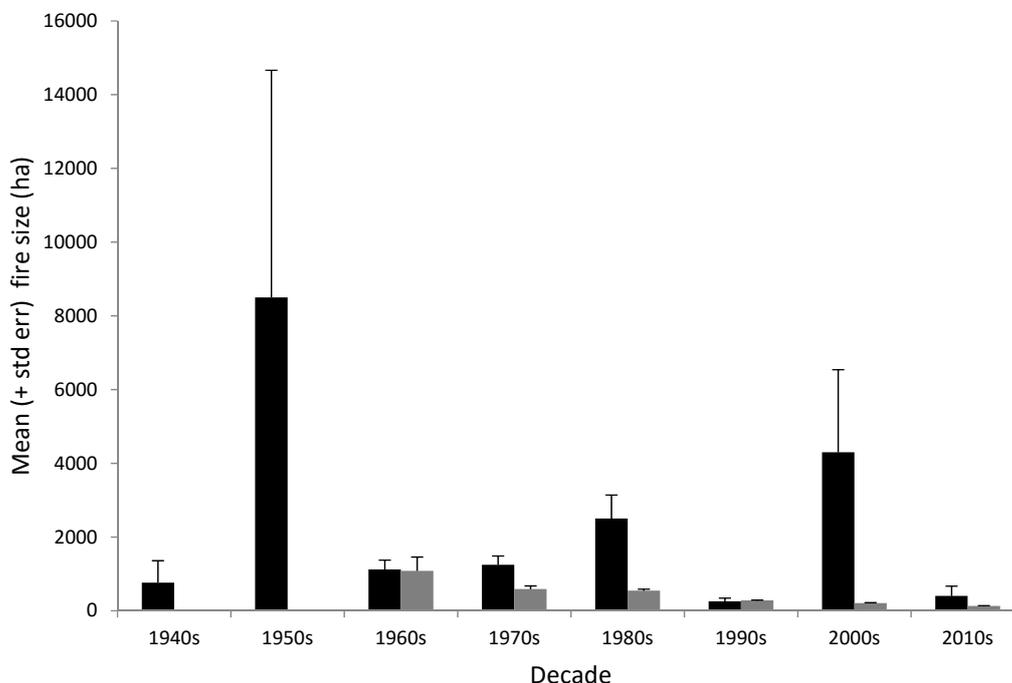


**Figure 4.1.** Fire history of Foothills Forest. Different colours represent foothills vegetation last burnt in each decade since 1900 (the start of fire mapping). Pale grey depicts Foothills Forests for which no fire has been mapped over this period.

Of the area mapped as having burnt since 1900, around three-quarters (~ 2,900,000 ha) was burnt most recently by bushfire, with the remainder (~ 860,000 ha) burnt during planned burns. In the period for which more reliable fire mapping is available (1980s onwards), planned burning has accounted for a greater area burnt than bushfire in most decades (Figure 4.2). The exception to this pattern is the period 2000-2009, in which a series of very extensive bushfires occurred in the Great Dividing Range region. Fire mapping data indicate that since the 1960s the average size of planned burns has decreased (Figure 4.3). However, this may be an artefact of improved mapping of smaller fires over time.

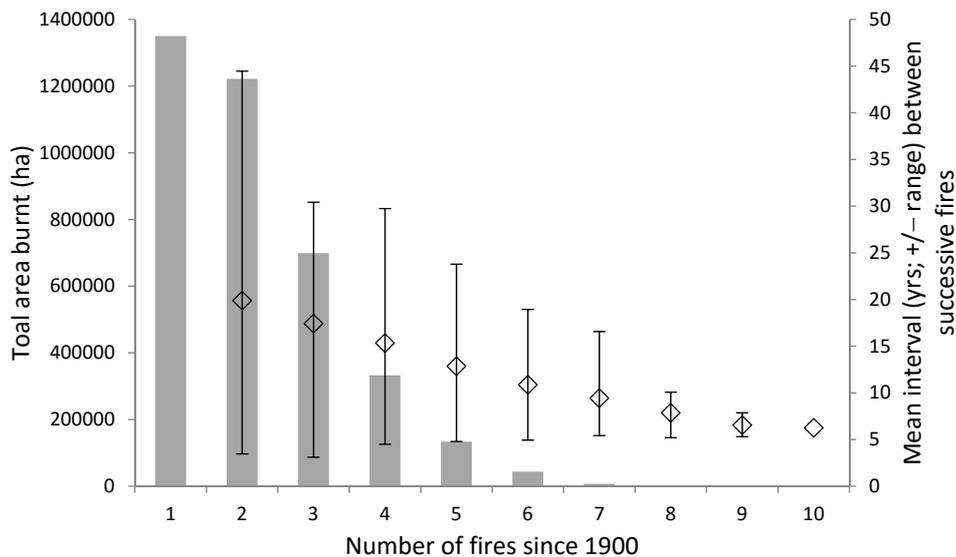


**Figure 4.2.** Overall extent (ha) of Foothills Forest vegetation burnt by bushfire (shown in black) and planned burns (grey) in each decade since 1900 (includes vegetation reburnt in subsequent fires).



**Figure 4.3.** Mean size (+ standard error) of bushfires (shown in black) and planned burns (grey) in each decade since 1940. Data for bushfires mapped in the following decades have not been included: 1920s (only mapped fire=21020 ha), 1930s (mean fire size=120780 ha (+/- 113176)).

Most of the Foothills Forest vegetation mapped as being burnt since 1900 has experienced only one (36%) or two (32%) fires over this period (Figure 4.4). For areas mapped as being burnt more than once since 1900, the average inter-fire interval ranges between about 6 and 20 years (Figure 4.4). Approximately 10% of the study area has experienced fire whilst below the average tolerable fire interval for foothills EVDs (approximately 14 years). However, it is important to note that these figures under-estimate inter-fire intervals, as they do not include areas that were only burnt once, or not at all, since 1900. Planned burns accounted for a greater incidence of repeat fires over this period than bushfires. Bushfire comprised all mapped fires for only a small proportion (0.06%) of areas were burnt more than once since 1900. For areas burnt three or more times, planned fire contributed, on average, to over half (59%) of these successive fires.



**Figure 4.4.** Total area burnt (ha; columns) at different frequencies since 1900 and mean interval (years; diamonds, error bars show interval range) between successive fires for areas experiencing more than one fire over this period.

Land tenure appears to influence fire regimes within Foothills Forest, with public land being more likely to be burnt. Within the area in which fire history was analysed, 59% is public land (reserves or state forest) and 41% privately owned. Public land made up 79% of the area subject to at least one fire. Public land also had higher fire frequency, with 35% being burnt twice, and 39% burnt three or more times, compared to 22% and 4% respectively for private land. It is likely that a lower level of planned burning on private land accounts for this pattern.

Fire mapping data indicates that the Foothills Forest system has been subject to a diversity of fire regimes. Inter-fire interval in particular, varies widely. In contrast, there is less variation in time since fire, with the bulk of the system having been burnt either between 2000 and 2009 (42%), or apparently remaining unburnt since before 1900 (32%). This bifurcation of time since fire values may restrict the ability of managers to achieve desired growth stage distributions (time since fire category; see Sections 10, 11).

## 5 Data sets

### *Biological data*

The Foothills Fire and Biota project used existing data sets held by the project consortium organisations and DELWP. In all, eight sets of flora and fauna field survey data were used (Table 5.1). These were derived from major research or ecological monitoring programs carried out over the years 2007-2012. Data sets contained data from a total of 922 sites, with good replication of samples (> 200) covering a range of subsets of the biota (birds, mammals, flora, vegetation structure; Table 5.2).

### *Predictor variables*

Predictor variables used in analyses were grouped into several 'themes' representing key drivers within the Foothills Forest system (Table 5.3). Each theme was represented by several variables that were thought to be potentially related to biotic patterns. The predictor variables used in analyses varied amongst the studies within the project, depending on the hypotheses being tested.

Data on predictor variables relating to field survey sites were extracted from spatial layers held by DELWP and ARI. Of these, fire history was clearly crucial to the project. This was sourced from the DELWP FireHat database. While this database is a valuable resource, we note several limitations. Firstly, the accuracy and completeness of fire mapping declines the further back in the record one goes. It seems likely this issue applies more to smaller bushfires and planned burns than major bushfires. However, ground truthing of fire history was carried out as part of field surveys from which data were used, so for the current study it is likely that time since fire, at least, is generally correct. Another issue with the available fire mapping is the limited extent (spatially and temporally) of severity mapping. This is problematic as both fire intensity/severity and intra-fire patchiness are known to be biologically important (Robinson *et al.* 2014). We accounted for fire severity by using fire type (bushfire versus planned burn) as a surrogate in models. We acknowledge that this is an imperfect solution, as severity within fire types (particularly bushfires) is variable, however, our results suggest that fire type does (at least approximately) indicate severity (see Section 8.1).

**Table 5.1.** Data sets used in the project. Data sources: ARI = Arthur Rylah Institute, DELWP = Department of Environment, Land water and Planning, DU = Deakin University, LTU = La Trobe University, UM = University of Melbourne. Y = data included in data set.

Source	Project	N sites	Pre-Post burn	Flora	Vegetation structure	Fuel	Camera	Diurnal Birds	Spotlighting	Bats	Trapping
<b>ARI</b>	Gippsland retrospective	68		Y	Y	Y	Y	Y		Y	
<b>ARI</b>	HawkEye Gippsland	50		Y	Y	Y	Y	Y		Y	
<b>ARI</b>	HawkEye Otways	24		Y	Y	Y	Y	Y			Y
<b>ARI</b>	Pre- and Post- fire Flora Monitoring	498	Y	Y							
<b>ARI/UM</b>	FESA (Wombat Forest)	25	Y	Y	?	Y		Y		Y	
<b>UM</b>	HawkEye Otways	129		Y	Y		Y	Y			Y
<b>UM</b>	Otways Fire, Landscape Pattern and Biodiveristy	32	Y		Y			Y			Y
<b>LTU/DU</b>	Faunal Refuges Project	96		Y	Y		Y	Y	Y		Y

**Table 5.2.** Number of samples available for analyses for biotic groups and features.

Biotic group/feature	Number of samples*
Mammals (Elliott trapping)	367
Mammals (camera trapping)	919
Mammals (spotlighting)	285
Birds	1822
Flora	808
Vegetation structure/habitat	393
Tree attributes (diameter, density etc.)	554
Litter	535
Fuel load/hazard	343

\*Includes repeat surveys within sites

**Table 5.3.** Predictor variables within ecological driver 'themes'.

Drivers	Variables	Description
FIRE	Time since fire	Years since fire
	Inter fire interval	Mean interval between fires
		Minimum interval between fires
		Has the minimum tolerable fire interval been breached? (y/n)
	Variation in interval between fires	
	Fire frequency	Number of fires occurring at site
	Fire severity	Fire type (bushfire, planned burn)
	Spatial attributes (within 500m, 1km, 2km of site)	Amount of recently burnt ( $\leq 3$ years) vegetation
Diversity of fire ages (richness and Shannon H index)		
CLIMATE AND RAINFALL	Drought	Rainfall deficit (annual rain – 30yr average) in year of survey
		Rainfall deficit (annual rain – 30yr average) in year prior to survey
	Post fire rain	Rainfall in year following fire (for sites $\leq 5$ years old)
	Average rainfall	July rainfall (to represent broader geographic gradients)
ENVIRONMENT AND TOPOGRAPHY	Topographic position	Visible sky
	Moisture availability	Terrain wetness (wetness 2)
		Stream density
Spatial attributes (within 500m, 1km, 2km of site)	Calculated for each of the three measures above	
VEGETATION TYPE	Vegetation type	Ecological Vegetation Division
	Productivity	Measure of vegetation foliage cover
	Spatial attributes (within 500m, 1km, 2km of site)	EVD (clumped to represent dry, moist, wet, heathy communities)
HABITAT VARIABLES	Differs between data sets	Differs between data sets
BIOTIC PROCESSES	Predation	Camera trapping data on fox occurrence
		Predictive layer of predator occurrence

## 6 Approach to analyses

Details of analyses are given in study summaries and appendices below. However, there are some common features of the analyses that were a product of the nature of the data used. There was some variation in methods across the data sets. For example, some flora surveys recorded plant cover, while others recorded density; for fauna surveys there was variation in the duration of sampling (e.g. trap nights per sample). One strategy to account for this was to reduce the data to presence/absence, rather than abundance. In addition, most analyses used a mixed-effects modelling approach, in which random terms were included in models to account for methodological variation amongst data sets, as well as issues such as lack of independence of sub-samples within sites.

## 7 Advantages and limitations of data and analytical approach

There are positive and negative aspects to the approach of combining data sets that was taken in this project. The main advantages were:

- Large sample size of combined data sets provided greater power to detect trends.
- Greater spatial scale allowed system-wide analysis.
- Combined data sets provided greater replication of samples across a broader range of values of predictor variables, in particular fire history.

Limitations of the approach include:

- *Increased 'noise' in data.* While difficult to quantify, we expect that combining data across a wide spatial and temporal range may increase unexplained variation (error) in analyses. Some of this variation is 'natural' in the sense that it arises from the inherent complexity of the system and increases with spatial and temporal scale of data sets. In addition, noise also arises from differences in survey protocols, observers etc. across the data sets.
- *Loss of information in reducing data to presence/absence.* As described above, in many analyses, biotic data were converted to presence/absence to allow the data sets to be combined. This means that, potentially, ecologically meaningful patterns in species abundances were not detected. However, for species with low prevalence (the case for most of the species considered in our analyses) presence-absence is highly correlated with abundance. Therefore, we considered that the advantages of being able to combine data sets and include more species in analyses outweighed the disadvantage of information loss.

- *Time taken to organise data.* Combining and collating the data sets prior to analyses proved to be a major task. This included: examining data collection protocols to determine where it was appropriate to combine data sets; standardising variable labels; extracting attributes of sites from spatial data sets (e.g. fire history, climate and topographic variables). In addition, it was necessary to check that there was sufficient replication of samples across the fire history and other gradients to carry out the planned analyses.
- *More complex analyses.* As discussed above, combining the data sets necessitated the use of more complex statistical techniques in order to account for patterns in the data that may have arisen due to differences in sampling techniques.

### **Lessons for future monitoring programs**

On the basis of our experience with the range of data sets used in the project, we recommend the following issues be considered for any future large-scale survey or monitoring program.

- *Survey design.* The FFB project utilised data sets from projects that were mostly designed to answer specific questions about biota/environment relationships. While it eventuated that we were able to combine a number of data sets to address numerous questions at a system-wide scale, there was no guarantee that this would be the case. A future monitoring program should be designed such that sampling adequately encompasses a broad range of the variation in key predictor variables, with sufficient control sites and replication within, for example, vegetation types and regions. Since the design of a monitoring program should be driven by the questions that it seeks to address, the questions need to be carefully and clearly framed to meet current and future needs.
- *Robust protocols for data management and curation.* As noted above, data management was a time consuming aspect of the project. Well managed data sets would minimise lead time prior to analysis. We recommend that responsibility for managing data sets be assigned to a specific person or unit within DELWP as a core part of their job description (including allocating required funding).
- *Data custodianship and access.* Some potentially useful data sets were unavailable either because they had not been managed so as to be readily usable, or permission to use them was not granted. In other cases, it was initially unclear who (individuals and organisations) had custodianship over data sets. The data management issue has been discussed above. It would also be useful to create a register of data sets held by DELWP, including meta-data, custodianship and whether there are restrictions on data availability to DELWP staff and/or external research partners.
- *Standardised predictors.* For a retrospective or space for time study, such as the FFB project, there are a multitude of potential predictor variables available, mostly derived from spatial data, capturing aspects of climate, topography, vegetation, soils etc. While any of these variables may potentially exhibit relationships with biotic patterns, only a few will represent actual mechanisms of biota/environment interactions. For a monitoring program aimed at providing regular reporting on biodiversity, it would be beneficial to identify a short list of key predictor variables to use in analyses. Ideally, in addition to reflecting ecological mechanisms, variables on this list would not be correlated with each other and be would represented by data that are easily collected.
- *Rare species.* General biological survey data, as was available for the Foothills Fire and Biota project, usually include few records of rare species. However, these species will clearly be of interest to land managers. Therefore, consideration needs to be given to incorporating targeted surveys for these species into monitoring programs.

## 8 Relationships between biota and fire and environmental drivers

### 8.1 Vegetation structure

We investigated the relationships of Foothills Forest vegetation structure (cover of vegetation strata) to attributes of the fire regime. These analyses examined the following questions:

1. What is the relative influence of fire and environmental drivers on vegetation structure in Foothills Forests?
2. Is post-fire change in vegetation structure affected by the attributes of preceding fires?
3. Does the severity of the most recent fire alter the effects of preceding fires on vegetation structure?

#### Methods

Vegetation structure was represented by the proportional cover of vegetation in each height stratum. These data were also used to calculate the Shannon-Wiener diversity index ( $H'$ ), as a measure of the structural diversity of vegetation. Fire regime attributes considered were time since fire, inter-fire interval, Fire Frequency and Fire Type (bushfire/planned burn; a surrogate for severity), while environmental variables were rainfall (mean July rainfall), topographic wetness, topographic position (visible sky), and vegetation type. For these analyses EVDs were classified into three groups: Driest (Grassy/Heathy Dry Forest, Tall Mixed Forest), Dry (Foothills Forest, Forby Forest), Mesic (Moist Forest, Riparian (Higher Rainfall)).

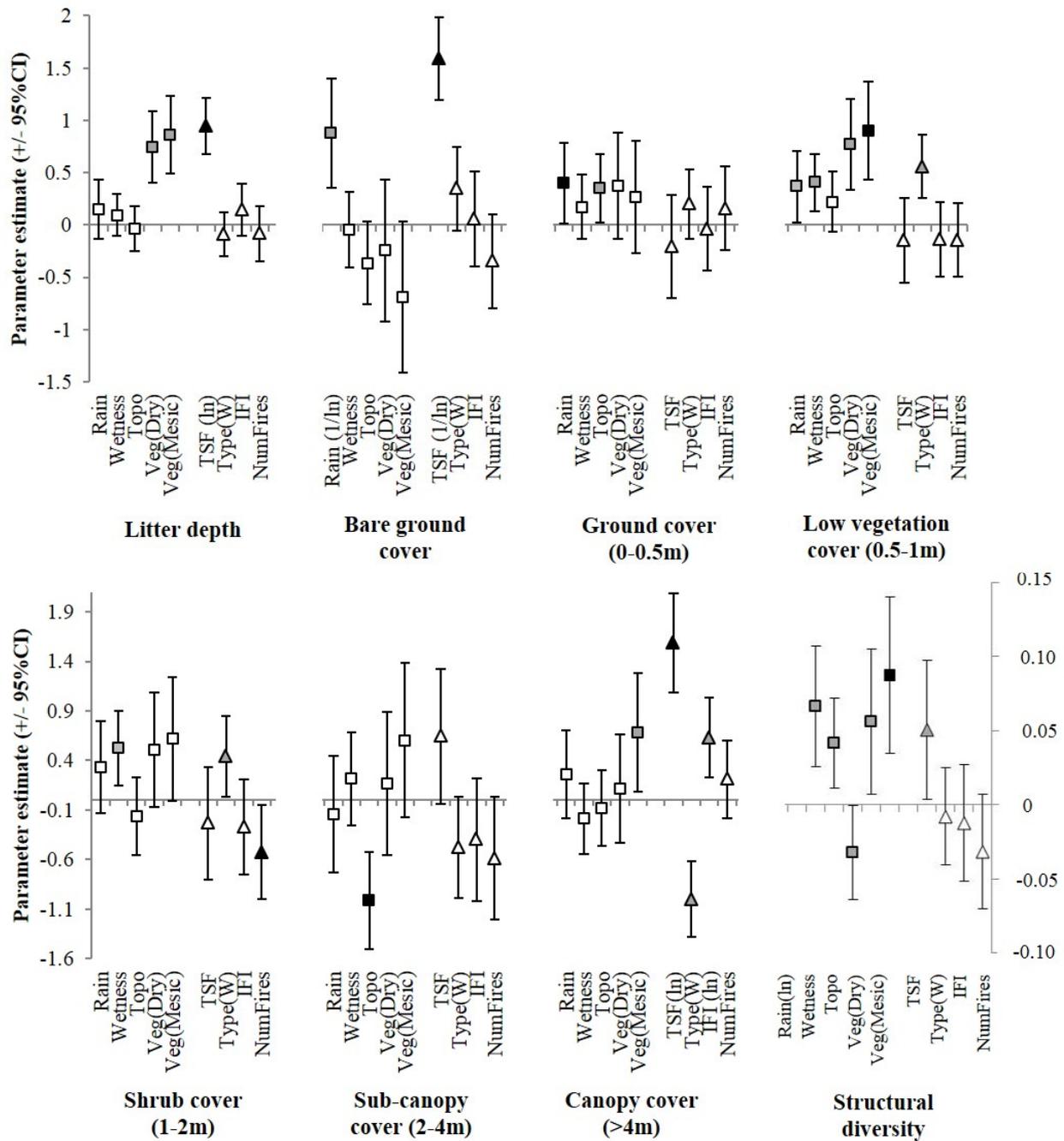
Generalised linear mixed models (GLMM) were used to assess the relative influence of fire history and environmental attributes on vegetation structure. For each response variable, all predictor variables (transformed if exploratory analyses indicated this was necessary) were included in a single 'global' model. Parameter estimates, and associated standard errors, were used to compare effect sizes between variables. To examine questions 2 and 3, models with and without interaction terms for relevant fire variables were compared.

#### Results

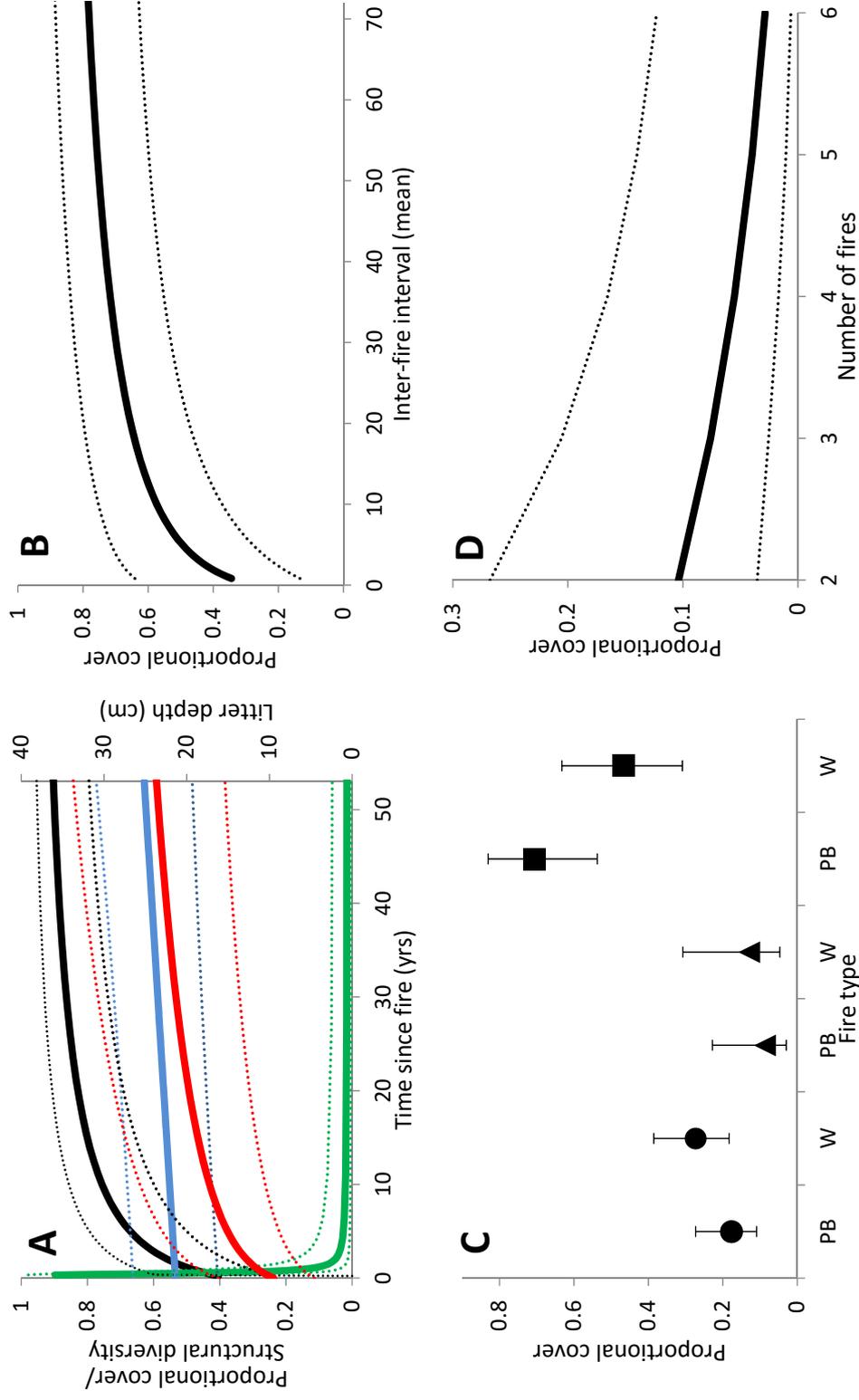
Both fire and environmental drivers affected vegetation structure in Foothills Forests (Figure 8.1.1). Fire was the strongest driver of litter depth and the cover of bare ground, shrub (1-2m), and canopy (>4m) strata. Litter depth and canopy cover increased with time since fire; the cover of bare ground decreased (Figure 8.1.2). Shrub cover decreased with increasing fire frequency. Model results also revealed additional effects of fire history attributes on vegetation structure. Fire severity affected the cover of low (0.5-1m), shrub (1-2m) and canopy (>4m) vegetation: bushfire resulted in increased cover of low and shrub vegetation, and decreased canopy cover, relative to planned burning. In addition, canopy cover increased with inter-fire interval (IFI), and structural diversity increased with time since fire (TSF; Figure 8.1.2).

Environmental drivers had the strongest influence on the cover of ground (0-0.5m), low (0.5-1m), and sub-canopy (2-4m) strata, and overall structural diversity. Influential characteristics included mean annual rainfall (Rain), topographic variation (Topo) and vegetation type (Figure 8.1.1).

Variables representing fire and environmental attributes accounted for a relatively limited amount of variation in response variables: the highest  $R^2$  value for any global model was 16%.

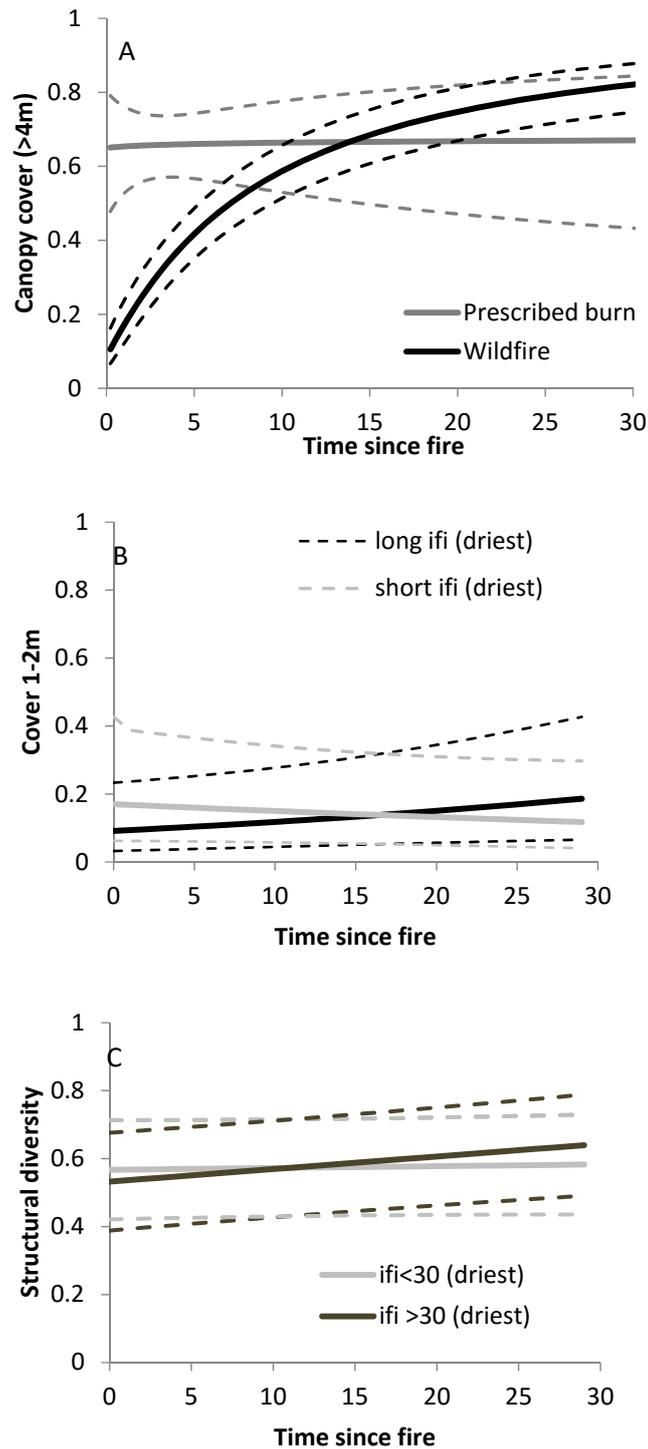


**Figure 8.1.1.** Parameter estimates ( $\pm$  95% confidence intervals) from generalised linear mixed models for the relationship between environmental (square symbols) and fire (triangles) drivers and different attributes of vegetation structure. Filled symbols indicate parameter estimates for which the 95% confidence intervals do not include zero, with black indicating the variable with the largest parameter estimate (i.e. strongest effect on associated response variable).



**Figure 8.1.2.** Predicted relationships between vegetation attributes and fire history measures. (A) Time since fire and canopy (black) and bare ground (green) cover, litter depth (red), and structural diversity (blue). (B) Inter-fire interval and canopy cover. (C) Fire type (PB=planned burn; W=wildfire/bushfire) and low (0.5-1; circles), shrub (1-2m; triangles) and canopy (>4m, squares) cover. (D) Number of fires and shrub cover (1-2m). Dashed lines/error bars show 95% confidence intervals.

We found clear support for an interaction between fire severity ('Type') and post-fire change in canopy cover (>4m). Canopy cover increased with time since bushfire, but showed no change with time since planned fire (Figure 8.1.3).



**Figure 8.1.3.** Predicted relationships between fire history attributes and a) canopy cover (>4m) b) cover mid shrubs (1-2m) and c) structural diversity from models with interaction terms for fire history attributes. Plots illustrate legacy effects of preceding fire history on post-fire change in associated vegetation structural attributes; all plots show predicted values for the driest vegetation type.

In contrast, the severity of the most recent fire (Type) did not alter the effect of past fires, as represented by fire frequency and inter-fire interval, on any attribute of vegetation structure examined (Appendix 1). However, it is important to note that fire frequency and inter-fire interval were not found to have a strong effect on any response variable in these models, limiting the capacity to adequately examine this question.

## Discussion

Vegetation structure in Foothills Forests is influenced by both fire history and environmental characteristics, with the relative strength of these factors differing between vegetation attributes. Fire was the strongest driver of ground strata (litter depth, cover of bare ground), and shrub and canopy layers, with time since fire being a particularly important influence. The cover of ground and canopy strata, and overall structural diversity, increased with time since fire. These findings have implications for habitat suitability for associated fauna in Foothills Forests. For example, increased abundances of both ground- and canopy-foraging bird species in unburnt patches, relative to surrounding recently burnt vegetation, is thought to be related to the habitat characteristics – specifically food resources – provided by older Foothills Forest vegetation (Robinson *et al.* 2014).

We found minimal evidence that past fires influenced post-fire succession (Question 2) or that fire severity overrides the effect of preceding fires (Question 3). In accord with previous studies (e.g. Sitters *et al.* 2015) in which time since fire and fire severity affected most vegetation attributes, these findings indicate that the primary way fire affects the structure of Foothills Forests is via the influence of fire events (e.g. time since fire, the severity of the most recent fire). In contrast, measures of longer-term fire regimes (e.g. fire frequency and interval) exert a weaker influence.

Up to three-quarters of the variation in vegetation structure remained unexplained by fire and environmental drivers. This unexplained variation may in part be due to ‘noise’ in the data sets, as discussed above (Section 7). However, it is also likely to indicate that additional factors shape Foothills Forest vegetation, many of which are not amenable to land management. Empirical data contributing to these analyses were drawn from a large geographic area (155 000 km<sup>2</sup>), and so factors operating at a finer scale (e.g. local disturbance histories) may also affect vegetation structure. These results do not suggest that (planned) fire in this system will have no effect on native biodiversity (see Aponte *et al.* 2014 for empirical evidence to the contrary). However, they do highlight that fire management must be undertaken in the context of a range of other factors. For example, topographic position has been found to modify the effects of fire severity on coarse woody debris (Bassett *et al.* 2015) and vegetation structure (Collins *et al.* 2012) in this, and other eucalypt dominated forests, respectively.

Our findings show that fire history attributes have a variable effect on the structure of Foothills Forest vegetation: some vegetation attributes do not appear to be strongly affected by fire whereas others, notably ground and canopy strata, show stronger relationships. Particularly important were attributes of the most recent fire, which influenced vegetation structure for several decades. While the relatively weak effects of fire on some vegetation attributes may be related to the variance inherent in data drawn from a large study area, they indicate that at a broad scale, structural attributes of Foothills Forest vegetation appear to be relatively resilient to the effects of fire.

## 8.2 Flora

Patterns of floristic composition (presence/absence of species within samples) and species richness in relation to fire regime and environmental variables were examined. The aim of the analysis was to determine the relative influence of the various fire and environmental predictors on flora.

### Methods

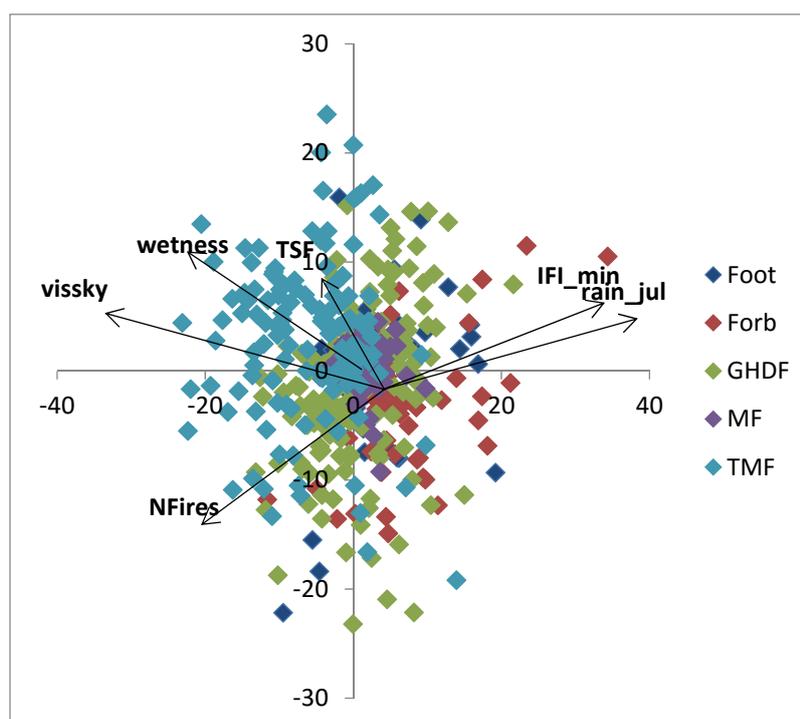
Fire regime attributes considered were time since fire (TSF), minimum inter-fire interval (IFI), and fire frequency (number of recorded fires), while environmental variables were rainfall (mean July rainfall), topographic wetness, topographic position (visible sky), and vegetation type (EVD). Study region was also included as a predictor variable in analyses. Sample effort ( $\log_e$  of sample plot size) was included in species richness analyses to account for the effect of this variable on the observed number of species within samples.

Floristic composition was analysed using multivariate techniques (ordination, PERMANOVA). Boosted regression tree (BRT) analysis was used to examine the relative influence of predictor variables on overall species richness, as well as species richness within functional groups. Functional classification of species was taken from the Victorian Plant Vital Attributes database (DELWP unpublished). Within this database plant functional groups are defined primarily on the basis of regeneration characteristics, based on the Vital Attributes system proposed by Noble and Slatyer (1980). The functional groups that occurred within the current data set with sufficient frequency for analysis were: seeder herbs, resprouter herbs, seeder shrubs (short juvenile), seeder shrubs (long juvenile), resprouter shrubs (short juvenile) and resprouter shrubs (long juvenile).

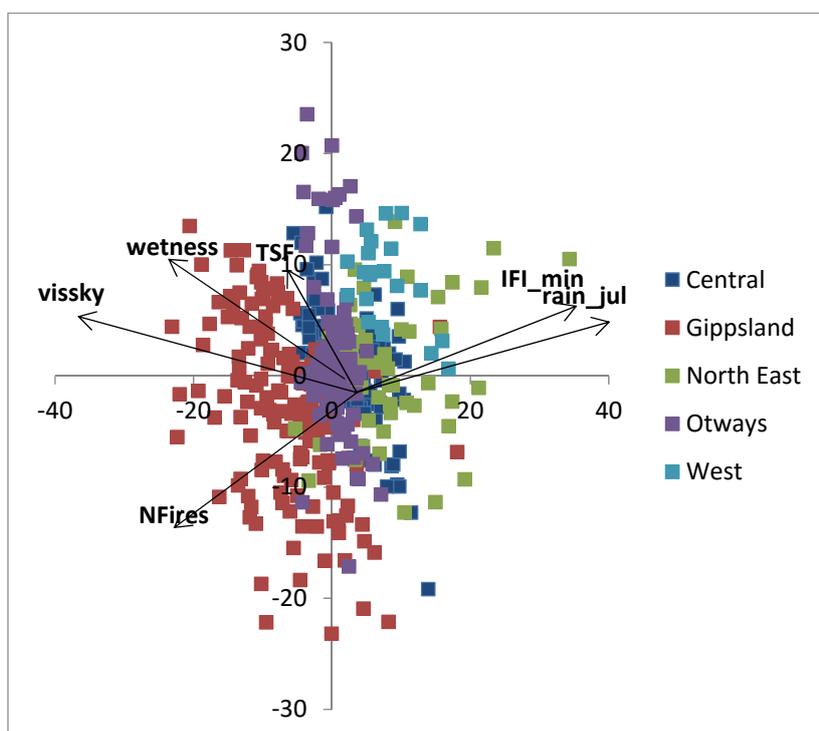
### Results

#### *Floristic composition*

Ordination of the overall data set indicated (unsurprisingly) that floristic composition varied amongst EVDs and regions (Figures 8.2.1, 8.2.2; ordination stress = 0.18). Grassy-heathy dry forest and tall mixed forest were the most compositionally distinct EVDs. Sites within other EVDs tended to be placed more closely together, reflecting their greater similarity in species composition.



**Figure 8.2.1.** NMDS ordination of samples and vectors for predictor variables, with samples classified by EVD. Foot = Foothills Forest, Forb = forby forest, GHDF = grassy-heathy dry forest, MF = moist forest, TMF = tall mixed forest.



**Figure 8.2.2.** NMDS ordination of samples and vectors for predictor variables, with samples classified by study region.

Floristic composition was most strongly related to region, with EVD and July rainfall the next best predictors (Table 8.2.1). Amongst fire regime variables, composition was most strongly related to time since fire.

**Table 8.2.1.** Results of PERMANOVA analysis of composition in relation to predictor variables. Bold  $R^2$  value indicates significant relationship to composition ( $P < 0.05$ )

	df	SS	MS	F	$R^2$
Time since fire	1	5858	5858.2	11.55	<b>0.02</b>
Minimum IFI	1	4253	4253	8.39	<b>0.01</b>
Number Fires	1	849	848.6	1.67	0.00
July rainfall	1	13333	13333.2	26.30	<b>0.04</b>
Wetness	1	7531	7531	14.85	<b>0.03</b>
Visible Sky	1	3726	3726.4	7.35	<b>0.01</b>
Region	3	38052	12684	25.02	<b>0.13</b>
EVD	4	15143	3785.8	7.47	<b>0.05</b>
Residuals	419	212440	507		0.71

When analysed separately within each EVD, composition was most strongly related to region in all EVDs except grassy-heathy dry forest and moist forest (Table 8.2.2). Time since fire had a significant influence on composition in all EVDs and was the strongest predictor, apart from region, in Foothills Forest and moist forest. Interestingly, the rank order of time since fire  $R^2$  values across EVDs was consistent with that of 'mesic-ness', that is the degree to which the forest understorey is typically dominated by mesic shrub and small tree species (moist>foothills>tall mixed>forby>grassy-heathy). Minimum inter-fire interval was equally influential with time since fire and July rainfall in Foothills Forest and was also significantly related to composition in grassy-heathy and tall mixed forests. Composition was related to the number of fires in foothills and grassy-heathy forest. Topographic wetness and visible sky were related to composition in all EVDs except Foothills Forest, however, these topographic variables were not the strongest predictors of composition in any forest type.

**Table 8.2.2.** R<sup>2</sup> values of predictors from PERMANOVA analyses within EVDs. Bold type indicates P < 0.05.

	Foothills	Forby	Grassy-heathy dry	Moist	Tall mixed
Time since fire	<b>0.09</b>	<b>0.04</b>	<b>0.02</b>	<b>0.10</b>	<b>0.03</b>
Minimum IFI	<b>0.10</b>	0.01	<b>0.02</b>	<b>0.02</b>	<b>0.06</b>
Number Fires	<b>0.04</b>	0.01	0.01	0.02	0.01
July rainfall	<b>0.05</b>	<b>0.04</b>	<b>0.13</b>	0.02	<b>0.07</b>
Wetness	0.02	<b>0.02</b>	<b>0.01</b>	<b>0.05</b>	<b>0.02</b>
Visible Sky	0.02	<b>0.02</b>	0.01	<b>0.02</b>	<b>0.02</b>
Region	<b>0.17</b>	<b>0.19</b>	<b>0.10</b>	<b>0.04</b>	<b>0.13</b>

### *Functional group relationships to fire regime variables*

The strongest predictors for total species richness were region, vegetation type and TSF. TSF was also amongst the strongest predictors for species richness in all functional groups, contributing 13-46% of explained variation across groups (Table 8.2.3). Overall species richness and richness within functional groups generally exhibited similarly shaped relationships to time since fire, which consisted of a spike immediately post-fire, a trough approximately two years post fire, then a rise to a plateau at about 10 years post-fire (see Figure 6; Appendix 2). Inter-fire interval was amongst the strongest predictors for seeder shrubs and resprouting shrubs with a long juvenile period. Topographic position (visible sky) and July rainfall were amongst the strongest predictors for all functional groups except resprouting shrubs with long juvenile period, and seeder shrubs in the case of July rainfall. Number of fires, region and vegetation type were relatively weak predictors for all response variables. Similarly sample effort made only a minor contribution to explained deviance in all models.

**Table 8.2.3.** Deviance explained and contributions of predictors from boosted regression tree models of species richness. Bold type indicates the three strongest predictors within each model.

	% dev explained	% contribution to explained variance								
		TSF	IFI	NFIRE	WET	VISSKY	RAIN_AV	REGION	VEG	SAMP_EFFORT
Total richness	58.04	<b>12.9</b>	6.6	0.9	10	8.6	11.8	<b>27.9</b>	<b>13.6</b>	7.8
Shrub	24.85	<b>38.4</b>	9.5	0.8	8.1	<b>16.4</b>	<b>10.2</b>	6.3	8.3	2.1
Seed Shrub	27.32	<b>45.8</b>	<b>9.3</b>	1.1	6.0	<b>17.8</b>	8.8	4.0	5.2	2.0
Seed Shrub LJ	37.6	<b>35.4</b>	9.3	2.4	9.8	<b>20.0</b>	<b>11.6</b>	3.8	5.2	2.6
Seed Shrub SJ	15.75	<b>46.1</b>	7.9	3.9	7.3	<b>13.3</b>	<b>11.7</b>	2.1	6.6	1.0
Resprout shrub	10.75	<b>32.5</b>	5.1	1.5	12.7	<b>13.2</b>	<b>23.6</b>	5.4	5.3	0.7
Resprout shrub LJ	14.73	<b>29.3</b>	<b>16.3</b>	1.5	<b>18.6</b>	11.2	10.3	5.1	5.6	2.0
Resprout shrub SJ	15.07	<b>35.9</b>	5.8	0.6	11.5	<b>17.5</b>	<b>17.1</b>	4.0	7.2	0.4
Herb	43.17	<b>30.3</b>	11.0	0.7	11.4	<b>12.9</b>	<b>18.8</b>	3.8	9.2	1.9
Resprout Herb	38.16	<b>29.6</b>	11.2	0.6	12.5	<b>12.8</b>	<b>18.5</b>	4.0	8.5	2.5
Seed Herb	35.97	<b>31.3</b>	11.3	1.0	11.7	<b>13.9</b>	<b>17.3</b>	2.7	9.3	1.5

## Discussion

Our results show that floristic composition within the Foothills Forest system is influenced by fire regime, topography, climate and bioregional factors. However, the relative influence of these drivers varies amongst the constituent vegetation communities comprising Foothills Forest. Overall species richness, and species richness within plant functional groups, were related to time since fire, with other fire regime

variables generally having little influence. Topographic position (as indicated by visible sky) and rainfall were also relatively strong predictors of functional group richness.

Overall, and in most EVDs, region was a strong determinant of floristic composition. It is likely that differences in soils amongst regions accounts for a substantial part of this variation. Within the study region, the best soils (“very fertile loams”) predominate in the North-east, with somewhat less productive soils in Gippsland, the Otway Ranges and the Central Highlands and relatively poor soils in the West (Gibbons and Rowan 1993). A transition from mesic or grassy/forby to more sclerophyllous vegetation would be expected across such a soil fertility gradient (Beadle 1953), though this may be subtle within particular vegetation communities (EVDs). Regional differences in composition may also be due to large-scale biogeographic patterns. For example, sites in the North-east and West regions of the study area are situated on the northern fall of the Great Dividing Range, and therefore have floristic affinities with inland forests and woodlands. Similarly, Gippsland (particularly in the east) is floristically similar to sub-coastal New South Wales (Costermans 1994).

Amongst the fire regime variables considered, time since fire had the most consistent influence on composition. It is notable that the strength of the relationship of composition to time since fire across EVDs appeared to be positively correlated with the gradient from ‘drier’ to ‘wetter’ communities. This suggests that compositional changes associated with post-fire succession are more subdued in drier forest types. This may be because, where moisture is limiting, there is less potential for succession to a more mesic community and hence composition remains more stable over time. The strong relationship of composition within grassy-heathy dry forest to rainfall further suggests the primary influence of moisture availability, rather than fire regime, in this community

The generally observed pattern of the relationship of overall and functional group species richness to time since fire (a spike immediately post-fire, a trough approximately two years post fire, then a rise to a plateau at about 10 years post-fire) accords with the pattern of vegetation regeneration in Foothills Forest. Fire prompts a mass regeneration event, with most species rapidly re-establishing from seed and/or vegetatively, including a suite of ‘weedy’ colonising species. As most plants are small, many species can be present within sample plots. However, as plants grow the vegetation rapidly comes to be dominated by dense stands of relatively few highly competitive species (e.g. eucalypt saplings, *Goodenia ovata*, *Pomaderris aspera*). Over time, as regenerating shrubs and trees begin to thin out, the ground layer again opens up, creating niches for low-growing herbaceous species, leading to a rise in species richness.

The fire regime attributes of minimum inter-fire interval and number of fires had less consistent influence on composition, relative to time since fire, although inter-fire interval was amongst the strongest predictors of composition in Foothills Forest (EVD) and tall mixed forest. Similarly inter-fire interval and number of fires were generally relatively weak predictors of species richness. The interaction of inter-fire interval and fire frequency (number of fires) with plant life histories can be a key driver of vegetation composition (Keith *et al.* 2002). It is unclear why these variables are more influential in some EVDs than others. However, the overall weaker relationships of inter-fire interval and number of fires to composition, relative to time since fire, may indicate that the foothills system is resilient to variation in these drivers, at least within the range of values examined in the current study. The foothills system has been exposed to a diversity of fire regimes over at least the last century (and probably considerably longer), and therefore it is unlikely that many constituent species have narrow fire regime requirements for their persistence. However, variation in fire regimes may result in marked changes in *relative abundance* of species, which could not be detected with the current data set. In the communities examined many species are near ubiquitous, but can vary markedly in abundance.

Across the foothills system as a whole, biogeographical influences are the strongest drivers of composition. System-wide relationships of composition to fire regime variables are likely to reflect associations of fire regimes with vegetation communities. However, within communities (EVDs) there are relatively strong effects of fire regime on composition. The most pronounced of these appears to be successional change with increasing time since fire.

## 8.3 Birds

We examined the relationships between different site and landscape elements of fire regime on bird functional guilds. This approach is useful for communicating a generalised pattern of functional responses to disturbance (Moretti and Legg 2009), particularly where there may be insufficient data for analysing trends at the species level. Functional guilds have been identified for birds in Foothills Forest in relation to their nesting and feeding attributes (Loyn 1998). These attributes represent distinct habitat preferences that may be differentially affected by the biological legacy of fire regimes.

Recent work on avian responses to fire regimes in Foothills Forest has indicated that bird abundance is higher with longer time since fire (Robinson *et al.* 2014). However, the effects of *fire frequency* on birds are less well understood. The extensive data sets available in this project provided a good opportunity to test the hypothesis that fire frequency affects bird abundance. We predicted that increasing fire frequency (number of recorded fires) would decrease the abundance of birds in all guilds except for those feeding or nesting on open ground. This prediction is based on the observation that more frequent fire creates a more open forest understorey (see Section 8.1 above). Furthermore, we predicted that there would be an interactive effect of fire frequency and fire severity, such that the effects of repeat fires would be more pronounced where the most recent fire was a bushfire, rather than a (generally) less severe planned burn. Lastly, we examined the relative importance of site and landscape attributes of fire regimes on abundance within bird functional guilds.

### Methods

The combined project data sets provided 1432 bird counts across 431 sites and included survey data from the years 2010-2013. Data were selected where the bird survey technique was consistent and provided a count of individuals for each species per survey. Observer, time of day, breeding season and a combined metric comprising season (breeding / non-breeding) and survey year were included as random factors.

The data for bird species were then aggregated into one of 13 functional guilds to provide a measure of abundance per bird count for each of 10 feeding guilds: frugivores (F), nectarivores (N), carnivores (CV), insectivores (feeding from: bark (B), canopy (C), open ground under trees (OT), damp ground (DG), trees and shrubs (TS)), granivores (feeding from ground (GG) or shrubs/trees (GST)); and three nesting guilds: ground nesters (G), large hollow nesters (LH) and small hollow nesters (SH). To assess the consistency of population responses among members of the nectarivore functional guild seven species were analysed individually. These were Brown-headed Honeyeater, Crescent Honeyeater, Eastern Spinebill, Red Wattlebird, White-eared Honeyeater, White-naped Honeyeater and Yellow-faced Honeyeater.

Fire history attributes (Table 8.3.1) for each site / survey combination were extracted by intersecting a spatial layer of fire history information (FireHatv2013, DEPI) with site temporal and spatial information using GIS.

**Table 8.3.1.** Site and landscape attributes of fire regime considered in analysis of bird functional guilds.

Scale	Variable abbreviation	Variable description
Site	TSF	Time Since the last Fire event
	NF	Number of Fires since 1970*
	FT	Fire Type. Bushfire (BF) and planned fire (PF) <sup>†</sup>
	GS	Growth Stages of the vegetation ( <i>sensu</i> Cheal 2010)
	GS2	Growth Stages of the vegetation: early/mature-late (derived from Cheal 2010)
Landscape	AB500m, AB1km, AB2km	Area Burnt at different radii from site within 3 years of the survey
	FAR500m, FAR1km, FAR2km	Fire Age Richness at different radii from site
	SFD500m, SFD1km, SFD2km	Shannon Fire Diversity at different radii from site

\*Reliability of fire records is unknown prior to 1970. <sup>†</sup>This simple dichotomy was necessitated due to the paucity of accurate fire severity mapping prior to 2000.

To investigate which element and scale of the fire regime accounted for the greatest proportion of the variance in abundance of each bird guild, 18 alternative models (general linear mixed models GLMMs) were evaluated using a model selection process. Site scale models (n=9) were: TSF, TSF<sup>2</sup>, TSF+ TSF<sup>2</sup>, NF, NF<sup>2</sup>, NF+NF<sup>2</sup>, FT, GS, GS<sup>2</sup>. Quadratic terms (e.g. TSF<sup>2</sup>) were used to consider non-linear relationships. Models (n=9) with landscape-scale fire variables were: AB (500m, 1km, 2km), FAR (500m, 1km, 2km), SFD (500m, 1km, 2km). The model with the lowest AIC (Akaike’s Information Criteria) value was selected. AIC is a measure of model explanatory power relative to model complexity (parsimony) – lower AIC values indicate ‘better’ models. In addition, separate GLMMs were also used to test for interactive effects of fire type (FT) and the number of fires (NF) on bird guilds and species. Four random factors were included in all models: SurvyrBreed (6 categories representing surveys in breeding / non-breeding season and unique year combinations), MornAft (survey conducted in morning or afternoon), Site (observational level term for each of 431 sites), and Observer (to account for differences between 29 observers).

## Results

### *Relationship between bird guilds and number of fires*

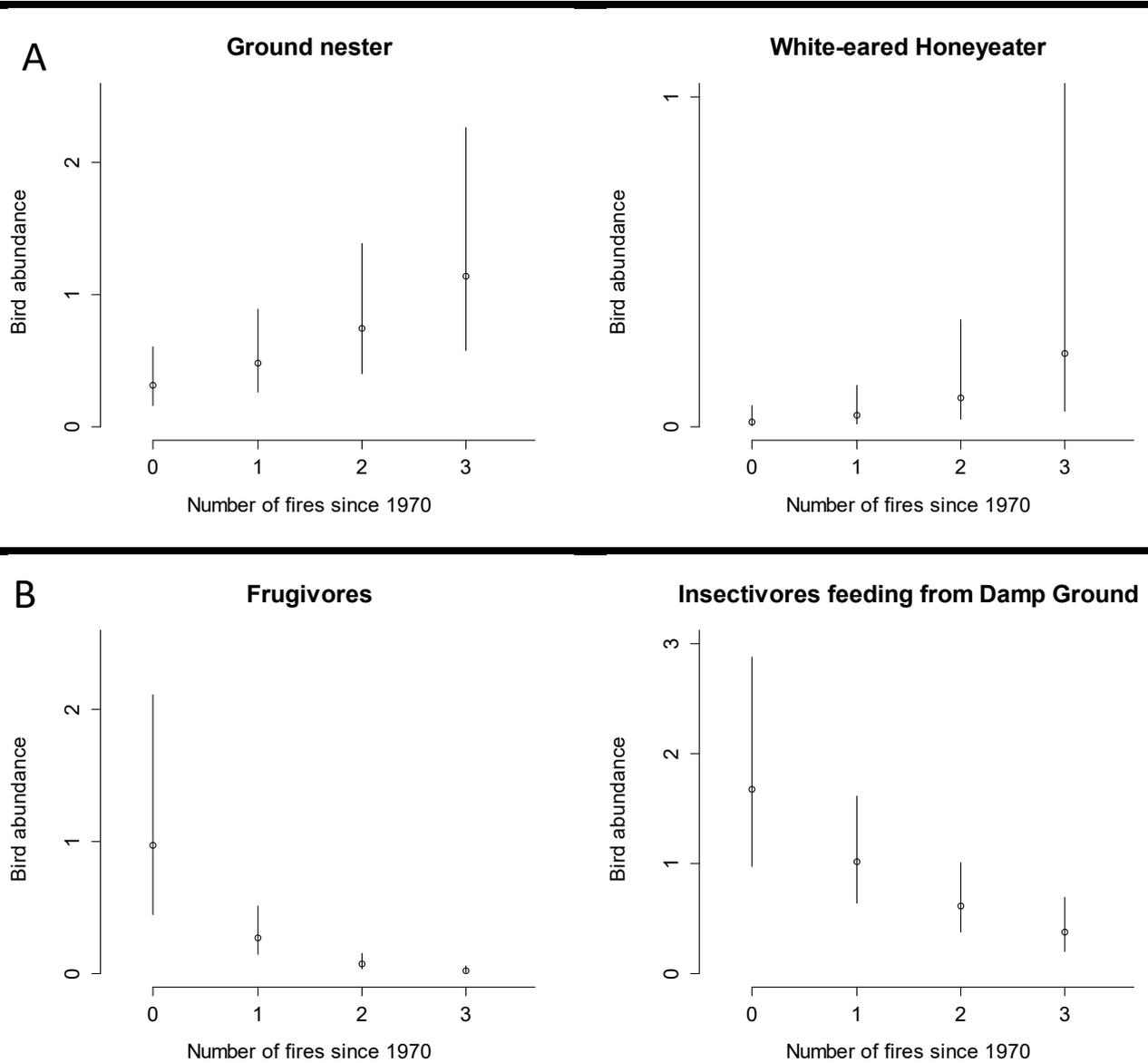
In general, the abundance of birds per guild was negatively related to the number of fires, except for the guilds representing birds feeding or nesting in areas with sparsely vegetated ground cover (insectivores feeding in open ground under trees (OT), granivores feeding from ground (GG) and ground nesters (G); Table 8.3.2). These three guilds were found to have similar or slightly increasing abundance with increasing number of fires (Figure 8.3.1, panels A and C). In contrast, six guilds that are not associated with open ground (insectivores feeding on: damp ground (DG), in trees and shrubs (TS), on bark (B); frugivores (F); large hollow (LH) and small hollow (SH) nesters) all showed decreasing abundance with increasing numbers of fires since 1970 (Figure 8.3.1, panels B and D). Three guilds (carnivores (CV), insectivores feeding in the canopy (C) and granivores feeding in shrubs and trees (GST)) did not show any clear response to increasing number of fires (Table 8.3.2).

**Table 8.3.2.** Relationship between the abundance of birds in each functional guild and the number of fires since 1970. Arrows indicate if the term (NF, NF<sup>2</sup>) was significantly positive or negative (p< 0.05). Non-significant relationships are denoted with ns.

Group	Guild	Relationship between guild and the number of fires since 1970	
		linear term (NF)	quadratic term (NF <sup>2</sup> )
Feeding	Insectivore: Open ground under Trees		↑
	Insectivore: Damp Ground under Trees	↓	
	Insectivore: Bark		↓
	Insectivore: Canopy	ns	
	Insectivore: Trees and Shrubs	↓	
	Nectatvore	↓	
	Frugivore	↓	
	Granivore: Shrubs /Trees	ns	
	Granivore: Ground	↑	
	Carnivore	ns	
Nesting	Ground	↑	
	Large Hollows	↓	
	Small hollows		↓

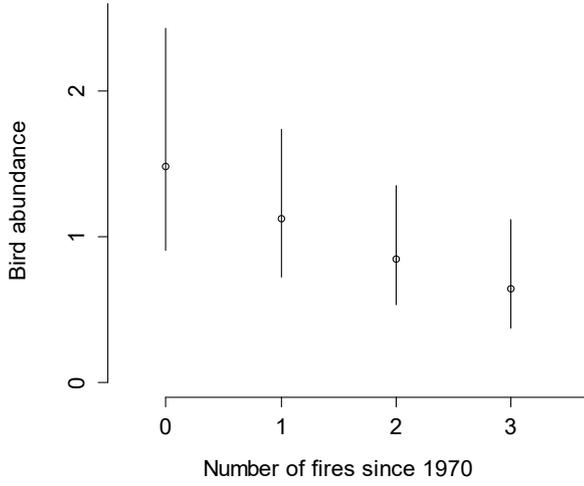
### Relationship between bird guilds and time since fire

Only four guilds showed a significant positive relationship to time since (frugivores (F), nectarivores (N), granivores feeding from ground (GG) and large hollow nesters (LH); Figure 8.3.1, panels E and G). Ground nesters (G) was the only guild to show a negative relationship to time since fire (Figure 8.3.1, panel F). The remaining eight guilds showed no clear relationship with time since fire.

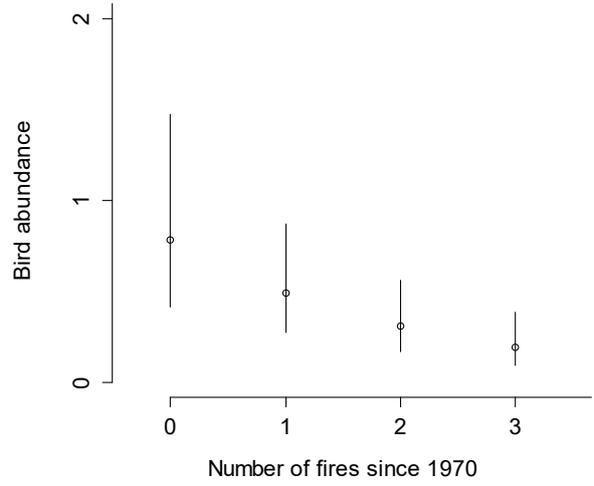


**B** cont.

**Large Hollow nester**

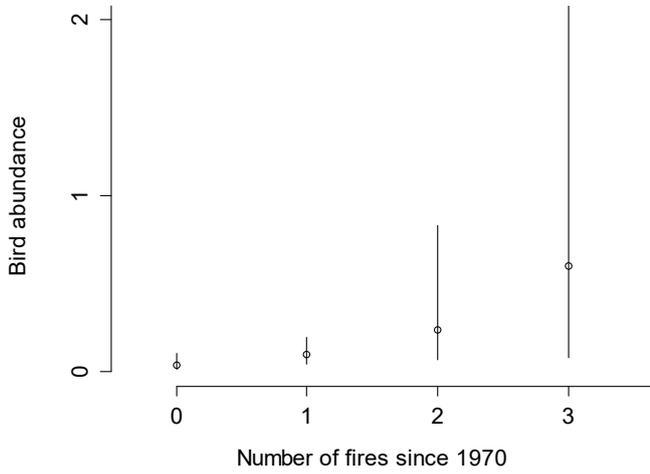


**Yellow-faced Honeyeater**

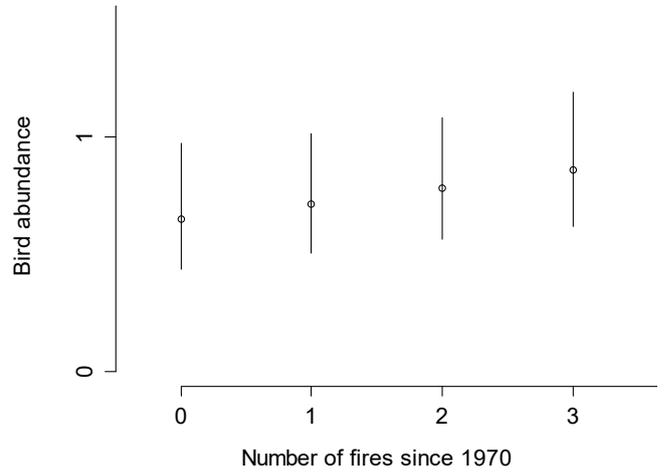


**C**

**Granivore feeding from Ground**

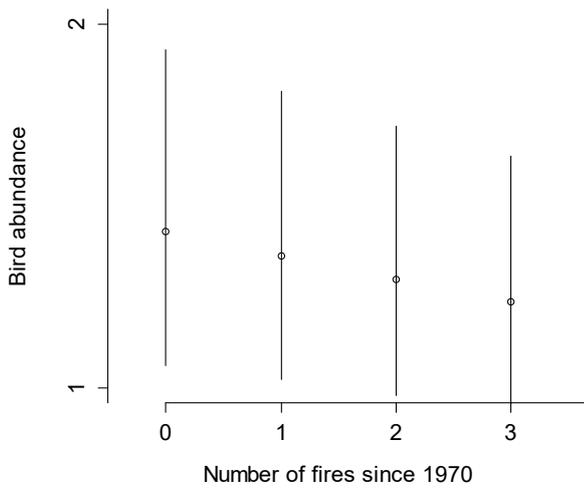


**Insectivore foraging from Open ground under Trees**

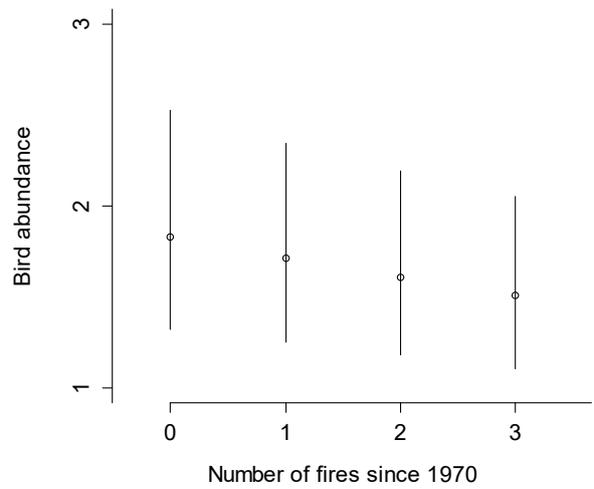


**D**

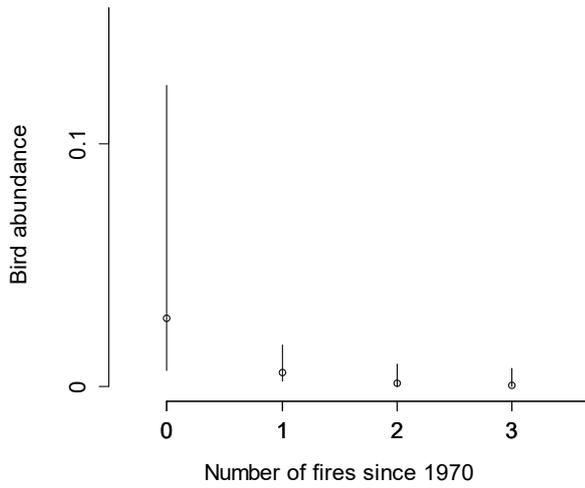
**Insectivore foraging from Bark**



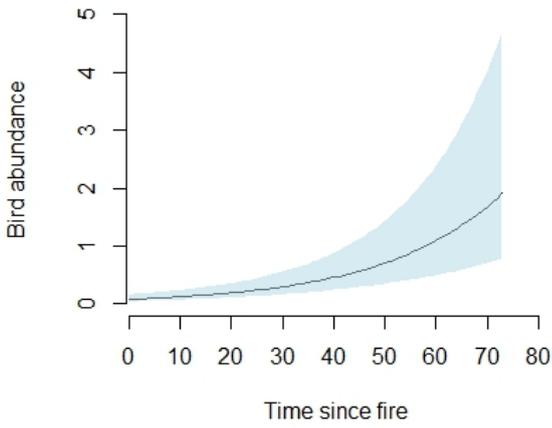
**Small Hollow nester**



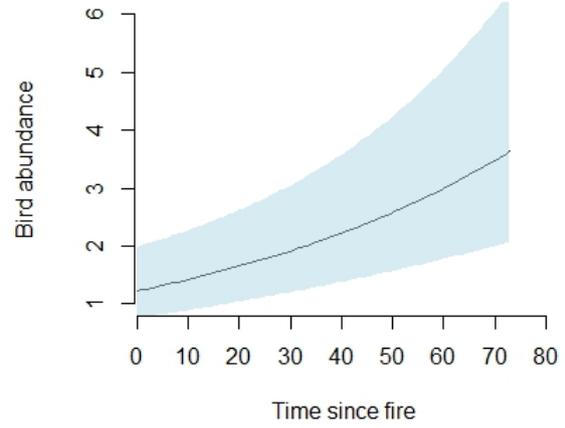
**D** cont. **White-naped Honeyeater**



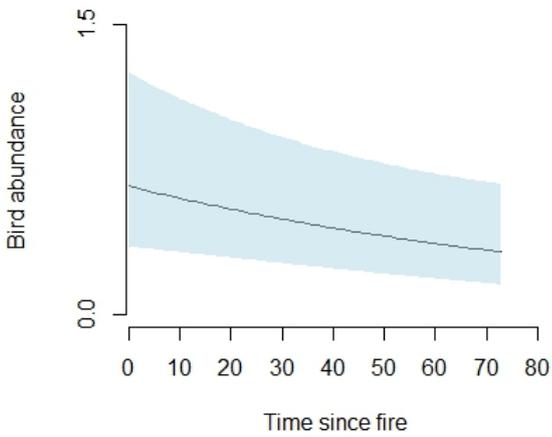
**E** **Frugivore**



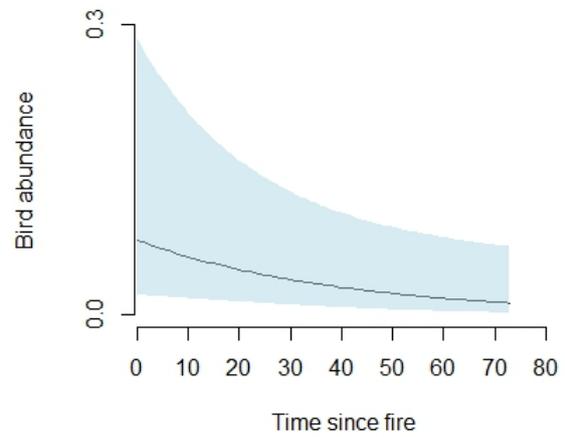
**Nectarivore**

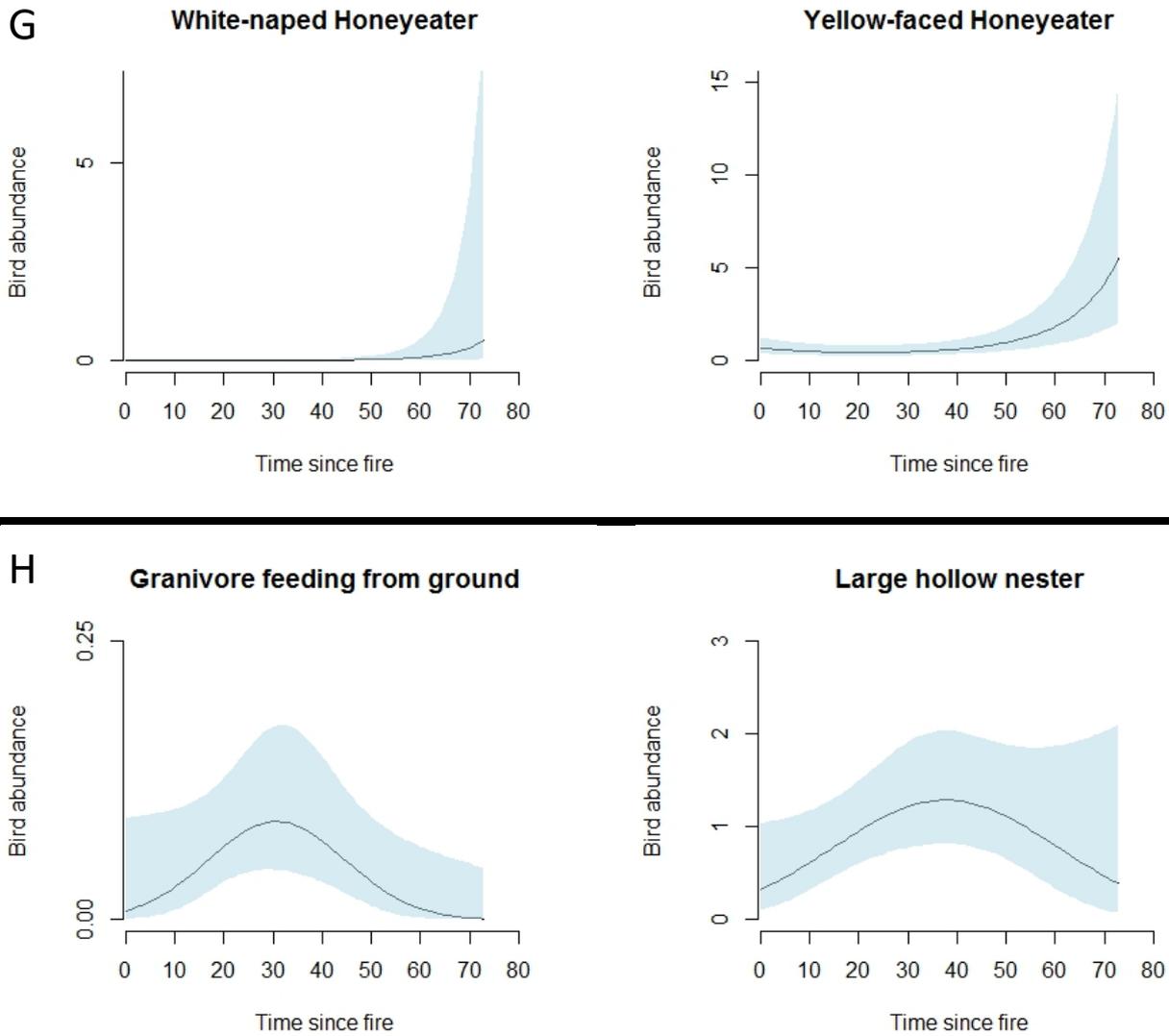


**F** **Ground Nester**



**White-eared Honeyeater**

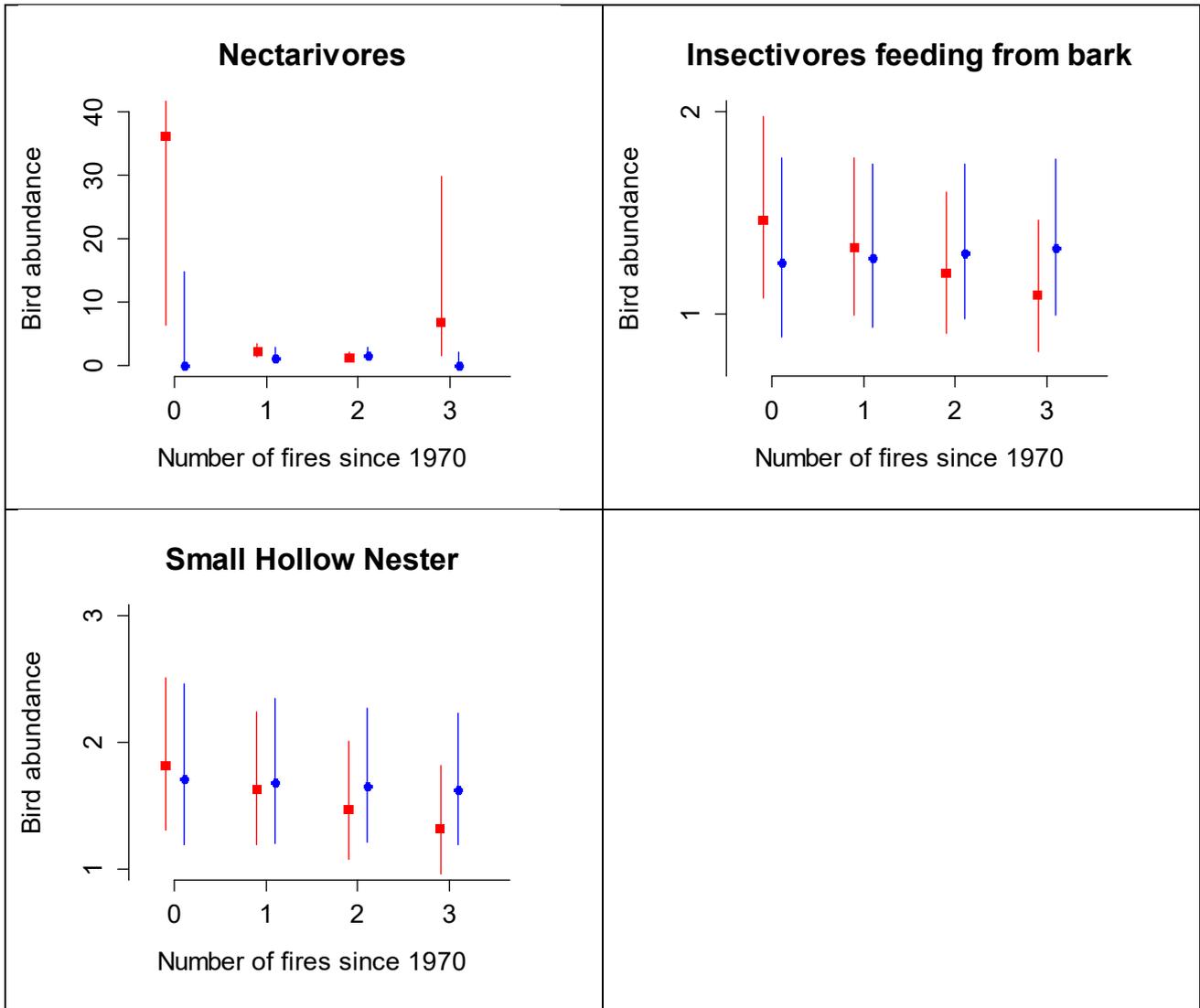




**Figure 8.3.1.** Relationships of functional guild and species to number of fires and time since fire. Panel A: Significant (positive) term for the number of fires since 1970. Panel B: Significant (negative) term for the number of fires since 1970. Panel C: Significant (positive) term for the number of fires since 1970<sup>2</sup> (quadratic term). Panel D: Significant (negative) term for the number of fires since 1970<sup>2</sup> (quadratic). Panel E: Significant (positive) term for time since fire. Panel F: Significant (negative) term for time since fire. Panel G: Significant (positive) term for time since fire<sup>2</sup> (quadratic). Panel H: Significant (negative) term for time since fire<sup>2</sup> (quadratic).

*Interaction between fire type and the number of fires*

There was an interactive effect of number of fires and fire type for three functional guilds (insectivores feeding on bark (B), small hollow nesters (SH) and nectarivores (N); Figure 8.3.2). Where bushfire was the most recent fire, increasing the number of fires in the last 40 years was associated with fewer birds in B and SH guilds. Conversely, where planned fire was the last fire, there was a more muted response to increasing number of fires in the last 40 years in B and SH guilds. Nectarivores showed a complex response following bushfire, whereby abundance declined with one and two fires (compared to no fire since 1970) but then slightly increased with three fires. In comparison, the response of nectarivores to planned fire was muted, with only a slight decline in abundance with three fires compared to one or two fires (Figure 8.3.2). None of the guilds had significant values for an interaction between time since fire and fire type.



**Figure 8.3.2.** Interaction between fire type and number of fires since 1970 for bird guilds. Red squares and blue triangles are mean predicted values for bushfire and planned fire respectively. Red lines and blue vertical lines are upper and lower confidence intervals for bushfire and planned fire respectively. Where “0” fires since 1970, fire type was based on that of the most recent recorded fire before 1970.

### Importance of fire regime at different scales

Outputs from the AIC model selection process (Appendix 3) showed that there were alternative models that performed comparably well with the lowest ranked model. However, the models within 2 AIC of the lowest ranking model generally comprised site based elements of fire regime or only landscape elements of fire regime which provides information about the importance of scale. For instance two of the ground feeding and ground nesting guilds (insectivores feeding on open ground under trees (OT) and ground nesters (G) were better explained by site elements of fire regime (FT, and NF; Table 8.3.3).

Nectarivores were also most strongly associated with site scale elements of fire regime (TSF). Number of fires (linear or quadratic terms) was the most parsimonious model for three guilds (insectivores feeding in trees and shrubs (ST), frugivores (F) and small hollow nesters (SH).

Five guilds (insectivores feeding in: damp ground (DG), bark (B), canopy (C), carnivores (CV) and large hollow (LH) nesters) were more strongly predicted by landscape scale elements of fire regime, particularly the “area of recently burnt vegetation within 2km of the site.” However, model selection indicated that in some cases alternative landscape scale fire measures (e.g. richness and diversity of age classes) performed comparably well, (i.e. were within 2 AIC of the best model). The two granivore guilds (GST and GG) had site or landscape scale predictors within 2 AIC of the model with the lowest AIC value (Table 8.3.3).

**Table 8.3.3.** Model with the lowest AIC for the abundance of birds in each functional guild.

Group	Guild	Scale	Model with lowest AIC	Coefficient	LCL	UCL	RC	RM	
Feeding	Insectivore: Open ground under Trees	Site	FT(BF/PF)	0.45(PF)*	0.19	0.72	0.68	0.04	
	Insectivore: Damp ground under Trees	Landscape	AB2km	-0.59*	-0.48	-0.71	0.61	0.08	
	Insectivore: Bark	Landscape	AB2km	-0.17*	-0.31	-0.03	0.27	0.06	
	Insectivore: Canopy	Landscape	AB2km	-0.36*	-0.48	-0.25	0.66	0.09	
	Insectivore: Trees and Shrubs	Site	NF	-0.25*	-0.39	-0.10	0.62	0.05	
	Nectarivore		Site	TSF	-0.43	-0.99	0.12	0.66	0.05
				+TSF <sup>2</sup>	0.81*	0.26	1.37		
	Frugivore		Site	NF	-1.09*	-1.43	-0.75	0.59	0.08
	Granivore: Shrubs / Trees	Landscape	FAR1km	0.11	-0.05	0.28	0.27	0.03	
	Granivore: Ground	Landscape	FAR2km	-0.46*	-0.85	-0.06	0.20	0.02	
Carnivore	Landscape	AB2km	-0.28*	-0.48	-0.08	0.39	0.05		
Nesting	Ground	Site	NF	0.37*	0.21	0.52	0.46	0.04	
	Large Hollows	Landscape	AB2km	-0.51*	-0.69	-0.33	0.6	0.07	
	Small hollows	Site	NF <sup>2</sup>	-0.16 *	-0.26	-0.06	0.38	0.05	

Coefficients with \* are significant terms

See Table 8.3.1 for explanation of different predictors in each model. Note that AIC model selection process indicates that there were alternative models that performed comparably well. These results are provided in detail in Appendix 3. LCL and UCL denote lower and upper 95% confidence limits respectively. RC: R-square conditional (variation explained by both fixed and random effects), RM: R-square marginal (variation explained by fixed effects).

## Discussion

### *Fire frequency*

The abundance of birds in functional feeding and nesting guilds in Foothill Forests showed contrasting responses to different elements of the fire regime, providing new information for fire management. In particular, most guilds were found to be negatively associated with the number of fires in the last 40 years, except for those feeding or nesting in open ground. For instance, the abundance of birds that feed in damp ground under trees halved with three fires in 40 years (reflecting a mean fire interval of 13 years) compared with sites with no fires in the previous 40 years. This is likely to be related to the loss of critical habitat resources. For example, repeated fires in mixed species eucalypt forest have been shown to reduce coarse woody debris (Aponte *et al.* 2014), which may diminish foraging substrates for birds that feed on insects in damp ground under trees since older logs have capacity for intense water reabsorption at later decay stages (Pichler *et al.* 2012). Frugivores were found to be the functional guild most sensitive to repeated fires, with their abundance approximately halving after a single fire in 40 years compared with unburnt sites. This pattern is likely to be the result of reduction in abundance of fruiting shrubs, in particular *Coprosma quadrifida*. Our analyses of fire effects on vegetation structure showed that increased fire frequency reduced cover in the 1-2 m vegetation stratum, which is mostly comprised of shrubs (see Section 8.1).

### *Fire type*

We used fire type as a proxy for fire severity to reflect broad differences between bushfires (generally higher severity) and planned fire (generally lower severity). Fire severity is important as it mediates the biological legacies post fire thereby influencing habitat quality. In most cases higher severity fires result in more extensive combustion of understorey vegetation so may show stronger relationships with ground nesting and ground feeding birds. Our analyses showed that only one guild, birds that prefer to forage in open ground under trees, had fire type as the most parsimonious model. This is consistent with higher severity bushfires removing ground fuel thereby improving foraging opportunities for this guild.

Few guilds showed evidence of an interaction between the number of fires and fire type suggesting that, in most cases, fire severity does not mediate the effects of repeat fires. However, the interaction was significant for birds feeding on insects from bark: this guild showed a stronger decline in abundance with repeated fires where the last fire was a bushfire, compared with where the last fire was a planned fire. This pattern may reflect greater removal of bark foraging substrate following generally higher severity bushfires compared to generally lower severity planned fire.

The complex response of nectarivores to fire type and number of fires may partly relate to the different habitat preferences of most honeyeaters compared to White-eared Honeyeater. It is known to favour shrubby regrowth 5-10 years after logging or fire, whereas most other honeyeaters prefer mature forest (Loyn 1980). Our results suggest that subdividing the guild or tracking individual species would provide more meaningful trends.

### *Scale of fire regime*

The importance of the scale of a fire regime also varied between guilds. Model selection showed that some guilds had more parsimonious relationships with site scale elements of fire regime (ground nesters, small hollow nesters, frugivores, nectarivores, insectivore feeding in open-ground under trees, insectivore feeding in trees or shrubs) whereas several other guilds has more parsimonious models with landscape scale elements of fire regime (large hollow nesters, carnivores, insectivores feeding on damp ground and insectivores feeding in the canopy).

Of the site-based elements of fire regime, time since fire was the most parsimonious model for only one guild (nectarivores) but also showed significant relationships with four other guilds (frugivore, nectarivore, ground nester, granivore feeding from ground and large hollow nester). In comparison models containing number of fires was the most parsimonious model for four guilds (ground nesters, small hollow nesters, frugivores, and insectivores feeding in trees and shrubs) and showed significant relationships with four other guilds (insectivores feeding from damp ground nectarivores, granivores feeding from ground and large hollow nesters). These results suggest that investigating the effects of repeat fires may be more valuable in understanding the persistence of most bird guilds compared to that associated with time since fire.

Of the landscape scale elements of fire regime the negative effects of large recent fires (three years prior to the survey) was the most parsimonious model for five guilds (insectivores feeding from each of damp ground, bark and canopy; large hollow nesters; and carnivores). This pattern is consistent with results from wetter forests in the Central Highlands of Victoria and may stem from the burned areas around sites acting as population sinks (Lindenmayer *et al.* 2014). In addition to this, birds are mobile and many species have large home ranges and these attributes may be better reflected by landscape variables than site variables. This was also shown in previous studies where substantial reductions in bird populations were detected in the three years following severe bushfire (Loyn 1997).

### *Other factors*

A substantial proportion of the variance of the models of all bird functional guilds was explained by season, observer and time of day, and suggests that future monitoring and research could be improved by greater consistency in those aspects, or at least by ensuring that these factors are accounted for when tracking trends over time. Overall there was little variance explained by the fire regime elements contained in the models of bird abundance, indicating that other environmental drivers, such as elevation and topography, may be more influential in explaining the patterns observed (Lindenmayer *et al.* 2014). Inaccuracies in fire history, particularly in relation to fire severity, may also have obscured the true signal of fire regime on birds.

### **Conclusion**

Analysis of bird data from Foothill Forests suggest that differences in site and landscape elements of fire regime are associated with differences in abundance of several nesting and feeding guilds. The relationship between the extent of recent fires at the landscape scale and abundance of several guilds suggests this particular element of fire regime may provide a useful indicator of ecosystem trends. Considering that the minimum tolerable fire interval in foothill forest vegetation spans from 8-25 years (average ~ 13; Cheal 2010), there is a risk that continued burning at this rate uniformly across the state could substantially reduce the abundance of at least two functional guilds (frugivores and birds foraging on damp ground under trees). Increasing the minimum interval between fires is more likely to support the objective of maintaining or improving ecosystem resilience. Where shorter intervals are deemed necessary to support other objectives, consideration should be given to the proximity of these more frequently burnt patches to areas burnt at longer intervals, as these less frequently burnt patches may support the persistence of these more fire sensitive guilds.

## 8.4 Mammals

We investigated the relationships of native terrestrial mammal species richness (total richness and richness of critical weight range (35 and 5500g) species) and occurrence (presence/absence) of individual species to fire regime and environmental and vegetation variables. The aim was to determine the relative influence of these potential drivers on mammal communities. Mammal data were derived from camera trapping surveys.

### Methods

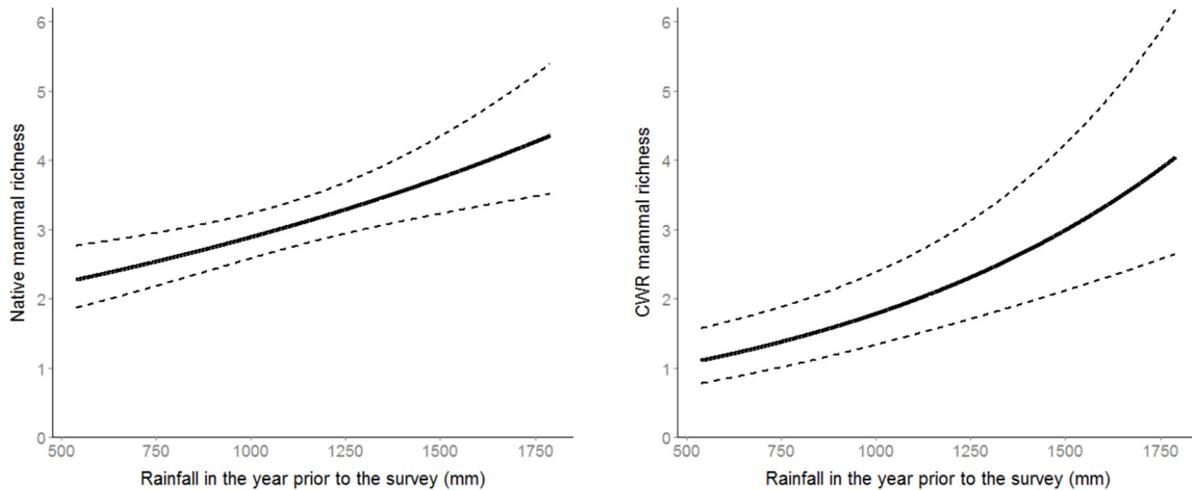
For species richness, a two stage approach to modelling was used. First, univariate models were constructed relating richness to a range of variables representing different metrics of three 'themes': rainfall, topography and vegetation. The best model from each theme was selected using AIC and the covariate in this model was used in multivariate GLMMs that also included the fire regime variables TSF, number of fires since 1970 and fire age diversity and richness within a 500m radius of sites, plus study region. A model selection procedure was used to identify the combination of variables that best explained variation in species richness (for details of analyses see Appendices 4 and 5).

Species occurrence analyses similarly involved several steps. In this case, generalised additive mixed models (GAMMs) were used for modelling. First, univariate models of species presence/absence in relation to several fire regime attributes were constructed. The fire regime attributes examined were: TSF, fire type, mean IFI, number of fires, and diversity and richness of fire ages within 1000m radius of sites. A second set of models examined the influence of TSF, EVD group (foothills = Foothills Forest, tall mixed forest, forby forest; heathy = grassy/heathy dry forest, damp scrub, heathland (sands), moist = moist forest, riparian (higher rainfall), treed swampy wetland), presence/absence of predators (cats or foxes; from camera data), cumulative proportion of mean annual rainfall (CPMR; see Appendix 5 for details), visible sky, relative cover and survey effort (camera trap nights per site). Lastly, species' occurrences were modelled in relation to vegetation structural variables (see Section 8.1).

### Results

#### *Species richness*

Species richness of native mammals ranged from 0 to 9 (median 3) from a pool of 23 species across all sites. Richness of critical weight range species (CWR) ranged from 0 to 7 (median 2) from a pool of 16 species across all sites. None of the four fire variables tested appeared in the best models for predicting either native or CWR richness. The best model for native richness contained relative cover and rainfall at a site in the year prior to the survey. Species richness was positively related to total rainfall at a site in the year prior to the survey (Figure 8.4.1a) and negatively related to relative cover at a site. The best model for CWR richness contained rainfall at a site in the year prior to the survey alone, with the relationship again being positive (Figure 8.4.1b).



**Figure 8.4.1.** Modelled relationship between rainfall and richness of (a) native mammal species, (b) critical weight range mammal species. Dashed lines are 95% confidence intervals.

### Species occurrence

Across all study programs, including repeat surveys at some sites and multiple cameras at sites, a total of 23,931 effective camera trap nights were conducted. From this effort a total of 30 mammal species were detected (22 native, 8 introduced). Many of these species were encountered infrequently (<10% of sites). Analysis was conducted for six species, chosen according to three factors: (1) availability of sufficient data (detected at >10% of sites); (2) presence across whole study area; and (3) considered likely to be regularly attracted to baits, thereby making camera traps a reliable survey method. These species were *Antechinus* sp. (records of *Antechinus agilis* and *A. swainsonii* combined due to difficulty in differentiating between these species from camera images; 158 sites; 34%), Bush Rat (*Rattus fuscipes*; 218; 47%), Short-beaked Echidna (*Tachyglossus aculeatus*; 182; 39%), Mountain Brushtail Possum (*Trichosurus cunninghami*) (95; 20%), Common Brushtail Possum (*Trichosurus vulpecula*; 99; 21%), and Black Wallaby (*Wallabia bicolor*; 361; 78%).

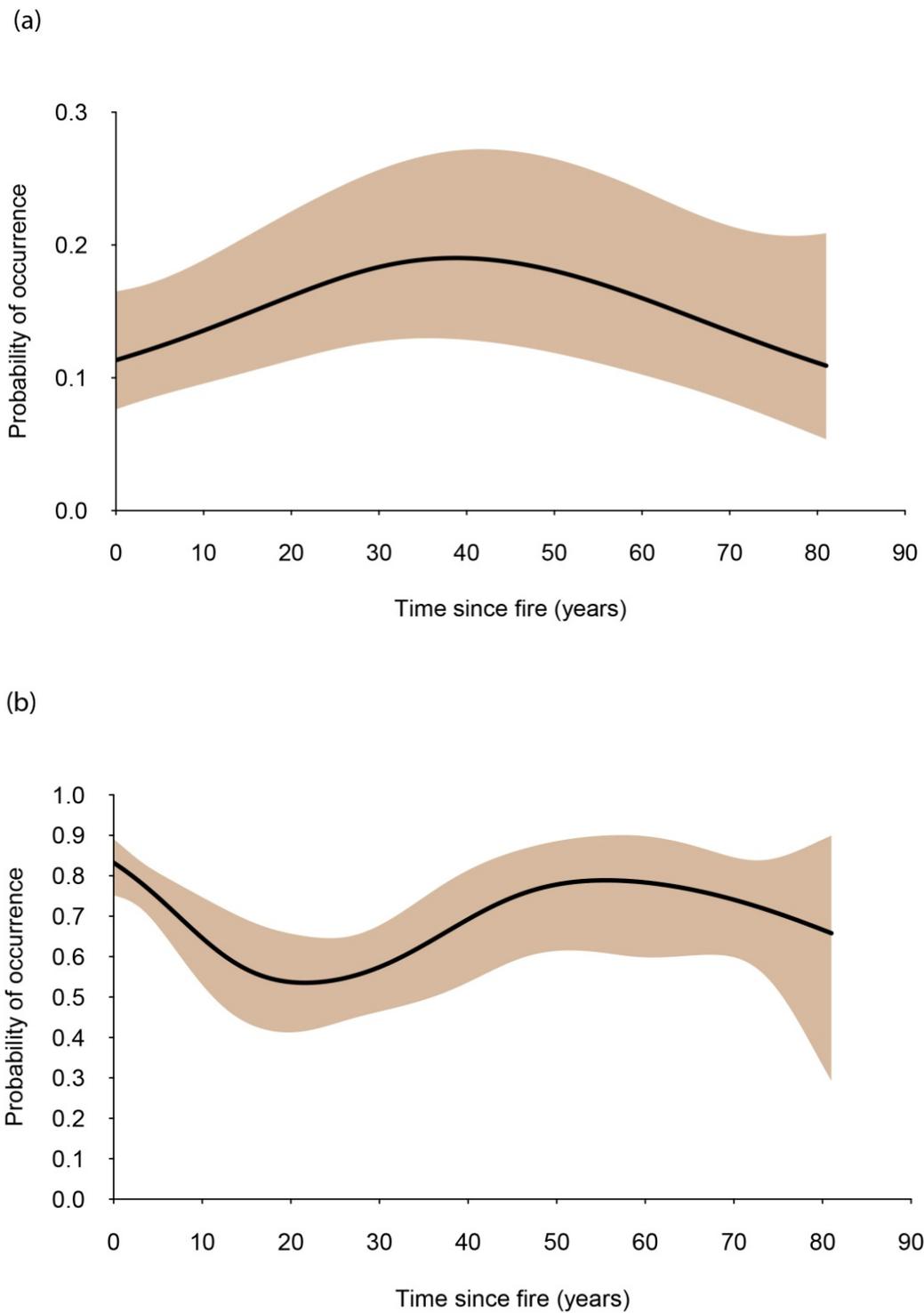
At least one fire variable had a significant influence on the occurrence of each of the mammal species investigated. However, different species responded to different variables (Table 8.4.1). The variance explained by models was universally low, being 5% or less in all but one case (Table 8.4.1). Thus, although influential fire variables were identified, the majority of variance remained unexplained for all species.

Time since fire was identified as an influence on both *T. aculeatus* and *W. bicolor*. GAMM predicted values indicate that the probability of occurrence of *T. aculeatus* increases with time since fire up to 40 years but then gradually declines as sites which remain unburnt for longer periods (Figure 8.4.2a). The occurrence of *W. bicolor* initially declines in the first 20 years post-fire but then increases to reach a peak after 50-60 years (Figure 8.4.2b). Mean inter-fire interval was an influence on three species: *Antechinus* sp., *T. cunninghami* and *W. bicolor*. Probability of occurrence of *Antechinus* sp. was predicted to decline linearly with increasing mean inter-fire interval, while the reverse was true for *W. bicolor* (Figure 8.4.3a and b). *T. cunninghami* displayed a slight increase in occurrence with increasing mean inter-fire interval but confidence intervals were large. The number of fires experienced by a site had a positive linear influence on *Antechinus* sp. and a negative linear influence on *T. aculeatus*. Fire type (planned burn or bushfire) was found to influence *R. fuscipes*, *T. aculeatus* and *T. vulpecula* (Table 8.4.1). Probability of occurrence was predicted to be higher for both *R. fuscipes* and *T. aculeatus* when the most recent fire was a bushfire, while *T. vulpecula* was less likely to occur at sites where the most recent fire was a bushfire. The two spatial fire variables (diversity of fire ages and richness of fire ages within 1000 m buffer) were not found to influence any species (Table 8.4.1).

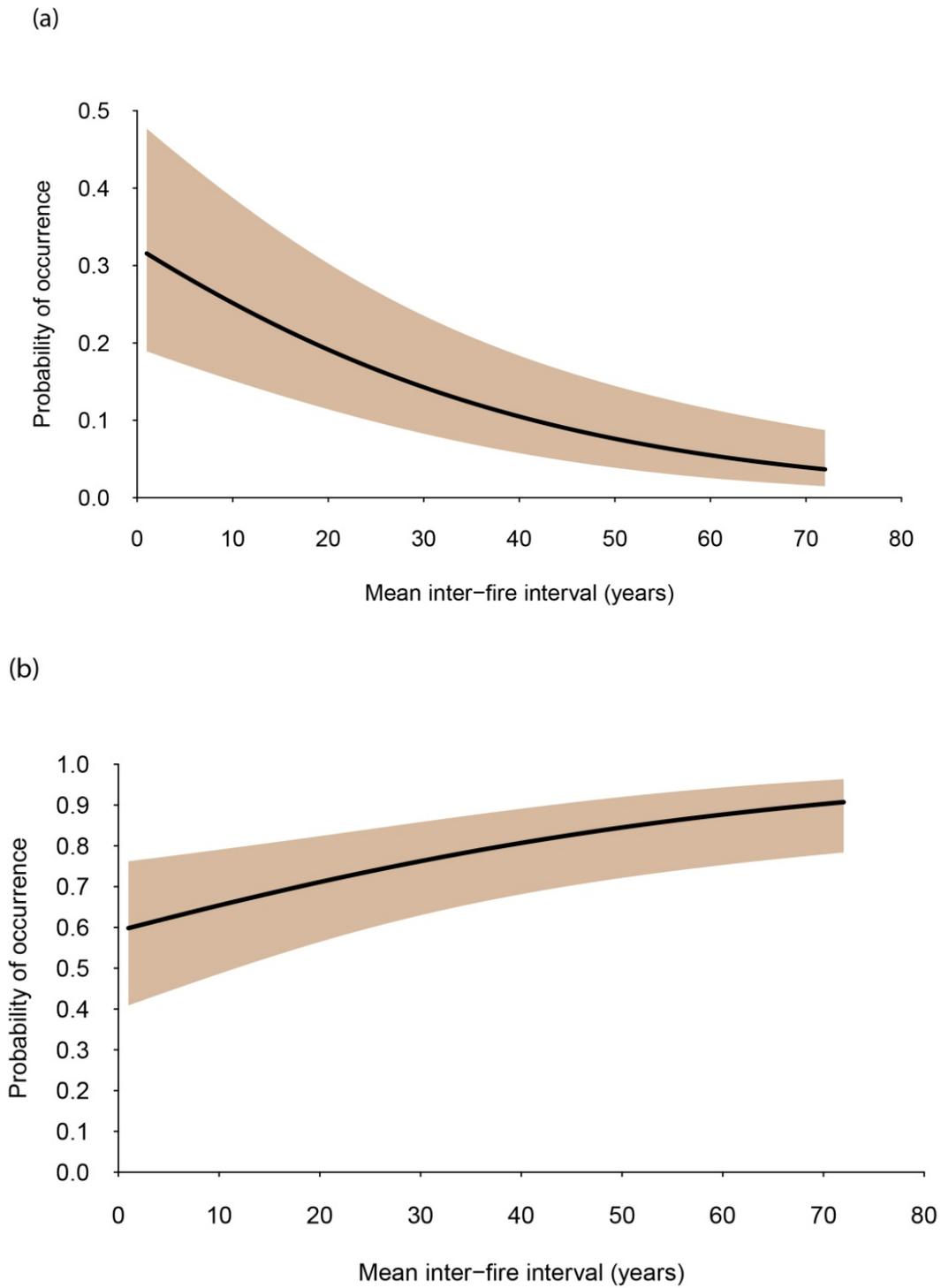
**Table 8.4.1.** Results from univariate generalised additive mixed models of the relationship of the presence/absence of mammal species to fire variables. Influential variables ( $P < 0.05$ ) are in bold. Variance explained ( $D^2$ ) is shown only for influential variables.  $s(\dots)$  = explanatory variable was fitted as a non-parametric smoothing term e.d.f. = estimated degrees of freedom for each smoothing term.

Species	Fire variable	e.d.f.	P-value	$D^2$
<i>Antechinus sp</i>	$s(\text{Time since fire})$	1.0	0.087	
	Fire type: bushfire <sup>a</sup>	n.a. <sup>b</sup>	0.870	
	<b><math>s(\text{Mean inter-fire-interval})</math></b>	<b>1.0</b>	<b>0.000</b>	0.02
	<b><math>s(\text{Number of fires})</math></b>	<b>1.0</b>	<b>0.001</b>	<0.01
	$s(\text{Diversity of fire ages (1000 m)})$	1.0	0.400	
	$s(\text{Richness of fire ages (1000 m)})$	1.0	0.525	
<i>Rattus fuscipes</i>	$s(\text{Time since fire})$	1.0	0.486	
	<b>Fire type: bushfire<sup>a</sup></b>	<b>n.a.<sup>b</sup></b>	<b>0.001</b>	<0.01
	$s(\text{Mean inter-fire-interval})$	1.0	0.557	
	$s(\text{Number of fires})$	1.0	0.986	
	$s(\text{Diversity of fire ages (1000 m)})$	1.0	0.764	
	$s(\text{Richness of fire ages (1000 m)})$	1.0	0.185	
<i>Tachyglossus aculeatus</i>	<b><math>s(\text{Time since fire})</math></b>	<b>2.1</b>	<b>0.022</b>	0.05
	<b>Fire type: bushfire<sup>a</sup></b>	<b>n.a.<sup>b</sup></b>	<b>0.005</b>	0.04
	$s(\text{Mean inter-fire-interval})$	1.0	0.551	
	<b><math>s(\text{Number of fires})</math></b>	<b>1.0</b>	<b>0.042</b>	0.03
	$s(\text{Diversity of fire ages (1000 m)})$	1.0	0.672	
	$s(\text{Richness of fire ages (1000 m)})$	1.0	0.910	
<i>Trichosurus cunninghami</i>	$s(\text{Time since fire})$	1.0	0.939	
	Fire type: bushfire <sup>a</sup>	n.a. <sup>b</sup>	0.052	
	<b><math>s(\text{Mean inter-fire-interval})</math></b>	<b>1.0</b>	<b>0.000</b>	0.02
	$s(\text{Number of fires})$	1.0	0.505	
	$s(\text{Diversity of fire ages (1000 m)})$	1.0	0.382	
	$s(\text{Richness of fire ages (1000 m)})$	1.0	0.711	
<i>Trichosurus vulpecula</i>	$s(\text{Time since fire})$	1.0	0.057	
	<b>Fire type: bushfire<sup>a</sup></b>	<b>n.a.<sup>b</sup></b>	<b>0.001</b>	0.11
	$s(\text{Mean inter-fire-interval})$	1.0	0.125	
	$s(\text{Number of fires})$	1.0	0.575	
	$s(\text{Diversity of fire ages (1000 m)})$	1.0	0.181	
	$s(\text{Richness of fire ages (1000 m)})$	1.0	0.533	
<i>Wallabia bicolor</i>	<b><math>s(\text{Time since fire})</math></b>	<b>3.3</b>	<b>0.000</b>	0.05
	Fire type: bushfire <sup>a</sup>	n.a. <sup>b</sup>	0.771	
	<b><math>s(\text{Mean inter-fire-interval})</math></b>	<b>1.0</b>	<b>0.002</b>	<0.01
	$s(\text{Number of fires})$	1.0	0.460	
	$s(\text{Diversity of fire ages (1000 m)})$	1.0	0.174	
	$s(\text{Richness of fire ages (1000 m)})$	2.1	0.046	

<sup>a</sup> The reference category for the categorical variable 'Fire type' was 'planned burn'. <sup>b</sup> Estimated degrees of freedom not applicable for parametric terms (i.e., explanatory variables not fitted with smoothing terms).



**Figure 8.4.2.** Predicted probability of occurrence as a function of time since fire for: (a) *Tachyglossus aculeatus*; and (b) *Wallabia bicolor*. Predictions were generated from generalised additive mixed models containing a single fire variable. Shaded areas represent 95% confidence intervals for predictions.



**Figure 8.4.3.** Predicted probability of occurrence as a function of mean inter-fire interval for: (a) *Antechinus sp.*; and (b) *Wallabia bicolor*. Predictions were generated from generalised additive mixed models containing a single fire variable. Shaded areas represent 95% confidence intervals for predictions.

Models including time since fire and other environmental variables had low explanatory power, with the variance explained by models being  $\leq 5\%$  for all species (Table 8.4.2). Again, taxon-specific responses to variables were observed. TSF was found to influence *Antechinus* sp. only. TSF interacted with EVD for this species; the probability of occurrence of *Antechinus* sp. declines with increasing time since fire in the foothills group, but no relationship with time since fire was evident in the other EVD groups (Figure 8.4.4). Irrespective of time since fire, *Antechinus* sp. was more likely to occur in the moist EVD group. A similar association was observed for *R. fuscipes*, while *W. bicolor* was less likely to occur in the moist EVD group compared to the foothills and heathy groups. EVD group was also influential for *T. vulpecula*, with this species having a higher probability of occurrence in the heathy group. The topographic position of a site (visible sky) was important for *Antechinus* sp., *R. fuscipes*, *T. aculeatus* and *W. bicolor* (Table 2). *Antechinus* sp., *T. aculeatus* and *W. bicolor* displayed a positive relationship with this variable, indicating a preference for sites on less undulating terrain. In contrast, *R. fuscipes* was more likely to occur at sites with a low percentage of visible sky (i.e., gullies). The relative cover of canopy trees (relative cover) at a site was a positive influence on the occurrence of *R. fuscipes* and *T. cunninghami*, but a negative influence on *T. vulpecula*. *Antechinus* sp. and *W. bicolor* responded to introduced predators, with both species less likely to occur at sites where such predators were detected. Finally, the cumulative proportion of long-term mean annual rainfall (CPMR) at a site was important only for *Antechinus* sp., with probability of occurrence of this species increasing with higher rainfall in the preceding three years.

**Table 8.4.2.** Results from generalised additive mixed models used to investigate the relative influence of time since fire and other environmental variables on the presence/absence of mammal species. A single model including all variables was fitted for each species. Influential variables ( $P < 0.05$ ) are emboldened. Variance explained ( $D^2$ ) is shown for each model.  $s(\dots)$  Indicates that an explanatory variable was fitted as a non-parametric smoothing term. e.d.f. represents the estimated degrees of freedom for each smoothing term.

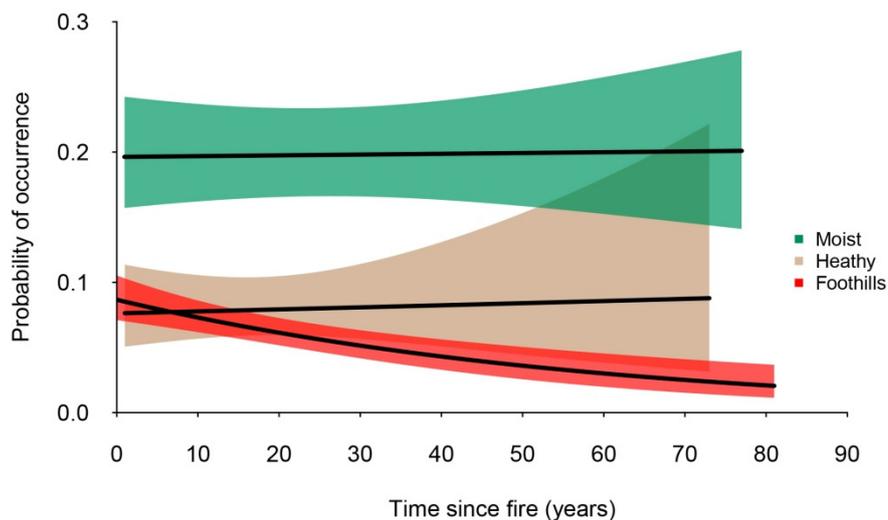
Species	Explanatory variable	e.d.f.	$P$ -value	$D^2$
<i>Antechinus sp</i>	<b><math>s(\text{Time since fire: foothills})</math></b>	<b>1.0</b>	<b>0.030</b>	0.05
	$s(\text{Time since fire: heathy})$	1.0	0.910	
	$s(\text{Time since fire: moist})$	1.0	0.957	
	EVD group: heathy <sup>a</sup>	n.a. <sup>c</sup>	0.398	
	<b>EVD group: moist<sup>a</sup></b>	<b>n.a.<sup>c</sup></b>	<b>0.000</b>	
	<b>Predator p/a: present<sup>b</sup></b>	<b>n.a.<sup>c</sup></b>	<b>0.024</b>	
	<b><math>s(\text{CPMR})</math></b>	<b>1.0</b>	<b>0.000</b>	
	<b><math>s(\text{Visible sky})</math></b>	<b>1.0</b>	<b>0.007</b>	
	$s(\text{Relative cover})$	1.0	0.145	
	$s(\text{Survey effort})$	1.0	0.689	
<i>Rattus fuscipes</i>	$s(\text{Time since fire: foothills})$	1.0	0.620	0.02
	$s(\text{Time since fire: heathy})$	1.0	0.216	
	$s(\text{Time since fire: moist})$	1.0	0.506	
	EVD group: heathy <sup>a</sup>	n.a. <sup>c</sup>	0.139	
	<b>EVD group: moist<sup>a</sup></b>	<b>n.a.<sup>c</sup></b>	<b>0.000</b>	
	Predator p/a: present <sup>b</sup>	n.a. <sup>c</sup>	0.815	
	$s(\text{CPMR})$	1.0	0.151	
	<b><math>s(\text{Visible sky})</math></b>	<b>1.0</b>	<b>0.035</b>	
	<b><math>s(\text{Relative cover})</math></b>	<b>1.0</b>	<b>0.000</b>	
	$s(\text{Survey effort})$	1.0	0.246	
<i>Tachyglossus aculeatus</i>	$s(\text{Time since fire: foothills})$	1.8	0.096	0.05
	$s(\text{Time since fire: heathy})$	1.0	0.220	
	$s(\text{Time since fire: moist})$	1.0	0.735	
	EVD group: heathy <sup>a</sup>	n.a. <sup>c</sup>	0.943	
	EVD group: moist <sup>a</sup>	n.a. <sup>c</sup>	0.198	
	Predator p/a: present <sup>b</sup>	n.a. <sup>c</sup>	0.136	
	$s(\text{CPMR})$	1.0	0.607	
	<b><math>s(\text{Visible sky})</math></b>	<b>1.0</b>	<b>0.025</b>	
	$s(\text{Relative cover})$	1.0	0.222	
	<b><math>s(\text{Survey effort})</math></b>	<b>1.0</b>	<b>0.001</b>	
<i>Trichosurus cunninghami</i>	$s(\text{Time since fire: foothills})$	1.0	0.541	<0.01
	$s(\text{Time since fire: heathy})$	1.0	0.634	
	$s(\text{Time since fire: moist})$	1.0	0.567	
	EVD group: heathy <sup>a</sup>	n.a. <sup>c</sup>	0.766	
	EVD group: moist <sup>a</sup>	n.a. <sup>c</sup>	0.904	
	Predator p/a: present <sup>b</sup>	n.a. <sup>c</sup>	0.163	
	$s(\text{CPMR})$	1.0	0.174	
	$s(\text{Visible sky})$	1.0	0.241	

Species	Explanatory variable	e.d.f.	P-value	D <sup>2</sup>
	<b>s(Relative cover)</b>	<b>1.0</b>	<b>0.005</b>	
	<b>s(Survey effort)</b>	<b>2.3</b>	<b>0.035</b>	
<i>Trichosurus vulpecula</i>	s(Time since fire: foothills)	1.0	0.106	<0.01
	s(Time since fire: heathy)	1.0	0.536	
	s(Time since fire: moist)	1.0	0.353	
	<b>EVD group: heathy<sup>a</sup></b>	<b>n.a.<sup>c</sup></b>	<b>0.000</b>	
	EVD group: moist <sup>a</sup>	n.a. <sup>c</sup>	0.228	
	Predator p/a: present <sup>b</sup>	n.a. <sup>c</sup>	0.182	
	s(CPMR)	1.0	0.204	
	s(Visible sky)	1.0	0.211	
	<b>s(Relative cover)</b>	<b>1.0</b>	<b>0.001</b>	
	<b>s(Survey effort)</b>	<b>1.0</b>	<b>0.003</b>	
<i>Wallabia bicolor</i>	s(Time since fire: foothills)	1.0	0.583	<0.01
	s(Time since fire: heathy)	1.0	0.051	
	s(Time since fire: moist)	1.0	0.059	
	EVD group: heathy <sup>a</sup>	n.a. <sup>c</sup>	0.082	
	<b>EVD group: moist<sup>a</sup></b>	<b>n.a.<sup>c</sup></b>	<b>0.015</b>	
	<b>Predator p/a: present<sup>b</sup></b>	<b>n.a.<sup>c</sup></b>	<b>0.043</b>	
	s(CPMR)	1.0	0.074	
	<b>s(Visible sky)</b>	<b>1.0</b>	<b>0.000</b>	
	s(Relative cover)	1.0	0.116	
	<b>s(Survey effort)</b>	<b>2.0</b>	<b>0.000</b>	

<sup>a</sup> The reference category for the categorical variable 'EVD group' was 'foothills'

<sup>b</sup> The reference category for the categorical variable 'Predator p/a' was 'absent'

<sup>c</sup> Estimated degrees of freedom not applicable for parametric terms (i.e., explanatory variables not fitted with smoothing terms).



**Figure 8.4.4.** Predicted probability of occurrence of *Antechinus* sp. as a function of time since fire for each EVD group. Predictions were generated from a generalised additive mixed model. Shaded areas represent  $\pm 1$  standard error for predictions.

Models investigating relationships between the presence-absence of species and site habitat variables explained  $\leq 3\%$  of variance in the data for all species (see Table 3, Appendix 5). Structural diversity of vegetation was a positive influence on the occurrence of *R. fuscipes*, as was cover of vegetation  $>0.5 < 1.0$  m and  $>4.0$  m. Occurrence of *Antechinus* sp. displayed a negative association with litter depth. Bare ground cover and vegetation cover  $>4.0$  m were both identified as influences on the occurrence of *T. cunninghami*, but confidence intervals around predictions were wide in both cases. Increasing cover of vegetation  $<0.5$  m was found to be a positive influence on the occurrence of *W. bicolor*. None of the habitat variables considered were important for *T. aculeatus* or *T. vulpecula*.

## Discussion

We were unable to detect a relationship between species richness and fire history at either the site or landscape scale in this study. This is despite a relatively large number of sites and a relatively restricted suite of vegetation types. These findings are at odds with other studies examining mammal responses to fire regimes (e.g. Olsen *et al.* 2003, Kelly *et al.* 2010). However, these studies considered arid zone small mammal communities that primarily consisted of marsupials in the family Dasyuridae (Kelly *et al.* 2010) and rodents (Order: Rodentia, both studies). In this study we considered a broader assemblage of forest species including rodents and dasyurids but also macropods, bandicoots and possums. It may be that these forest dwelling species respond differently (or not at all) to fire history compared with arid zone species. In addition, our findings indicate that individual species respond differently to fire, possibly depending on their habitat and feeding requirements. These different responses may cancel each other out, leading to no overall pattern in species richness in relation to fire regime.

The occurrence of the mammal species investigated was influenced by a variety of factors. When analysed in isolation, fire variables were influential for all species. The type of the most recent fire (planned burn/ bushfire) influenced the occurrence of three species, indicating that the severity of fire events is important. Interestingly, two of the species that responded to this variable were more likely to occur when the most recent fire was a bushfire. Mean inter-fire interval also influenced the occurrence of three species, while time since fire and the total number of fires experienced at a site each influenced two species. Although analysed separately, these latter three variables tend to be at least partly inter-related (e.g. sites not burnt for decades have generally not experienced a high number of total fires, and sites with long mean inter-fire intervals have also generally not experienced a high number of total fires).

When analysed in conjunction with broad environmental variables, time since fire was found to only influence *Antechinus* sp., and only in the foothills EVD group. Environmental variables such as EVD group, topographic position (visible sky) and relative canopy cover were more consistently identified as predictors of mammal occurrence. Similarly, rainfall and vegetation cover were identified as the best predictors of mammal richness. This suggests that, across the large spatial and temporal extent of this study, the influence of fire is not as pervasive as that of other environmental variables for the mammal species considered.

## 9 Cross-taxonomic comparison of relationships between species and fire regime components

The analyses presented in Section 8 focussed on the responses of particular biotic groups to fire and environmental variables. In this section we compare responses to fire regime variables across taxonomic groups. First, we develop statistical models of species' responses to fire regime characteristics. This includes quantifying the role of the time since last fire, the interval between multiple fires and the type of fire (low intensity planned burns and high intensity bushfire). Second, we compare and synthesize the responses to fire of three taxonomic groups: birds, small mammals and plants.

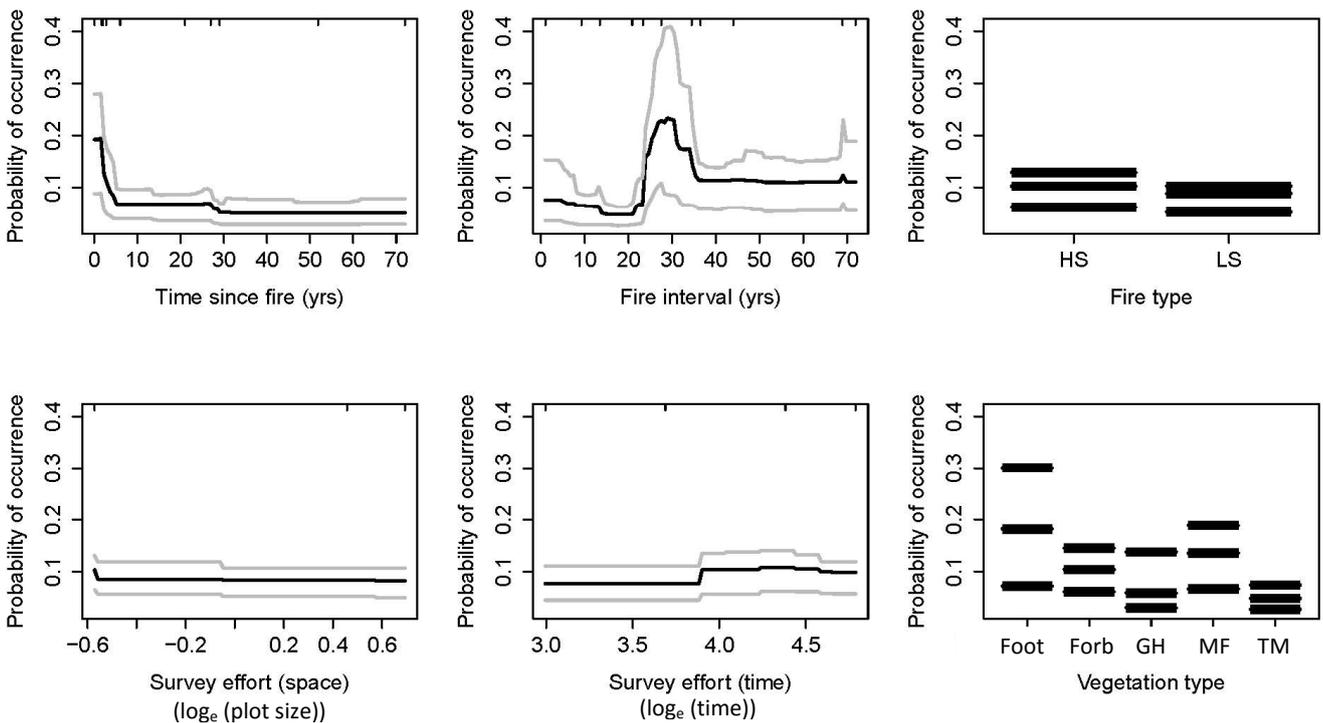
We used boosted regression trees (BRTs) to determine animal and plant responses to fire regimes. BRTs provide a flexible framework to model non-linear relationships and to account for interactions between variables (Elith *et al.* 2008; Appendix 6). We modelled the response variable as the presence or absence of a species at a site. The model for each species included five or six predictor variables (depending on the taxonomic group). For all groups, we included time since last fire, mean interval between fires and type of fire (lower intensity planned burns or higher intensity bushfire). We also included EVD as a predictor to account for local variation in vegetation. Additional variables relating to survey effort, that were specific to each taxonomic group, were included in models to account for spatial and temporal variation in sampling. For plants and birds, spatial survey effort was included in the models as  $\log_e$  (area of survey plot). For plants, temporal survey effort was represented by the models as the number of times a quadrat was surveyed. For birds, temporal survey effort was modelled as  $\log_e$  (total survey time). For small mammals, survey effort was included in the model as  $\log_e$  (number of camera trap nights).

We modelled all bird, small mammal and plant species that occurred at  $\geq 10\%$  of sites and that were present in at least three sub regions in our study area (including the Otway Ranges, Central, West, Gippsland and North East regions). We assessed the strength of these relationships by plotting model predictions and by calculating the percentage deviance explained by each model. The ability of models to accurately discriminate between a species' presence and absence was assessed using the area under the curve (AUC) of a receiver operating characteristic (ROC) plot. Models with AUC values of  $< 0.7$ ,  $0.7-0.9$  and  $> 0.9$  were interpreted as offering poor, useful and very good discrimination, respectively (Pearce and Ferrier 2000).

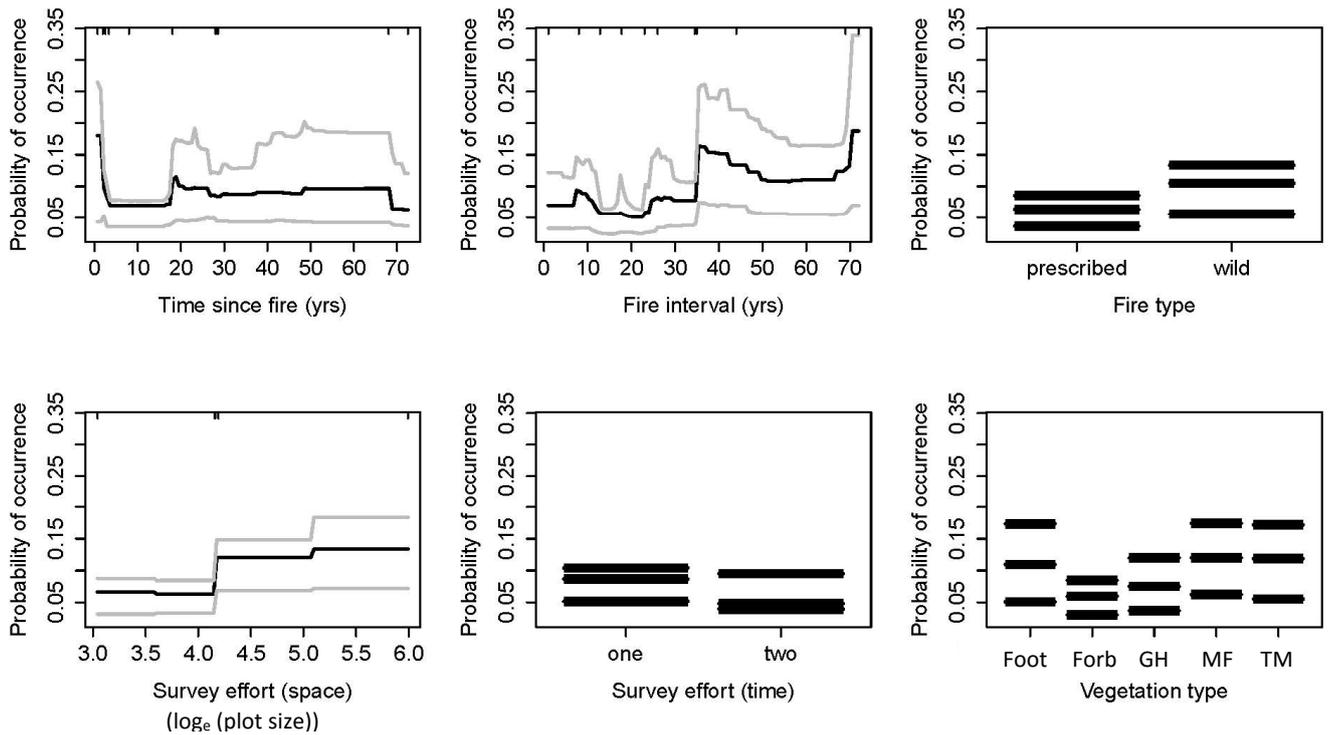
## Results

### Individual species

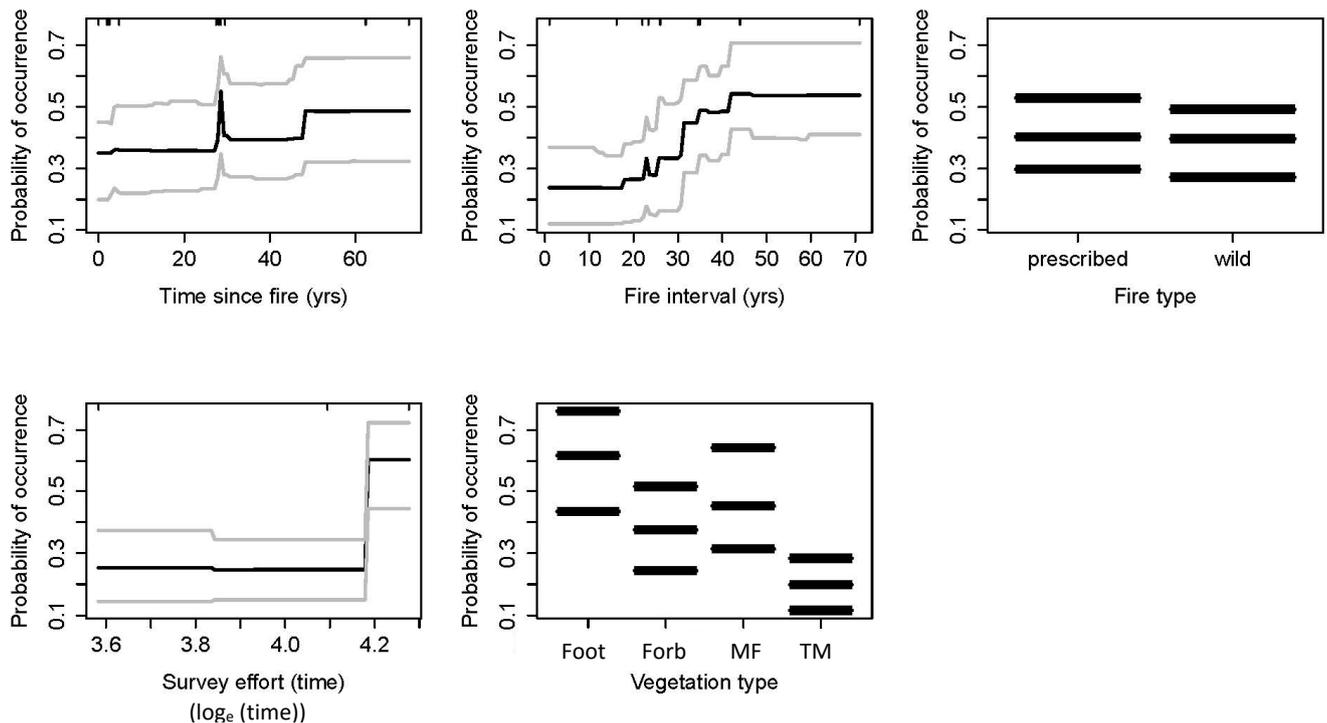
Boosted regression tree models had good predictive capacity ( $AUC \geq 0.7$ ) for 20 of 31 bird species, 66 of 85 plant species, and four of four small mammal species (Table 1, 2 and 3; Appendix 6). The explanatory power of the BRTs ranged from 4-73% (deviance explained). The fire responses of three exemplar species are shown in Figures 9.1, 9.2 and 9.3. The Flame Robin *Petroica phoenicea* was most strongly associated with time since fire; probability of occurrence was higher in early successional vegetation (< 5 years; Figure 9.1). *Spyridium parvifolium* was most strongly associated with inter-fire interval. It reached highest probability of occurrence in longer fire intervals between fires (>35 years; Figure 9.2). The Bush Rat *Rattus fuscipes* reached its highest probability of occurrence in older growth stages (>20 years) with long intervals between fires (Figure 9.3).



**Figure 9.1.** Response of Flame Robin *Petroica phoenicea* to fire regime characteristics, vegetation type and survey effort. For continuous predictors, solid black lines are predictions of the probability of occurrence from a boosted regression tree model and solid grey lines are bootstrapped 95% confidence intervals. For categorical predictors, middle bar represents predicted probability of occurrence, outer bars represent 95% confidence intervals. Fire type: HS = bushfire, LS = planned fire. Vegetation type = EVDs; Foot = Foothills Forest, Forb = Forby Forest, GH = Grassy/Heathy Dry Forest, MF = Moist Forest, TM = Tall Mixed Forest.



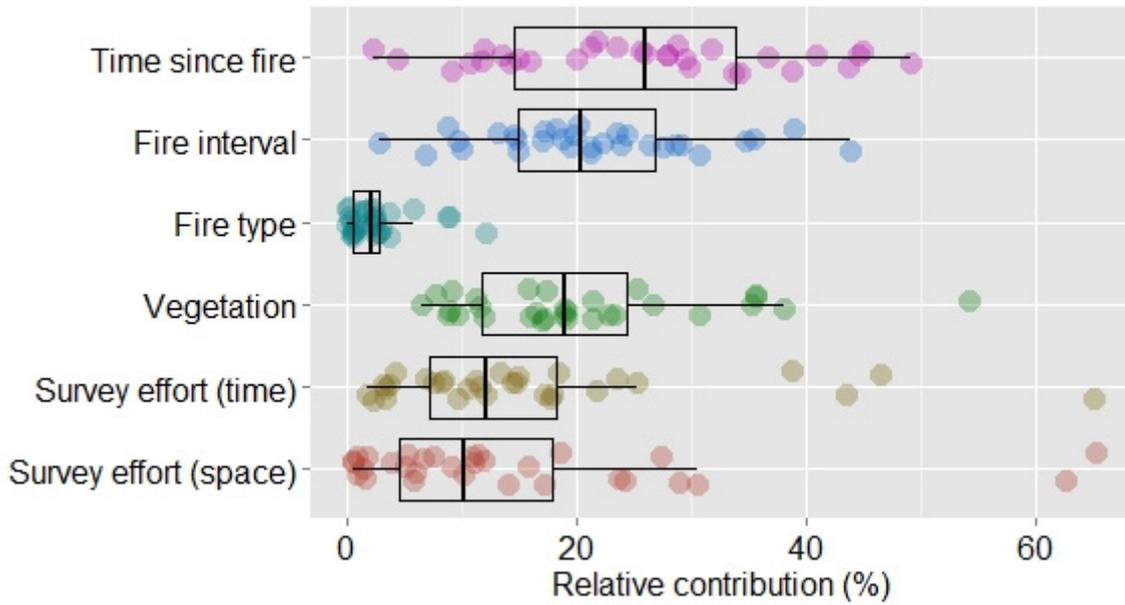
**Figure 9.2.** Response of *Spyridium parvifolium* to fire regime characteristics, vegetation type and survey effort. For continuous predictors, solid black lines are predictions of the probability of occurrence from a boosted regression tree model and solid grey lines are bootstrapped 95% confidence intervals. For categorical predictors, middle bar represents predicted probability of occurrence, outer bars represent 95% confidence intervals. Fire type: HS = bushfire, LS = planned fire. Vegetation type = EVDs; Foot = Foothills Forest, Forb = Forby Forest, GH = Grassy/Heathy Dry Forest, MF = Moist Forest, TM = Tall Mixed Forest.



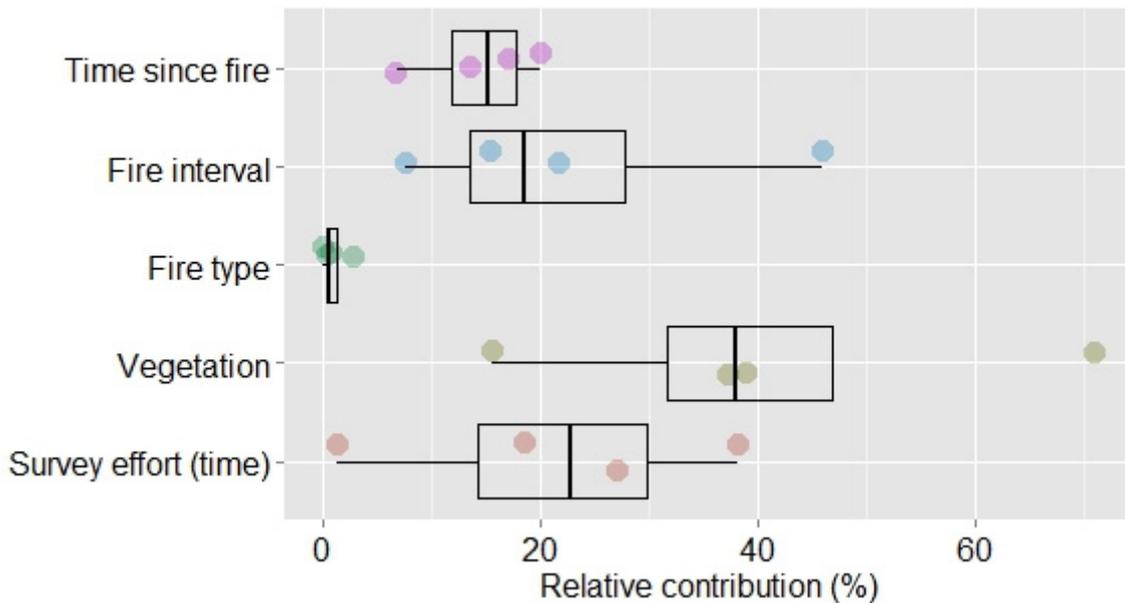
**Figure 9.3.** Response of Bush Rat *Rattus fuscipes* to fire regime characteristics, vegetation type and survey effort. For continuous predictors, solid black lines are predictions of the probability of occurrence from a boosted regression tree model and solid grey lines are bootstrapped 95% confidence intervals. For categorical predictors, middle bar represents predicted probability of occurrence, outer bars represent 95% confidence intervals. Fire type: HS = bushfire, LS = planned fire. Vegetation type = EVDs; Foot = Foothills Forest, Forb = Forby Forest, MF = Moist Forest, TM = Tall Mixed Forest.

### Multitaxa comparisons

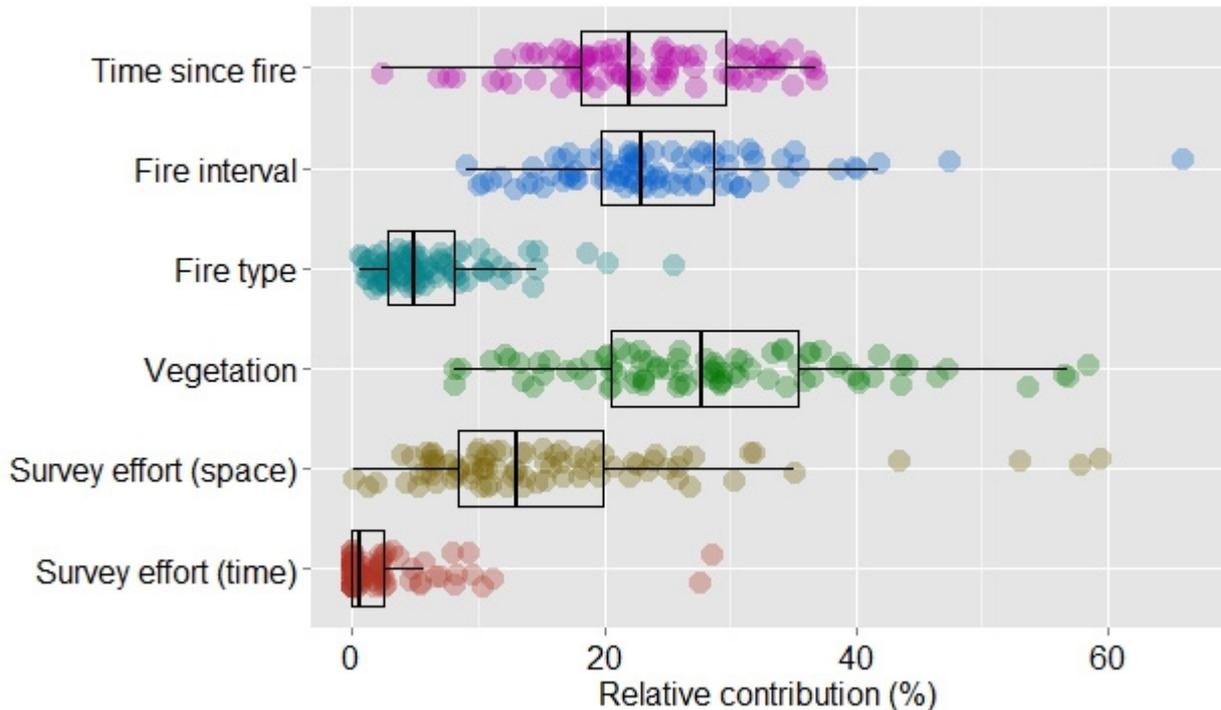
The explanatory power of the boosted regression tree models was higher for plants (av. % Dev = 34.3; min-max % Dev 11.8-73.2) than for birds (av. % Dev = 18.9; min-max % Dev 4.0-50.9) and mammals (av. % Dev = 26.8; min-max % Dev 24.0-29.1). For birds, time since fire made the largest contribution to model explanatory power (Figure 9.4). Twelve of 30 bird species showed a strong response to time since fire (>5% deviance explained attributed to time since fire: Table 1, Appendix 6). Fire interval also had a strong influence on the distribution of many species of birds (Figure 9.4). For small mammals, time since fire and inter-fire interval both made a large contribution to model explanatory power (Figure 9.5). Two of four small mammal species showed a strong response to time since fire and inter-fire interval (>5% deviance explained attributed to these two variables: Table 5). However, local vegetation type was the biggest influence on small mammals (Figure 9.5). Unsurprisingly, plant species occurrence was most strongly associated with vegetation type (Figure 9.6). Inter-fire interval and time since fire had similar levels of influence, while the influence of fire type was relatively low. Fifty-two of 85 plant species showed a strong response to inter-fire interval (>5% deviance explained attributed to inter-fire interval). Fire type (planned burn vs. bushfire) had a relatively small influence across all groups.



**Figure 9.4.** The relative influence of fire regime characteristics, vegetation type and survey effort on birds. Within each category, points represent the response of a single species to a predictor variable. Relative contribution (%) is defined as the influence of each predictor on total model explanatory power for a given species. A higher relative contribution represents a stronger effect of that variable. Box plots highlight the median response to each predictor variable.



**Figure 9.5.** The relative influence of fire regime characteristics, vegetation type and survey effort on small mammals. Within each category, points represent the response of a single species to a predictor variable. Relative contribution (%) is defined as the influence of each predictor on total model explanatory power for a given species. A higher relative contribution represents a stronger effect of that variable. Box plots highlight the median response to each predictor variable.



**Figure 9.6.** The relative influence of fire regime characteristics, vegetation type and survey effort on plants. Within each category, points represent the response of a single species to a predictor variable. Relative contribution (%) is defined as the influence of each predictor on total model explanatory power for a given species. A higher relative contribution represents a stronger effect of that variable. Box plots highlight the median response to each predictor variable.

## Discussion

The importance of fire regime characteristics depended on the focal group considered. Plants were more strongly influenced by inter-fire interval than were birds and mammals. For more mobile groups, such as birds, time since the last fire was a stronger influence than inter-fire interval. Inappropriate fire regimes have been linked to population declines of mammals, birds and plants worldwide. However, the relationship between fire regimes and biodiversity varies between different ecosystems, groups and species. We have shown that multiple aspects of the fire regime influence the distribution of birds, plants and small mammals.

Management objectives should be based on the requirements of species in a community. We recommend managers of temperate Foothills Forests set biodiversity objectives based on both fire events (e.g. growth stage distributions) and inter-fire interval (e.g. tolerable fire intervals) to guide fire management. In addition to temporal properties of fire regimes that are amenable to manipulation, an understanding of how other processes (such as variation in local vegetation) affect biodiversity is also essential for conservation management.



## Methods

Non-linear regression (Generalised Additive Models, GAM) was used to model the occurrence (presence/absence) of 116 native plant, bird and mammal species in relation to time since fire for Forby Forest, Foothills Forest, Grassy/Heathy Dry Forest, Moist Forest, and Tall Mixed Forest. Patterns of post-fire change across species, up until 72 years post-fire, were compared with current growth stage groupings (Cheal 2010).

Models were fitted for species recorded at >10% of taxon-specific survey sites. Further, to ensure models were based on data with sufficient geographic spread across the region, only plant and bird species recorded in the Central and Otway regions, plus at least one other (i.e.  $\geq$  three regions in total), were considered. This additional requirement was not imposed on mammal species due to the reduced geographic coverage of the live-trapping data set.

Model output was assessed to identify species showing a significant ( $P < 0.05$ ) relationship with time since fire. This was done for each EVD separately. Plots of significant time since fire relationships were then collated and visually assessed. Visual assessment included classifying response shapes into one of five *a priori* groups (see Watson *et al.* 2012, Nimmo *et al.* 2012; Figure 10.1). 'Incline' species increase with time since fire; 'decline' species decrease with time since fire; 'irruptive' species show highest occurrence directly after fire, followed by a marked decline in older vegetation; 'plateau' species increase in occurrence with time since fire until a certain point, with little change thereafter; 'bell' species show a marked peak in occurrence in between periods of lower occurrence (inverse-bell species show the opposite pattern – higher occurrence in young/old vegetation). Time since fire relationships were also examined to identify periods during which many species showed a) their highest predicted occurrence or b) changes in predicted occurrence, such as might indicate distinct growth stages. These methods follow those used to examine growth stages in mallee vegetation (Avitabile 2014).

## Results

The occurrence of 116 species was modelled in relation to time since fire and EVD: 81 plants, 33 birds, and two mammals. Overall, there were relatively few significant time since fire relationships across EVDs (103 [18%] of 578 relationships: Table 2; Appendix 7). This result is likely related, in part, to very limited availability of data across the 72-year chronosequence for some EVDs (see Appendix 7). Notably, birds were surveyed at relatively few sites in Grassy/Heathy Dry Forest and Tall Mixed Forest, while the same was true for plants in Foothills Forest. Data for mammals were relatively limited in all EVDs (and not available for Grassy/Heathy Dry Forest). Given this, findings of no effect of time since fire may not accurately reflect true relationships in all cases. This interpretation is supported by 23 instances of species showing strong change in predicted occurrence across the post-fire chronosequence, despite the term for time since fire not being significant at the  $P = 0.05$  level (Table 1; Appendix 7).

### *Foothills Forest*

Eighteen of 116 species showed a significant relationship with time since fire in Foothills Forest; nine species showed a strong yet non-significant relationship (Table 2; Appendix 7). Results did not indicate notable trends/changes in the occurrence of species prior to ~10-years post-fire, such as would match the first three growth stages in this EVD. Nonetheless, most irruptive species were found to show a marked drop in predicted occurrence ~10-20 years post-fire, coinciding with the transition to adolescence that occurs at 10 years (Figure 10.1). Bell species showed a peak in predicted occurrence at around 30-40 years, corresponding with the transition between vigorous maturity and stasis. In addition, the plateau species also reached their maximum at around this point. Trends shown by inclining and declining species highlight that change continues to occur after 40 years post-fire.

### *Forby Forest*

Thirty-six of 116 species showed significant relationships with time since fire in Forby Forest: incline, decline and bell responses were most common (Table 2; Appendix 7). As with Foothills Forest, results did not reveal post-fire change to match the three earliest growth stages in this community. However, ~40 years post-fire was a common time for many bell species to peak in occurrence, corresponding with the transition between vigorous maturity and stasis (Figure 10.1). Linear patterns of change in inclining and declining species suggest ongoing change over 70 years.

### *Grassy/Heathy Dry Forest*

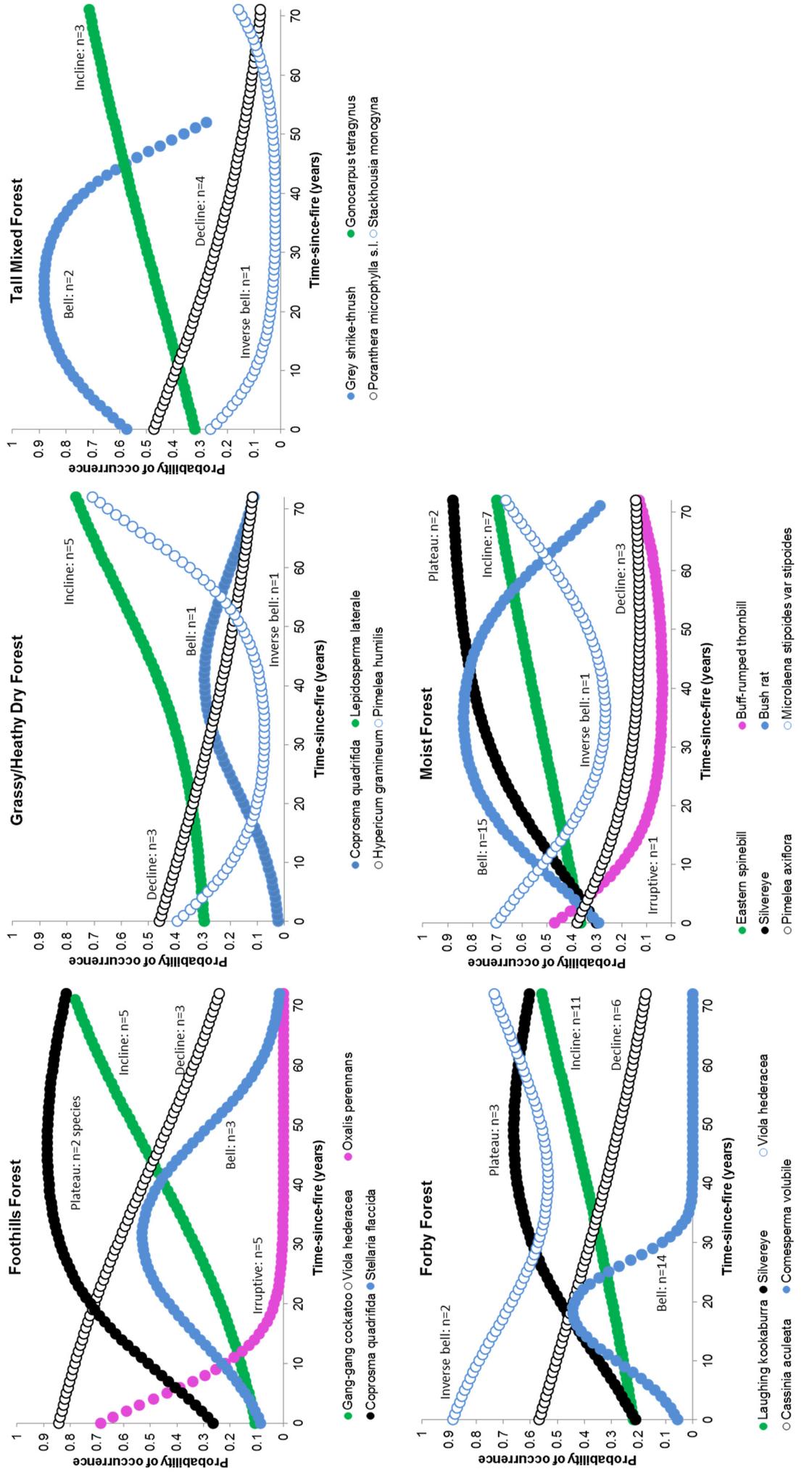
Ten of 114 species showed significant relationships with time since fire in Grassy/Heathy Dry Forest; six species showed a strong yet non-significant relationship (Table 2; Appendix 7). As for other EVDs, results did not highlight early patterns of change to correspond with the first three growth stages. While post-fire trends for a few species changed at around 30-40 years (e.g. peak for bell species, non-linear incline species), results did not suggest the occurrence of a distinct, short growth stage at this point (Figure 10.1). Rather, they were more consistent with a broad period of change between 30-40yrs. Inclining and declining species also highlight ongoing change over 70 years in Grassy/Heathy Dry Forest.

### *Moist Forest*

Twenty-nine of 116 species showed significant relationships with time since fire (Table 2; Appendix 7). Again, results did not reveal early post-fire changes consistent with the first three growth stages. Response curves for most bell and plateau species suggest change at around 30-40 years post-fire, however, some species did show earlier change as is more consistent with the transition between adolescence and maturity at 26 years. Inclining species suggest ongoing change over 70 years (Figure 10.1).

### *Tall Mixed Forest*

Ten species showed significant relationships with time since fire in Tall Mixed Forest, with a further eight species showing strong yet non-significant relationships (Table 2; Appendix 7). Again, our results did not reveal trends consistent with the first three growth stages. Some non-linear relationships (decline, inverse bell) suggested change broadly corresponding with transition at 20 years between vigorous maturity and stasis (Figure 10.1). Inclining and declining relationships again suggested ongoing change to provide support for the occurrence of older growth stages (e.g. stasis to waning transition) that data limitations otherwise prevent us from examining more comprehensively.



**Figure 10.1.** Predicted relationships between the occurrence of individual species and time since fire for each EVD. Relationships are derived from generalised additive models, and illustrate the different *a priori* response shapes identified for time since fire relationships: green = inclining species; white = declining species; pink = irruptive species; black = plateau species; blue = bell shaped species; and blue and white = inverse bell shaped species.

## Discussion

We found that a minority of the species analysed exhibited a statistically significant response to time since fire (9-25% of species across EVDs). However, as noted above, this may be due to limitations in data rather than indicating a lack of time since fire response.

Our results did not identify common patterns of change occurring earlier than 10 years post-fire, such as would be consistent with the first three growth stages identified for all EVDs. However, with the exception of seven bird sites in Tall Mixed Forest, no plant or bird data were available for the first (Renewal) stage in any EVD. Thus, we were not able to examine trends within the period covering these earliest growth stages. Similarly, our data did not extend beyond a time since fire of 72 years and therefore we had no ability to discern any species patterns that might be associated with the oldest growth stage (waning/senescent) in any EVD except Grassy/heathy Dry Forest.

However, given these limitations, considering species that did respond to TSF, across EVDs, it was possible to discern transitions in species' probability of occurrence at:

- ~10 years – irruptive species approaching minimum probability of occurrence.
- ~20-40 years – bell species peak, plateau species approaching maximum probability of occurrence, inverse bell approaching minimum.
- ~50-70 years – bell species approaching or reach minima (occurs at ~ 35 years in Forby Forest), inverse bell increase again.

Overall, the timing of transitions approximate the thresholds for growth stages defined by Cheal (2010) for which we had data (i.e. excluding youngest and oldest growth stages, as discussed above) in most EVDs. Therefore, our results provide qualified support for the existing growth stage definitions. In addition, we found no evidence suggesting alternative, more accurate growth stage definitions. However, our results do not indicate the need for a distinct short stage at time 35-45 years (Waning) in Grassy/Heathy Dry Forest, as is included in the Cheal (2010) definitions.

It is important to note that the use of a growth stage-based model to characterise fire regime-biota relationships involves imposing categories onto what are, in reality, continuous processes. In the current study this is illustrated by a number of species exhibiting constantly increasing or decreasing probability of occurrence across the available time since fire chronosequence, rather than exhibiting threshold (irruptive, plateau) or peak/trough (bell) responses.

Given the centrality of the growth stage concept to ecological fire management planning in Victoria, there is a need for further research to ensure that defined growth stages accurately reflect species responses to fire regimes. In particular, it would be beneficial to use field data on species occurrences across the time since fire chronosequence to test growth stage definitions and species response models derived from expert opinion.

# 11 Optimal growth stage structures for birds, small mammals and plants in Foothills Forest

A key measure of ecosystem resilience used in fire management planning in Victoria is the geometric mean of species' relative abundance (GMA or  $G$ ; Buckland *et al.* 2011). Recent work demonstrates that GMA is a useful measure of biodiversity in fire-prone ecosystems (Di Stefano *et al.* 2013; Kelly *et al.* 2015).

Defining fire management objectives based on GMA can assist managers in determining whether the vegetation growth stage structure is adequate to maintain habitat for fauna and flora species or whether management interventions such as planned burning are needed to create or protect key growth stages. As a relatively new approach, there is a need to better understand how to use GMA to set fire management objectives in a range of vegetation types, including Foothills Forests. Specifically, the application of GMA to the management of parks and reserves requires a detailed understanding of how sensitive GMA is to the plant and animal data and models that underpin the index; how differences between local forest vegetation types might influence the index; and how the index changes under alternative fire history scenarios in Foothills Forests.

Our analysis of GMA in Foothills Forest proceeded in three steps. First, we developed statistical models and predictions of species' responses to time since fire (see Section 9). Second, we determined the optimal allocation of growth stages ('growth stage structure', GSS) within each of the Foothills EVDs, for each taxonomic group, based on the geometric mean of species' probability of occurrence (correlated with relative abundance, see Appendix 6). Finally, we compared how the optimal allocation of growth stages changed between taxonomic groups and EVDs (see Appendix 8 for more detailed description of approach and methods). Note that the growth stage structure presented here is the 'optimal' GSS (sensu *DELWP Policy Position – Measuring Ecosystem Resilience in Strategic Bushfire Management Planning*; DELWP 2015). We have used the 'optimal' GSS, rather than the 'ecological' GSS, as the latter pertains to a particular region/landscape (Machunter *et al.* 2015). The various constraints that are introduced into the GSS analysis to produce the 'ecological' GSS reflect "landscape level ecological objectives" (DELWP 2015), while our analyses pertain to the foothills system as a whole. While analyses based on the optimal GSS may not be directly applicable to management planning, they still have value in elucidating the theoretical maximum value of GMA in the Foothills EVDs, the growth stages that are most critical for biota, and how biodiversity responds to alternative GSSs.

## Methods

We used boosted regression trees (BRTs) to determine animal and plant responses to time since fire, as described in Section 9 above. We modelled the response variable as the presence or absence of a species at a site. When species are at low prevalence, as was the case in our data set, probability of occurrence is highly correlated with relative abundance.

We classified each EVD into four growth stages based on Cheal 2010 (Table 11.1). Consultation with experts and local fire managers indicated that these modified growth stages represent distinct structural and floristic elements in Foothills Forests. Our aim was to determine the optimal combination of these growth stages for biodiversity conservation. It is important to note that we did not have samples representing the oldest growth stage in EVDs, as defined by Cheal (2010), with the exception of Grassy/Heathy Dry Forest.

**Table 11.1.** Vegetation growth stage definitions for each EVD. Terminology follows Cheal (2010). Juv = Juvenile. Ad = Adolescent, V Mat = Vigorous Maturity, Mat = Maturity, Sta = Stasis, Wan = Waning, Sen = Senescent.

EVD	Vegetation growth stages			
Foothills Forest	Juv (0-3 yrs)	Ad (4-10 yrs)	V Mat (11-40 yrs)	Sta (41-72 yrs)
Forby Forest	Juv (0-3 yrs)	Ad (4-10 yrs)	V Mat (11-40 yrs)	Sta (41-72 yrs)
Moist Forest	Juv (0-11 yrs)	Ad (12-26 yrs)	Mat 1 (27-50 yrs)	Mat 2 (51-75 yrs)
Tall Forest	Juv (0-3 yrs)	Ad (4-8 yrs)	Sta (9-60 yrs)	Wan (61-72 yrs)
Grassy/Heathy Dry Forest	Juv (0-3 yrs)	Ad (4-10 yrs)	Mat (11-35 yrs)	Sen (36-72 yrs)

For all species, we calculated the probability of occurrence in each age-class by averaging the boosted regression model predictions over the time period defined by each growth stage. The average probability of occurrence of species  $j$  in a given area depends on the proportion of that area that is in each growth stage. For example, in Forby Forest, the average probability of occurrence of each species depends on the area of vegetation classified as Juvenile ( $a_J$ ), Adolescent ( $a_A$ ), Vigorous Maturity ( $a_V$ ) and Stasis ( $a_S$ ). Thus, the probability of occurrence of species  $j$  in a given area of Forby Forest is:

$$P_j = q_{J,j}a_J + q_{A,j}a_A + q_{V,j}a_V + q_{S,j}a_S \quad (\text{Equation 1})$$

where  $q_{c,j}$  is the probability of occurrence of species  $j$  in age class  $c$  with  $c$  taking values of J, A, V or S.

The geometric mean, averaging over  $n$  species in the area, is  $GMA = \exp\left(\frac{1}{n} \sum_{j=1}^n \log P_j\right)$ . The age structure that maximizes the geometric mean will vary depending on the number of species that occur in particular growth stages and the probability of occurrence of those species in the different growth stages.

We used numerical optimization to determine the values of  $a_J$ ,  $a_A$ ,  $a_V$ , and  $a_S$  that maximizes the geometric mean (GMA), averaging across the species in the community, subject to the constraint  $a_J + a_A + a_V + a_S = 1$ . We calculated the optimal allocation of growth stages that maximized the geometric mean for birds and plants as separate groups, and for all groups (birds, small mammals and plants) combined.

## Results

### *What is the optimal growth stage structure for birds, plants and all taxonomic groups combined?*

The optimal growth stage structure for birds in Forby Forest was comprised of Stasis and Adolescent growth stages (Figure 11.1). A similar mix of growth stages, weighted towards a mix of Adolescent and/or older vegetation, maximized GMA for birds in each vegetation type (Figure 11.1; Tables 11.2-11.6). The optimal growth stage structure for plants in Forby Forest was comprised of Stasis and Juvenile vegetation (Figure 11.2). This optimal solution for plants was similar between all vegetation types and was weighted towards recently burnt vegetation (Figure 11.2; Table 11.2-11.6). For the combined bird, plant and small mammal data, the solution in all vegetation types that maximized GMA included a mix of Juvenile and the oldest growth stage in each EVD (Figure 11.3; Tables 11.2-11.6).

**Table 11.2.** Optimal vegetation growth stage structures for four growth stages in Foothills Forest EVD. B = Birds, M = Mammals, P = Plants. GMA= Geometric mean of species' relative abundance. TSF refers to members of a taxonomic group that showed a significant relationship with time since fire.

Data	Taxa	EVD	Proportion of growth stage				Optimal GMA
			Juv (0-3)	Ad (4-10)	V Mat (11-40)	Sta (41-72)	
All birds (32 species)	B	Foothills	0.00	0.75	0.00	0.25	0.35
TSF birds (7 species)	B	Foothills	0.33	0.22	0.00	0.45	0.34
All plants (85 species)	P	Foothills	0.78	0.00	0.00	0.22	0.17
TSF plants (57 species)	P	Foothills	0.81	0.00	0.00	0.19	0.18
All birds, mammals and plants (150 species)	B, M, P	Foothills	0.70	0.00	0.00	0.30	0.20
TSF birds, mammals and plants (97 species)	B, M, P	Foothills	0.74	0.00	0.00	0.26	0.19

**Table 11.3.** Optimal vegetation growth stage structures for four age-classes in Forby Forest EVD. B = Birds, M = Mammals, P = Plants. GMA= Geometric mean of species' relative abundance. TSF refers to members of a taxonomic group that showed a significant relationship with time since fire.

Data	Taxa	EVD	Proportion of growth stage				Optimal GMA
			Juv (0-3)	Ad (4-10)	V Mat (11-40)	Sta (41-72)	
All birds (32 species)	B	Forby	0.00	0.12	0.00	0.88	0.32
TSF birds (7 species)	B	Forby	0.26	0.00	0.00	0.74	0.28
All plants (85 species)	P	Forby	0.20	0.00	0.00	0.80	0.18
TSF plants (57 species)	P	Forby	0.32	0.00	0.00	0.68	0.18
All birds, mammals and plants (150 species)	B, M, P	Forby	0.10	0.00	0.00	0.90	0.20
TSF birds, mammals and plants (97 species)	B, M, P	Forby	0.21	0.00	0.00	0.79	0.18

**Table 11.4.** Optimal vegetation growth stage structures for four age-classes in Grassy Heathy Dry Forest EVD. Note: there are no mammal trapping data for this vegetation type. B = Birds, P = Plants. GMA= Geometric mean of species' relative abundance. TSF refers to members of a taxonomic group that showed a significant relationship with time since fire.

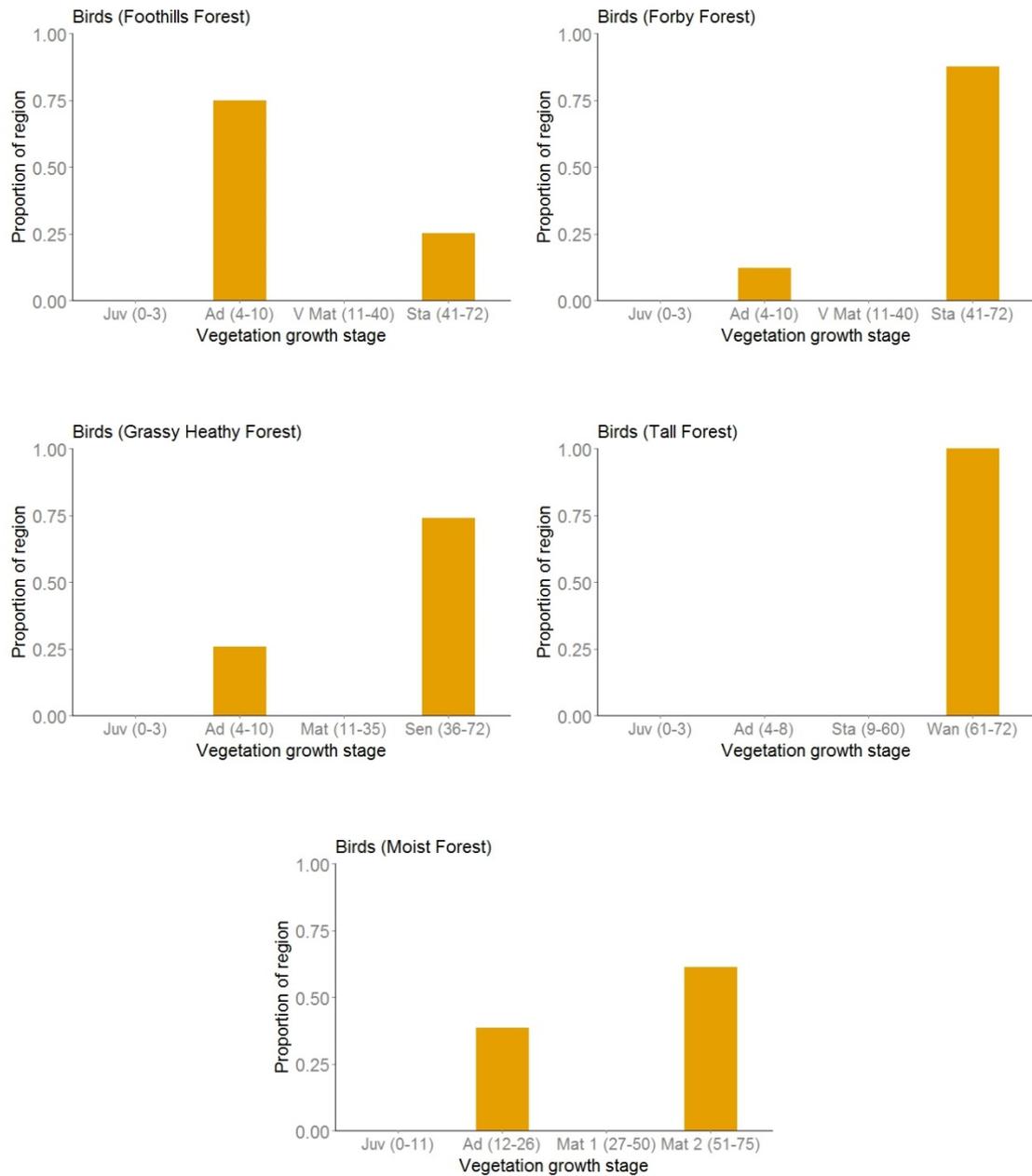
Data	Taxa	EVD	Proportion of growth stage				Optimal GMA
			Juv (0-3)	Ad (4-10)	Mat (11-35)	Sen (36-72)	
All birds (32 species)	B	Grassy	0.00	0.26	0.00	0.74	0.29
TSF birds (7 species)	B	Grassy	0.04	0.00	0.00	0.96	0.22
All plants (85 species)	P	Grassy	0.73	0.00	0.00	0.27	0.16
TSF plants (57 species)	P	Grassy	0.68	0.00	0.00	0.32	0.14
All birds, and plants (174 species)	B, M, P	Grassy	0.64	0.00	0.00	0.36	0.18
TSF birds and plants (94 species)	B, M, P	Grassy	0.60	0.00	0.00	0.40	0.15

**Table 11.5.** Optimal vegetation growth stage structures for four age-classes in Tall Mixed Forest EVD. B = Birds, M = Mammals, P = Plants. GMA= Geometric mean of species' relative abundance. TSF refers to members of a taxonomic group that showed a significant relationship with time since fire.

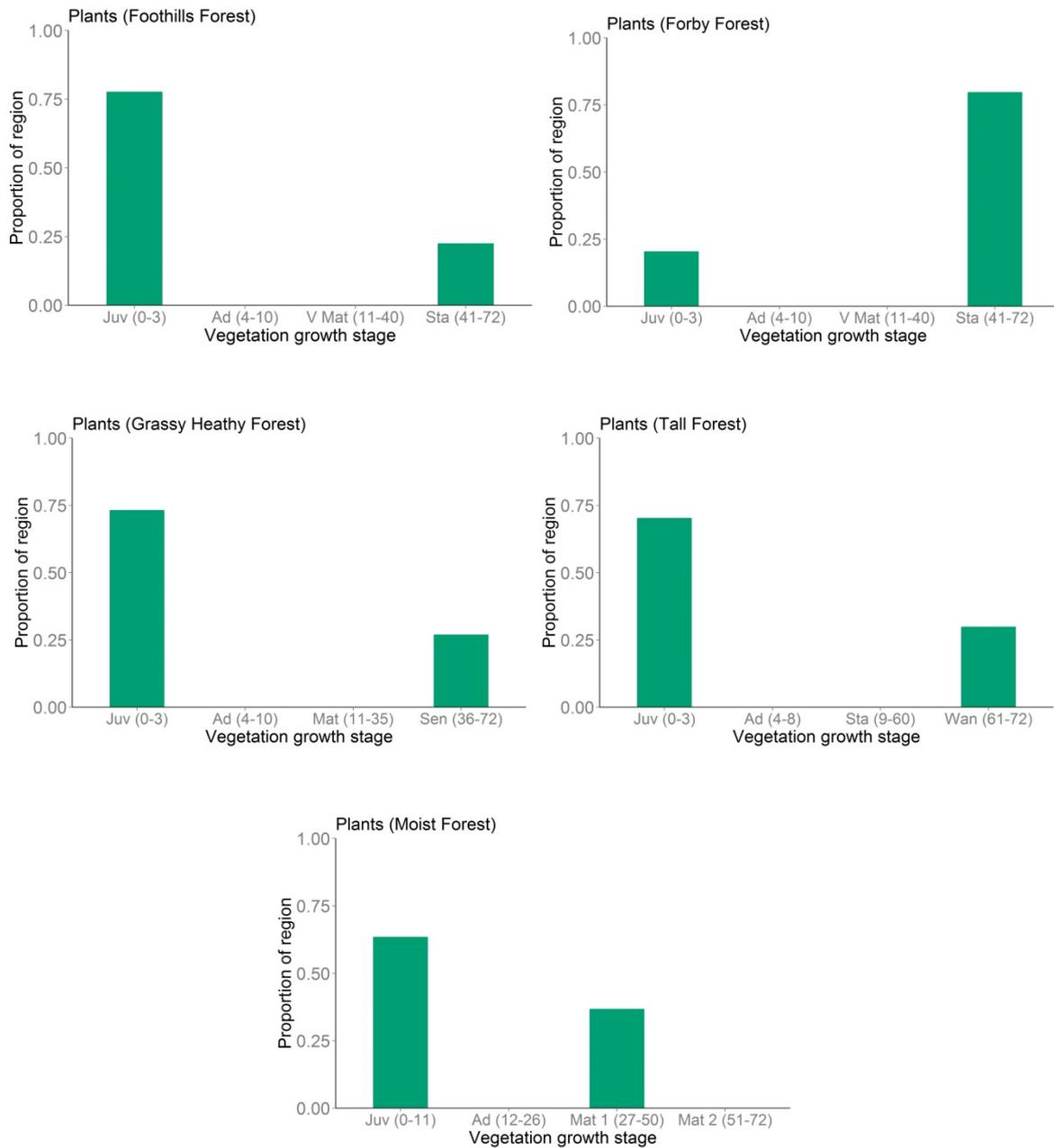
Data	Taxa	EVD	Proportion of growth stage				Optimal GMA
			Juv (0-3)	Ad (4-8)	Sta (9-60)	Wan (61-72)	
All birds (32 species)	B	Tall	0.00	0.00	0.00	1.00	0.31
TSF birds (7 species)	B	Tall	0.00	0.00	0.00	1.00	0.28
All plants (85 species)	P	Tall	0.70	0.00	0.00	0.30	0.17
TSF plants (57 species)	P	Tall	0.74	0.00	0.00	0.26	0.17
All birds, mammals and plants (150 species)	B, M, P	Tall	0.36	0.00	0.00	0.64	0.20
TSF birds, mammals and plants (97 species)	B, M, P	Tall	0.45	0.00	0.00	0.55	0.17

**Table 11.6.** Optimal vegetation growth stage structures for four age-classes in Moist Forest EVD. B = Birds, M = Mammals, P = Plants. GMA= Geometric mean of species' relative abundance. TSF refers to members of a taxonomic group that showed a significant relationship with time since fire.

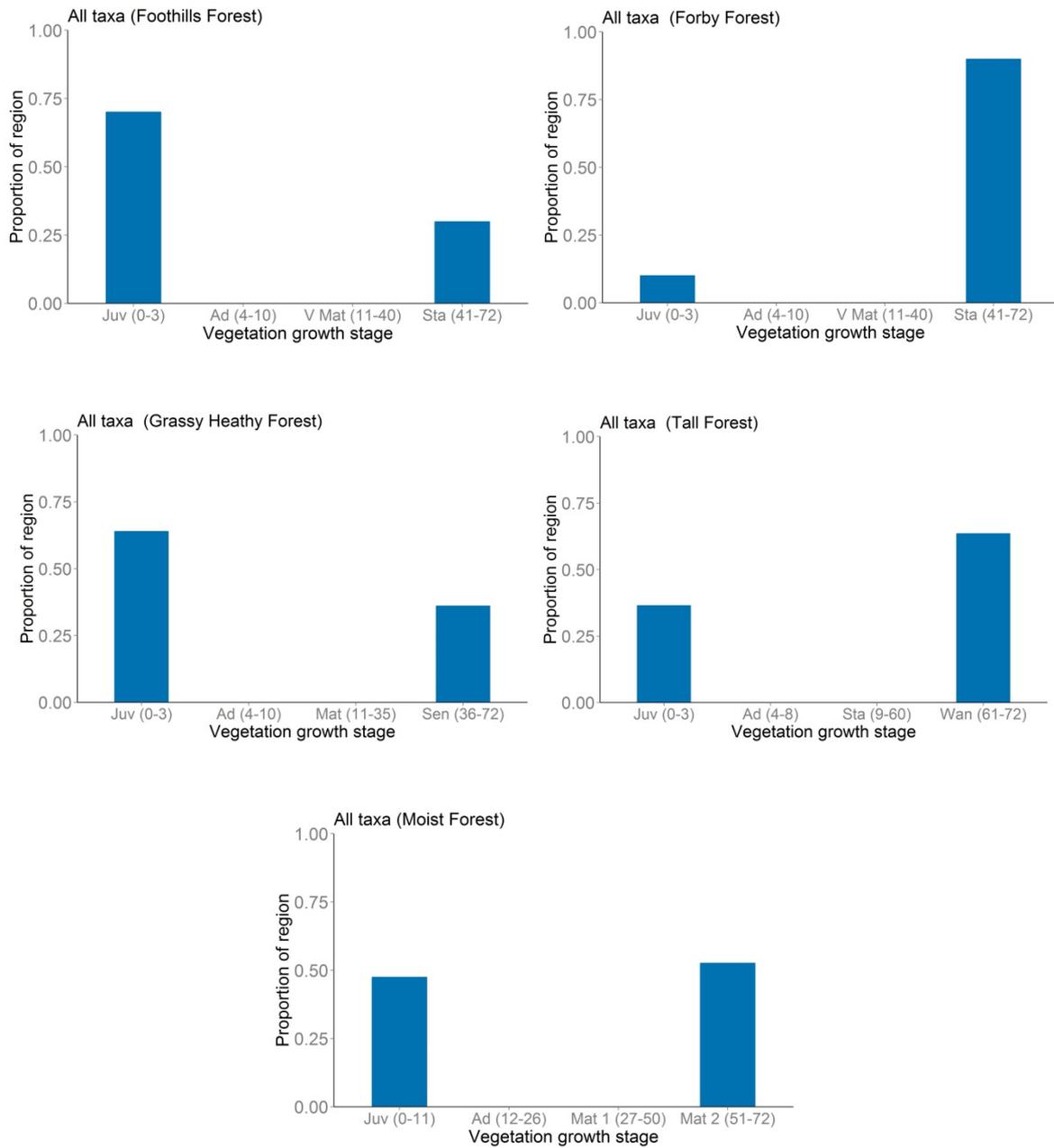
Data	Taxa	EVD	Proportion of growth stage				Optimal GMA
			Juv (0-11)	Ad (12-26)	Mat 1 (27-50)	Mat 2 (51-75)	
All birds (32 species)	B	Moist	0.00	0.39	0.00	0.61	0.35
TSF birds (7 species)	B	Moist	0.15	0.00	0.00	0.85	0.33
All plants (85 species)	P	Moist	0.63	0.00	0.37	0.00	0.16
TSF plants (57 species)	P	Moist	0.62	0.00	0.37	0.01	0.18
All birds, mammals and plants (150 species)	B, M, P	Moist	0.47	0.00	0.00	0.53	0.20
TSF birds, mammals and plants (97 species)	B, M, P	Moist	0.47	0.00	0.00	0.53	0.19



**Figure 11.1.** Vegetation growth stage structures that maximize geometric mean of species' relative abundance in the Foothills Forest EVDs. Results are for all birds at  $\geq 10\%$  of sites and that occur in three major sub regions.



**Figure 11.2.** Vegetation growth stage structures that maximize geometric mean of species' relative abundance in the Foothills Forest EVDs. Results are for all plants at  $\geq 10\%$  of sites and that occur in three major sub regions.



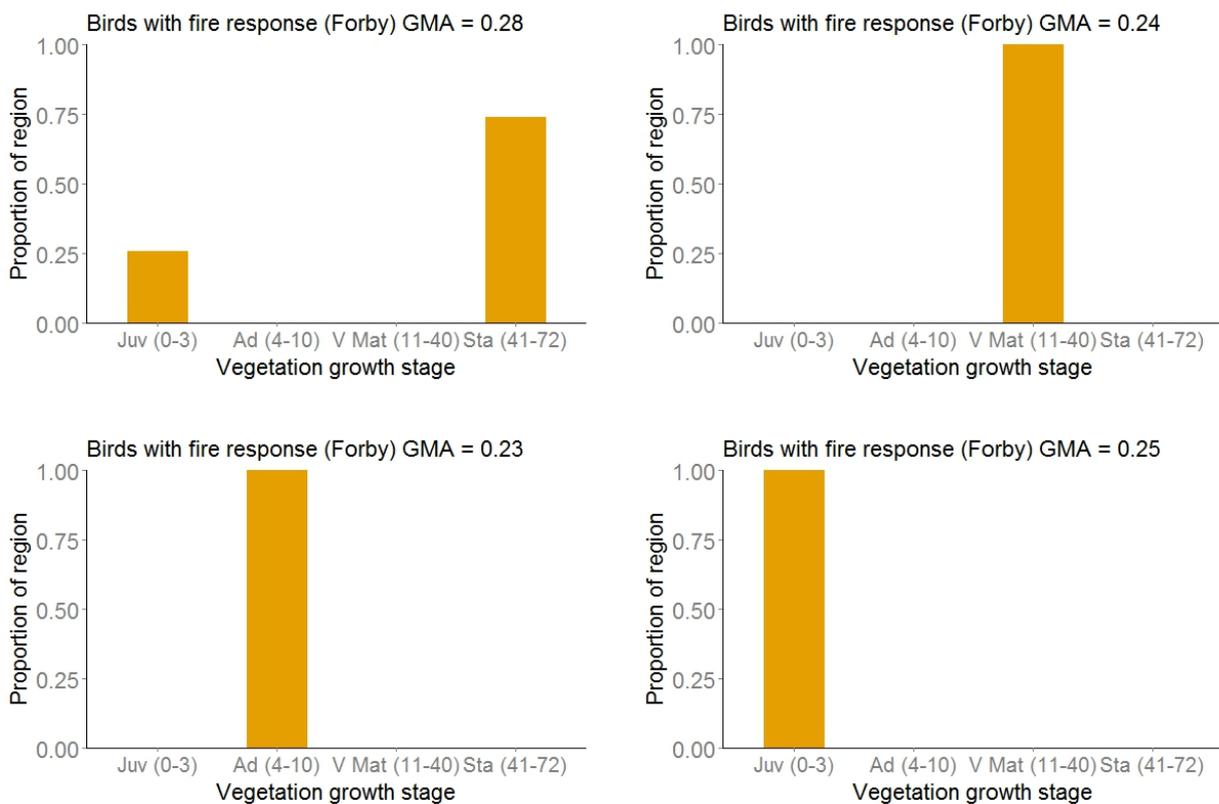
**Figure 11.3.** Vegetation growth stage structures that maximize geometric mean of species' relative abundance in the Foothills Forests EVDs. Results are for all birds, small mammals and plants at  $\geq 10\%$  of sites and that occur in three major sub regions.

*How sensitive is the GMA index to changes in the mix of growth stages?*

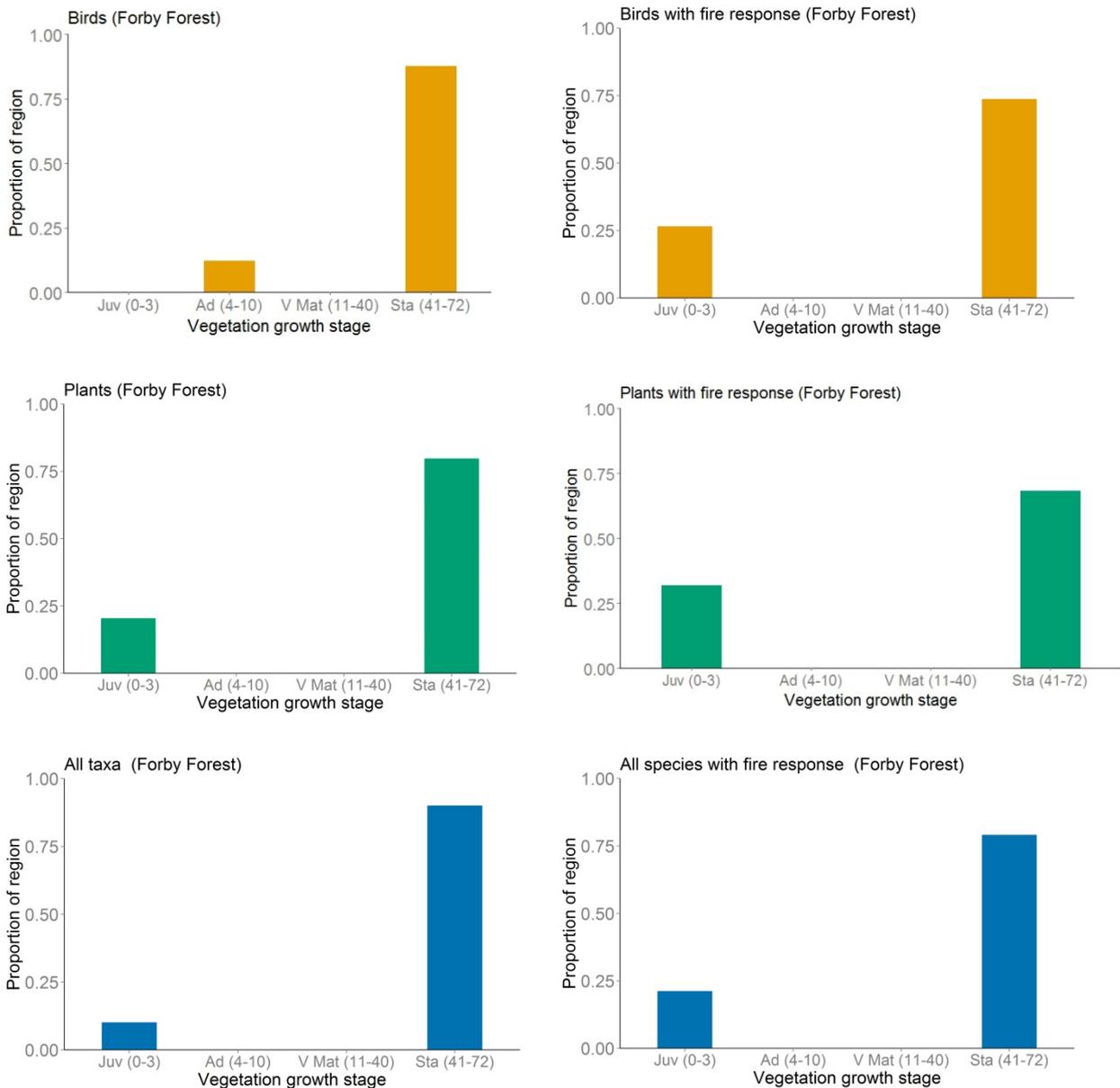
We explored how sensitive the GMA index was to alternative combinations of growth stages by focusing on seven bird species that were associated with time since fire. Several allocations of age-classes produced a similar value of GMA to the optimal solution (Figure 11.4). For example, a given area consisting entirely of the Vigorous Maturity growth stage produces a GMA of 0.25 in Forby Forest. The optimal solution for Forby Forest is similar: a GMA of 0.28. For birds associated with time since fire, the scenario that minimizes GMA in Forby Forest is an area made up entirely of Adolescent vegetation (Figure 11.4; GMA = 0.23). This represents an 18% decline in GMA from the optimal solution.

*How sensitive is GMA to the species included in the optimization procedure?*

We examined the sensitivity of GMA to the species included in the optimization procedure using Forby Forest vegetation as a case study. For each combination of taxonomic groups, there were minimal differences between the growth stages that maximized GMA when all species were included in the analysis versus that when only species associated with time since fire were included in the analysis (Figure 11.5).



**Figure 11.4.** Comparison of geometric mean of species’ relative abundance under alternative vegetation growth stage allocations. Results are for birds that respond strongly to time since fire in Forby Forest EVD.



**Figure 11.5.** Comparison of vegetation growth stage structures that maximize geometric mean of species' relative abundance for 'all species' and 'fire response' species. Results are shown for birds, plants and all taxonomic groups (birds, small mammals and plants) combined in the Forby Forest EVD.

## Discussion

Our results indicated that the vegetation growth stage structure that maximised GMA for birds, small mammals and plants included a mix of recently burnt vegetation and older vegetation. This overall pattern was similar across EVDs, although there was some variation in the proportions of growth stages. Separate analyses for birds and plants similarly showed that GMA of these groups was maximised with a mix of young and old vegetation. For birds, this mix tended to have a greater proportion of old than young vegetation, while for plants the reverse was true. These results appear to be driven by a relatively small number of species that are strongly associated with either early or late growth stages (see Section 10). Most species, while they may show a response to time since fire, occur across growth stages and so do not influence the optimisation outcome as strongly as these 'growth stage specialists'.

Separate analysis of the bird data showed that a range of alternative growth stage structures resulted in only minor reductions in GMA values relative to the optimal solution. Again, this is likely to be a reflection of most species persisting across a broad range of growth stages. This in turn suggests that most species in Foothills Forests are resilient to a variety of planned burning strategies. However, it should be noted that, due to the nature of the data sets used, only relatively common species were included in these analyses. Including less common species (e.g. threatened species) in determining optimum growth stage distributions would help ensure that fire management caters for the needs of these species. To achieve this, targeted surveys are likely to be necessary to obtain sufficient data on uncommon species. Analysis of single species responses to fire regimes are likely to be more informative than overall biodiversity measures, such as GMA, when considering species of particular management interest.

The analysis presented here dealt with the Foothills Forest system as a whole and it is somewhat unclear to what degree the results apply at finer scales (e.g. region, landscape). Exploratory analyses using data solely from the Otways region provided very similar results to those shown here, suggesting it is possible to use broad-scale models to guide more localised management. However, more work on the 'scalability' of this approach would be beneficial.

The 'optimal' growth stage distributions presented here indicate the importance of long unburnt sites for fauna. They also suggest a need for a relatively high proportion of young vegetation, especially for plants. This finding is in accord with numerous studies that have found plant richness to be higher soon after fire compared to later (Gill 1999; Penman *et al.* 2009). However, in the present study, many of the species driving this pattern are common 'weedy' (though native) species. Given that currently 42% of Foothills Forest has been burnt in the last 15 years it is likely that these species' requirements will be met incidentally by bushfire and planned burning, as well as other disturbances, and so burning specifically to cater for them is unlikely to be necessary, at least in the short to medium term.

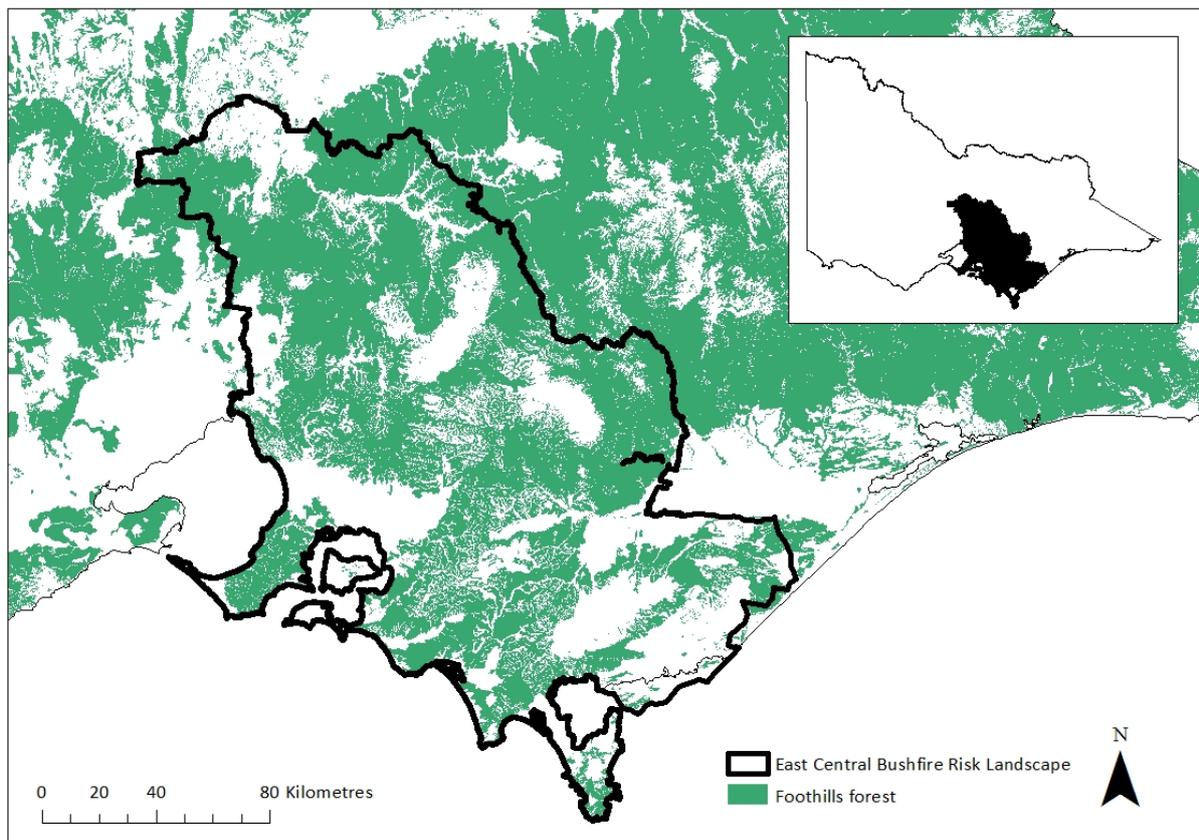
The issue of detectability also needs to be considered when deriving growth stage structures based on GMA of plants. Most flora survey data only records plants present above ground. However, many species will be present at, or readily migrate to, a site as seeds or other propagules, which germinate/sprout post-fire. Therefore, a growth stage structure analysis based on above ground flora, such as presented here, is likely to over-estimate the extent of 'young' growth stages required to ensure persistence of early successional plant species. These issues also pertain to determining tolerable fire intervals, i.e. considering only the above-ground flora will result in an under-estimation of maximum tolerable fire interval. Better information on propagule dynamics in relation to fire regimes is required to address these issues.

Detailed species distribution modelling shows that additional components of the fire regime influence the distribution of birds, small mammals and plants (see Section 9). GMA calculations are based on a single measure of the fire regime: time since fire. It is important that other attributes of the fire regime such as fire severity, inter-fire interval and the spatial characteristics of fire are also considered in fire management planning.

## 12 Biodiversity outcomes of fire management scenarios

Empirically-based predictive models of relationships between biota, fire and landscape have the potential to assist in translating research findings into a management context. The ability to predict the likely biotic consequences of manipulating fire regimes can provide managers with greater capacity to determine those regimes that are most likely to deliver desired biodiversity outcomes. In this regard, simulations, particularly in a spatially explicit setting, allow ‘experimentation’ with various fire regimes in order to identify those that are optimal for achieving management aims.

In the current project, we drew on the results of GMA optimisation (see Section 11) to assess the biodiversity outcomes of four fire management scenarios within the East Central Bushfire Risk Landscape (ECBRL) area (Figure 12.1). The ECBRL is an important area for fire management in Victoria, as it contains areas with high levels of risk to human life from bushfires, as well as substantial areas of intact native vegetation and hence high biodiversity values (DEPI 2014).



**Figure 12.1.** Location of the East Central Bushfire Risk Landscape.

## Methods

The scenarios examined represent a range of ‘real life’ fire management options, devised in collaboration with ECBRL staff and in the context of the ECBRL strategic plan. They encompass a range of spatial extent and fire return intervals of planned burning (Table 12.1). It should be noted that the scenarios do not include any bushfire and so represent a ‘minimum’ level of fire in the study landscape. They include a ‘status quo’ scenario (Scenario 1), based on DELWP’s approach to planned burning prior to formulation of the ECBRL strategic plan. Scenarios 2-4 represent options for targeted planned burning in Priority Fuel Management Areas (i.e. areas where planned burning produces the greatest reduction in residual risk), including with (Scenario 2) and without (Scenarios 3, 4) a target of burning 5% of public land annually. The scenarios were transferred to a spatial context using the Woodstock software (Remsoft 2014), which generated spatial layers for time since fire, inter-fire interval and number of fires across the ECBRL area for 40 years from 2014 (Woodstock modelling carried out by Owen Salkin, Natural Systems Analytics). The time since fire layers were used to calculate the proportion of each Foothills Forest EVD within the four growth stages used in the GMA optimisation analysis at five year time steps over the 40 years of the scenarios. Using the species models from the optimisation analysis (Sections 9, 11), we then calculated GMA for birds, plants and all species (birds, plants and mammals). As described above, the data used to generate these models only cover a chronosequence up to 72 or 75 years (depending on EVD). In the current analysis, we included the area within EVDs that was mapped as older than this within the oldest growth stage considered in our GMA optimisation analysis. The trajectory of GMA over the 40 years was plotted, allowing comparison of patterns of change in GMA amongst scenarios and with optimum GMA.

An additional aim of these analyses, stemming from discussions with biodiversity managers in the ECBRL, was to assess effects of fire scenarios on individual species of interest, in particular rare or threatened species. However, the data sets used in the current project do not contain sufficient records of rare/threatened species for this to be possible. We were able to assess the responses of species that: (1) exhibited relatively strong responses to time since fire in the growth stage analysis; (2) collectively, were representative of the range of time since fire response curve shapes (see Section 9 above). The reasoning here is that examining changes in predicted occurrence of these ‘responsive’ species under the various scenarios may provide an indication of potential responses of rare/threatened species. The species examined in this part of the analysis were those with the strongest time since fire responses within the Moist Forest EVD (response shape in brackets): *Billardiera mutabilis* (bell-shaped), Crescent honeyeater (incline), *Cassinia aculeata* (irruptive), Silvereeye (plateau), Spotted pardalote (decline).

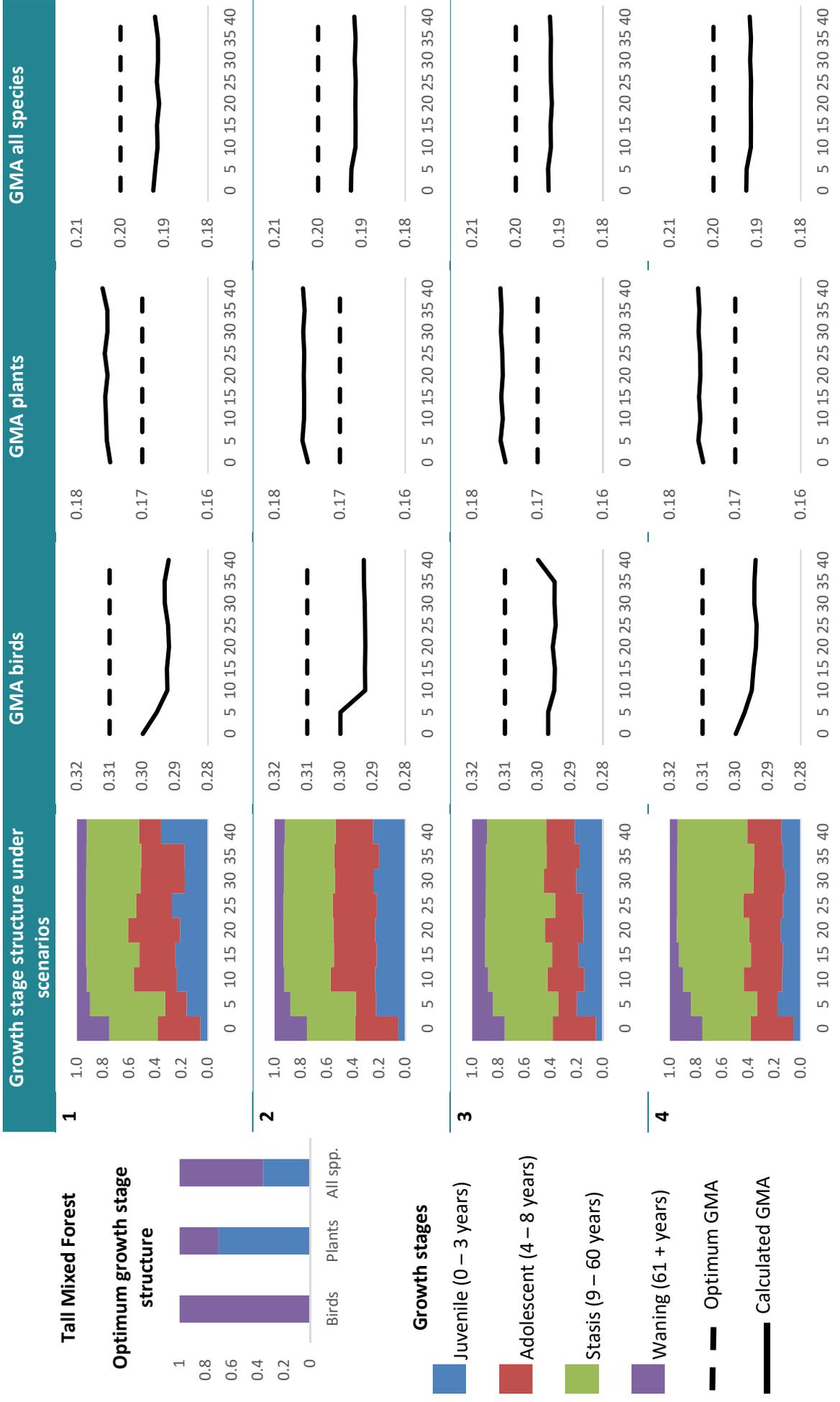
**Table 12.1.** Alternative scenarios for fire management in the East Central Bushfire Risk Landscape. APZ = Asset protection zone, BMZ = Bushfire Moderation Zone, LMZ = Landscape Management Zone, PFMA = Priority Fuel Management Area, TFI = Tolerable fire interval.

Scenario	Definitions used to generate scenario spatial layers in Woodstock
1. Status Quo	<ul style="list-style-type: none"> <li>i) Current Zoning</li> <li>ii) Implement Burn Plan 15 for years 1-3</li> <li>iii) From year 4 burn as close as possible to 32,500ha/annum</li> <li>iv) Burn APZ at 4 year cycle (~3,710 ha/annum)</li> <li>v) Burn BMZ at 8 year cycle (~20,830 ha/annum)</li> <li>vi) From year 4 LMZ outside PFMA managed to be burnt as close as possible to midpoint TFI and always above min TFI (~7,960ha/annum)</li> <li>vii) District targets pro rata of zone and frequency</li> </ul>
2. Priority Fuel Management Areas with 5% target	<ul style="list-style-type: none"> <li>i) Implement Burn Plan 15 for years 1-3</li> <li>ii) From year 4 burn as close as possible to 32,500ha/annum</li> <li>iii) From year 4 burn PFMA at 10 year cycle (~14,460ha/ annum)</li> <li>iv) Burn APZ outside of PFMA at 4 year cycle (~1,840 ha/annum)</li> <li>v) Burn BMZ outside of PFMA at 8 year cycle (~11,580 ha/annum)</li> <li>vi) From year 4 LMZ outside PFMA managed to be burnt as close as possible to midpoint TFI and always above min TFI (~4,630ha/annum)</li> <li>vii) District targets pro rata of zone and frequency</li> </ul>
3. Priority Fuel Management Areas without 5% target	<ul style="list-style-type: none"> <li>i) After year 1 treat as close to 25,200ha per annum</li> <li>ii) Burn APZ/BMZ/LMZ within PFMA at 12 year cycle (~14,460ha/ annum)</li> <li>iii) Burn APZ outside of PFMA at 4year cycle (~1,470ha/annum)</li> <li>iv) Burn BMZ outside of PFMA at 12 year cycle (~7,720ha/annum)</li> <li>v) LMZ outside PFMA managed to be burnt as close as possible to midpoint TFI and always above min TFI (~1530ha/annum)</li> <li>vi) Use Burn Plan 15 as is for first year – then reduced ~25,200 for years 2 and 3, then Woodstock to schedule ~25,200 until year 40</li> <li>vii) District targets pro rata of zone and frequency</li> </ul>
4. Priority Fuel Management Areas without 5% target	<ul style="list-style-type: none"> <li>i) APZ in and out of PFMA burnt at 8 year cycle (~1,860 ha/annum)</li> <li>ii) BMZ out of PFMA burnt at 15 year cycle (~6,130 ha/annum)</li> <li>iii) BMZ and LMZ within PFMA burnt at 15 year cycle (~1,170 ha/annum)</li> <li>iv) LMZ outside PFMA managed to be burnt as close as possible to midpoint TFI and always above min TFI (~6,150ha/annum)</li> <li>v) Use Burn Plan 15 as is for first year – then reduced ~25,200 for years 2 and 3, then Woodstock to schedule ~25,200 until year 4</li> <li>vi) District targets pro rata of zone and frequency</li> </ul>

## Results

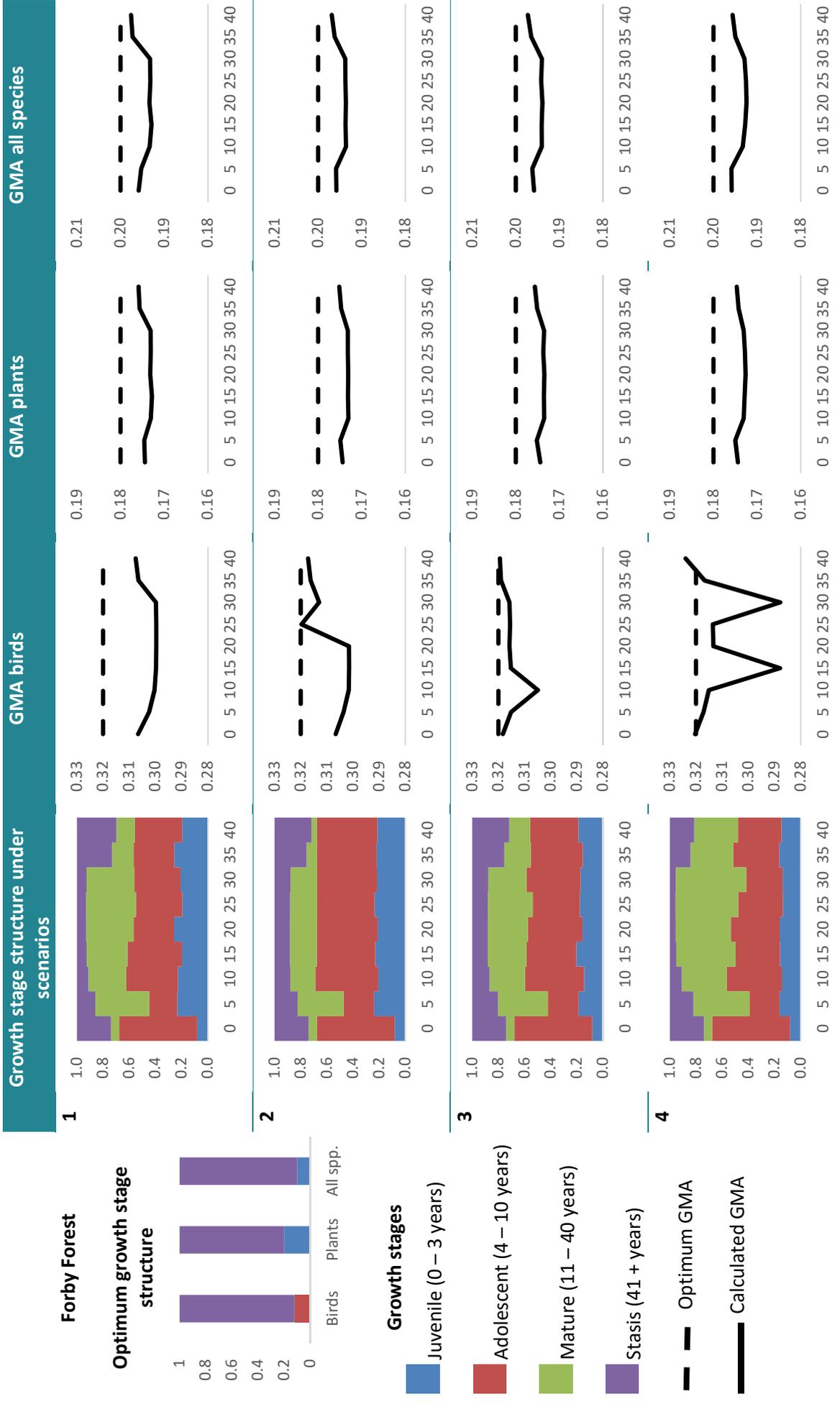
The degree to which the planned burning scenarios produced differences in trajectories of growth stage distributions varied amongst EVDs. This is most likely a reflection of the degree to which EVDs were subject to (simulated) burning. However, in no case did growth stage trajectories within EVDs vary dramatically amongst scenarios. The pattern of growth stage distributions was similar across all scenarios in Tall Mixed Forest, Moist Forest and Grassy/Heathy Dry Forest EVDs (Figures 12.2, 12.5 and 12.6). In the Foothills Forest EVD, scenario 4 resulted in a preponderance of vegetation in the mature growth stage (11-40 years since fire) from 20 to 40 years, while under the other scenarios there was a more even mix of growth stages over the course of the simulations. In Forby Forest, after 40 years, scenario 4 resulted in about 40% of vegetation being in the youngest two growth stages (i.e. <11 years since fire), while at 40 years under scenarios 1-3 the youngest growth stages made up 60-70% of vegetation.

Within EVDs, GMA values and trajectories were similar across scenarios for birds, plants and all species (Figure 12.2-12.6). In no case did the scenarios produce a distribution of growth stages similar to the optimal distribution. Despite this, GMA values under the scenarios were very close (all within 0.05 and many within 0.01) to optimum GMA values. Similarly, probability of occurrence values for species that are responsive to time since fire varied little under the various scenarios, and were always close to those under the optimal growth stage distribution (Figure 12.7).

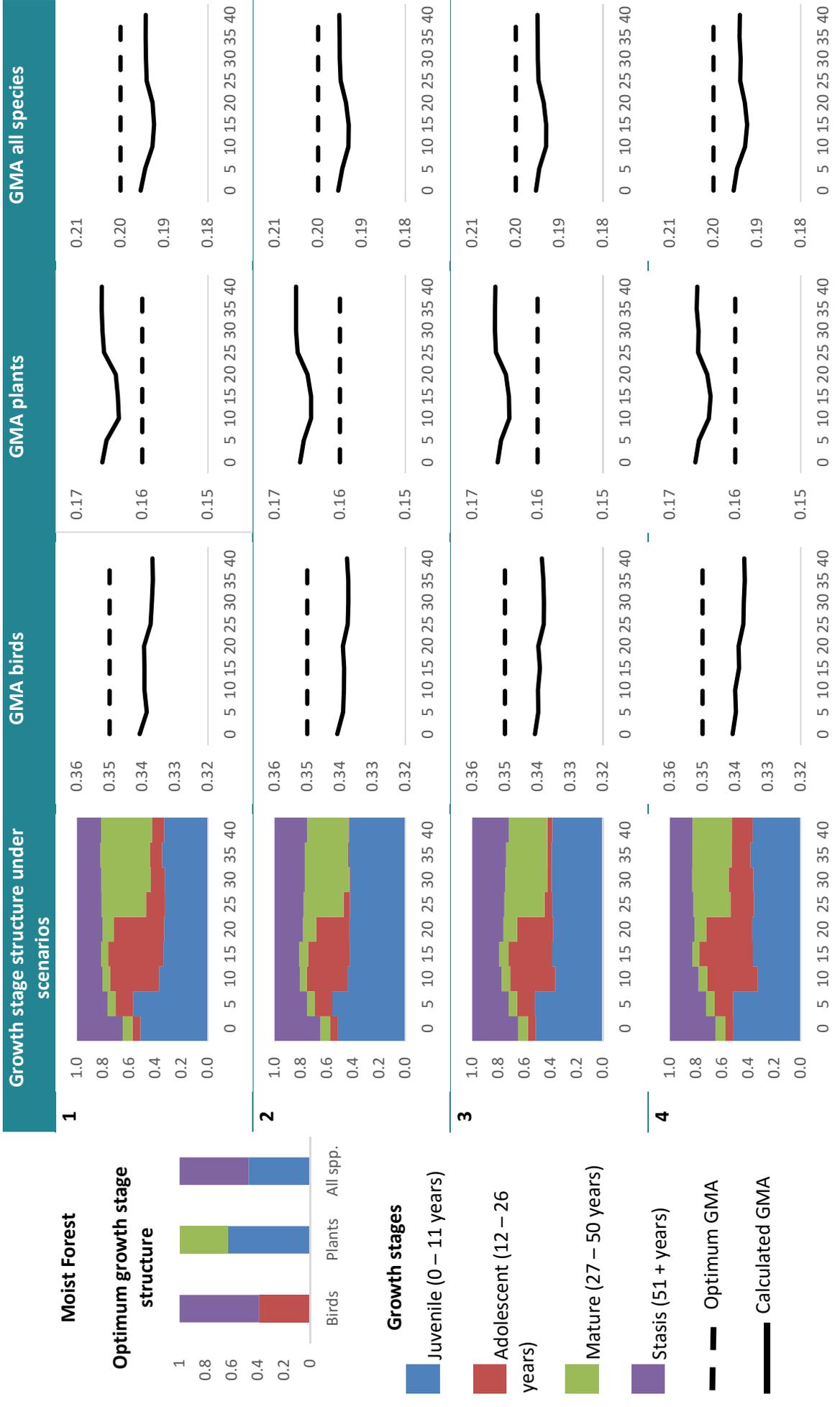


**Figure 12.2.** Optimal proportional growth stage distribution and 40 year trajectories of growth stage distribution and GMA for birds, plants and all species under management scenarios for Tall Mixed Forest EVD. Note that where the calculated GMA is greater than the optimum GMA, this is due to rounding error in calculations.

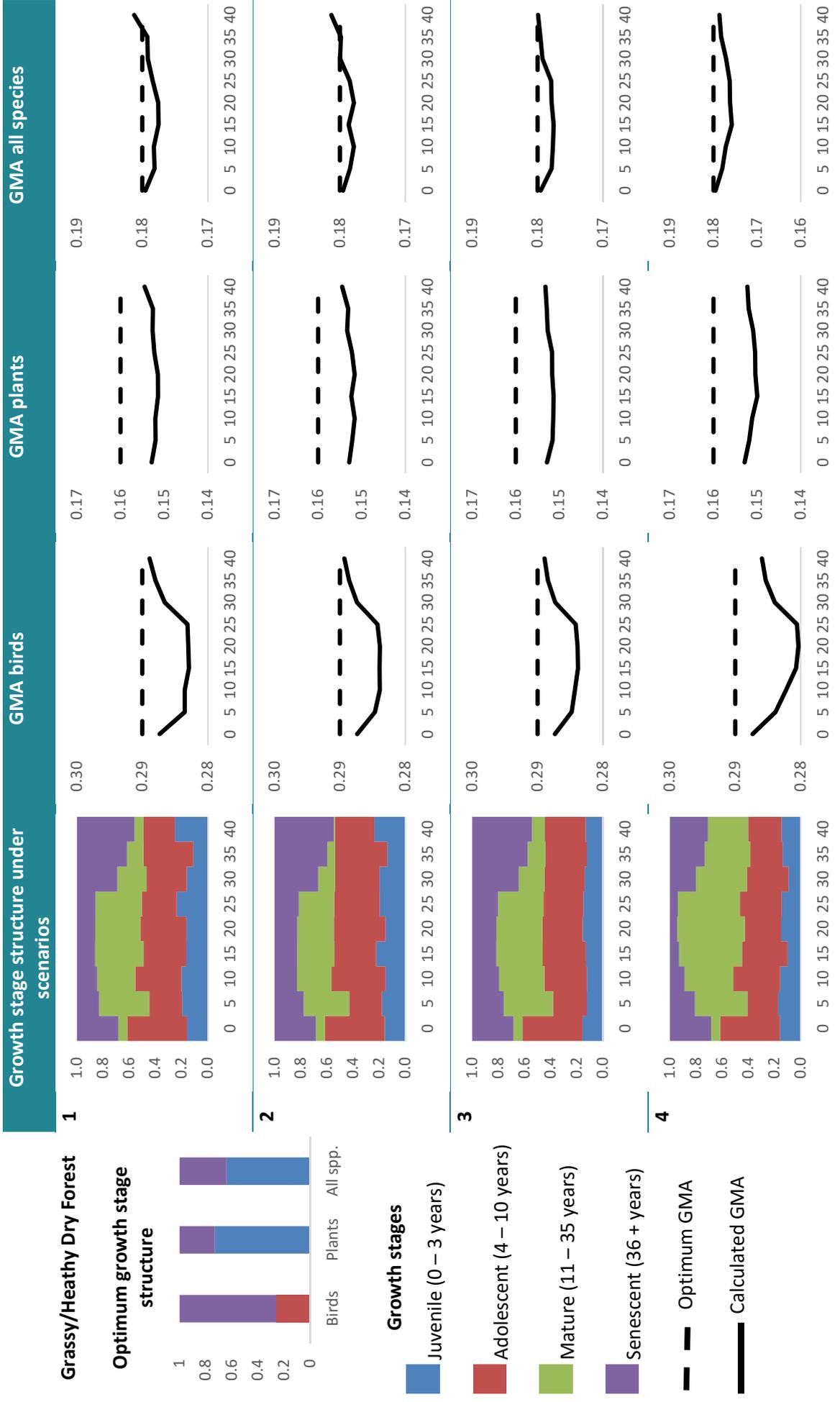




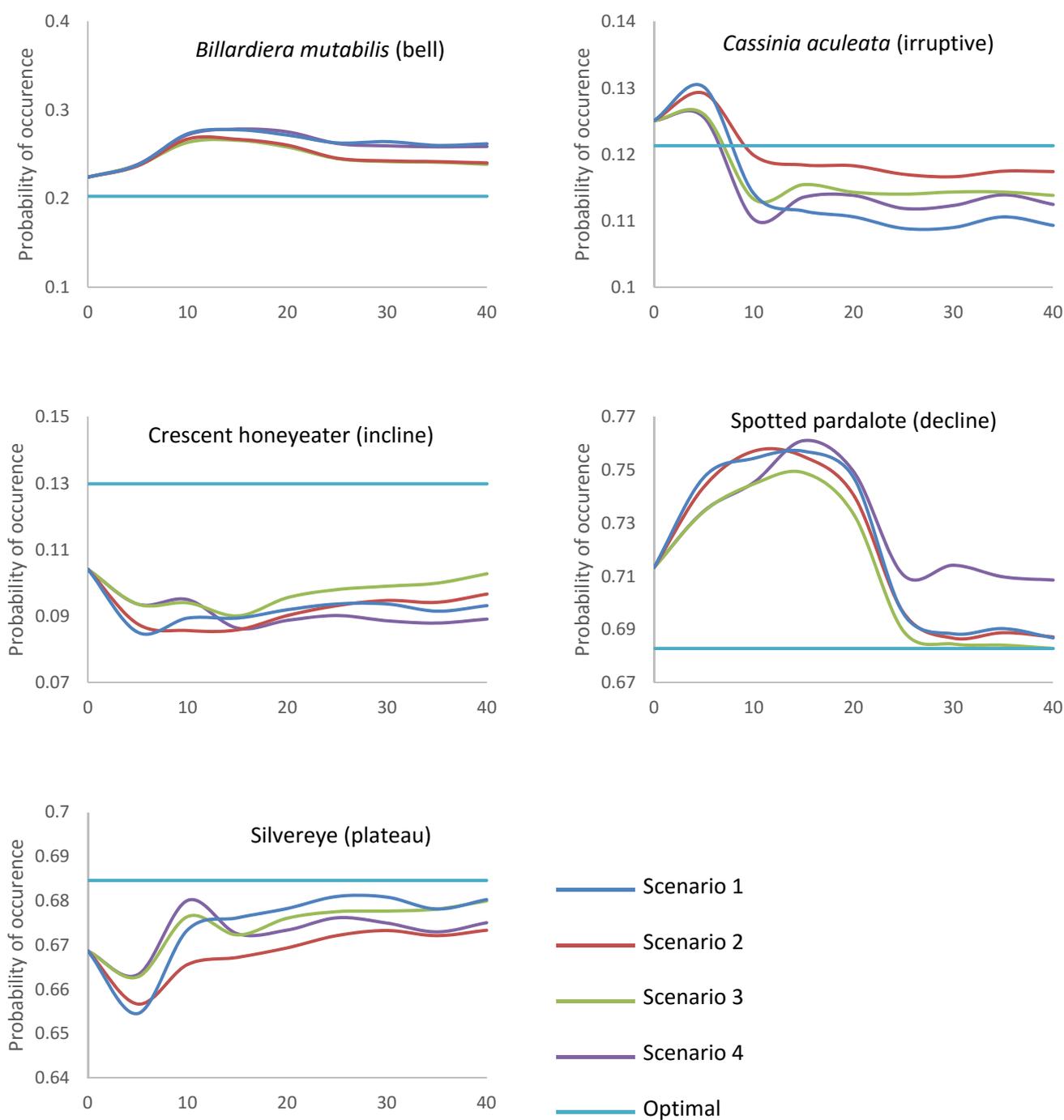
**Figure 12.4.** Optimal proportional growth stage distribution and 40 year trajectories of growth stage distribution and GMA for birds, plants and all species under management scenarios for Forby Forest EVD. Note that where the calculated GMA is greater than the optimum GMA, this is due to rounding error in calculations.



**Figure 12.5.** Optimal proportional growth stage distribution and 40 year trajectories of growth stage distribution and GMA for birds, plants and all species under management scenarios for Moist Forest EVD. Note that where the calculated GMA is greater than the optimum GMA, this is due to rounding error in calculations.



**Figure 12.6.** Optimal proportional growth stage distribution and 40 year trajectories of growth stage distribution and GMA for birds, plants and all species under management scenarios for Grassy/Heathy Dry Forest EVD. Note that where the calculated GMA is greater than the optimum GMA, this is due to rounding error in calculations.



**Figure 12.7.** Change in the probability of occurrence of time since fire responsive species in Moist Forest EVD under hypothetical fire management scenarios (see Table 12.1). ‘Optimal’ indicates a species’ probability of occurrence under a Moist Forest growth stage distribution that maximises GMA (see Section 11). Text in brackets indicates species time since fire response type.

## Discussion

The most striking result of these analyses is the very minor variation in GMA and the probability of species occurrence across and within the various scenarios. In itself, this result may be interpreted as being a product of the generally small differences in growth stage structures between scenarios. However, within EVDs, the GMA and species occurrence values under the scenarios also differed little from those under optimal growth stage distributions. These observations suggest, as noted above, that most species in Foothills Forest (at least amongst those species that could be analysed here) are resilient to quite marked variation in the range of growth stages that were able to be examined. While the occurrence of a number of species varies with time since fire, few are growth stage 'specialists', such that they are present only in particular growth stages. Therefore, it appears that, even in 'landscapes' with growth stage distributions that differ markedly from optimal, few species disappear altogether, and hence the overall GMA is maintained.

The scenarios presented here are 'spatially explicit' in the sense that they are derived from rule-based modelling of fire management in a real landscape. However, GMA and species occurrence values were calculated on the basis of simply summing the area of the landscape within each growth stage under each scenario. The analysis did not account for the size or spatial arrangement of resulting patches of vegetation within growth stages. These obviously are important issues for the persistence of species in landscapes subject to fire (e.g. Sitters *et al.* 2014). Developing methods that more fully incorporated the spatial attributes of fire regimes into modelling biodiversity outcomes would represent a major step forward in fire ecology and management.

# 13 Conclusion

## A resilient system?

The Foothills Fire and Biota project represents one of the few attempts to analyse biotic responses to fire regimes across an extensive and varied forest system. Overall the project results indicate that the Foothills Forest system is relatively resilient, in the sense that it appears most common species are likely to persist on a landscape scale if subjected to fire regimes within the range of historical variation.

Our findings are somewhat in contrast with those from similar studies in other fire-prone systems. For example, in Mallee and heathy woodlands, biodiversity, as indicated by GMA, was more strongly affected by variation in growth stage distribution than Foothills Forest in the current study (Di Stefano *et al.* 2013; Kelly *et al.* 2014). This difference may be due to the differences in the effects of fire between systems. In Mallee woodland, fires are stand replacing, in that there is near complete removal of existing vegetation, including eucalypt stems and important habitat features such as *Triodia* hummocks. Vegetation recovery takes several decades, and the abundance of many habitat features is strongly related to time since fire (Haslem *et al.* 2010). In contrast, in Foothills Forest even high-severity fires are not stand replacing, with most trees surviving. Fire has only a minor impact on the availability of habitat features such as logs (Bassett *et al.* 2015), and, beyond the first 2-3 years post-fire, the relationship of vegetation structure to time since fire is muted (Bassett *et al.* in prep.; Section 8.1 above). Therefore, many important structural elements within Foothills Forests persist through fire events and/or are only loosely coupled to time since fire (Swan *et al.* 2015). This in turn may explain why there are relatively few 'growth stage specialist' species in the system (Sections 10, 11) and the minor changes in GMA with variation in growth stage distribution (Sections 11, 12) observed in the current study.

Foothills Forest also differs from systems such as Mallee woodland in that fire severity can vary markedly, both within and between fires. Previous studies in Foothills Forest have shown that fire severity is a strong determinant of the effects of fire on biota (Robinson *et al.* 2014; Bassett *et al.* 2015; Chia *et al.* 2015). In the current study we have also found that fire severity can mediate the effects of other fire regime attributes (time since fire, fire frequency) on biota (Sections 8.1, 8.3). This has implications for predicting biodiversity on the basis of time since fire or growth stages – the trajectory of species populations over time may be quite different after a canopy consuming fire compared to after a patchy understory fire. There is a risk of over-stating a species' or communities resilience to high severity fire, if much of the data upon which such conclusions are based is drawn from sites that experienced low severity fire. A lack of historic mapping of fire severity makes incorporating its effects into analyses of biotic responses to fire difficult.

Another factor in the apparent resilience of Foothills Forest to variation in fire regimes may be the inherent environmental variability within the system. For example, previous studies have demonstrated that topography plays a major role in mediating the effects of fire on biota (Robinson *et al.* 2014). Similarly, variation in topography, (micro-)climate and vegetation type may influence both fire regimes and their impacts on habitat and species (Leonard *et al.* 2014; Sitters *et al.* 2014). In the current study we found that biotic responses to fire regime attributes varied amongst EVDs (Sections 8.1, 8.2, 10). In addition, environmental variables such as topography and rainfall often had equal or greater influence on the biota than the fire regime (Sections 8.1, 8.2, 8.4).

## Caveats

It is important to note some caveats to the overall finding regarding the resilience of the Foothills Forest system, related to the nature of the data used in the project. First, there was only sufficient data for analyses on relatively common and widespread species. Therefore, it remains unclear whether rare or threatened species also exhibit resilience to variable fire regimes, or indeed are rare/threatened because they have a narrower range of tolerance in this regard than more common species. Studies focussed on elucidating the fire responses of rare and threatened species would help resolve this knowledge gap.

The second caveat relates to the lack of samples with time since fire of greater than 72 years in most EVDs (75 in Moist Forest). This limitation meant that we were unable to examine any biotic patterns related to the oldest growth stage in all but one EVD. Our results indicate that sites with long time since fire are important for a number of species, in particular birds (Sections 10, 11), suggesting that sites with time since fire beyond the range of our chronosequence will also have high conservation value, but in the absence of data this remains speculative.

A third caveat relates to the necessity of using presence/absence data for most analyses. While we have documented relationships of species occurrence to fire regime and other variables, examining patterns of species abundances to fire regimes may provide a richer understanding of biota/fire relationships within the Foothills Forest system.

Finally, the data analysed within this project were mostly collected from large, contiguous blocks of Foothills Forest in National Parks and State Forest. However, a substantial proportion of the Foothills Forest system is fragmented and/or adjoins agricultural or urban areas. The fragmentation and associated degradation of forest introduces a suite of additional influences on biota (e.g. exotic plants and animals), which may interact with fire regimes, resulting in different outcomes for biota than those observed in more intact forest. Given that the interface between forest and urban areas is a key area for using planned burning to reduce risk to human life and assets, there is a need for research aimed at better understanding of these issues.

### **Project questions revisited**

The project aimed to address three overarching questions. Concise answers, based on our findings, are as follows:

*1. How does the interplay of landscape, vegetation and fire history influence biodiversity?*

Our results confirm that the fire regime is a key driver in the system, influencing the occurrence of numerous species and hence the composition of biotic communities (Section 8). However, for many elements of the biota, the influence of the fire regime is equalled or outweighed by that of climate, topography and biogeography. In addition, a number of species exhibited differing responses to fire regime variables in different EVDs (Section 10), indicating interactive effects of the fire regime with other drivers.

*2. How is biodiversity related to vegetation growth stages (a function of time since fire), and what mix of growth stages should managers aim for in order to conserve biodiversity?*

A number of biotic groups and some habitat structural variables exhibited relationships to time since fire (Sections 8, 9). We examined species time since fire response patterns in relation to current definitions of growth stages for Foothills Forest EVDs. For the growth stages for which data was available, the timing of transitions in species occurrences approximated current growth stage definitions. We found no evidence suggesting alternative growth stage definitions (Section 10).

We found that a mix of young and old growth stages maximised biodiversity (as indicated by GMA) in Foothills Forest EVDs. However, GMA changed little even when the growth stage structure varied considerably from the optimum (Sections 11, 12), indicating the possible resilience of the Foothills Forest system discussed above. This suggests managers may have a degree of latitude with regard to the mix of growth stages in Foothills Forest, at least with regard to conserving a suite of more common species. However, our results also indicate the importance of older vegetation, which, if lost, will, by definition, take decades to replace.

We considered the relationships of biota to growth stages in a 'spatially explicit' setting, based on modelling of plausible fire management options in a real landscape. However, the analysis did not account for the size or spatial arrangement of resulting patches of vegetation within growth stages. Developing methods that more fully incorporate the spatial attributes of fire regimes is an important area for future research.

### 3. *How can managers best use fire to maintain and enhance biodiversity values within Foothills Forest?*

As noted above, managers may have greater latitude in using fire to preserve biodiversity in Foothills Forest than in other systems (e.g. Mallee woodland; Kelly *et al.* 2014), providing older vegetation is maintained. This may mean that ecological fire management could be targeted at maintaining species or habitats of particular interest or value. This could include more fire sensitive vegetation and associated fauna that exist within a matrix of Foothills Forest. In addition, while in our analyses the biota was most consistently related to time since fire, we also found relationships to other fire regime variables, such as inter-fire interval and fire frequency and severity (Sections 8, 9). Future research aimed at informing the incorporation of these parameters into ecological fire management in Foothills Forest would be beneficial, for example testing and refining tolerable fire interval guidelines.

In addition to addressing these questions, undertaking this project has provided some useful insights into designing and managing large scale ecological studies. Our approach of combining data sets provided greater analytical power, but also entailed some disadvantages, including increased 'noise' due to methodological variation, loss of information due to abundance data being converted to presence/absence and the need for more complex analyses. Most significantly, collating and managing the resultant large data sets proved very time consuming. Working with these data sets indicated some lessons for any future large scale monitoring programs. These include: the importance of systematic data management and curation; careful consideration of predictor variables; and the need for targeted surveys to capture data on rare or threatened species. In addition, a robust monitoring program also requires sound survey design, which flows from carefully framed questions that address processes of interest. These lessons are informing the development of improved monitoring guidelines currently being undertaken by La Trobe University and DELWP (Scientifically-based monitoring project).

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# Appendices

Appendix number	Associated Section	Title	Lead author
1	8.1	Fire events and regimes: do they interact to affect vegetation structure in temperate eucalypt forests?	Angie Haslem
2	8.2	Fire in the hills: Relationships of fire regimes and environmental drivers to floristic composition in eucalypt forests	Steve Leonard
3	8.3	Contrasting response of bird functional guilds to fire regime in Victorian Foothills Forest	Josephine MacHunter
4	8.4	The Influence of fire history on mammal species richness in eucalypt forests	Matthew Bruce
5	8.4	The influence of fire regime and environmental drivers on small mammals in foothill eucalypt forests	Greg Holland
6	9	The relative influence of fire regime characteristics on the distribution of birds, small mammals and plants in temperate Australian forests	Luke Kelly
7	10	Empirical assessment of the growth stages identified for foothill forest EVDs	Angie Haslem
8	11	Optimal fire histories for birds, small mammals and plants in temperate Australian forests	Luke Kelly

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