

FIRE REGIMES AND FAUNA IN MONTANE FORESTS IN THE NORTHERN AUSTRALIAN ALPS.

Anthony Corrigan

MRes. Thesis

Supervisors: David Nipperess and Julian Seddon

**Department of Biological Sciences
Faculty of Science and Engineering
Macquarie University**

October 2016

This thesis is written in the form of a journal article from Journal of Applied Ecology

Declaration

I wish to acknowledge the following assistance in the research detailed in this report:

Principal supervisor, David Nipperess assisted in the conceptualising this project, experimental design and statistical analyses.

Julian Seddon, technical supervisor helped with conceptualising the project, statistical analyses, experimental design and field work.

Both also contributed by commenting on drafts of this report. I am very grateful for their input and guidance.

All other research described in this report is my own original work.

Anthony Charles Corrigan

10 October 2016

Title page (journal format).

FIRE AND FOREST FAUNA IN THE NORTHERN AUSTRALIAN ALPS.

Anthony Corrigan, Julian Seddon and David Nipperess

Summary

1. The management of conservation lands in fire prone landscapes across the globe has set up a tension between the conservation of biodiversity and protection of human life and property. Compounding this tension is a lack of knowledge around fauna specific impacts potentially linked to fire management activities such as prescribed burning.
2. To determine if fire frequency and prior fire intensity had an influence on mammal assemblages, wet sclerophyll forests across gradients of these explanatory variables, were surveyed for mammals using camera traps. Univariate and multivariate analysis were applied to investigate relationships between mammal faunal phylogenetic diversity, functional diversity and habitat variable.
3. Species data were analysed to determine the relative phylogenetic diversity and functional diversity of fauna across the fire variable gradients at the site scale and the regional (beta) scale.
4. While the impact of increased fire severity appeared to reduce the occurrence of some taxa, functional diversity was maintained at the scales examined.
5. Policy implications. The findings of this study indicate that the phylogenetic measures and functional diversity measures can be applied to better understand mammal fauna community dynamics in relation to the impacts deterministic and stochastic fire events. In turn, fire management can be better informed to ensure biodiversity assets are maximised across ecological communities,

Word count

- summary - 161
- main text - 8,899
- acknowledgements - 166
- references, - 3369
- tables and figure legends - 973

The number of tables - 17

The number of figures - 24

The number of references 133.

Key words: Mammal fire response; Australian Alps; phylogenetic diversity; functional diversity.

Introduction.

In fire prone landscapes across the world, land managers have applied various treatments to vegetation to minimise the risk of the spread of wildfire that may impact humans and their built assets (Gill & Stephens 2009). The use of prescribed fire for this purpose in areas managed for biodiversity conservation creates a tension between management approaches that on the one hand aim to maximise biodiversity outcomes and on the other primarily aim to provide for the protection of life and property (E.g. Keith, Williams and Woinarski (2002); DellaSala *et al.* (2004); Gill and Stephens (2009); Driscoll *et al.* (2010a); Driscoll *et al.* (2010b)).

The impact of fire management activities on the specific habitat needs of fauna has not been adequately researched and further work is needed (Clarke 2008). Two elements of fire regime (*sensu* Gill 1975), fire frequency and fire intensity have been shown to influence vegetation structure and hence fauna habitat (Haslem *et al.* 2011; Collins *et al.* 2012). In this study, two variables are used to synthesise an environmental gradient where classes of fire severity (the loss or change in organic matter in a system through the impact of fire intensity – see Keeley 2009) and fire frequency, defined here as the number of fires since the year 1900. Two important elements of Gill's (1975) fire regime, time since fire and seasonality have not been included. Time since fire across the non-long unburned sites is taken to be 13 years as all those sites were impacted by the January 2003 Canberra bushfires, while the long unburned sites have no recorded fire history since 1900. The potential confounding effects of the contrast in time since fire between in burned and unburned sites will be discussed later. Seasonality data for the impacted sites is standardised to summer and unknown for the long unburned.

Fire, fauna and habitat

Several authors have raised the point that there is insufficient knowledge and research relating to fire management and fauna (Clarke 2008; MacHunter, Menkhorst & Loyn 2009; Driscoll *et al.* 2010b; Haslem *et al.* 2011). In recent years research in this area can be divided into general (Gill 1975) studies examining the impact of fire across faunal groups (E.g. Bradstock *et al.* 2005; Pryke & Samways 2012), the effect of fire on broad taxonomic groups (E.g. Lindenmayer *et al.* 2008; Westgate, Driscoll & Lindenmayer 2012; Sitters *et al.* 2014) and species specific studies aimed at understanding the interaction of fire and the ecology of individual species (E.g. Driscoll & Roberts 1997; Baker *et al.* 2010; Tuft, Crowther & McArthur 2012). In order to provide sufficient information for land managers to consider biodiversity conservation when planning fire management activities, an understanding of the needs of species and communities across

ecological and management scales is required (Gill 2008; Driscoll *et al.* 2010b; Giljohann *et al.* 2015).

Driscoll *et al.* (2010b) highlight three broad categories where knowledge is lacking in relation to the potential for inappropriate fire regimes to cause population extinctions: (1) developing a mechanistic understanding of species responses to fire regimes; (2) knowledge of how the spatial and temporal arrangement of fire regimes into fire mosaics influences the biota; and (3) understanding interactions of fire regimes with other processes that can either modify the response of species to particular fire regimes, or modify the regimes directly. The first category was considered for plants by Keith, Williams and Woinarski (2002), who recognised that monitoring at the species scale would be difficult and a framework that elucidated a smaller number of informative components was required. Such frameworks have been developed for plants (Kenny *et al.* 2004; Cheal 2010) although issues with their applicability are acknowledged (Kenny 2013). Cheal (2010) for example provides descriptive data for the habitat values on a community by community basis for each successional stage of those vegetation communities found throughout Victoria. However Clarke (2008) argues strongly that while no less important for conservation, plants and plant communities can be poor surrogates for the needs of fauna in relation to fire management. Categories 2 and 3 of Driscoll *et al.* (2010b) logically follow from the first. Without the mechanistic understanding on a species by species level, species within populations and across their range, the broader interactions between elements of fire regimes and those interactions with other ecological forces cannot be determined. To this end the fire ecology of individual species needs to be investigated, especially where entities are of conservation concern. The broader context however also requires ongoing attention and studies that focus on the most appropriate methods (E.g. Whelan *et al.* 2002; Giljohann *et al.* 2015) and those that find broad relationships across taxa (E.g. Haslem *et al.* 2011; Collins *et al.* 2012) should be conducted concurrently with those focussed at the community or species level.

At the site scale (as well as at greater scales), a fire regime that provides for a greater diversity of habitat elements should provide for greater phylogenetic diversity and functional diversity than those of less diverse habitat elements. An increase in fire frequency in wetter eucalyptus communities has been shown to alter vegetation structure by promoting the growth of mid storey shrub species (particularly Fabaceae) and reducing over storey canopy cover (Gill 1975 and references therein). Increasing fire severity similarly influences forest structure by reducing canopy and promoting shrub growth (Gill 1975). At low levels of fire severity it was anticipated

that habitat complexity was greatest due to the patchy nature of combustion and uneven impact on vertical plant structures. This would have promoted a mosaic of habitat features at the site scale.

In relation to the measures employed in this study, it was hypothesised that lower to intermediate levels of disturbance, would provide for the greatest heterogeneity in habitat. This in turn provided for greater opportunities for fauna and hence a prospect for greater phylogenetic diversity. Similarly from a functional perspective, greater habitat variation and complexity should provide for a greater number of species resulting in greater redundancy in ecosystem functional roles. It could be expected that more specialised species (those with one or few Key Functional Roles) would be more abundant in areas of low to moderate disturbance. Communities with multiple species performing similar ecological functions are likely to be more resilient to disturbance than those with single or few entities performing functions (Carmona et al. 2016)

The aim of this study is to make use of functional and biodiversity measures to quantify the impacts of fire regime through an improved understanding of the impacts on faunal diversity, faunal community resilience and fauna habitat. These data can then be synthesised to better inform the management of faunal resources in the montane forests of the ACT through greater understanding of the application of prescribed fire.

The use of phylogenetic diversity as a measure for biodiversity assessments.

The use of phylogenetic diversity (PD) as a biodiversity measure was first proposed by Vane-Wright, Humphries and Williams (1991) as a practical method to assess the relative merits of potential land conservation units constrained within resourcing limits. These authors recognised that the approach they were advocating was limited by its inability to discriminate between the relative values of taxa at different taxonomic levels. Faith (1992) showed that by summing lengths of relative paths in a cladogram, a measure of the relative diversity of each cladistic feature could be quantified. Faith further demonstrated the flexibility of the application of PD to taxonomic levels higher and lower than species. Petchey and Gaston (2002) extended the approach to functional data represented in a dendrogram, exemplifying functional relationships. A broad array of methodology surrounding PD has developed recently, with for example, an entire edition of *Ecology* dedicated to the matter see (Cavender-Bares, Ackerly & Kozak 2012) and associated papers for details).

The value of the PD approach for incorporating evolutionary history into biodiversity conservation and monitoring was the subject of a more recent paper by Faith (2013). He draws together a set of measures including resemblance in PD (Nipperess, Faith & Barton 2010), complementarity, endemism and Hill and Valley numbers (Chao, Chiu & Jost 2010) to calculate expected PD that can then be meaningfully applied to answer questions around conservation priorities. This feature of PD analysis can integrate biodiversity measures with extinction probabilities and account for phylogenetic evenness or rareness from datasets. Faith (2013) concludes the usefulness of PD as primary interpretation of data is only limited in its application by the imagination.

The acceptance of PD as a biodiversity measure was reviewed by Winter, Devictor and Schweiger (2013) who concluded that the measure had potential to be informative in reserve selection provided it was employed with a suite of other diversity measures, for example species richness and functional diversity. They also claimed that the application of PD was ambiguous due to the broad array of approaches available and the lack of clarity in the choice of these approaches by authors.

A further criticism to the approach of using PD as a community measure was its correlation with species richness and the similar trend of these measures to increase with sampling effort (Rodrigues, Brooks & Gaston 2005). In fact, these authors conclude that species richness is a reasonable surrogate for PD and, at least for reserve selection purposes, makes PD redundant. However, the employment of rarefaction to PD calculation (Nipperess & Matsen 2013; Chao *et al.* 2015; Nipperess 2016b) overcomes the problem of autocorrelation with species richness by precisely modelling the relationship of PD with sample size and provides a framework for the measurement of phylogenetic evenness, phylogenetic dispersion and other measures as outlined above. This extension of PD (Δ PD), represents the initial slope of the rarefaction curve and provides an index of PD that can be applied across a range of entropic scales - alpha, beta and gamma (see Jost 2006). Specifically, for the purposes of this study, a measure of phylogenetic dispersion, independent of species richness, is defined as the initial slope of a rarefaction curve describing the relationship between species richness and PD (figure 1).

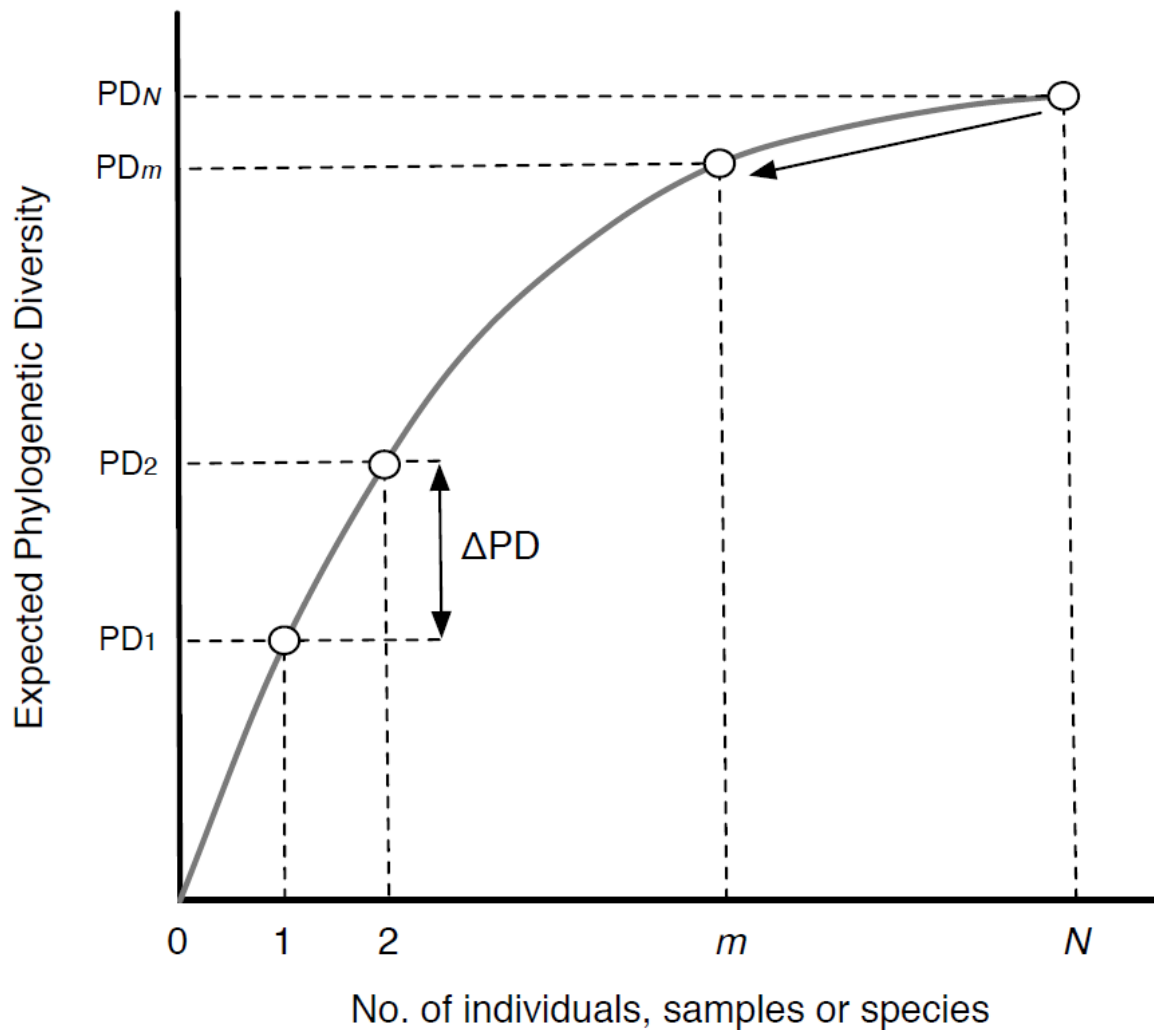


Figure 1. Phylogenetic Diversity (PD) and sampling depth. The x-axis shows the level of sampling, measured in accumulation units (individuals, samples or species) and the y-axis is expected PD. ΔPD is the expected increase in PD the randomly drawn first and second sampling units, PD_1 and PD_2 . PD_N is the PD for the complete data set (N), while PD_m the expected PD of the subset m , that has been rarefied (unidirectional arrow) through a process of randomised subsampling (reproduced from Nipperess 2016b).

The refined use of PD as described above can be applied to provide information relevant for the conservation of biodiversity and the management of fire in two ways. Firstly by understanding the relative phylogenetic structures in various successional stages within and across ecological communities, complementarity and rarness of these structures can be quantified (E.g. McNeely *et al.* 1990; Lean & Maclaurin 2016) and management actions taken to ensure they have representation within any management unit (Kraft & Ackerly 2010; Nipperess, Faith & Barton 2010; Rosauer *et al.* 2014; Yan *et al.* 2016). Secondly by understand how a disturbance gradient (in this case using fire variables) impact on phylogenetic structure, the relative robustness and resilience of communities can be determined in relation to their loss of biodiversity along that gradient. . This approach has been used in a number of studies (Helmus *et al.* 2010; Flynn *et al.* 2011; Nipperess *et al.* 2012; Qian & Zhang 2016). For example Kraft and Ackerly 2010, found that

phylogenetic measures relating to community structure varied at the plot and site scale in relation to habitat filtering in Amazon rain forest communities. Such habitat filtering of phylogenies has also been demonstrated at larger scales for example Yan *et al.* (2016) concluded in their study conducted across mainland China amongst terrestrial vertebrates environmental filtering along with speciation was important for determining species assemblages.

Functional diversity

Laureto, Cianciaruso and Samia (2015) reviewed the background of functional diversity (FD) and its usefulness as a concept for explaining species - habitat relationships as well as its utilisation in the evaluation of community responses to environmental gradients and perturbations. Since the early 1970's, trait based approaches have been used to investigate the impacts of environmental perturbation and species influences on ecosystem function (Hooper *et al.* 2002). This has been particularly true for plants where broad relationships have been established between plant traits and global environmental gradients (E.g. Reich, Walters & Ellsworth 1997; Deil 2005; Kattge *et al.* 2011). Petchey and Gaston (2002) proposed a methodology for calculating FD in a similar manner to that described for PD by Faith (1992) by summing the branch lengths of a dendrogram produced by cluster analysis on functional characters of component species. That approach has been utilised in this study but has been extended to create an index of functional dispersion (Δ FD) in the manner described for phylogenetic dispersion (Δ PD) by Nipperess (2016b).

Function is often considered analogous to traits or sets of traits that may infer functional characters (E.g. Carmona *et al.* 2016), however, as many have pointed out, (E.g. Bolnick *et al.* 2011; Auger & Shipley 2013; Jackson, Peltzer & Wardle 2013) there are issues surrounding intra specific trait variability that need to be resolved. The importance of intra specific trait variation also varies according to alpha, beta and gamma scales (Loreau *et al.* 2001; Chalmardrier *et al.* 2015). Carmona *et al.* (2016) have proposed a solution that is scale independent and have developed a group of functions they term "trait probability densities" and a framework that can be used to provide inputs into traditional FD indices such as richness, evenness and divergence. While more sophisticated than previous methodology, this approach is reliant (as are previous methods) on forming a strong link between traits and ecosystem function. Although this link may seem intuitive, a broad range of morphological, behavioural and physiological traits may be needed to adequately describe the functional role of a species in its environment (Nock, Vogt & Beisner 2001). The number and type of traits to be considered varies with scale, the organism or community of interest and type of interaction in question (Poos, Walker & Jackson 2009).

Rather than focussing at the trait level, Marcot and Vander Heyden (2001) developed Key Ecological Functions (KEF), a role based framework that can be constructed from the known life histories of species within a community. The “key” ecological function of a species is its main ecological role or roles within the community. The advantage of this approach is its ease of application – the links between the measures and actual function are explicit, and measures are meaningful in an ecosystem management context.

Marcot and Vander Heyden (2001) developed such a hierarchical framework of ecological functions for the Pacific north west of the United States. These functions included trophic relationships, nutrient cycling roles, organismal relationships, vertebrate disease vector and reservoir roles, soil relationships, wood structure relationships, water relationships and vegetation structural relationships. For the purposes of this study, a subset of these roles and relationships was constructed and these were assigned to mammal species by consulting published accounts of life history or other ecological data (see supplementary material). The subset represents a set of roles that represent energy and nutrient transfer through the mammal fauna of the ecosystem as well as ecosystem engineering functions. The KEF’s chosen for this study were depicted against each of the species recorded in the ACT’s wildlife atlas (ACT Government 2016) figure 2.

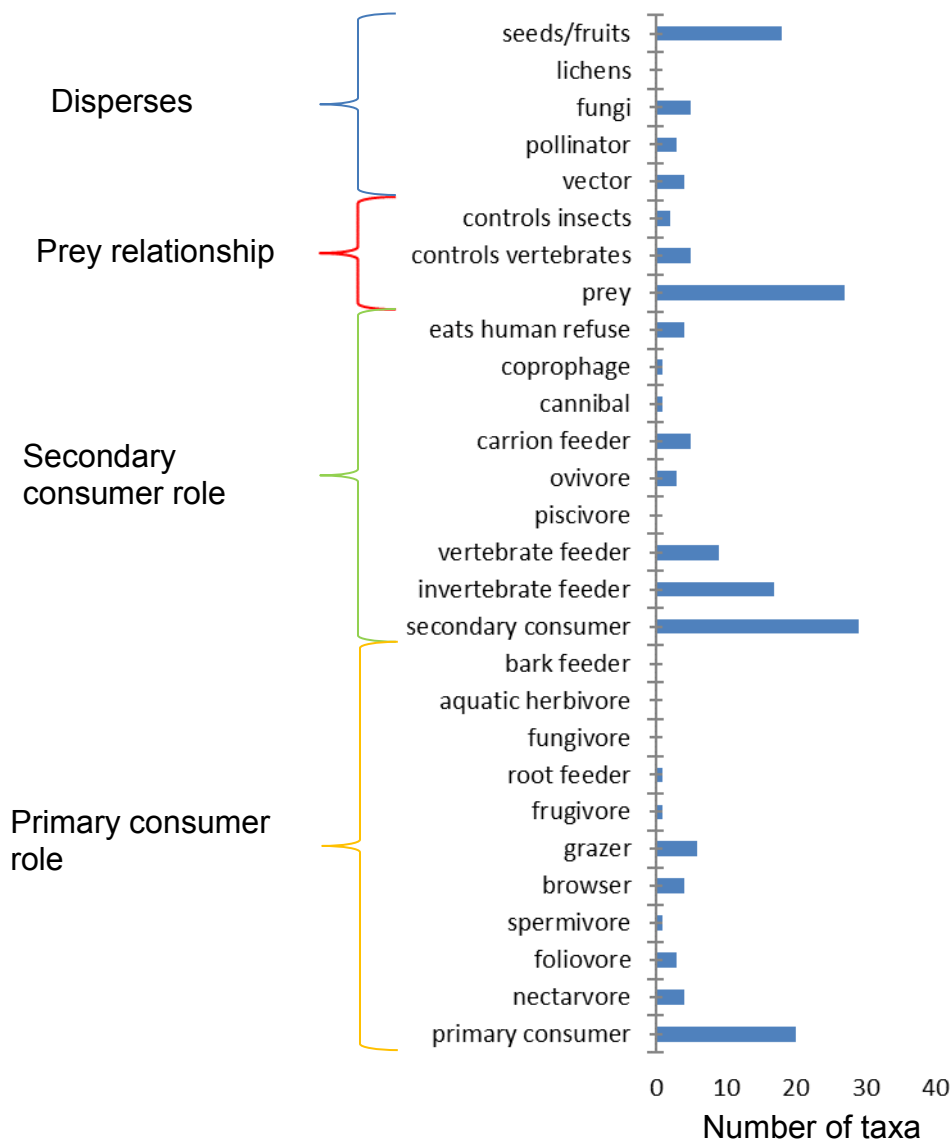


Figure 2. An example of a frequency histogram representing the number of species for each Key Ecological Function category drawn from species recorded in the ACT's Wildlife Atlas (ACT Government 2016) for Namadgi National Park and adapted from Marcot and Aubrey (2003). The broad KEF's of primary consumer, secondary consumer and prey are shown to indicate the overall contribution of those entities.

As vegetation plays an important role in the provision of habitat and resources for fauna, the impact of fire on the structure of the vegetation was considered to be an important variable for this study. The impact of fire on fauna habitat has been recently demonstrated in south eastern Australia (Nimmo *et al.* 2014; Swan *et al.* 2015). In particular, Swan *et al.* (2015) demonstrated that the impact of fire on habitat elements such as litter and shrub cover varied with vegetation type.

Study

This study was conducted within Tidbinbilla Nature Reserve (TNR) and Namadgi National Park (NNP) in the Australian Capital Territory that is located within the south east of Australian. These conservation reserves form part of Australian Alps National Parks System that encompasses the Australian Alps bioregion of Victoria, NSW and the ACT.

To better inform fire management in NNP and TNR and other conservation lands in the ACT the Environment and Planning Directorate undertakes monitoring and research into fire related activities of the ACT Parks and Conservation Service.

This study forms a component of a larger research framework (Mulvaney *et al.* 2014) being undertaken by the Conservation Research unit (CR), Environment and Planning Directorate of the ACT Government that seeks to provide an understanding of the influences of fire management activities on the biodiversity values of protected areas in the ACT. This information is then used to inform management decisions including the development of strategic fire management plans that prescribe a spatial and temporal mosaic of fuel reduction burning on conservation land. The research and monitoring efforts of CR were primarily focussed on providing data around the minimum tolerable fire interval for vegetation across forest communities until recent years (Kitchin 2008). As mentioned previously, several authors have commented that the needs of fauna had not been adequately considered by land managers when designing and implementing fire fuel management programs. ACT Government ecologists recognised this knowledge gap and have implemented the current research framework with the goal of integrating fauna requirements (Corrigan *et al.* 2015) with the requirements of other ecological assets for fire management.

The research framework has identified several priority forest communities in the conservation lands of the ACT, including tall, wet sclerophyll forests, snowgum woodlands, alpine bogs and mires and dry sclerophyll forests. Tall wet forest communities were chosen as the subject of this study as they represented a smaller proportion of the total area of the forests of interest, providing the opportunity to test the approaches detailed here. The findings of this study will inform the design of future studies in other, more widespread forest communities.

To test the hypothesis that fire severity (*sensu* Keeley 2009) and fire frequency (Gill 1975) shape vegetation structure and hence habitat for fauna, this study examined the phylogenetic structure of bird and mammal fauna across a pyrodiversity gradient in wet sclerophyll forests conservation

managed lands in the Australian Capital Territory (ACT). The relationship between community functional response and pyrodiversity were also studied, making use of an approach developed for studying mammal fauna population dynamics in the north west conifer forests of North America (Marcot & Vander Heyden 2001; Marcot & Aubrey 2003). The term “pyrodiversity variable” is used here to refer to the environmental gradients formed by fire severity and fire frequency.

Fire management in the Australian Alps and the ACT

Aboriginal use of fire in the Australian Alps at the time of European settlement was variously reported by early explorers and settlers (Zylstra 2006 and references therein) as being widespread, and applied throughout vegetation communities in montane and tableland areas. However more recently it is believed to have largely been restricted to the lower woodlands and grasslands (Zylstra 2006). With the advent of agriculture in this landscape in the mid to late 1800's, fire was seen as a method for renewing pasture and reducing the cover of trees and shrubs throughout the alps (Pryor 1939).

Pryor (1939) remarked that the fire frequency had apparently increased markedly since European settlement into the mountainous forests of the ACT and adjacent ranges. He proposed that fire had been an irregular event, occurring every 50-100 years prior to this period where lightning would have been the most likely ignition source. His assertions are supported by more recent analysis (Banks 1989; Zylstra 2006). From a reanalysis of Pryor's data in combination with other sources, Banks (1989) reported that across surveyed sites in the Brindabella Ranges (the western most range of the ACT), the mean fire interval increased from one fire in the entire 130 year period prior to 1860 to an average of one fire every 4.9 years in the period 1860-1973.

The ACT was declared a federal territory in 1902 and the Cotter River Catchment was set aside for as water catchment for the development and sustainment of the national capital, Canberra (IconWater 2016). From that time until the declaration of Namadgi National Park (NNP) in 1984, the catchment was largely managed for water catchment and some forestry production that involved the use of fire for hazard reduction and silviculture (ACT Government unpublished data). From 1984 until 2003 fire management in NNP was predominantly focused on the suppression of wild fire and some minor, experimental ecological burning (Don Fletcher pers. comm.).

Recommendations from the Victorian Royal Commission into the 1939 fires included more burning in forests for protection purposes and the establishment of a fire fighting authority (Stretton 1939). The use of prescribed fire to reduce the impact of wild fires has become a widely accepted

practice across Australia (Keith, Williams & Woinarski 2002; DellaSala *et al.* 2004; Parr & Anderson 2006; Gill 2008; Gibbons *et al.* 2012). Coroner Maria Dugan, inquiring into the 2003 Canberra fires, recommended that a hazard reduction program “involving regular and strategic burning in all areas of the ACT” be implemented (Australian Capital Territory & Doogan 2006). The use of prescribed fire is a key strategy for the ACT Parks and Conservation Service annual fuel management program, the Bushfire Operations Plan (BOP), (ACT Government 2015) in its application to Namadgi National Park and Tidbinbilla Nature Reserve where a mosaic burning approach as described in (Parr & Brockett 1999; Parr & Andersen 2006) has been adopted.

Green and Osborne (2012) also documented the increase of fire frequency during the late 19th and first half of the 20th centuries and raised concerns that wildfire frequency would increase with climate change. They highlighted that repeat fires in alpine and sub alpine communities in 2003, 2006 and 2007 were a phenomenon that is likely to increase with changing climate. Repeated high severity wildfire and subsequent habitat change are likely to push species that are already impacted by climate induced stressors to local extinction (Green & Osborne 2012).

As early as 1939, Pryor recognised that fire severity and fire frequency played a significant role in forest dynamics and in forest structure in particular. While his insight was clearly in relation to the indiscriminate nature of graziers use of fire, the impacts of repeated fire in forest communities have been echoed more recently (Penman *et al.* 2008; Fisher *et al.* 2009). He claimed that this indiscriminate burning of the upland forests of the ACT would result in the substantial loss of large trees from forest systems and those systems would be replaced by shrub lands:

“The extent of the degradation of the country will depend upon the frequency and severity of the burning, and will be somewhere between the limits of slightly damaged forest, more or less similar to the undisturbed vegetation, and a barren waste of eroded mountains with sparse plant cover.

Continued indiscriminate burning is fatal as a long-range policy of land use. It is self-destructive, and must ultimately be abandoned if the land is to remain productive.” (Pryor 1939, p. 37).

Pryor further described the reduction of several tree species resulting from repeated fire and his observations of long unburned forest patches as having “rather open, well-spaced undergrowth” of small shrubs and grasses.

The research question used to frame this research was: How does a gradient across fire severity and fire frequency influence faunal community assemblage metrics describing phylogenetic diversity, and ecosystem function?, and how do these explanatory variables influence habitat structural elements in wet forests?

Materials and methods.

Site stratification and selection– mammals.

This study was confined to wet sclerophyll communities within NNP and TNR in south eastern Australia (see figure 3). To achieve a balanced design for the project, planning involved finding replicate sites across pyrodiversity variables while minimising variance amongst other, potentially confounding variables such as aspect, slope and topographic position. All sites chosen in this study were either long unburned (greater than 50 years since last fire) or were burned in wildfires during January 2003. The sites had experienced no fire since 2003.



Figure 3. Tidbinbilla Nature Reserve (TNR) and Namadgi National Park (NNP) within the Australian Capital Territory (ACT) (left), and the location of the study area (red star) within Australia (right).

Terrain variables were stratified using Quantum GIS (QGIS 2011) by constructing a vector grid layer of 50*50 metre polygons and then extracting topographic data from raster layers supplied by the ACT Government. These were then transformed into three categories each for topographic position index (TPI) (valley, mid-slope or ridge) and slope, while aspect was generalised into 4 categories (see table 1). Fire severity was drawn from data originally prepared for the ACT Government immediately following the 2003 fires (figure 1). These data were interpreted from Landsat imagery and aerial photography (Barrett 2006). Three classes were synthesised in this study from the original five, (class 1) severity 0 (unburnt); (class 2) severity 1-3; and (class 3) severity 4 and 5.

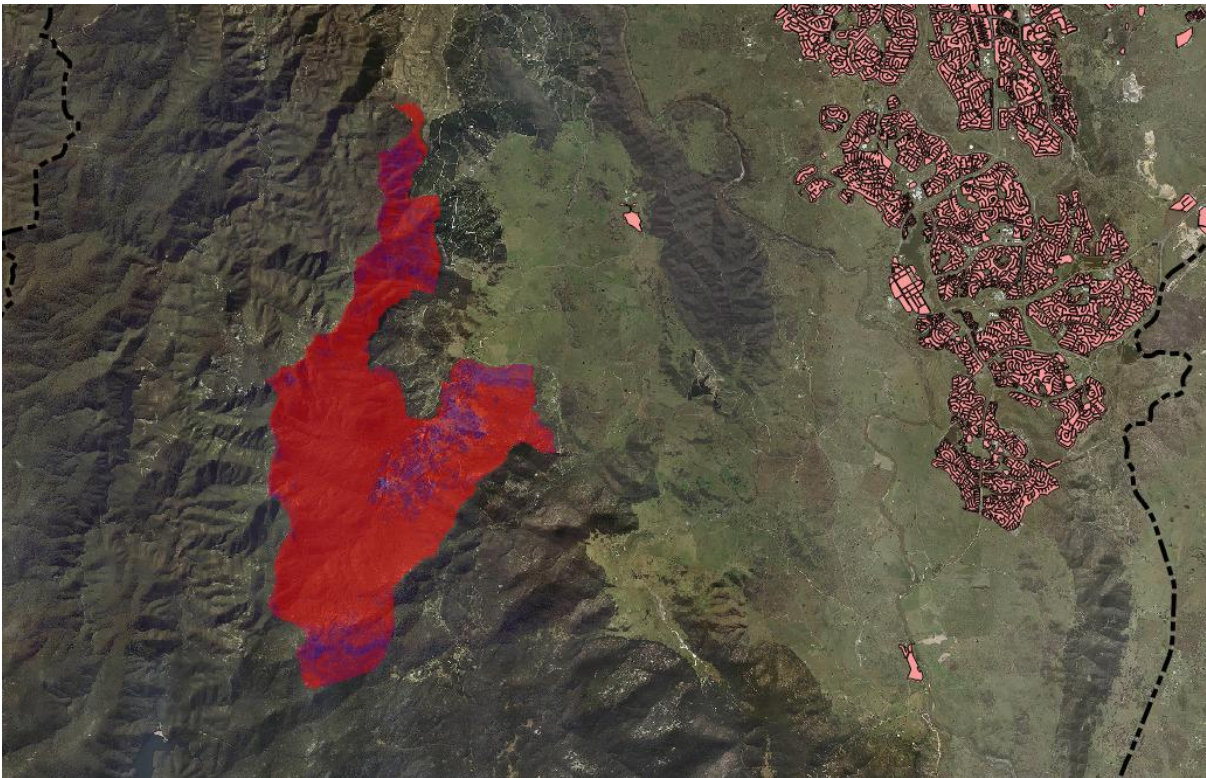


Figure 4. 2003 fire severity raster layer of Tidbinbilla Nature Reserve, west of southern Canberra. The fire severity is represented by a spectral gradient from blue to red, blue pixels are low severity, red are extreme.

Fire frequency classes were constructed from ACT Government fire history data (1900 – present) that had been compiled from a range of sources including contemporaneous maps, personal accounts and digital spatial data collected from fire grounds (ACT government unpublished data).

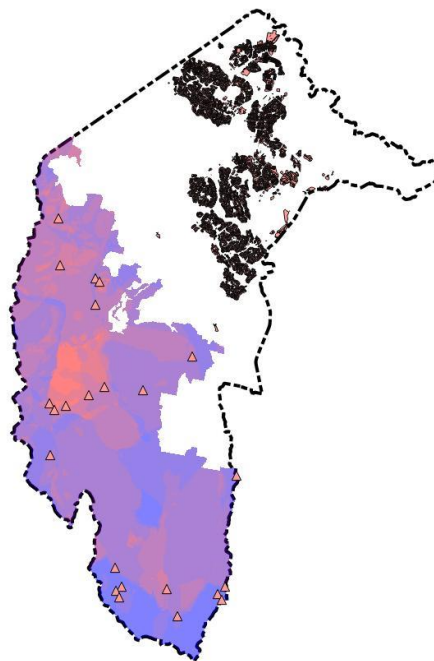


Figure 5. Mammal survey sites across (triangles) NNP and TNR represented here by a fire frequency raster image. Red signifies high fire frequency grading to blue, no recorded fires.

The three classes were defined for this study comprised (1) long unburnt (no fire recorded since 1900); (2) one, two or three fires since 1900; (3) four or more fires since 1900 (see figure 5).

Table 1. Variables used to stratify mammal sites based on explanatory variables. Six replicates with attributes from each row in the table were planned for this project, representing a total of 30 plots. The class code HM was not surveyed as insufficient replicates were found to be available after site inspections.

Number of sites	Frequency class	Severity class	Aspect class	Slope class	TPI class	Class code
6	1 (unburned)	1 (unburned)	3 (southerly)	2 (moderate)	2 (mid-slope)	UB
6	2 (1-3 fires)	2 (moderate)	3 (southerly)	2 (moderate)	2 (mid-slope)	MM
6	2 (1-3 fires)	3 (severe)	3 (southerly)	2 (moderate)	2 (mid-slope)	MH
6	3 (≥ 4 fires)	2 (moderate)	3 (southerly)	2 (moderate)	2 (mid-slope)	HM
6	3 (≥ 4 fires)	3 (severe)	3 (southerly)	2 (moderate)	2 (mid-slope)	HH

QGIS (QGIS 2011) was used to find candidate plot locations with the appropriate characteristics for each of the explanatory and terrain variables. This generated over 500, 000 candidate points from which geographically dispersed subsets were created. Individual sites were then randomly selected from these candidate points using the QGIS random points selection tool.

During field visits, sites were assessed against the stratification criteria and those not suitable were discarded and another candidate chosen from the random sample. Field inspection revealed that most candidate sites in the combination high severity (class 3) and moderate frequency (class 2) had been impacted by more recent planned burns. For this reason, that class of sites (HM) was dropped from the study. At completion of the field work, six replicate sites in each of classes UB, MH and HH and five replicates in class MM had been surveyed.

Bird data.¹

The Canberra Ornithologists Group (COG) had been collecting observational data for birds and creating a data set that includes areas of NNP and TNR. The group kindly agreed to share a subset of these data for this project and other ACT Government research purposes. The data were collected by members of COG in two hectare plots over 20 minute periods in NNP and TNR (Canberra Ornithologists Group 2016). This dataset included bird observations dating back to 1984, however to reduce the effect of time since the 2003 fires and for comparison with the

¹ The data collected here was found to be insufficient for meaningful analysis and the results are not reported. The methods are included here to comply with the MRes thesis instructions that encouraged students to include aspects of the project that were tried, even if meaningful results were not found.

mammal data (see below), data from the most recent year with sufficient data points for analysis (2014) was used here. Data for 2015 was included in the original set, however it was found to have insufficient observations within the target forest formation to be of use to this project.

The data were projected into a GIS layer and analysed at site level to elucidate stratification characteristics (QGIS 2011). Resultant records were filtered to remove those species that were not likely to be regular occupants of wet forest and potential vagrants. This was achieved by checking the species entries in Taylor (2013) to determine their habitat preferences and nesting habits to ascertain residency status. Table 3 below indicates the stratification variables applied to the COG bird observational data.

Table 2. Variables used to stratify bird sites based on explanatory variables.

Stratification variable	Measure	Field value
Vegetation type	Keith class	Tall wet forest
Season	Month	October
Fire severity	Class	3-5
Fire frequency	No. fires since 1900	1, 2, 3 and 6
Residency	Tall wet forest nesting	Species

The stratification yielded 148 records across 10 sites and included 27 species. Figure 6 shows the sites selected for this study.

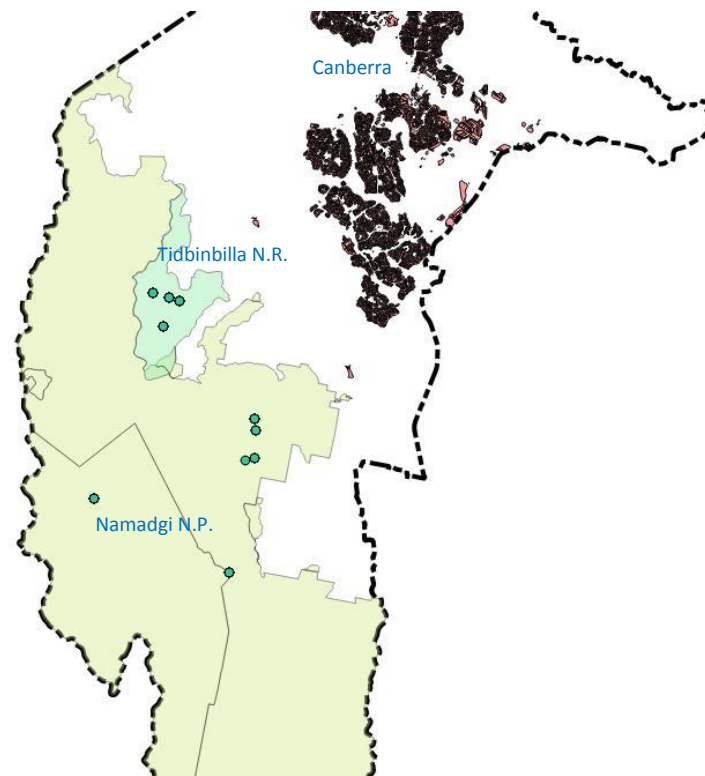


Figure 6. Canberra Ornithologists Group site locations for wet forest bird observation surveys conducted in 2014 used in this study.

After applying the filters and stratification, the available data for analysis were geographically restricted to the central area of NNP and to TNR. This feature of the data inevitably resulted in a restriction in the breadth of fire frequency and severity classes obtainable for analysis, in particular no data from unburned sites was available. Table 3 below indicates the fire variable stratification of the data.

Table 3. Fire variables used in the analysis of bird phylogenetic and functional diversity.

Fire class code	Fire severity (2003 fires)	Fire frequency (since 1900)	Replicate sites
PC32	2	2	1
PC42	4	2	2
PC46	4	6	1
PC51	5	1	1
PC52	5	2	3
PC53	5	3	1

Habitat structure survey.

At the time of the planning and field work component of this project, updated, fine scale vegetation mapping was being undertaken for the entire ACT by the Environment and Planning

Directorate of the ACT. To date, approximately half of NNP has been completed however earlier, coarser scale mapping for the balance of the area was available (Baines *et al.* 2013).

To assess the potential impact of fire variables on habitat elements, the datasets described in table 4 were collected for 19 of the 23 sites of this study (see table 1). Complete data was not collected for all sites as a result of the extreme weather experienced during the survey period (see below).

Table 4. Habitat data collected at each survey site of this project, see figure 4 for details of the plot layout. (After Mulvaney *et al.* 2014).

Dataset	Plot Size	Key Variables
Site data	50 x 20m	Plot location and orientation Aspect Slope
Habitat attributes	50 x 20m	Number of trees with hollows Total length of logs >10cm diameter Number of large trees >40cm dbh
Vegetation cover	50m line transect	Over-storey percent foliage cover Mid-storey percent foliage cover Low shrub cover Grass cover Sedge/rush cover Forb cover Bare ground cover Litter cover Litter depth

At each site, two plots and a transect were established to collect the data types described in table 4, site photographs were taken and the location and landscape details recorded (figure 6).

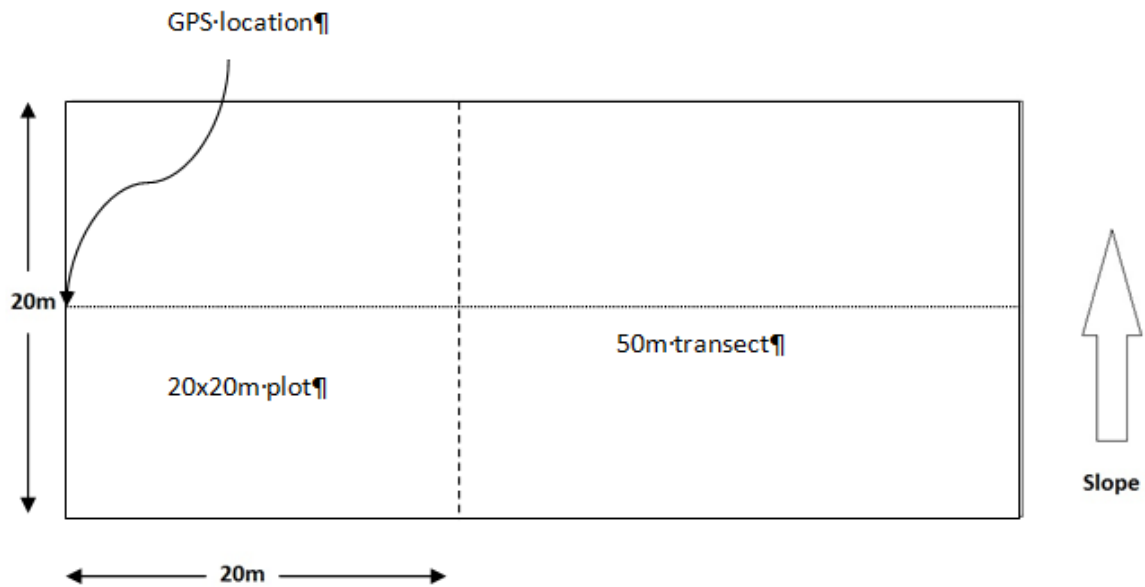


Figure 6. Plot layout for habitat survey undertaken at each site. Adapted from Mulvaney *et al.* (2014).

Details of field methods are listed in the appendix.

Camera trap survey.

The use of remote camera traps as a cost effective method for detecting wildlife in census and survey work has increased remarkably over recent decades (Fleming *et al.* 2014). As the use of cameras has increased in this field, the development of technique has also advanced (Rovero, Tobler & Sanderson 2010; Fleming *et al.* 2014; Si, Kays & Ding 2014; Swan *et al.* 2014; Meek, Ballard & Fleming 2015; Welbourne *et al.* 2015).

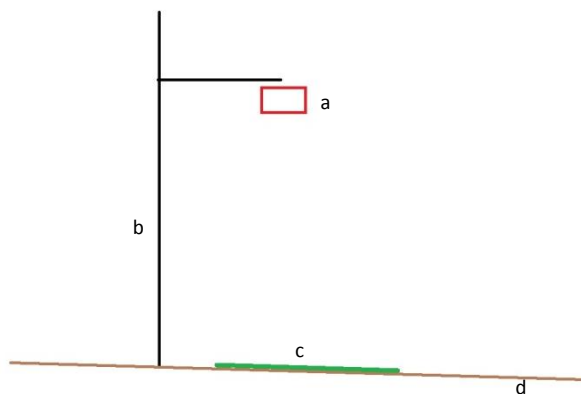


Figure 7. Side elevation diagram of the white flash camera (TYPE WF) deployment for sampling small mammals. The camera (a) is set on a steel post (b) ~0.8m above a cork tile (c) that is scented with rendered bacon fat and sesame oil. The camera is angled so as its detector is parallel to the ground surface (d).

The technique used here was adapted following (Welbourne 2013) and (Towerton *et al.* 2011) to detect small mammal species as well as larger fauna such as macropods, felids and canids. This

method makes use of at least two cameras at each site, one infrared instrument (TYPE IR) (Reconyx PC900) set to monitor a game trail for larger species and the second, a white flash camera (TYPE WF) (Reconyx HC550) , vertically mounted, 800mm above the ground with an attractant to monitor smaller species (figure 7). For the TYPE WF two attractants were applied to a commercially produced cork flooring tile that was anchored to the ground using steel pins. The attractants were commercially produced sesame oil to target herbivores and omnivores (Chang's, Auburn NSW), following the findings of (Diete *et al.* 2016) and rendered bacon fat (to target carnivores), produced by frying approximately 40g of bacon fat in 2 mL of sunflower oil (Crisco) until 20mL of fluid is produced. The TYPE IR camera was mounted to a suitable tree at ~1.5 m above the ground and angled towards the ground to detect fauna moving along the trail. Both types of cameras were set to their most sensitive motion detection setting and programmed to take 5 successive images each time they were triggered.



Figure 8. The image on the left was taken by the TYPE WF camera trap and shows the cork tile on which two separate attractants have been placed. The image on the right was taken by the TYPE IR camera trap, designed to detect larger species in this case *Vulpes vulpes*.

Camera trap deployment time for detecting small mammal resident species within a site was found by De Bondi *et al.* (2010) to be five nights and Si, Kays and Ding (2014) found a trade-off between trap nights and additional sites for optimal detection. In this study, traps were planned to be deployed for a minimum of 7 nights and a maximum of 14 nights however, heavy snow and rain was experienced during the fieldwork period that impeded access to the sites resulting in highly variable effort across sites (7-73 nights with an average of 22.5 nights). The winter was the third wettest on record (Bureau of Meteorology 2016) and access along some roads within Namadgi was denied by the ACT Parks and Conservation Service from early June due the potential damage to those roads. The lack of access in the south of the park resulted in some site data not being collected. Only 19 of 23 sites had complete habitat data at the end of field work.

Statistical analysis.

All statistical analysis were conducted in the data analysis software R (R Development Core Team 2016).

Habitat data.

The elements of the habitat data (table 6) were analysed using analysis of variance for any relationship across pyrodiversity classes as well as multivariate techniques. Data were z-transformed to overcome differences in scales.

Variation in trapping effort.

For the reasons outlined above, the number of nights cameras were deployed varied substantially, with a minimum of 7 nights, a maximum of 73 nights and a mean of 22.5 nights the minimum number of trap nights was 32. To ensure that this variation did not contribute to some systematic bias, the relationships between explanatory variables, diversity measures and trap nights were tested using Pearson's product-moment correlation and linear modelling. To further guard against trap night induced bias, rarefaction techniques (Simberloff 1972; Heck, van Belle & Simberloff 1975) were applied to find the expected richness of each site for a specific number of randomly selected trap nights. Expected richness (S) for a given sampling effort is the sum of probabilities (p) of each species occurring in a subset of m accumulation units (Chiarucci *et al.* 2008; Nipperess 2016b), equation 1:

$$E[S]_m = \sum_i^s m p_i \quad (1)$$

In this case p is calculated in the square brackets in equation 2, where N is the number of trap nights, n_i is the total number of nights for species i summed across all cameras for each site and m is the minimum number of trap nights across all sites.

$$E[S]_m = \sum_i^s \left[1 - \frac{\binom{N - n_i}{m}}{\binom{N}{m}} \right] \quad (2)$$

Phylogenetic and species richness analysis

Mammals.

To analyse the phylogenetic characteristics of the mammal communities detected during the survey, a published mammal supertree (Kuhn, Mooers & Thomas 2011) was obtained from internet resources. This supertree was originally published by Bininda-Emonds *et al.* (2007) and had been updated by Fritz, Bininda-Emonds and Purvis (2009). In order to resolve the numerous polytomies in this supertree, Kuhn, Mooers and Thomas (2011) generated a set of 100 equally plausible trees using a constant rate birth-death process in conjunction with Monte Carlo Markov chain algorithms. Phylogenetic data was then analysed utilising R packages 'ape' (Paradis, Claude & Strimmer 2004), 'vegan' (Oksanen *et al.* 2007), and the stand-alone functions 'phylorare' (Nipperess 2014) and 'phylocurve' (Nipperess 2016a).

Phylogenetic response to the fire variables was assessed using the phylogenetic dispersion index across the fire variable classes. The methods of Nipperess (2016b) were used to calculate phylogenetic dispersion indexes at each site class, UB, MM, MH and HH and 'phylocurve' was used to produce rarefaction curves. In this case incidence data (presence) by site was compared to the 100 trees of Kuhn, Mooers and Thomas (2011) and the mean rarefied dispersion (ΔPD) calculated.

Functional response was similarly calculated between sites and an analysis of variance model of the outputs was generated using analysis of variance. Species richness was also calculated at the site scale and the same analysis of variance was applied as well as the post hoc test, Tukey's honest significant difference (HSD) using expected richness data to overcome potential trap night bias.

Species compositional data were analysed for differences amongst fire variables using vegan package functions. Initially Nonmetric Multidimensional Scaling (NMDS), a form of ordination describing the relationships within and between alpha and beta data groups was used to identify potentially significant differences in fire class values. PERMANOVA was used to test the assumption of homogeneity within data groups in relation to multivariate dispersion, in the context of a resemblance measure, in this case Euclidian distance. Permutation tests utilising analysis of variance were then applied to determine the significance between data groups.

Birds

The phylogenetic features of the bird data set were analysed in a similar manner to that of the mammal data. In this case, the supertree (1,000 trees) of Jetz *et al.* (2012) was used to create a phylogenetic tree for the filtered COG bird data. Phylogenetic dispersion vales for each pyro-class were calculated and rarefaction curves produced.

Species richness metrics were produced using the methods outlined for mammals (above).

Functional analysis

Functional dendrograms (figure 10) for mammal and bird functional diversities were generated using data from published sources of functional traits (see supplementary material).

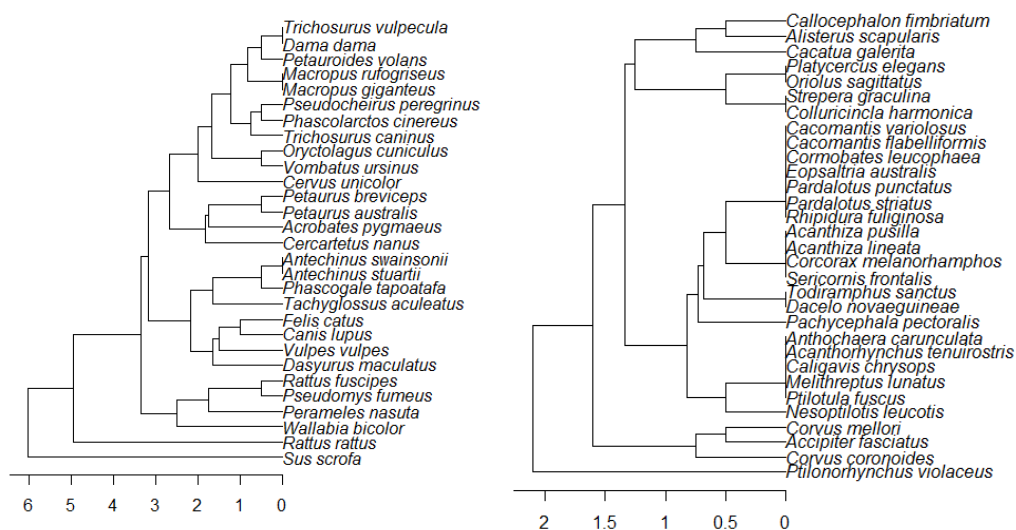


Figure 10. Functional dendrograms for mammals (left) and birds (right) generated using the Vegan Package (Oksanen *et al.* 2007). Scales indicate manhattan distance.

For mammals, Key Ecological Functions (KEF) (Marcot & Vander Heyden 2001; Morrison, Marcot & Mannan 2012) were assigned from published ecological data for each of the species recorded to be present in NNP or TNR from ACT Wildlife Atlas data (see reference table in appendix N). As each of these functional traits were recorded as present or absent, a distance matrix was constructed using Manhattan distance. These distance values were then used to construct a functional dendrogram. To do this a analysis was performed using a cluster analysis(Oksanen *et al.* 2007) in this case, making use of Ward's minimum variance methods (Ward Jr 1963) to construct

the dendrogram. This was in turn used to generate the functional dispersion index in the same manner the phylogenetic dispersion index was calculated, for each site and across explanatory variable classes.

For birds the functional measure for KEF was based on diet traits drawn from Garnett *et al.* (2015). The categories used to construct the functional measure were major dietary components of one or more of the following: fruit, nectar, seeds, foliage, corms, invertebrates, vertebrates and carrion.

The aov function in R was utilised to construct a linear model using analysis of variance (ANOVA) to evaluate any significant difference in functional dispersion across sites and explanatory variable classes. This was appropriate given the balance across the data. Post-hoc analysis, Tukey's Honest Significant Difference was conducted to identify pairwise relationships between fire classes.

Pearson's product-moment coefficient was used to examine any correlative relationship between phylogenetic dispersion and functional dispersion at the site level.

Results

Sites were surveyed and camera traps deployed during the period 15th of March to 29th of August 2016. A total of 23 sites were surveyed across the explanatory variable classes. Weather was a major barrier to site access that limited the collection of quality habitat data from all sites surveyed for mammal presence (19). Camera trapping yielded a total of 7,860 images and 358 mammal detections of 18 species.

Mammals

The mammal species detected are detailed in table 5 below.

Table 5. Mammal species detected across the 23 sites in this study, 18 species were recorded. Nomenclature and authorities follow (Van Dyck & Strahan 2008).

Species	Common name
<i>Antechinus agilis</i> (Waterhouse, 1840)	Agile antechinus
<i>Antechinus swainsonii</i> (Macleay, 1840)	Dusky antechinus
<i>Canis lupus</i> (Linnaeus 1758)	Dingo
<i>Cercartetus nanus</i> (Geoffroy and Desmarest 1817)	Eastern pygmy possum
<i>Felis catus</i> (Linnaeus 1758)	Feral cat
<i>Macropus giganteus</i> (Shaw 1790)	Eastern grey kangaroo
<i>Macropus rufogriseus</i> (Desmarest 1817)	Red-necked wallaby
<i>Oryctolagus cuniculus</i> (Linnaeus 1758)	Rabbit
<i>Pseudocheirus peregrinus</i> (Boddaert 1785)	Common ringtail possum
<i>Rattus fuscipes</i> (Waterhouse 1839)	Bush rat
<i>Rattus rattus</i> (Linnaeus 1758)	Black rat
<i>Sus scrofa</i> (Linnaeus 1758)	Pig
<i>Tachyglossus aculeatus</i> (Shaw 1792)	Echidna
<i>Trichosurus cunninghami</i> (Lindenmayer, Dubach and Viggers 2002)	Mountain brush tallied possum
<i>Trichosurus vulpecula</i> (Kerr 1792)	Common brush tailed possum
<i>Vombatus ursinus</i> (Shaw 1800)	Wombat
<i>Vulpes vulpes</i> (Linnaeus 1758)	Red fox
<i>Wallabia bicolor</i> (Desmarest 1804)	Swamp wallaby

In all the results listed below, the abbreviations for each explanatory variable class are the class codes in tables 1 and 3 in the methods section above.

Table 6. The number of sites of mammal observations by explanatory variable site classes. The numbers in parentheses indicate the number of sites in that class.

Species	UB (6)	MM (5)	MH (6)	HH (6)
<i>Antechinus agilis</i>	0	0	4	2
<i>Antechinus swainsonii</i>	2	1	0	2
<i>Canis lupus</i>	1	1	0	0
<i>Cercartetus nanus</i>	0	0	1	0
<i>Felis catus</i>	3	0	0	0
<i>Macropus giganteus</i>	2	3	1	1
<i>Macropus rufogriseus</i>	6	4	4	1
<i>Oryctolagus cuniculus</i>	2	1	0	0
<i>Pseudocheirus peregrinus</i>	0	0	1	0
<i>Rattus fuscipes</i>	0	0	2	2
<i>Rattus rattus</i>	0	0	2	0
<i>Sus scrofa</i>	1	0	0	1
<i>Tachyglossus aculeatus</i>	3	1	2	0
<i>Trichosurus cunninghami</i>	3	1	0	0
<i>Trichosurus vulpecula</i>	0	4	5	1
<i>Vombatus ursinus</i>	1	2	5	0
<i>Vulpes vulpes</i>	0	1	1	0
<i>Wallabia bicolor</i>	4	3	5	2

Habitat data.

The habitat data from the 19 sites represented the following number of replicates in each explanatory variable class: HH, 6; MH, 5; MM, 4 and UB, 4.

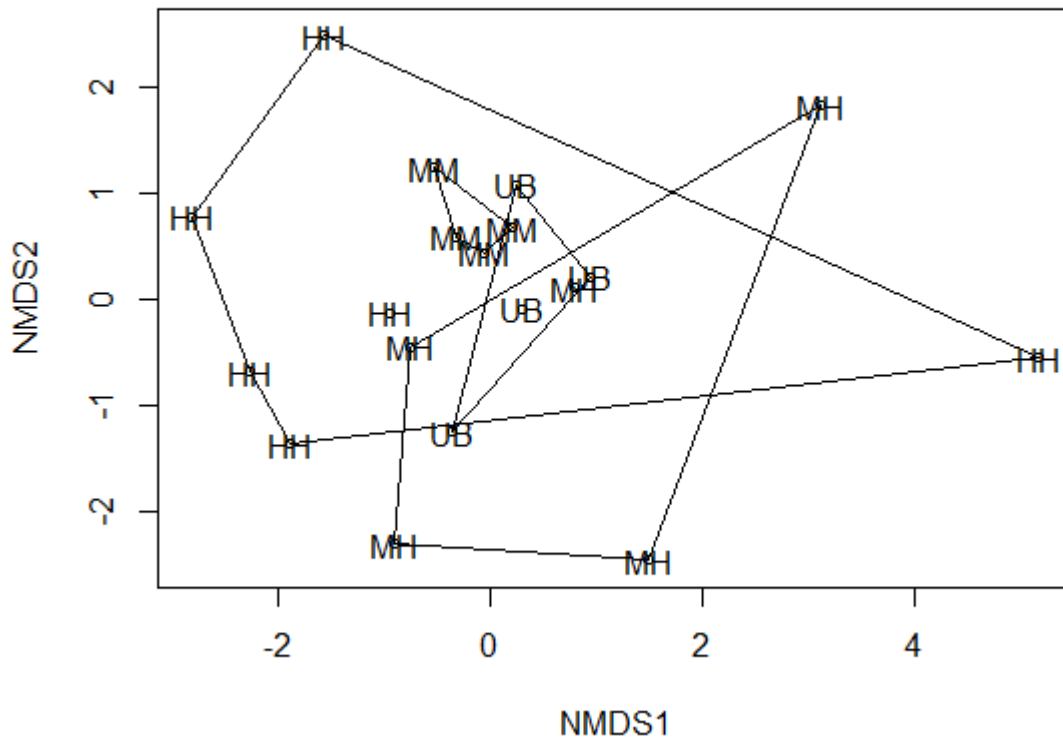


Figure 11. NMDS ordination plot depicting habitat data grouped by the explanatory variable (Run 4, stress 0.15, Procrustes: rmse 0.0537215 max resid 0.20).

The more highly disturbed sites (classes HH and MH) displayed a greater diversity in habitat values, while the less disturbed sites (classes MM and UB) were more similar within classes.

A Permutational Multivariate Analysis of Variance (PERMANOVA) using Euclidian distance indicated significant differences between site classes ($R^2=0.26$, $p>0.006$, $df=3$). Tukey's Honest Significant Differences indicated which pairs of classes were significantly different in multivariate dispersion (table 6).

Table 7. Tukey's Honest Significant Differences for z-transformed, positive adjusted habitat variables. All pairs are not significantly different in multivariate dispersion.

Classes	Difference	lower	upper	p adjusted
MH-HH	-0.008	-0.061	0.044	0.967
MM-HH	0.044	-0.014	0.102	0.172
UB-HH	0.0327	-0.025	0.091	0.398
MM-MH	0.052	-0.004	0.108	0.070
UB-MH	0.041	-0.015	0.097	0.194
UB-MM	-0.011	-0.073	0.050	0.949

Given that homogeneity of multivariate dispersion is an assumption of the PERMANOVA test, only those pairs of classes that were not significantly different were tested for pairwise differences in species composition (using ANOVA with Bonferroni correction for multiple comparisons – table 7). In this case, all possible pairs of classes were tested because no comparison was significantly ($p \leq 0.05$) different in multivariate dispersion.

Table 8 Bonferroni corrected p – values for each pair of explanatory variable classes using PERMANOVA (999 permutations).

Classes	Bonferroni-corrected p-value	R^2	Degrees of freedom
MH-HH	0.028	0.14	1
MM-HH	0.048	0.14	1
UB-HH	0.028	0.14	1
MM-MH	0.052	0.14	1
UB-MH	0.028	0.14	1
UB-MM	0.004	0.14	1

Pairwise tests indicated that each group was significantly different in habitat characteristics to each of the others, except the pair MM, MH.

Individual habitat elements.

Only two of the elements contributing to the habitat measures (table 4) were found to have a significant contribution to relationships with explanatory variable classes. These were over storey cover ($f=5.42$, $p=0.01$, $df=3$) and shrub cover ($f=5.08$, $p=0.01$, $df=3$), figures 12 and 13.

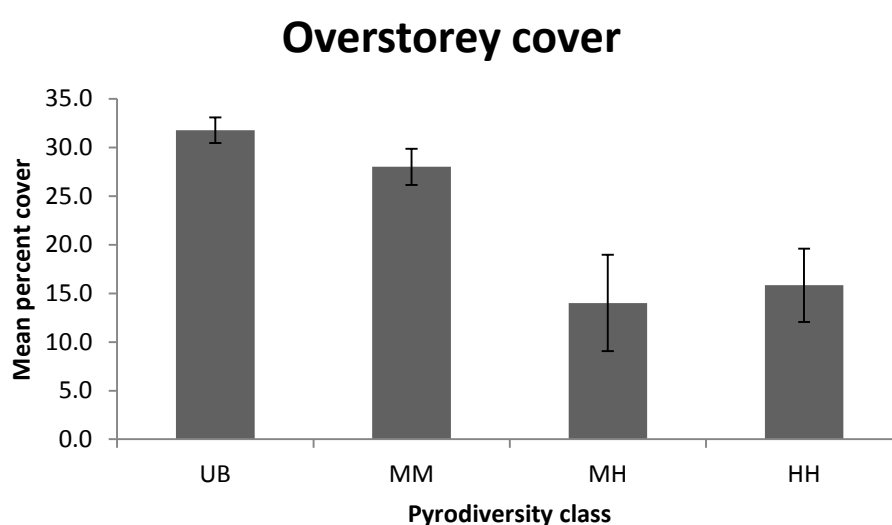


Figure 12. Mean percent cover of over storey vegetation in relation to explanatory variable classes. Error bars indicate SE.

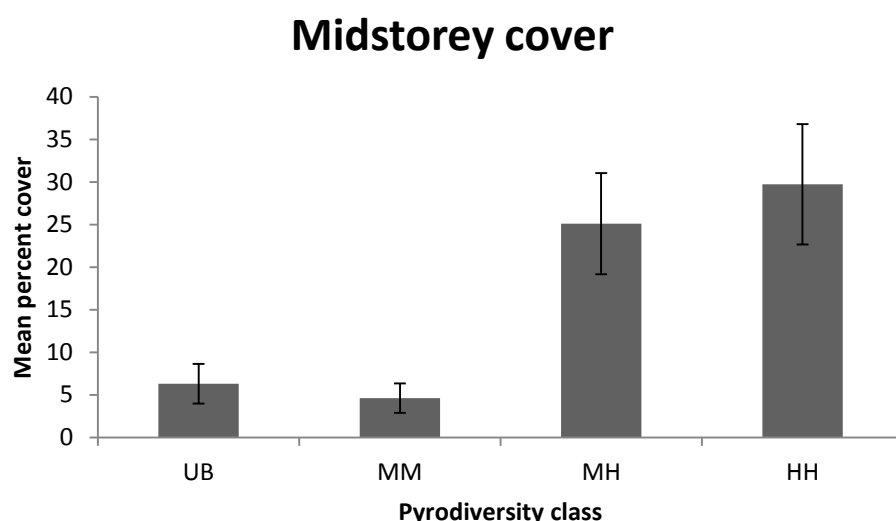


Figure 13. Mean percent cover of over mid storey vegetation in relation to explanatory variable classes. Error bars indicate SE.

This result indicates that the higher disturbance categories were having a greater impact on the canopy of over storey species. Increased mid storey cover could have been an artefact of reduced over storey shading and competition and potentially greater germination of these species post fire.

Variation in trapping effort.

To test for the impact of the previously reported variation in trapping effort on the calculated fauna assembly attributes, correlative relationships between trap nights and species richness, trap nights and phylogenetic dispersion and trap nights and functional dispersion were examined.

Table 9. Pearson's product-moment correlation for calculated fauna assembly attributes and number of traps nights (camera trap effort).

Attribute	t- value	Degrees of freedom	p- value	Correlation
Species richness	1.59	21	0.13	0.33
Phylogenetic dispersion	-0.98	19	0.34	-0.22
Functional dispersion	-0.59	19	0.56	-0.13

As the p value was < 0.2 for species richness, the decision to use rarefaction to correct for the difference in trap nights across sites was justified. Expected species richness values were calculated for all sites for a minimum number of trap nights (26) and these values were used

instead of observed species richness in subsequent analyses. No significant relationship between phylogenetic dispersion and trap nights and functional dispersion and trap nights was observed.

Phylogenetic dispersion at the habitat scale

For mammals across sites the PD data indicated a trend for greater phylogenetic dispersion index (ΔPD) in the long unburnt sites decreasing with increased fire frequency and severity.

Table 10. Mean Phylogenetic Dispersion Indexes across explanatory variable classes.

	UB	MM	MH	HH
ΔPD	112.66	112.12	100.73	91.02

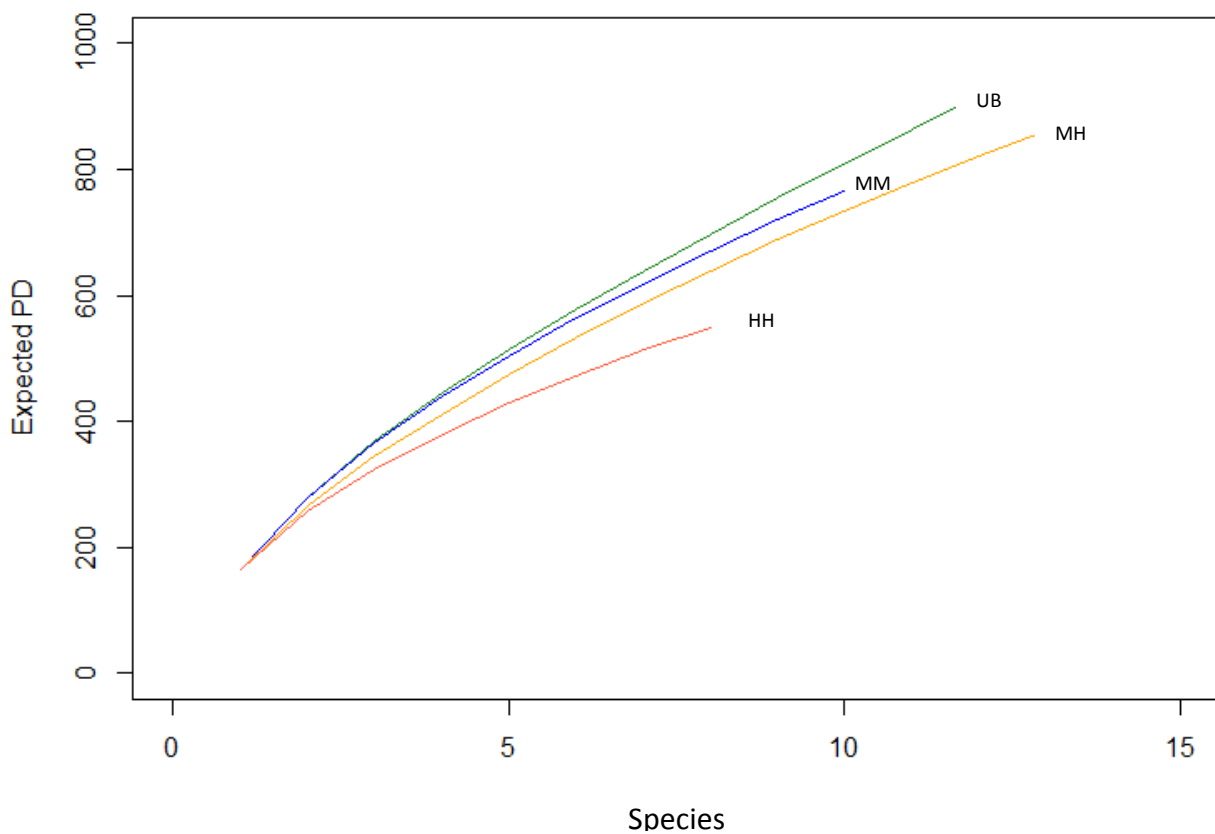


Figure 14. Rarefaction curves for expected phylogenetic diversity (PD) of mammals in each explanatory variable class: (a) UB, (b) MM, (c) MH and (d) HH. The figure indicates a steady increase in the expected PD from the more greatly disturbed sites to the less disturbed sites.

Mammal phylogenetic diversity at the site level.

Mammal PDI at the site scale exhibited no statistically significant difference (ANOVA $df=3$, $F=0.48$, $p=0.7$) between explanatory variable classes however the mean values were highest in the unburned sites and lowest in the highly impacted, higher frequency sites.

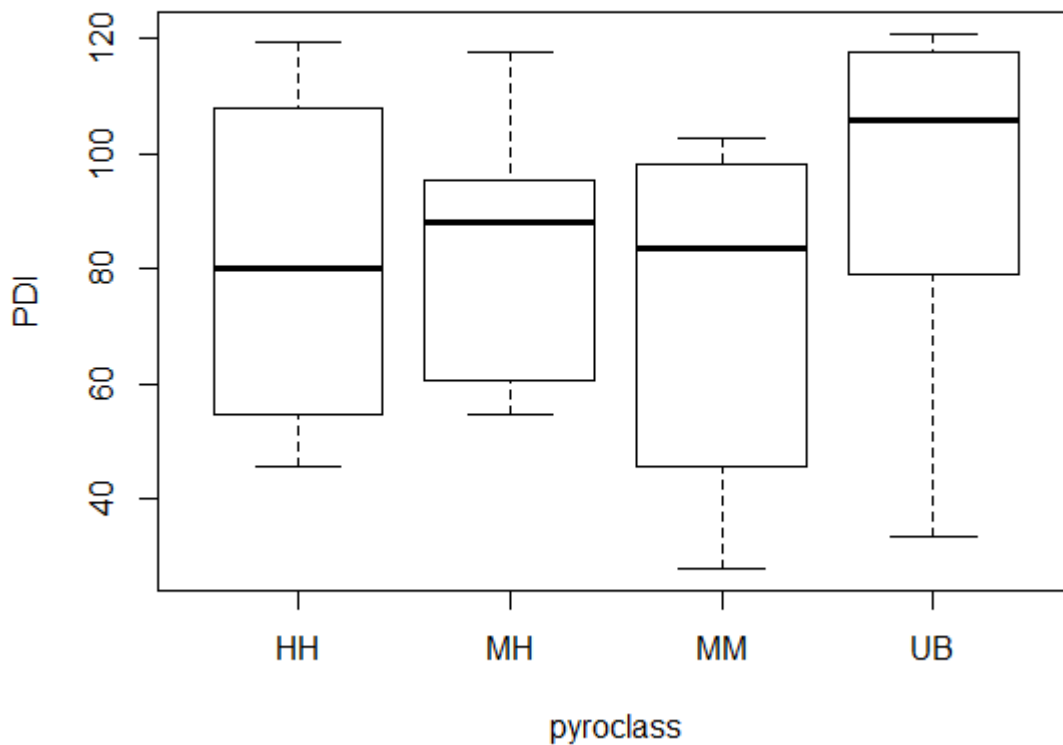


Figure 15. Boxplot of PDI of mammals across explanatory variable classes at the site level. No significant relationship was detected (ANOVA $df=3$; $f=0.48$; $p=0.7$)

Species richness

Analysis of variance for the expected richness relationship with explanatory variable site classes was highly significant ($f=5.85$, $p=0.005$, $df=3$)

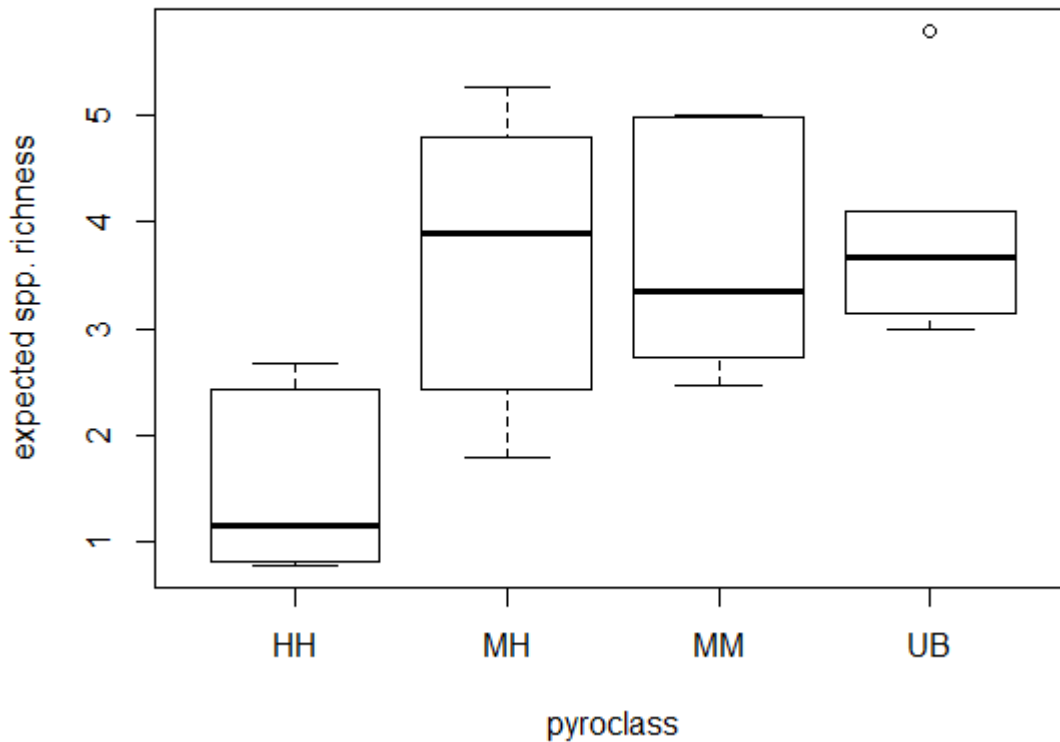


Figure 16. Boxplot indicating expected species richness of mammals across explanatory variable classes (ANOVA $df=3$; $f=5.85$; $p=0.005$)

Analysis of variance indicated that there was a significant difference in expected species richness and Tukey's HSD was used for pairwise testing to determine the significant differences between each of the classes.

Table 11. Tukey's HSD pairwise tests for significant differences in expected species richness between explanatory variable classes. Differences between HH and the other three classes were significant.

Classes	Difference	lower	upper	p adjusted
MH-HH	2.173	0.320	4.026	0.018
MM-HH	2.207	0.264	4.150	0.023
UB-HH	2.395	0.542	4.247	0.009
MM-MH	0.034	-1.909	1.977	1.000
UB-MH	0.222	-1.631	2.075	0.986
UB-MM	0.188	-1.755	2.131	0.993

Species composition in relation to site classes.

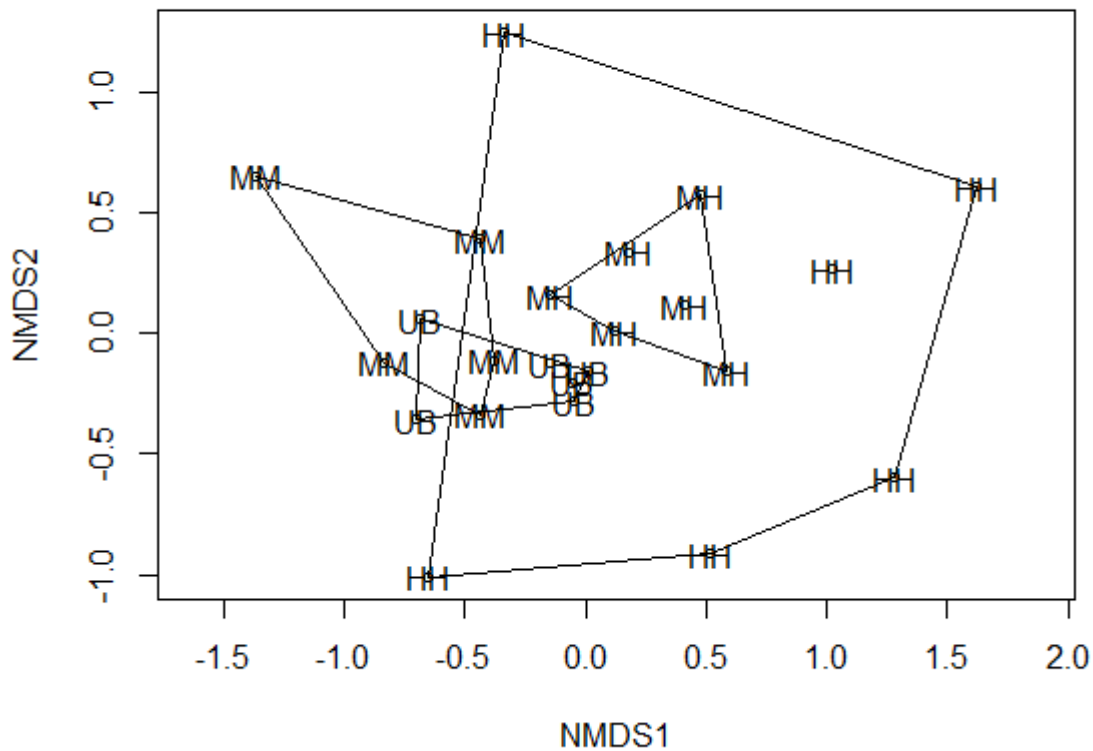


Figure 17. Ordination plot (Run 5 stress 0.1246812) exhibiting site level species composition of mammals across explanatory variable classes.

A Permutational Multivariate Analysis of Variance (PERMANOVA) using the Bray – Curtis dissimilarity across site classes indicated significant differences in species composition between explanatory variable classes ($R^2=0.33$, $p>0.001$, $df=3$). Because of the observed relationship between species richness and survey effort, Bray-Curtis distance was calculated from a site by species matrix of probabilities rather than abundances. Each cell in the matrix was calculated from the rarefaction formula and was the probability of selecting that species from that site when the data were rarefied to the minimum number of 26 trap nights. The explanatory variable classes were significantly different in multivariate dispersion of species composition, indicating that classes differed in beta-diversity among sites (Permutational test of multivariate dispersion; $F=4.21$, $p=0.03$, $df=3$). Tukey's Honest Significant Differences (Tukey's HSD) indicated which pairs of classes were significantly different in multivariate dispersion (table 10).

Table 12. PERMANOVA test for multivariate dispersion, four of the six classes tests were shown to be testable in a pairwise comparison.

Classes	Difference	lower	upper	p adjusted
MH-HH	-0.25	-0.49	0.00	0.05
MM-HH	-0.16	-0.41	0.09	0.32
UB-HH	-0.28	-0.52	-0.04	0.02
MM-MH	0.09	-0.17	0.34	0.78
UB-MH	-0.03	-0.28	0.21	0.98
UB-MM	-0.12	-0.38	0.13	0.55

Given that homogeneity of multivariate dispersion is an assumption of the PERMANOVA test, only those pairs of classes that were not significantly different were tested for pairwise differences in species composition (using PERMANOVA with Bonferroni correction for multiple comparisons – table 11).

From the testable pairs of site classes

Table 13. Bonferroni-corrected p-values indicated that these pairs were significantly differentiated.

Classes	Bonferroni-corrected p-value	R ²	Degrees of freedom
MM-HH	0.004	0.33	1
MM-MH	0.004	0.33	1
UB-MH	0.004	0.33	1
UB-MM	0.004	0.33	1

The compositional differences between site classes are shown from the occurrence data represented in Table 14.

Table 14. Occurrence data for mammal species across explanatory variable gradients.

Site class	UB	MM	MH	HH
Macropus_giganteus	1	1	1	1
Macropus_rufogriseus	1	1	1	1
Tachyglossus_aculeatus	1	1	1	0
Vombatus_ursinus	1	1	1	0
Antechinus_swainsonii	1	1	0	1
Wallabia_bicolor	1	1	1	1
Trichosurus_vulpecula	1	1	1	1
Trichosurus_caninus	1	0	1	0
Pseudocheirus_peregrinus	0	0	1	0
Cercartetus_nanus	0	0	1	0

Antechinus_stuartii	0	0	1	1
Rattus_rattus	0	0	1	0
Rattus_fuscipes	0	0	1	1
Felis_catus	1	0	0	0
Vulpes_vulpes	0	1	1	0
Canis_lupus	1	1	0	0
Sus_scrofa	1	0	0	1
Oryctolagus_cuniculus	1	1	0	0

Functional dispersion

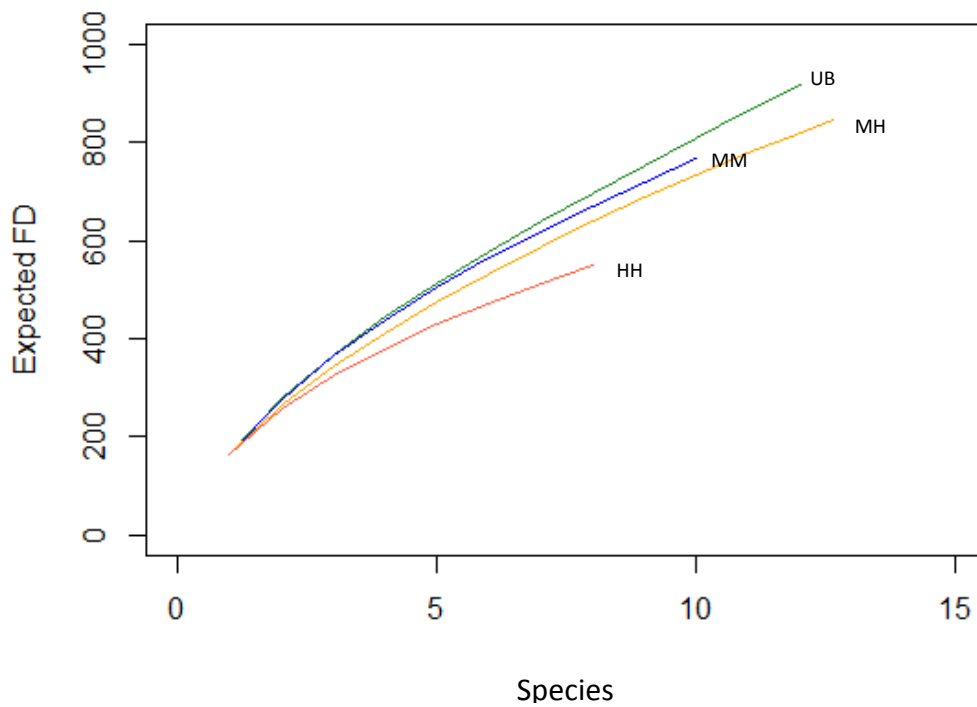


Figure 18. Rarefaction curves for expected Functional Diversity (FD) of mammals in each explanatory variable class.

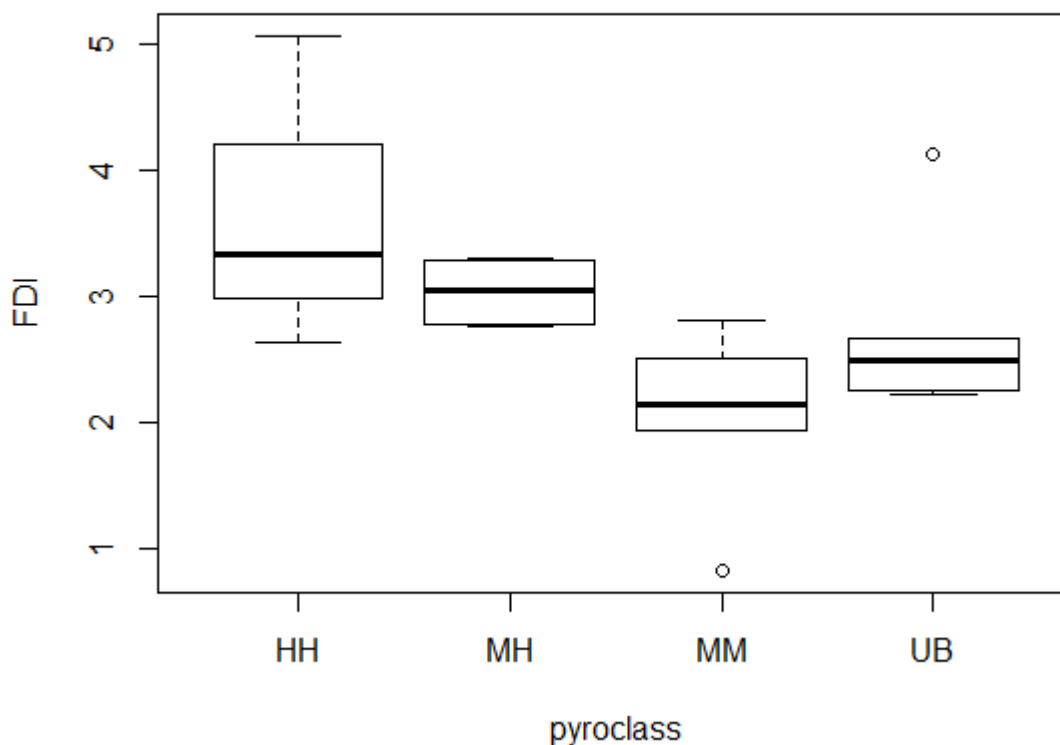


Figure 19. Functional dispersion of mammals across explanatory variable classes, classes HH and MM were found to be significantly different ($F=3.58$, $p=0.037$, $df=3$).

Analysis of variance indicated that across explanatory variable classes there were significant differences in functional diversity indexes, ($F=3.58$, $p=0.037$, $df=3$). Tuckey's HSD was again used to determine the pairwise difference between explanatory variable classes, the results are shown in table 15.

Table 15. Differences in in functional dispersion across explanatory variable classes. The only significant difference was found between classes MM and HH.

Classes	Difference	lower	upper	p adjusted
MH-HH	-0.64324	-2.09709	0.810613	0.596343
MM-HH	-1.63363	-3.13516	-0.1321	0.030646
UB-HH	-0.97261	-2.42646	0.481239	0.261259
MM-MH	-0.99039	-2.23539	0.254612	0.145473
UB-MH	-0.32937	-1.51644	0.85769	0.856191
UB-MM	0.661017	-0.58399	1.906019	0.449753

At the site level, functional dispersion and phylogenetic dispersion were found not correlated using Pearson's product-moment correlation ($r=0.55$, $p=0.30$ $df=16$).

Birds

Early analysis of the bird data indicated that there was insufficient samples and limited replication within and across classes of stratification. No further analysis was conducted using these data and no meaningful results were produced.

Discussion.

Species detected.

Of the 18 mammal species detected by camera trapping, nine were marsupials, eight were eutherians and one was a monotreme. This sample was a broad representation of the class Mammalia which accounted for the observed, relatively high expected ΔPD values compared to those values for the birds.

Following stratification and filtering, 27 species of birds were drawn from COG observations for this study, these are listed in the supplementary information.

Habitat and explanatory variables

The habitat structure was clearly differentiated across the classes, indicating that measured variables responded to fire and were therefore appropriate for this component of the study. The variation between each of the classes was likely to be driven by one or more of the measured habitat elements. A strong positive trend in shrub cover was indicated in figure N (results) with increased explanatory variable class (UB \rightarrow HH), while over storey cover declined across the same gradient. No other trends were seen in the habitat elements.

Explanatory variable gradients and fauna.

In this study, a contrast in the phylogenetic response to explanatory variable variables was seen between mammals and birds.

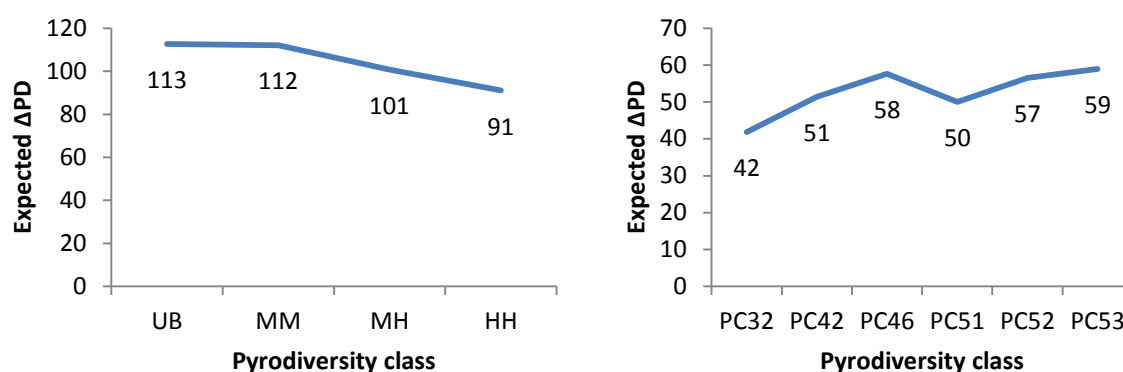


Figure 19. Expected phylogenetic dispersion values for mammals (left), and birds (right). The two groups exhibit contrasting response trends to increased fire frequency and severity. See table 3 for explanatory variable class codes.

For mammals there was a statistically non-significant trend to lower ΔPD with increased fire frequency and severity, while with birds, a statistically non-significant trend of increasing in ΔPD with increases in fire frequency and severity. It is possible that the peaks and trough in the right hand graph in figure 14 reflect the importance of frequency (the second digit in the explanatory variable code). However the small sample size in this data set does not allow for further investigation here, but could be investigated further with more sites and observations targeted specifically at researching this feature.

The trend in reduced ΔPD in mammals could suggest a decline in evolutionary potential for those sites severely burnt and with higher fire frequency (Nipperess & Matsen 2013). However this is dependent on the scale of the disturbance, the proximity of refugia and the dispersal characteristics of fauna (Berry *et al.* 2015). Of note however that there is still a signal in the species richness data some 13 years after the most recent fire. This coincides with the habitat data, where the HH class is significantly distinct from all other classes, possibly indicating that the fewer species prefer this habitat.

In contrast to ΔPD , ΔFD was highest in the most impacted site class, HH implying that the reduced numbers of species were individually contributing a greater number of KEF's than those with greater species richness. A close inspection of the species that were detected at these sites indicated that some of them had relatively high numbers of KEF's, *Sus scrofa*, *Wallabia bicolor* and *Rattus fuscipes*. The higher KEF value indicate more generalist species while more specialised species have lower KEF values (Marcot & Aubrey 2003), table 16.

Table 16. Mammal species detected in explanatory variable class HH and the numbers of KEF's that they represent.

Species	Common name	Number of KEF's
<i>Macropus giganteus</i>	Eastern grey kangaroo	3
<i>Macropus rufogriseus</i>	Red-necked wallaby	3
<i>Antechinus swainsonii</i>	Dusky antechinus	3
<i>Wallabia bicolor</i>	Swamp wallaby	9
<i>Trichosurus vulpecula</i>	Common brush-tailed possum	3
<i>Antechinus agilis</i>	Agile antechinus	3
<i>Rattus fuscipes</i>	Bush rat	7
<i>Sus scrofa</i>	Pig	12

Also observed was the absence of introduced predators in site classes MH and HH. No larger predators were detected at HH sites, and only one MH site recorded *Vulpes vulpes*, red fox. The use of burned areas soon after fire by introduced predators has been reported in south-eastern Australia (E.g. Robley 2013; Payne *et al.* 2014), and given the time since fire reported here, the observations of this study do not contradict earlier findings. It does suggest that these areas are less used by these predators through subsequent successional stages. It may however represent a particular habitat opportunity or other resource that is linked to the successional stage of higher explanatory variable class coded sites (Chia *et al.* 2016). Robinson *et al.* (2014) found that compared to long unburned sites, fire reduced species richness. Changes in species richness were not statistically significant between explanatory variable classes in this study, but generally trended upwards (see figure N). The increased mid storey cover in the higher explanatory variable classes observed in the mammal sites may provide greater cover and other resources for a broader range of bird species.

Of further interest were the records of *Felis catus*, feral cats that were only observed in unburned sites and not in any other explanatory variable class. The increased scrubbyness of the classes where cats were not observed may be a factor in their absence.

Synthesis and applications

The results of this study indicate that fire frequency and severity have an impact on fauna habitat (figures 11, 12, 13; table7) and that in turn impacts phylogenetic (and functional structure of faunal communities figures 14, 18,20 and 23). The implications for management are therefore to provide habitat opportunities for those potentially unique species compositions through heterogeneous fire regimes.

Several studies have been undertaken investigating the impact of disturbance on the phylogenetic structure of communities (E.g. Dinnage 2009; Helmus *et al.* 2010; Brunbjerg *et al.* 2012; Ding *et al.* 2012), but relatively few have vertebrates as their target species. Of those studies the majority are focused on tropical birds (E.g. Gomez *et al.* 2010; Gianuca *et al.* 2014; Klingbeil & Willig 2016). The use of ΔPD as measure of community structure, biodiversity and beta-scale evolutionary potential across environmental gradients shows considerable promise for vertebrate taxa (see Nipperess 2016b).

The use of KEF's to characterise the functional roles of entities within communities or ecosystems has the advantage of easily being derived from the known ecology and life history information of those entities. The approach can be extended to investigate various community characteristics related to function using the same set of measures as for biodiversity (E.g. Shannon's entropy, functional evenness, functional dissimilarity, etc.). In relation to threatened or declining species the imperilled functions of those entities can also easily be identified (Marcot & Aubrey 2003).

The premise that patch mosaic burning maximises biodiversity has been questioned by some authors (E.g. Parr & Anderson 2006; Taylor *et al.* 2012) however a range of vegetation within various post-fire stages could be managed to provide the maximum diversity across communities (E.g. Richards, Possingham & Tizard 1999; McCarthy 2012; Di Stefano *et al.* 2013). The scale at which this needs to be undertaken is dependent on the biological need of those entities within the community. Allowing for dispersion from minimum viable areas that in turn support minimum viable populations can be a starting point for considering the scale of disturbance. (Wisz *et al.* 2013)

It is important to note that the relative proportion of long unburned in a community needs to be higher than other classes so as to provide a buffer against unplanned fire impacting areas stochastically (McCarthy 2012).

The effectiveness of the strategy of landscape scale mosaic burning for fuel reduction in preventing severe fires crossing the urban interface and impacting human life and property is a matter of debate in the literature (Gibbons *et al.* 2012; Attiwill & Adams 2013; Gill, Stephens & Cary 2013; McCaw 2013). Gibbons *et al.* (2012) found that prescribed burning was only effective in mitigating house loss when conducted close to the build asset. They suggest burning at distances greater than several hundred metres from assets is ineffective as a measure for preventing urban impacts. Further, Tolhurst and McCarthy (2016) found that the influence of

previous fire to the severity of wildfires was minimal during weather conditions that were conducive to high forest fire danger indexes (FFDI) >25 ; and that in milder condition (FFDI <25) fuel reduction burning less than three years old was most likely to reduce fire severity. They also found that fuel reduction burns less than 10 years old reduced fire severity to an extent that canopy loss was reduced (again if the FFDI was less than 25). The effectiveness of hazard reduction burning, given that fire behaviour leading to loss of life and urban destruction generally occur at FDI's of greater than 50 (Blanchi, Leonard & Leicester 2006; Gibbons *et al.* 2012) is therefore questionable. Keith, Williams and Woinarski (2002); Gill (2008); Di Stefano *et al.* (2013); Driscoll *et al.* (2016) present examples of fire management approaches that allow for management strategies to recognise and integrate biodiversity values at landscape scales. These approaches provide planning frameworks and tools that assess the relative efficacy of fuel treatment strategies in the context of their potential impact on biodiversity needs and values.

Limitations and further research

The effectiveness of the measures used here could be further tested using larger data sets. The bird data here was not collected specifically for this study and therefore does not conform to rigorous design that is fit for purpose. The number of replicates sites restricted the analyses and conclusions that were able to be extrapolated from these data.

For mammals the use camera detection methods limited the resultant data to presence only. While it was possible to recognise some individuals of some species, there was insufficient "mark – recapture" incidents across species and individuals to provide any meaningful abundance data. While some methods relating detection probability to abundance (E.g. Rowcliffe *et al.* 2008) the type of data available from this study precluded their use. Without abundance data studies was limited in their ability to produce meaningful information that relate to the importance or rareness of individual species through measures such as evenness or geometric mean abundance.

As mentioned earlier, it is intended that the approaches used here will be implemented across other vegetation communities in NNP and TNR, and these data will be used to further evaluate the effectiveness of the approaches. It is hoped that this set of studies will provide for the establishment of longer term studies advocated by Lindenmayer *et al.* (2016).

The complex relationships between the biota and fire regime elements (Gill 1975) and their interactions with each other have been the subject of a range of studies (E.g. Gill & Catling 2002; Hobbs 2002; Keith, McCaw & Whelan 2002; Lindenmayer *et al.* 2016). The attributes of time since fire and seasonality of fire were not expressly considered in this study. However the time since last fire was standardised to the 2003 fire (16 years) and no sites were chosen that had burned since that time. Seasonality data for fires prior to the 1980's was not reliable (in many cases only the year of the fire was recorded) and therefore not used in this study, but is clearly important for both plants and animals as populations may be more vulnerable to its effects during their reproductive season (Gill 1975; Knox & Clarke 2006). Inter-fire interval (the mean period between fires where the fire frequency is greater than 1) was also not considered here.

There was also insufficient data to test the relative impact of fire frequency and fire severity. An experimental approach that incorporated forest type could be more definitive than the space for time approach used here in providing such data.

Authors Contributions

ACC conceived the project and the survey design, conducted the analyses, conducted the field work and wrote the manuscript.

DAN helped conceive the project and the survey design, wrote the code for the analyses and provided critical comment on the manuscript.

JAS helped conceive the project and the survey design, contributed to the field work and provided critical comment on the manuscript.

All authors contributed critically to the drafts and gave final approval for publication.

This study was conducted under University of Canberra Animal Ethics approval CEAE15-17.

Acknowledgements.

Firstly I would like to acknowledge the substantial help and support provided by my supervisors, David Nipperess and Julian Seddon.

The Conservation Research unit of the ACT Government's Environment and Planning Directorate allowed me to conduct this research as part of my duties as the Fire Ecologist in the unit. It also provided study leave for writing and attendance at Macquarie University as required.

The Canberra Ornithologist's Group, particularly Chris Davey for facilitating the use of their bird data for NNP and TNR.

For assistance in the field, I wish to thank Luke Johnston, Nick Wilson, Jenny Smits, Adam Groth, Jack Corrigan, Annette Wrightson, Beth Corrigan and Lauren Corrigan. Annette Wrightson proof read the manuscript.

Mellissa Snape provided statistical advice and intellectual discussion.

Don Fletcher provided information around the early management of NNP and TNR.

Margaret Kitchin and Annie Lane granted permission for the study to be undertaken.

The ACT Government provided support through the provision of vehicles, field equipment and field time for ACC.

References

- ACT Government (2015) Bushfire Operations Plan. ACT Government
http://www.environment.act.gov.au/parks-conservation/bushfire_management/fuel_management/bushfire-operations-plan, Canberra.
- ACT Government (2016) ACT Wildlife Atlas Records. (ed. A. Government).
<https://www.data.act.gov.au/Environment/ACT-Wildlife-Atlas-Records/e9ux-7djy>.
- Attiwill, P.M. & Adams, M.A. (2013) Mega-fires, inquiries and politics in the eucalypt forests of Victoria, south-eastern Australia. *Forest Ecology and Management*, **294**, 45-53.
- Auger, S. & Shipley, B. (2013) Inter-specific and intra-specific trait variation along short environmental gradients in an old-growth temperate forest. *Journal of Vegetation Science*, **24**, 419-428.
- Australian Capital Territory & Doogan, M. (2006) *The Canberra Firestorm: Inquests and Inquiry Into Four Deaths and Four Fires Between 8 and 18 January 2006*. ACT Coroners Court.
- Baines, G., Webster, M., Cook, E., Johnston, L. & Seddon, J. (2013) *The Vegetation of the Kowen, Majura and Jerrabomberra Districts of the Australian Capital Territory*.
- Baker, J., Whelan, R.J., Evans, L., Moore, S. & Norton, M. (2010) Managing the Ground Parrot in its fiery habitat in south-eastern Australia. *Emu*, **110**, 279-284.
- Banks, J. (1989) A history of forest fire in the Australian Alps. *The scientific significance of the Australian Alps*, 265-280.
- Barrett, T. (2006) Modelling burn severity for the 2003 NSW/ACT wildfires using landsat imagery. *Life In A Fire-Prone Environment: Translating Science Into Practice*. Brisbane, 6–9 June 2006
- Berry, L.E., Driscoll, D.A., Stein, J.A., Blanchard, W., Banks, S.C., Bradstock, R.A. & Lindenmayer, D.B. (2015) Identifying the location of fire refuges in wet forest ecosystems. *Ecological Applications*, **25**, 2337-2348.
- Bininda-Emonds, O.R., Cardillo, M., Jones, K.E., MacPhee, R.D., Beck, R.M., Grenyer, R., Price, S.A., Vos, R.A., Gittleman, J.L. & Purvis, A. (2007) The delayed rise of present-day mammals. *Nature*, **446**, 507-512.
- Blanchi, R., Leonard, J.E. & Leicester, R.H. (2006) Lessons learnt from post-bushfire surveys at the urban interface in Australia. *Forest Ecology and Management*, **234**, S139.
- Bolnick, D.I., Amarasekare, P., Araújo, M.S., Bürger, R., Levine, J.M., Novak, M., Rudolf, V.H., Schreiber, S.J., Urban, M.C. & Vasseur, D.A. (2011) Why intraspecific trait variation matters in community ecology. *Trends in Ecology & Evolution*, **26**, 183-192.
- Bradstock, R.A., Bedward, M., Gill, A.M. & Cohn, J.S. (2005) Which mosaic? A landscape ecological approach for evaluating interactions between fire regimes, habitat and animals. *Wildlife Research*, **32**, 409-423.
- Brunbjerg, A.K., Borchsenius, F., Eiserhardt, W.L., Ejrnæs, R. & Svenning, J.C. (2012) Disturbance drives phylogenetic community structure in coastal dune vegetation. *Journal of Vegetation Science*, **23**, 1082-1094.
- Bureau of Meteorology (2016) Canberra in winter 2016: A wet winter with warm nights. pp. Report. Commonwealth of Australia.
- Canberra Ornithologists Group (2016).
- Carmona, C., de Bello, F., Mason, N. & Lepš, J. (2016) Traits Without Borders: Integrating Functional Diversity Across Scales. *Trends in Ecology & Evolution*.
- Cavender-Bares, J., Ackerly, D.D. & Kozak, K.H. (2012) Integrating ecology and phylogenetics: the footprint of history in modern-day communities. *Ecology*, **93**.
- Chalmandrier, L., Münkemüller, T., Devictor, V., Lavergne, S. & Thuiller, W. (2015) Decomposing changes in phylogenetic and functional diversity over space and time. *Methods in Ecology and Evolution*, **6**, 109-118.
- Chao, A., Chiu, C.-H. & Jost, L. (2010) Phylogenetic diversity measures based on Hill numbers. *Philosophical Transactions of the Royal Society of London B: Biological Sciences*, **365**, 3599-3609.
- Chao, A., Chiu, C.H., Hsieh, T., Davis, T., Nipperess, D.A. & Faith, D.P. (2015) Rarefaction and extrapolation of phylogenetic diversity. *Methods in Ecology and Evolution*, **6**, 380-388.
- Cheal, D.C. (2010) *Growth stages and tolerable fire intervals for Victoria's native vegetation data sets*. Victorian Government Department of Sustainability and Environment.

- Chia, E.K., Bassett, M., Leonard, S.W., Holland, G.J., Ritchie, E.G., Clarke, M.F. & Bennett, A.F. (2016) Effects of the fire regime on mammal occurrence after wildfire: Site effects vs landscape context in fire-prone forests. *Forest Ecology and Management*, **363**, 130-139.
- Chiarucci, A., Bacaro, G., Rocchini, D. & Fattorini, L. (2008) Discovering and rediscovering the sample-based rarefaction formula in the ecological literature. *Community Ecology*, **9**, 121-123.
- Clarke, M.F. (2008) Catering for the needs of fauna in fire management: science or just wishful thinking? *Wildlife Research*, **35**, 385-394.
- Collins, L., Bradstock, R.A., Tasker, E.M. & Whelan, R.J. (2012) Impact of fire regimes, logging and topography on hollows in fallen logs in eucalypt forest of south eastern Australia. *Biological conservation*, **149**, 23-31.
- Corrigan, A., Seddon, J., Johnston, L. & Kitchin, M. (2015) Fire Ecology Program Update 2015, Research Report 2015/4. *Environment and Planning Directorate, ACT Government, Canberra*.
- De Bondi, N., White, J.G., Stevens, M. & Cooke, R. (2010) A comparison of the effectiveness of camera trapping and live trapping for sampling terrestrial small-mammal communities. *Wildlife Research*, **37**, 456-465.
- Deil, U. (2005) A review on habitats, plant traits and vegetation of ephemeral wetlands—a global perspective. *Phytocoenologia*, **35**, 533-706.
- DellaSala, D.A., Williams, J.E., Williams, C.D. & Franklin, J.F. (2004) Beyond smoke and mirrors: a synthesis of fire policy and science. *Conservation Biology*, **18**, 976-986.
- Di Stefano, J., McCarthy, M.A., York, A., Duff, T.J., Slingo, J. & Christie, F. (2013) Defining vegetation age class distributions for multispecies conservation in fire-prone landscapes. *Biological conservation*, **166**, 111-117.
- Diete, R.L., Meek, P.D., Dixon, K.M., Dickman, C.R. & Leung, L.K.-P. (2016) Best bait for your buck: bait preference for camera trapping north Australian mammals. *Australian Journal of Zoology*.
- Ding, Y., Zang, R., Letcher, S.G., Liu, S. & He, F. (2012) Disturbance regime changes the trait distribution, phylogenetic structure and community assembly of tropical rain forests. *Oikos*, **121**, 1263-1270.
- Dinnage, R. (2009) Disturbance alters the phylogenetic composition and structure of plant communities in an old field system. *PloS one*, **4**, e7071.
- Driscoll, D.A., Bode, M., Bradstock, R.A., Keith, D.A., Penman, T.D. & Price, O.F. (2016) Resolving future fire management conflicts using multicriteria decision making. *Conservation Biology*, **30**, 196-205.
- Driscoll, D.A., Lindenmayer, D.B., Bennett, A.F., Bode, M., Bradstock, R.A., Cary, G.J., Clarke, M.F., Dexter, N., Fensham, R., Friend, G., Gill, M., James, S., Kay, G., Keith, D.A., MacGregor, C., Possingham, H.P., Russel-Smith, J., Salt, D., Watson, J.E.M., Williams, D. & York, A. (2010a) Resolving conflicts in fire management using decision theory: asset-protection versus biodiversity conservation. *Conservation Letters*, **3**, 215-223.
- Driscoll, D.A., Lindenmayer, D.B., Bennett, A.F., Bode, M., Bradstock, R.A., Cary, G.J., Clarke, M.F., Dexter, N., Fensham, R., Friend, G., Gill, M., James, S., Kay, G., Keith, D.A., MacGregor, C., Russel-Smith, J., Salt, D., Watson, J.E.M., Williams, R.J. & York, A. (2010b) Fire management for biodiversity conservation: key research questions and our capacity to answer them. *Biological conservation*, **143**, 1928-1939.
- Driscoll, D.A. & Roberts, J.D. (1997) Impact of fuel-reduction burning on the frog *Geocrinia lutea* in southwest Western Australia. *Australian Journal of Ecology*, **22**, 334-339.
- Faith, D.P. (1992) Conservation evaluation and phylogenetic diversity. *Biological conservation*, **61**, 1-10.
- Faith, D.P. (2013) Biodiversity and evolutionary history: useful extensions of the PD phylogenetic diversity assessment framework. *Annals of the New York Academy of Sciences*, **1289**, 69-89.
- Fisher, J.L., Loneragan, W.A., Dixon, K., Delaney, J. & Veneklaas, E.J. (2009) Altered vegetation structure and composition linked to fire frequency and plant invasion in a biodiverse woodland. *Biological conservation*, **142**, 2270-2281.
- Fleming, P., Meek, P., Ballard, G., Banks, P., Claridge, A., Sanderson, J. & Swann, D. (2014) *Camera Trapping: Wildlife Management and Research*. CSIRO PUBLISHING.
- Flynn, D.F., Mirotchnick, N., Jain, M., Palmer, M.I. & Naeem, S. (2011) Functional and phylogenetic diversity as predictors of biodiversity–ecosystem-function relationships. *Ecology*, **92**, 1573-1581.

- Fritz, S.A., Bininda-Emonds, O.R. & Purvis, A. (2009) Geographical variation in predictors of mammalian extinction risk: big is bad, but only in the tropics. *Ecology letters*, **12**, 538-549.
- Garnett, S.T., Duursma, D.E., Ehmke, G., Guay, P.-J., Stewart, A., Szabo, J.K., Weston, M.A., Bennett, S., Crowley, G.M. & Drynan, D. (2015) Biological, ecological, conservation and legal information for all species and subspecies of Australian bird. *Scientific data*, **2**.
- Gianuca, A.T., Dias, R.A., Debastiani, V.J. & Duarte, L.D. (2014) Habitat filtering influences the phylogenetic structure of avian communities across a coastal gradient in southern Brazil. *Austral Ecology*, **39**, 29-38.
- Gibbons, P., Van Bommel, L., Gill, A.M., Cary, G.J., Driscoll, D.A., Bradstock, R.A., Knight, E., Moritz, M.A., Stephens, S.L. & Lindenmayer, D.B. (2012) Land management practices associated with house loss in wildfires. *PloS one*, **7**, e29212.
- Giljohann, K., McCarthy, M., Kelly, L. & Regan, T. (2015) Choice of biodiversity index drives optimal fire management decisions. *Ecological Applications*, **25**, 264-277.
- Gill, A.M. (1975) Fire and the Australian flora: a review. *Australian forestry*, **38**, 4-25.
- Gill, A.M. (2008) *Underpinnings of Fire Management for Biodiversity Conservation in Reserves: Fire and Adaptive Management*. Department of Sustainability and Environment.
- Gill, A.M. & Catling, P.C. (2002) Fire regimes and biodiversity of forested landscapes of southern Australia. *Flammable Australia: the Fire Regimes and Biodiversity of a Continent* (eds R.A. Bradstock, J.E. Williams & A.M. Gill), pp. 351-369. Cambridge University Press.
- Gill, A.M. & Stephens, S.L. (2009) Scientific and social challenges for the management of fire-prone wildland–urban interfaces. *Environmental Research Letters*, **4**, 034014.
- Gill, A.M., Stephens, S.L. & Cary, G.J. (2013) The worldwide “wildfire” problem. *Ecological Applications*, **23**, 438-454.
- Gomez, J.P., Bravo, G.A., Brumfield, R.T., Tello, J.G. & Cadena, C.D. (2010) A phylogenetic approach to disentangling the role of competition and habitat filtering in community assembly of Neotropical forest birds. *Journal of animal ecology*, **79**, 1181-1192.
- Green, K. & Osborne, W. (2012) Field guide to wildlife of the Australian snow country.
- Haslem, A., Kelly, L.T., Nimmo, D.G., Watson, S.J., Kenny, S.A., Taylor, R.S., Avitabile, S.C., Callister, K.E., Spence-Bailey, L.M. & Clarke, M.F. (2011) Habitat or fuel? Implications of long-term, post-fire dynamics for the development of key resources for fauna and fire. *Journal of Applied Ecology*, **48**, 247-256.
- Heck, K.L., van Belle, G. & Simberloff, D. (1975) Explicit calculation of the rarefaction diversity measurement and the determination of sufficient sample size. *Ecology*, **56**, 1459-1461.
- Helmus, M.R., Keller, W.B., Paterson, M.J., Yan, N.D., Cannon, C.H. & Rusak, J.A. (2010) Communities contain closely related species during ecosystem disturbance. *Ecology letters*, **13**, 162-174.
- Hobbs, R. (2002) Fire regimes and their effects in Australian temperate woodlands. *Flammable Australia: the Fire Regimes and Biodiversity of a Continent* (eds R.A. Bradstock, J.E. Williams & A.M. Gill), pp. 305-326. Cambridge University Press.
- Hooper, D., Solan, M., Symstad, A., Diaz, S., Gessner, M., Buchmann, N., Degrange, V., Grime, P., Hulot, F. & Mermillod-Blondin, F. (2002) Species diversity, functional diversity and ecosystem functioning. *Biodiversity and ecosystem functioning: synthesis and perspectives*, 195-208.
- IconWater (2016), pp. Icon Water presents 100 years of Canberra's water story. Icon Water.
- Jackson, B.G., Peltzer, D.A. & Wardle, D.A. (2013) The within-species leaf economic spectrum does not predict leaf litter decomposability at either the within-species or whole community levels. *Journal of Ecology*, **101**, 1409-1419.
- Jetz, W., Thomas, G., Joy, J., Hartmann, K. & Mooers, A. (2012) The global diversity of birds in space and time. *Nature*, **491**, 444-448.
- Jones, K.E., Bielby, J., Cardillo, M., Fritz, S.A., O'Dell, J., Orme, C.D.L., Safi, K., Sechrest, W., Boakes, E.H. & Carbone, C. (2009) PanTHERIA: a species-level database of life history, ecology, and geography of extant and recently extinct mammals. *Ecology*, **90**, 2648-2648.
- Jost, L. (2006) Entropy and diversity. *Oikos*, **113**, 363-375.
- Kattge, J., Diaz, S., Lavorel, S., Prentice, I., Leadley, P., Bönisch, G., Garnier, E., Westoby, M., Reich, P.B. & Wright, I. (2011) TRY—a global database of plant traits. *Global change biology*, **17**, 2905-2935.

- Keeley, J.E. (2009) Fire intensity, fire severity and burn severity: a brief review and suggested usage. *International Journal of Wildland Fire*, **18**, 116-126.
- Keith, D.A., McCaw, W.L. & Whelan, R.J. (2002) Fire regimes in Australian heathlands and their effects on plants and animals. *Flammable Australia: the Fire Regimes and Biodiversity of a Continent* (eds R.A. Bradstock, J.E. Williams & A.M. Gill), pp. 199-237. Cambridge University Press.
- Keith, D.A., Williams, J.E. & Woinarski, J.C.Z. (2002) Fire management and biodiversity conservation: key approaches and principles. *Flammable Australia: the Fire Regimes and Biodiversity of a Continent* (eds R.A. Bradstock, J.E. Williams & A.M. Gill), pp. 401-425. Cambridge University Press.
- Kenny, B. (2013) Fire Interval Guidelines-What's Missing? *9th Biennial Nature Conservation Council of NSW Bushfire Conference, Sydney*. < http://www.nature.org.au/media/1735/kenny_ncc2013_fire-intervals.pdf.
- Kenny, B., Sutherland, E., Tasker, E. & Bradstock, R. (2004) Guidelines for ecologically sustainable fire management. *NSW National Parks and Wildlife Service, Hurstville*.
- Kitchin, M. (2008) Ecological Targets for Planned Fire Management - Deriving Fire Thresholds for the Conservation of Plant Species. Research and Planning, Parks Conservation and Lands, ACT Government, Canberra.
- Klingbeil, B.T. & Willig, M.R. (2016) Community assembly in temperate forest birds: habitat filtering, interspecific interactions and priority effects. *Evolutionary Ecology*, 1-20.
- Knox, K. & Clarke, P. (2006) Fire season and intensity affect shrub recruitment in temperate sclerophyllous woodlands. *Oecologia*, **149**, 730-739.
- Kraft, N.J. & Ackerly, D.D. (2010) Functional trait and phylogenetic tests of community assembly across spatial scales in an Amazonian forest. *Ecological Monographs*, **80**, 401-422.
- Kuhn, T.S., Mooers, A.Ø. & Thomas, G.H. (2011) A simple polytomy resolver for dated phylogenies. *Methods in Ecology and Evolution*, **2**, 427-436.
- Laureto, L.M.O., Cianciaruso, M.V. & Samia, D.S.M. (2015) Functional diversity: an overview of its history and applicability. *Natureza & Conservação*, **13**, 112-116.
- Lean, C. & Maclaurin, J. (2016) The Value of Phylogenetic Diversity. *Biodiversity Conservation and Phylogenetic Systematics: Preserving our evolutionary heritage in an extinction crisis* (eds R. Pellens & P. Grandcolas), pp. 19-37. Springer International Publishing, Cham.
- Lindenmayer, D.B., Blanchard, W., MacGregor, C., Barton, P., Banks, S.C., Crane, M., Michael, D., Okada, S., Berry, L. & Florance, D. (2016) Temporal trends in mammal responses to fire reveals the complex effects of fire regime attributes. *Ecological Applications*, **26**, 557-573.
- Lindenmayer, D.B., MacGregor, C., Welsh, A., Donnelly, C., Crane, M., Michael, D., Montague-Drake, R., Cunningham, R.B., Brown, D., Fortescue, M., Dexter, N., Hudson, M. & Gill, A.M. (2008) Contrasting mammal responses to vegetation type and fire. *Wildlife Research*, **35**, 395-408.
- Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Grime, J., Hector, A., Hooper, D., Huston, M., Raffaelli, D. & Schmid, B. (2001) Biodiversity and ecosystem functioning: current knowledge and future challenges. *science*, **294**, 804-808.
- MacHunter, J., Menkhorst, P. & Loyn, R. (2009) Towards a Process for Integrating Vertebrate Fauna into Fire Management Planning. Arthur Rylah Institute for Environmental Research.
- Marcot, B. & Aubrey, K. (2003) The functional diversity of mammals in coniferous forests of Western North America. *Mammal Community Dynamics: Management and Conservation in the Coniferous Forests of Western North America*. Cambridge University Press. Boston, 631-664.
- Marcot, B.G. & Vander Heyden, M. (2001) Key ecological functions of wildlife species. *communities*, **1**, 14-41.
- McCarthy, M. (2012) Review of resilience concepts and their measurement for fire management. *Department of Sustainability and Environment, Victorian State Government, Melbourne, Australia*.
- McCaw, W.L. (2013) Managing forest fuels using prescribed fire—a perspective from southern Australia. *Forest Ecology and Management*, **294**, 217-224.
- McNeely, J.A., Miller, K.R., Reid, W.V., Mittermeier, R.A. & Werner, T.B. (1990) Conserving the world's biodiversity. *IUCN: Gland, Switzerland*.
- Meek, P.D., Ballard, G.-A. & Fleming, P.J.S. (2015) The pitfalls of wildlife camera trapping as a survey tool in Australia. *Australian Mammalogy*, **37**, 13.

- Morrison, M.L., Marcot, B. & Mannan, W. (2012) *Wildlife-habitat relationships: concepts and applications*. Island Press.
- Mulvaney, J., Corrigan, A., Seddon, J., Johnston, L. & Kitchin, M. (2014) Prescribed monitoring program 2013-2014. (ed. A.G. Environment and Planning Directorate). Unpublished report ACT Government.
- Nimmo, D., Kelly, L., Farnsworth, L., Watson, S. & Bennett, A. (2014) Why do some species have geographically varying responses to fire history? *Ecography*, **37**, 805-813.
- Nipperess, D.A. (2014) Phylorare.
- Nipperess, D.A. (2016a) Phylocurve.
- Nipperess, D.A. (2016b) The Rarefaction of Phylogenetic Diversity: Formulation, Extension and Application. *Biodiversity Conservation and Phylogenetic Systematics*, pp. 197-217. Springer.
- Nipperess, D.A., Beattie, A.J., Faith, D.P., Ginn, S.G., Kitching, R.L., Reid, C.A., Russell, T. & Hughes, L. (2012) Plant phylogeny as a surrogate for turnover in beetle assemblages. *Biodiversity and Conservation*, **21**, 323-342.
- Nipperess, D.A., Faith, D.P. & Barton, K. (2010) Resemblance in phylogenetic diversity among ecological assemblages. *Journal of Vegetation Science*, **21**, 809-820.
- Nipperess, D.A. & Matsen, F.A. (2013) The mean and variance of phylogenetic diversity under rarefaction. *Methods in Ecology and Evolution*, **4**, 566-572.
- Nock, C.A., Vogt, R.J. & Beisner, B.E. (2001) Functional Traits. *eLS*. John Wiley & Sons, Ltd.
- Oksanen, J., Kindt, R., Legendre, P., O'Hara, B., Stevens, M.H.H., Oksanen, M.J. & Suggests, M. (2007) The vegan package. *Community ecology package*.
- Paradis, E., Claude, J. & Strimmer, K. (2004) APE: analyses of phylogenetics and evolution in R language. *Bioinformatics*, **20**, 289-290.
- Parr, C. & Brockett, B. (1999) Patch-mosaic burning: a new paradigm for savanna fire management in protected areas? *Koedoe*, **42**, 117-130.
- Parr, C.L. & Andersen, A.N. (2006) Patch mosaic burning for biodiversity conservation: a critique of the pyrodiversity paradigm. *Conservation Biology*, **20**, 1610-1619.
- Parr, C.L. & Anderson, A.N. (2006) Patch mosaic burning for biodiversity conservation: a critique of the pyrodiversity paradigm. *Conservation Biology*, **20**, 1610-1619.
- Payne, C.J., Ritchie, E.G., Kelly, L.T. & Nimmo, D.G. (2014) Does fire influence the landscape-scale distribution of an invasive mesopredator? *PloS one*, **9**, e107862.
- Penman, T.D., Binns, D.L., Shiels, R.J., Allen, R.M. & Kavanagh, R.P. (2008) Changes in understorey plant species richness following logging and prescribed burning in shrubby dry sclerophyll forests of south-eastern Australia. *Austral Ecology*, **33**, 197-210.
- Petchey, O.L. & Gaston, K.J. (2002) Functional diversity (FD), species richness and community composition. *Ecology letters*, **5**, 402-411.
- Poos, M.S., Walker, S.C. & Jackson, D.A. (2009) Functional-diversity indices can be driven by methodological choices and species richness. *Ecology*, **90**, 341-347.
- Pryke, J.S. & Samways, M.J. (2012) Differential resilience of invertebrates to fire. *Austral Ecology*, **37**, 460-469.
- Pryor, L. (1939) The bush fire problem in the Australian Capital Territory. *Australian forestry*, **4**, 33-38.
- QGIS (2011) QGIS Development Team, 2015. *Quantum GIS Geographic Information System*. Open Source Geospatial Foundation Project. <http://qgis.osgeo.org>.
- Qian, H. & Zhang, J. (2016) Are phylogenies derived from family-level supertrees robust for studies on macroecological patterns along environmental gradients? *Journal of Systematics and Evolution*, **54**, 29-36.
- R Development Core Team (2016) R: A language and environment for statistical computing. Vienna, Austria.
- Reich, P.B., Walters, M.B. & Ellsworth, D.S. (1997) From tropics to tundra: global convergence in plant functioning. *Proceedings of the National Academy of Sciences*, **94**, 13730-13734.
- Richards, S.A., Possingham, H.P. & Tizard, J. (1999) Optimal fire management for maintaining community diversity. *Ecological Applications*, **9**, 880-892.
- Robinson, N.M., Leonard, S.W., Bennett, A.F. & Clarke, M.F. (2014) Refuges for birds in fire-prone landscapes: the influence of fire severity and fire history on the distribution of forest birds. *Forest Ecology and Management*, **318**, 110-121.

- Robley, A., L. Woodford, M. Lindeman and R. Sorre (2013) ARI Technical Report 249 Influence of fire fox control and habitat on native mammals.
- Rodrigues, A., Brooks, T.M. & Gaston, K.J. (2005) Integrating phylogenetic diversity in the selection of priority areas for conservation: does it make a difference. *Phylogeny and conservation*, **8**, 101-119.
- Rosauer, D.F., Ferrier, S., Williams, K.J., Manion, G., Keogh, J.S. & Laffan, S.W. (2014) Phylogenetic generalised dissimilarity modelling: a new approach to analysing and predicting spatial turnover in the phylogenetic composition of communities. *Ecography*, **37**, 21-32.
- Rovero, F., Tobler, M. & Sanderson, J. (2010) Camera-trapping for inventorying terrestrial vertebrates. *Manual on field recording techniques and protocols for All Taxa Biodiversity Inventories and Monitoring. The Belgian National Focal Point to the Global Taxonomy Initiative*, pp. 100-128.
- Rowcliffe, J.M., Field, J., Turvey, S.T. & Carbone, C. (2008) Estimating animal density using camera traps without the need for individual recognition. *Journal of Applied Ecology*, **45**, 1228-1236.
- Si, X., Kays, R. & Ding, P. (2014) How long is enough to detect terrestrial animals? Estimating the minimum trapping effort on camera traps. *PeerJ*, **2**, e374.
- Simberloff, D. (1972) Properties of the rarefaction diversity measurement. *The American Naturalist*, **106**, 414-418.
- Sitters, H., Christie, F.J., Di Stefano, J., Swan, M., Penman, T., Collins, P.C. & York, A. (2014) Avian responses to the diversity and configuration of fire age classes and vegetation types across a rainfall gradient. *Forest Ecology and Management*, **318**, 13-20.
- Stretton, L.E.B. (1939) Report of the Royal Commission to inquire into the causes of and measures taken to prevent the bush fires of January, 1939 and to protect life and property and the measures to be taken to prevent bush fires in Victoria and to protect life and property in the event of future bush fires. Government Printer, Melbourne.
- Swan, M., Christie, F., Sitters, H., York, A. & Di Stefano, J. (2015) Predicting faunal fire responses in heterogeneous landscapes: the role of habitat structure. *Ecological Applications*, **25**, 2293-2305.
- Swan, M., Di Stefano, J., Christie, F., Steel, E. & York, A. (2014) Detecting mammals in heterogeneous landscapes: implications for biodiversity monitoring and management. *Biodiversity and Conservation*, **23**, 343-355.
- Taylor, M.D., Nicholas (2013) *Field Guide to the Birds of the ACT, 2nd Edition*. National Parks Association of the ACT Inc.
- Taylor, R.S., Watson, S.J., Nimmo, D.G., Kelly, L.T., Bennett, A.F. & Clarke, M.F. (2012) Landscape-scale effects of fire on bird assemblages: does pyrodiversity beget biodiversity? *Diversity and Distributions*, **18**, 519-529.
- Tolhurst, K.G. & McCarthy, G. (2016) Effect of prescribed burning on wildfire severity: a landscape-scale case study from the 2003 fires in Victoria. *Australian forestry*, **79**, 1-14.
- Towerton, A.L., Penman, T.D., Kavanagh, R.P. & Dickman, C.R. (2011) Detecting pest and prey responses to fox control across the landscape using remote cameras. *Wildlife Research*, **38**, 208-220.
- Tuft, K.D., Crowther, M.S. & McArthur, C. (2012) Fire and grazing influence food resources of an endangered rock-wallaby. *Wildlife Research*, **39**, 436-445.
- Van Dyck, S. & Strahan, R. (2008) *The mammals of Australia*. New Holland Pub Pty Limited.
- Vane-Wright, R.I., Humphries, C.J. & Williams, P.H. (1991) What to protect?—Systematics and the agony of choice. *Biological conservation*, **55**, 235-254.
- Ward Jr, J.H. (1963) Hierarchical grouping to optimize an objective function. *Journal of the American statistical association*, **58**, 236-244.
- Welbourne, D. (2013) A method for surveying diurnal terrestrial reptiles with passive infrared automatically triggered cameras. *PloS one*, **6**, e18965.
- Welbourne, D.J., MacGregor, C., Paull, D. & Lindenmayer, D.B. (2015) The effectiveness and cost of camera traps for surveying small reptiles and critical weight range mammals: a comparison with labour-intensive complementary methods. *Wildlife Research*, **42**, 414.
- Westgate, M.J., Driscoll, D.A. & Lindenmayer, D.B. (2012) Can the intermediate disturbance hypothesis and information on species traits predict anuran responses to fire? *Oikos*, **121**, 1516-1524.
- Whelan, R.J., Rodgers, L., Dickman, C.R. & Sutherland, E.F. (2002) Critical life cycles of plants and animals: developing a process-based understanding of population changes in fire-prone landscapes.

- Flammable Australia: the Fire Regimes and Biodiversity of a Continent* (eds R.A. Bradstock, J.E. Williams & A.M. Gill), pp. 94-124. Cambridge University Press.
- Winter, M., Devictor, V. & Schweiger, O. (2013) Phylogenetic diversity and nature conservation: where are we? *Trends in Ecology & Evolution*, **28**, 199-204.
- Wis, M.S., Pottier, J., Kissling, W.D., Pellissier, L., Lenoir, J., Damgaard, C.F., Dormann, C.F., Forchhammer, M.C., Grytnes, J.A. & Guisan, A. (2013) The role of biotic interactions in shaping distributions and realised assemblages of species: implications for species distribution modelling. *Biological Reviews*, **88**, 15-30.
- Yan, C., Xie, Y., Li, X., Holyoak, M. & Zhang, Z. (2016) Species co-occurrence and phylogenetic structure of terrestrial vertebrates at regional scales. *Global Ecology and Biogeography*.
- Zylstra, P. (2006) *Fire history of the Australian Alps: prehistory to 2003*. Department of the Environment and Water Resources.

Supplementary material

Table S1. Summary of camera trap deployments and site locations for mammal survey.

Site name	Site code	x	y	Date deployed	Date recovered	Number of wf traps	Number of ir traps	Total trap nights
TNR01	23223	671428	6071164	16/03/2016	23/03/2016	5	2	49
TNR02	23223	671353	6074763	17/03/2016	29/03/2016	5	2	84
NC01	33223	666433	6082903	03/04/2016	18/04/2016	3	1	60
GC01	13221	675006	6032937	05/04/2016	24/04/2016	3	1	76
CR01	23223	690530	6047966	20/04/2016	05/05/2016	3	1	60
CR02	23223	688971	6032949	21/04/2016	05/05/2016	3	1	56
CF01	23223	666624	6076504	04/05/2016	28/05/2016	3	1	96
TNR03	23223	671911	6074338	13/05/2016	26/06/2016	3	1	176
SC01	13221	674103	6035549	18/05/2016	03/07/2016	3	1	184
MG01	23222	665312	6050779	29/06/2016	08/08/2016	1	1	80
CG01	33223	670495	6058963	30/06/2016	05/08/2016	1	1	72
ST01	33223	672666	6060068	01/07/2016	31/08/2016	1	1	122
MT01	23222	684506	6064204	12/07/2016	08/08/2016	1	1	54
OR01	23222	677890	6059642	12/07/2016	08/08/2016	1	1	54
BR01	23222	681078	6032651	21/07/2016	13/08/2016	1	1	46
SS01	13221	674146	6032414	31/07/2016	20/08/2016	1	1	40
SS02	13221	674589	6031541	31/07/2016	20/08/2016	1	1	40
LH01	33223	665185	6057778	10/08/2016	27/08/2016	1	1	34
DC01	33223	665842	6056977	10/08/2016	27/08/2016	1	1	34
DC01	33223	665841	6056971	10/08/2016	27/08/2016	1	1	34
DC02	33223	667433	6057507	10/08/2016	27/08/2016	1	1	34
DC02	33223	667433	6057507	10/08/2016	27/08/2016	1	1	34
LHC01	13221	688037	6031940	12/08/2016	29/08/2016	1	1	34

LHC01	13221	688040	60319 41	12/08/2016	29/08/2016	1	1	34
LHC02	13221	688550	60311 60	12/08/2016	29/08/2016	1	1	34
NCK01	23222	682564	60289 50	13/08/2016	29/08/2016	1	1	32

Table S2. Field methods after (Mulvaney *et al.* 2014).

Variable/attribute	Method
PFC Canopy	50m transect. At each 5 metre interval make an assessment of the PFC using the “hole in the hands” estimation method.
Canopy height (median)	50x20. Estimate the median canopy height in metres. If 2 or more distinct sub strata exist, record for each as O1, O2...On.
Canopy Dieback	50x20. Estimate the average health of overstorey species: 1 – healthy tree, 2- foliage beginning to die from tips of branches, some thinning or ‘sickness’ of leaves and some partly dead branches, 3 – as with 2 but with greater loss of foliage and some completely dead branches, 4 – most of the epicormic foliage has died, 5- dead.
Species dominance and evenness	50x20. Record and identify the ≤ 3 most dominant species in the overstorey, record total species count of overstorey species and the number of individuals of each species. Can be done in conjunction with tree hollow assessment
PFC midstorey	50m transect. Visually estimate percent foliage cover every 5 metres of transect.
Midstorey species richness	20x20. Record and identify the ≤ 3 most dominant species in the midstorey , record total species count of midstorey species and the number of individuals of each species.
PFC grasses	At 50 points (i.e. every 1m) along 50m transects record whether grass intersects that point. Note, multiple ‘hits’ (i.e. multiple grasses) at a point count as one hit only. Divide the total of ‘hits’ by the number of points measured along the transects (i.e. 150). Multiple transects are usually required to sufficiently encompass ground stratum cover heterogeneity.
PFC shrubs	At 50 points (i.e. every 1m) along 50m transects record whether shrub intersects that

	point. Note, multiple 'hits' (i.e. multiple grasses) at a point count as one hit only. Divide the total of 'hits' by the number of points measured along the transects (i.e. 150). Multiple transects are usually required to sufficiently encompass ground stratum cover heterogeneity.
PFC other ground stratum Cryptogam, %rock, bare ground, litter cover.	At 50 points (i.e. every 1m) along 50m transects record whether a ground element intersects that point. Note, multiple 'hits' (i.e. multiple grasses) at a point count as one hit only. Divide the total of 'hits' by the number of points measured along the transects (i.e. 150). Multiple transects are usually required to sufficiently encompass ground stratum cover heterogeneity.
Litter depth	50m transect. At each metre interval measure the depth of litter using a ruler.
Length of fallen logs (CWD) over 0.1m dia.	This is the total length of logs at least 10cm diameter and at least 0.5m long. The diameter is estimated with a measuring tape (or callipers if available) held horizontally immediately above the log and the length is estimated to the nearest metre by measuring with a tape, or pacing along the part of the log that is at least 10cm diameter. If estimating length by pacing then the actual length of a sample of logs should be measured regularly with a tape so the assessor can calibrate their own estimate derived from pacing. Only those parts of logs lying within the plot are measured.

Table S3. Mean and standard errors for each of the habitat variables in each of explanatory variable classes.

Habitat variable	UB		MM		MH		HH	
	mean	s.e.	mean	s.e.	mean	s.e.	mean	s.e.
bare_ground	0.5	0.29	0	0	0.2	0.20	0.8	0.65
cryptogam	0.25	0.25	0	0	2.6	1.69	0	0
CWD_length	78.5	44.06	29.5	14.66	68.4	23.49	56.7	8.99
GC_Fern	0	0	0.5	0.50	3.2	2.27	4.3	3.59
GC_Grass	18.5	7.46	31.8	1.38	15.6	6.52	22.2	5.44
GC_Herb-dicot	1	0.41	3.0	1.47	3.0	1.79	7.0	1.46
GC_litter	48.5	0.65	49.3	0.25	37.4	7.16	46.7	0.92
GC_Sedge/Rush	1.5	1.19	4.0	1.78	1.0	0.55	1.7	0.42

GC_Woody	2.5	1.04	3.8	1.31	1.2	0.58	6.5	3.92
live_tree_shrub	0	0	0	0	0.8	0.49	7.2	2.51
mean_litter_depth	3.975	0.56	4.089	0.83	5.7	1.00	5.056	1.8
mean_MS_cover	6.3	2.33	4.611	1.74	25.1	5.94	29.7	7.06
OS_cover	31.8	1.31	28	1.87	14.0	4.95	15.8	3.76
rock	0.75	0.48	0.5	0.29	2.6	1.47	2.5	1.34

KEF data sources.

Table S4. Avian dietary preferences assigned from Garnett *et al.* (2015).

Species	Key dietary preference							
	Fruit	Nectar	Seeds	Foliage	Corms	Invertebrates	Vertebrates	Carrion
<i>Acanthiza lineata</i>	0	0	1	0	0	1	0	0
<i>Acanthiza pusilla</i>	0	0	1	0	0	1	0	0
<i>Acanthorhynchus tenuirostris</i>	0	1	0	0	0	1	0	0
<i>Accipiter fasciatus</i>	0	0	0	0	0	0	1	1
<i>Alisterus scapularis</i>	1	0	1	0	0	0	0	0
<i>Anthochaera carunculata</i>	0	1	0	0	0	1	0	0
<i>Cacatua galerita</i>	0	0	1	0	1	0	0	0
<i>Cacomantis flabelliformis</i>	0	0	0	0	0	1	0	0
<i>Cacomantis variolosus</i>	0	0	0	0	0	1	0	0
<i>Caligavis chrysops</i>	0	1	0	0	0	1	0	0
<i>Callocephalon fimbriatum</i>	0	0	1	0	0	0	0	0
<i>Colluricincla harmonica</i>	1	0	1	0	0	1	1	0
<i>Corcorax melanorhamphos</i>	0	0	1	0	0	1	0	0
<i>Cormobates leucophaea</i>	0	0	0	0	0	1	0	0
<i>Corvus coronoides</i>	0	0	1	0	0	1	1	1
<i>Corvus mellori</i>	0	0	0	0	0	1	1	1
<i>Dacelo novaeguineae</i>	0	0	0	0	0	1	1	0
<i>Eopsaltria australis</i>	0	0	0	0	0	1	0	0
<i>Melithreptus lunatus</i>	0	1	0	0	0	1	0	0
<i>Nesoptilotis leucotis</i>	1	1	0	0	0	1	0	0
<i>Oriolus sagittatus</i>	1	0	1	0	0	1	0	0
<i>Pachycephala pectoralis</i>	1	0	0	0	0	1	0	0
<i>Pardalotus punctatus</i>	0	0	0	0	0	1	0	0
<i>Pardalotus striatus</i>	0	0	0	0	0	1	0	0
<i>Platycercus elegans</i>	1	0	1	0	0	1	0	0
<i>Ptilonorhynchus violaceus</i>	1	1	0	1	0	0	0	0
<i>Ptilotula fuscus</i>	0	1	0	0	0	1	0	0
<i>Rhipidura fuliginosa</i>	0	0	0	0	0	1	0	0
<i>Sericornis frontalis</i>	0	0	1	0	0	1	0	0
<i>Strepera graculina</i>	1	0	1	0	0	1	1	0
<i>Todiramphus sanctus</i>	0	0	0	0	0	1	1	0

Table S5. Mammal Key Ecological Functions. Note that not all species listed were detected in this study

Species	Key Ecological Function determined from (Van Dyck & Strahan 2008; Jones <i>et al.</i> 2009)																								
	seeds/fruits	fungi	pollinator	vertebrate disease	insects	controls	controls vertebrates	root feeder	frugivore	grazer	browser	spernivore	foliovore	nectarvore	prey	carrion feeder	ovivore	vertebrate feeder	invertebrate feeder	Constructs burrows	Digs soil	Secondary burrow user	Creates wallows	timber	Damages
<i>Cervus unicolor</i>	1	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	1	1	
<i>Dama dama</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	
<i>Sus scrofa</i>	1	1	0	1	0	1	1	1	1	1	1	0	0	0	1	1	0	1	1	0	1	0	1	1	
<i>Canis lupus</i>	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	1	0	1	1	0	0	0	0	0	
<i>Felis catus</i>	0	0	0	1	0	1	0	0	0	0	0	0	0	0	1	0	0	1	1	0	0	0	0	0	
<i>Vulpes vulpes</i>	0	0	0	1	1	1	0	0	0	0	0	0	0	0	1	1	0	1	1	0	0	1	0	0	
<i>Antechinus stuartii</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	1	0	0	0	0	0	
<i>Antechinus swainsonii</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	1	0	0	0	0	0	
<i>Dasyurus maculatus</i>	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	1	1	1	1	0	0	0	0	0	
<i>Phascogale tapoatafa</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	1	1	0	0	0	0	0	
<i>Acrobates pygmaeus</i>	0	0	1	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	
<i>Macropus giganteus</i>	1	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	
<i>Macropus rufogriseus</i>	1	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	
<i>Petauroides volans</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	
<i>Petaurus australis</i>	1	0	1	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	1	0	0	0	0	1	
<i>Petaurus breviceps</i>	1	0	1	0	0	0	0	0	0	0	0	0	0	1	1	0	1	0	1	0	0	0	0	1	
<i>Cercartetus nanus</i>	1	0	1	0	0	0	0	0	1	0	0	1	0	1	1	0	0	0	1	0	0	0	0	0	
<i>Phascolarctos cinereus</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	
<i>Pseudocheirus peregrinus</i>	1	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	
<i>Trichosurus caninus</i>	1	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	1	0	0	0	0	0	
<i>Trichosurus vulpecula</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	
<i>Vombatus ursinus</i>	1	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	1	1	0	0	0	

Species	Key Ecological Function determined from (Van Dyck & Strahan 2008; Jones <i>et al.</i> 2009)																							
	seeds/fruits	fungi	pollinator	vertebrate disease	insects	controls	vertebrates	root feeder	frugivore	grazer	brower	spermiivore	foliovore	nectarvore	prey	carrion feeder	ovivore	vertebrate feeder	invertebrate feeder	Constructs burrows	Digs soil	Secondary burrow user	Creates wallows	Damages timber
<i>Wallabia bicolor</i>	1	1	0	0	0	0	0	0	0	1	1	1	0	0	1	1	0	0	1	0	1	0	0	0
<i>Oryctolagus cuniculus</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	1	0	0	0
<i>Tachyglossus aculeatus</i>	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	1	0	0	0
<i>Perameles nasuta</i>	1	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1	1	0	1	0	0	0
<i>Pseudomys fumeus</i>	1	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	1	1	0	0
<i>Rattus fuscipes</i>	1	1	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	1	0	1	1	0	0
<i>Rattus rattus</i>	1	1	0	1	0	0	0	0	1	0	0	1	0	0	1	0	1	1	1	1	1	1	0	0

Table S6. Number of detections of each species by site.

Site	Macropus_giganteus	Macropus_rufogriseus	Tachyglossus_aculeatus	Vombatus_ursinus	Antechinus_swainsonii	Wallabia_bicolor	Trichosurus_vulpecula	Trichosurus_caninus	Pseudocheirus_peregrinus	Cercartetus_nanus	Antechinus_stuartii	Rattus_rattus	Rattus_fuscipes	Felis_catus	Vulpes_vulpes	Canis_lupus	Sus_scrofa	Oryctolagus_cuniculus
BR01	2			4	2	2	8									1		
CF01						4	3			2	3							
CG01					6	1												
CR01		3	1	4		7	8			2					1			
CR02		3		2		4	8				2	1						
DC01													4					
DC02											1							

8

GC01	8				3	11	1				1					3
LH01	1								1						1	
LHC01	1				1	2	2									
LHC02	3				2	5										
MG01	4	4			1	2										
MT01	2	4	2	4												
NC01					2				7		12					
NCK01	1				1			1				1				3
OR01		4		3			4									
SC01		15		5		5	7	8			1					
SS01	2	9		3	1		4				1			1		
SS02	3	1	1				2									
ST01	1						3									
TNR01		1				6	1				6					
TNR02				2		8			1							
TNR03	3	5		14		25	1		1		6	1	4	1	2	
BR01	2			4	2	2	8								1	
CF01						4	3		2	3						
CG01					6	1										
CR01		3	1	4		7	8		2						1	
CR02		3		2		4	8			2	1					
DC01												4				
DC02									1							
GC01		8				3	11	1				1				3
LH01		1								1					1	
LHC01		1				1	2	2								
LHC02		3				2	5									
MG01	4	4				1	2									
MT01	2	4	2	4												
NC01						2				7		12				
NCK01	1				1			1					1			3

OR01		4		3		4								
SC01		15		5		5	7	8				1		
SS01	2	9		3	1		4					1		1
SS02	3	1	1				2							
ST01	1						3							
TNR01		1				6	1				6			
TNR02				2		8			1					
TNR03	3	5		14		25	1		1	6	1	4	1	2

Instructions for authors, Journal of Applied Ecology are available at the following web page. Their lengthiness prevents their full reproduction in full here.

<http://www.journalofappliedecology.org/view/0/authorGuideline.html#typesofpaperpublished>