SPATIAL AND TEMPORAL DRIVERS OF MAMMAL AND REPTILE DISTRIBUTIONS IN FIRE-PRONE LANDSCAPES

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Abstract

Fire is a key disturbance process in >50% of terrestrial ecosystems where species are often adapted to specific fire regimes. However, ongoing changes to fire regimes now represent a considerable threat to biodiversity, including mammals and reptiles in Australia. Despite this, there are large knowledge gaps around how different types of fire – or 'pyrodiversity' – affect biodiversity and how fire interacts with other key drivers of animal populations. This thesis explored the drivers of mammal and reptile distributions in fire-prone mallee woodlands, to answer key questions of wider relevance to fire-prone ecosystems.

First, I explored the influence of daily meteorological conditions on small mammal capture rates during field surveys. I built regression models of the daily capture rates of different species and families against six meteorological variables to determine favourable conditions for ecological surveys. Capture rates of all species were influenced by a least one meteorological variable, although responses to specific conditions varied between taxa. Small mammal surveys should be conducted in a variety of weather conditions to ensure a representative sample of the local community and my results can be used to determine conditions favourable to a single species or family.

Second, I explored how pyrodiversity influences the distributions of whole animal assemblages in conjunction with other environmental gradients. I built species distribution models for 17 mammals and 54 reptiles against different measures of fire and mapped the distribution of each species across a >100 000 km^2 region. Native mammals showed a variety of responses. Microbats were more likely to occur as time since fire increased, whereas rodent occurrence was positively correlated with recently burned areas. Occurrence of small dasyurid marsupials was positively correlated with the area and configuration of older post-fire age classes, showing the importance of spatial context in understanding the fire responses of animals. Reptile responses to pyrodiversity were more consistent, however there was variation within taxonomic groups regarding the importance of different fire elements. Many reptiles occurred most frequently between 0-50 years post-fire, coinciding with peaks in hummock grass growth. However, configuration of this key habitat component was important to reptiles, and planning for appropriate configurations of fire-derived habitat should be incorporated into fire

management for reptile conservation. Maps of species distributions can be used to better incorporate the needs of individual species into fire and conservation planning.

Lastly, I used a field experiment to test different mechanisms for how fine-scale patterns of fire influence animal populations in recently burnt environments. Small mammal, reptile and invasive predator activity was monitored in the year following a large planned burn to explore how the amount and configuration of burnt and unburnt habitat affects recovery. Reptile assemblages varied between burnt and unburnt sites, and reptile species richness increased at sites with higher amounts and connectivity of unburnt vegetation. Mammals did not have clear relationships with spatial patterns of planned fire, however fox activity increased after the fire at all sites. Small mammals appear resilient to this scale of planned fire and retaining well-connected unburnt refuges is important for maintaining reptile diversity.

Overall, my work helps to define the types of fire that benefit mallee mammal and reptile assemblages, and this thesis highlights how this knowledge can be used to improve fire management for biodiversity conservation.

Declaration

This is to certify that:

i. The thesis comprises only my original work towards the Doctor of Philosophy, except where indicated in the preface

ii. Due acknowledgement has been made in the text to all other material used; and

iii. The thesis is fewer than 100 000 words in length, exclusive of tables, maps, bibliographies and appendices

Katharine Senior

20/11/2020

Preface

This thesis has been supported through collaboration with various co-authors from the University of Melbourne, La Trobe University and the University of Leida (Spain), detailed below. To facilitate journal publications, the four data chapters that present my PhD research (Chapters 2-5) have been written as independent studies and therefore contain some unavoidable repetition. The publication status is as follows:

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Contributions: KS, JR and LK conceived the ideas. LK collected the data with colleagues from the Mallee Fire and Biodiversity and Mallee Hawkeye projects. JR and KS organised the data. KS performed the analysis with input from LK and MM. KS led the writing of the manuscript and revisions. All authors contributed critically to writing.

K. Senior contribution: 70%

Chapter 3

Senior, K.L., Giljohann, K.M., McCarthy, M.A., Rainsford, F.W. and Kelly, L.T. (in review). Predicting mammal responses to pyrodiversity: from microbats to macropods.

Contributions: KS led study conception, with contributions from LK and MM. KS, FR, KG and LK collaboratively conceived the methodology and models. KS led data acquisitions and analysis, with assistance from FR, KG and Simon Kapitza (the University of Melbourne). KS, LK and KG interpreted the results. KS led the writing of the manuscript and revisions. All authors contributed critically to writing.

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Chapter 4

Senior, K.L., Giljohann, K.M., McCarthy, M.A., Rainsford, F.W. and Kelly, L.T. (in prep*)*. Mapping desirable forms of pyrodiversity for snakes and lizards.

Contributions: KS led study conception, with contributions from LK and MM. KS, FR, KG and LK collaboratively conceived the methodology and models. KS led data acquisitions and analysis, with assistance from FR, KG and Simon Kapitza (the University of Melbourne). KS, LK and KG interpreted the results. KS led the writing of the manuscript, with input from LK, KG and MM.

K. Senior contribution: 75%

Chapter 5

Senior, K.L., Giljohann, K.M., McCarthy, M.A. and Kelly, L.T. (in prep). A field test of mechanisms underpinning animal persistence in recently burned landscapes.

Contributions: KS led the study conception, with input from KG, LK and MM. KS and LK designed the methodology. KS conducted the fieldwork (including site selection) with assistance from LK and volunteers. KS performed the GIS work and data analysis. KS interpreted the results, with assistance from KG, LK and MM. KS led the writing of the manuscript. All authors contributed critically to writing.

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Chapter 1 – Introduction

Fire-prone ecosystems

Fire is a major disturbance process in many ecosystems around the world (Bond and Keeley 2005; Archibald *et al.* 2013; He *et al.* 2019). Fire-dependent ecosystems cover >50% of the world's terrestrial land surface (Shlisky *et al.* 2007) and include several global biodiversity hotspots (Rundel *et al.* 2018). However, patterns of fire are changing and the risk of large and severe wildfire is predicted to increase with climate change in many Mediterranean-type ecosystems (Adams 2013; Mitchell *et al.* 2014; Wimberly and Liu 2014; Boer *et al.* 2016). Recent wildfire seasons in Australia ("Black Summer" 2019-2020 fire season; Boer *et al.* 2020) and California (2018 and 2020 wildfire seasons; Williams *et al.* 2019) are unprecedented in terms of area burnt and highlight the risk that more frequent wildfires pose to ecosystems. In other cases, such as savannas in Africa and South America, reduced fire activity can modify ecosystems and reduce biodiversity (Durigan and Ratter 2016; Probert *et al.* 2019; Durigan 2020).

Species that have evolved in areas with recurrent fire are often adapted to require specific fire regimes for their survival (Parr and Andersen 2006; Codding *et al.* 2014; Jones *et al.* 2020; Berlinck and Batista 2020). Fire regimes are the long-term patterns of fire experienced by an ecosystem or place, and are measured in terms of individual fire elements (or traits), such as the time in between fires, fire frequency, severity, season, and the extent, configuration and diversity of fires in a landscape (Gill 1975; Gill and Allan 2008). Globally, fire regimes are changing due to human activity including new land uses (Pausas and Keeley 2014; Baillie and Bayne 2019), modified fire management practices (Costa *et al.* 2020; Schmidt and Eloy 2020), climate change (Williams *et al.* 2019; Maxwell *et al.* 2020; Xu *et al.* 2020), habitat fragmentation (Parsons and Gosper 2011; Yezzi *et al.* 2019) and the introduction of invasive plants and animals (Radford *et al.* 2015; Fusco *et al.* 2019; Hradsky 2020).

Inappropriate fire regimes, often considered to be fire regimes outside of historical patterns, now represent a considerable threat to global biodiversity (Braun de Torrez *et al.* 2018; Carbone *et al.* 2019; Costa *et al.* 2020; Roberts *et al.* 2020). This includes a suite of vertebrates, including 16% of mammals and 14% of reptiles listed as threatened by the International Union for Conservation of Nature (Kelly *et al.* 2020). In Australia,

inappropriate fire regimes are a growing threat to the conservation of reptiles and small mammals (Woinarski *et al.* 2015a; Woinarski *et al.* 2015b). Thus, there is a pressing need to understand how different elements of fire regimes influence animals so that fire can be appropriately managed for biodiversity conservation (Maravalhas and Vasconcelos 2014; Legge *et al.* 2015; Kelly *et al.* 2017a; Wibisono *et al.* 2018; Volkmann *et al.* 2020).

Fire does not act alone; animal distributions are driven by complex relationships between climate, geology, a range of disturbances, and biological interactions (Driscoll *et al.* 2010; Grundel *et al.* 2015; Giljohann *et al.* 2017; Beale *et al.* 2018; Dorph *et al.* 2020). In arid ecosystems, for example, rainfall varies considerably from year to year and this shapes fire activity, food and shelter resources and post-fire recovery of plants and animals (Letnic and Dickman 2010; Plavsic 2014; Greenville *et al.* 2016a; Bennison *et al.* 2018; Westerhuis *et al.* 2020). Additionally, biotic interactions such predation, herbivory and competition interact with fire to influence animal populations (Fox 1982; Letnic *et al.* 2004; Fuhlendorf *et al.* 2010; McGregor *et al.* 2016b; Geary *et al.* 2018). For example, activity or abundance of predators can increase in recently burnt environments and this can have negative impacts on vertebrate populations (Frank *et al.* 2014; Hovick *et al.* 2017; Hradsky 2020). Responses to fire can also vary spatially, with both local (e.g. dunes and swales) and regional conditions (e.g. aridity gradients) influencing fire responses (Nimmo *et al.* 2014; Verdon *et al.* 2019; Burns and Phillips 2020). Understanding how fire operates in relation to other environmental processes is a key knowledge gap that need to be addressed for effective fire management of vertebrates from a range of taxonomic groups.

On pyrodiversity

The inherent variation in fire regimes creates patterns of fire in a landscape, called ‘pyrodiversity’. Here, I define pyrodiversity as spatial and temporal variation in fire regimes, including both variation within and between fire regimes (Martin and Sapsis 1991; Faivre *et al.* 2011; Tingley *et al.* 2016; Kelly *et al.* 2017a; Hempson *et al.* 2017; Bliege Bird *et al.* 2018; Stillman *et al.* 2019; Kelly *et al.* 2020). Variation in fire history – i.e. conditions generated by a fire regime at a particular location, such as the time, severity and extent of the most recent fire – also contributes to pyrodiversity in a landscape (Faivre *et al.* 2011; Kelly *et al.* 2015; Stillman *et al.* 2019; Steel *et al.* 2019).

A central theme in fire ecology is the idea that pyrodiversity supports biodiversity (Martin and Sapsis 1991). For example, mixed severity fire that creates interspersed patches burnt at low severity (surface fuel consumed) to high severity (canopy fuel consumed) leads to diverse bird assemblages in North American forests (Tingley *et al.* 2016; Taillie *et al.* 2018). Diversity in post-fire age classes that contain different food and shelter resources increases bird diversity in temperate forests of south-eastern Australia (Sitters *et al.* 2014). However, not all forms of pyrodiversity are ecologically meaningful, and pyrodiversity per se does not always promote diverse assemblages of animals (Parr and Andersen 2006). For example, intermediate variation in fire frequency benefits threatened species of mammal in savanna landscapes of northern Australia, but very high levels of variation in fire frequency do not (Davies *et al.* 2018). Mid to late successional vegetation is disproportionately important for birds, reptiles and small mammals in semi-arid woodlands, whereas areas with increasing diversity in age classes do not necessarily enhance animal populations (Kelly *et al.* 2012; Kelly *et al.* 2015; Connell *et al.* 2017). Rather than repeatedly asking the general question 'does pyrodiversity beget biodiversity', asking more precise questions offers opportunities to explore the many different forms of pyrodiversity that influence animals (Parr and Andersen 2006; Kelly *et al.* 2017a; Beale *et al.* 2018).

Variation in fire regime characteristics, including the time in between fires, fire frequency, severity, season, extent, configuration and spatial diversity underpins pyrodiversity (Faivre *et al.* 2011; Farnsworth *et al.* 2014; Kelly *et al.* 2017a; Steel *et al.* 2019; Stillman *et al.* 2019). Sources of temporal variation in fire patterns include time since the last fire, fire frequency, inter-fire interval and the season in which fire occurs. Much research on animal relationships with fire has focused on the responses of species to time since the last fire (hereafter "time since fire"). Time since fire can influence the availability of key resources, such as tree hollows (Haslem *et al.* 2012; Lindenmayer *et al.* 2012), grass and seeds (Murphy and Bowman 2007; Weier *et al.* 2018), and vegetation cover (Fox 1982; Coops and Catling 2000; Letnic *et al.* 2004; Di Stefano *et al.* 2011; Arthur *et al.* 2012). However, time since fire is not always the most influential aspect of a fire regime on animals (Driscoll *et al.* 2010; Lindenmayer *et al.* 2016a). For example, inter-fire interval (the time between multiple fires) is a useful descriptor of the distributions of vertebrates in dry eucalypt forests of south-eastern Australia (Kelly *et al.*

2017b). In North American forests, time since fire interacts with fire severity to influence avian diversity (Smucker *et al.* 2005; Taillie *et al.* 2018).

Fire frequency is another characteristic of fire regimes that contributes important temporal variation to pyrodiverse landscapes. Resources such as tree hollows take a long time to develop and frequent fires can reduce their availability (Haslem *et al.* 2012; Salmona *et al.* 2018). This has negative impacts on hollow dependent animals including birds, arboreal mammals and microbats (Lumsden and Bennet 1995; Lindenmayer *et al.* 2013; Connell *et al.* 2017). Focusing solely on time since fire would limit our understanding of hollow development (Haslem *et al.* 2012) and the availability of this resource in fire-prone landscapes.

The seasonality of fires is also important (Miller *et al.* 2019). A shift from small, early dry-season fires to large, late dry-season fires has been implicated in the decline of small mammals across much of northern Australia (Lawes *et al.* 2015), with increased fire frequency also a likely factor (Andersen *et al.* 2005; Andersen *et al.* 2012; Griffiths *et al.* 2015a; Andersen 2020). Seasonal timing of management actions such as prescribed fire can be important, with spring burns resulting in higher native bee abundances in tallgrass prairie compared to winter burns (Decker and Harmon-Threatt 2019). However, there are relatively few examples of studies that model the influence of fire frequency and fire season on animal distributions, and large knowledge gaps regarding how to predict animal responses to these fire regime elements.

Pyrodiversity also incorporates variation in spatial characteristics of fire regimes, such as the extent and configuration of burnt and unburnt vegetation, and spatial diversity in fire history (Bradstock *et al.* 1996; Faivre *et al.* 2011; Kelly *et al.* 2017a; Downing *et al.* 2019). For many species, large amounts of a single post-fire age class are required to maintain healthy populations (Taylor *et al.* 2013). For example, there is a positive correlation between large areas of long-unburnt vegetation and reptile and arboreal mammal populations in temperature eucalypt forests (Lindenmayer *et al.* 2013; Dixon *et al.* 2018). Other species require large patches of recently burnt vegetation, such as granivorous rodents that benefit from a post-fire flush of vegetation in temperate heathlands (Pereoglou *et al.* 2016). Some taxa, such as burrowing lizards, are not dependent on fire-derived habitat structure for shelter and may find a competitive

advantage in occupying recently burnt environments (Smith *et al.* 2013; Nimmo *et al.* 2013).

The configuration of fire-driven elements, in addition to the total area they comprise, can also be important (Brotons *et al.* 2005a). Adjacent patches of differing fire history can allow multiple species to coexist in a landscape, or a single species to meet different resource needs (Stillman *et al.* 2019). For example, the eastern chestnut mouse (*Pseudomys gracilicaudatus*) favours recently burnt environments, but requires a fine-scale burn configuration that incorporates shelter features such as rocks or collapsed burnt shrubs (Pereoglou *et al.* 2011; Pereoglou *et al.* 2016). Proximity to unburnt vegetation is a key driver for many species, affecting their ability to both utilise and recolonise burnt areas (Brotons *et al.* 2005b; Watson *et al.* 2012a; Law *et al.* 2018; Puig-Gironès *et al.* 2018; Simms *et al.* 2019). Importantly, unburnt vegetation can also be present within the burn perimeter of a fire, in patches which provide refuges for animals and aid recolonisation (Robinson *et al.* 2013; Leonard *et al.* 2014; Swan *et al.* 2016).

While these examples highlight the role of spatial and temporal variation in fire regimes in defining pyrodiversity-biodiversity relationships, most come from studies on a single fire regime variable, or small subset of measures. Studies that explore the effects of many different spatial and temporal fire regime attributes on biodiversity remain rare, despite being identified as a key knowledge gap by Driscoll et al. (2010) over a decade ago.

Fine-scale patterns of fire and the mechanisms through which they shape animal populations

Fire affects biodiversity at a range of scales, from regions and landscapes (10s to 100s of km^2), to local areas and microhabitats (e.g. 100 ha to individual plants). At finer scales, fire-derived habitat can be a strong influence on terrestrial fauna occurrence (Fox 1982; Catling *et al.* 2001; Di Stefano *et al.* 2011; Pinto *et al.* 2018; Dorph *et al.* 2020). In a range of temperate ecosystems, a relatively predictable post-fire vegetation succession takes place that affects vegetation structure and provides changing food and shelter resources (Fox 1982; Brehme *et al.* 2011; Arthur *et al.* 2012; Swan *et al.* 2015; Lindenmayer *et al.* 2016b). Conditions meet the needs of different mammal species at different times, allowing them to peak in abundance and outcompete other sympatric species when ideal habitat conditions are present, then decline in abundance as the conditions become more suitable for a competitor (Fox *et al.* 2003). However, this pattern

of predictable vegetation and animal post-fire succession – an idealized pattern described by the habitat accommodation model (Fox 1982) – is not displayed in all fire-prone ecosystems. For example, in arid and semi-arid regions of Australia, most reptiles respond to fire in a predictable succession in line with the habitat accommodation model (Letnic *et al.* 2004; Nimmo *et al.* 2012). However, while some small mammal species in these ecosystems respond to changes in vegetation structure associated with shelter, the whole community does not follow a predictable succession of replacement in dominant species (Letnic and Dickman 2010; Kelly *et al.* 2011). Small mammals in arid and semi-arid areas may not follow a predictable successional replacement sequence for several reasons. They move long distances and can shift their home ranges out of fire-affected areas (Dickman *et al.* 1995; Letnic 2001; Bos and Carthew 2007; Nimmo *et al.* 2019). Small mammals have life history strategies such as torpor and burrowing that buffer the immediate effects of fire on food and shelter resources (Körtner *et al.* 2007; Körtner *et al.* 2016; Geiser *et al.* 2018). Some species also respond more strongly to pulses in resources driven by rainfall than to fire-affected vegetation change (described in more detail below; Brown and Ernest 2002; Kelly *et al.* 2013; Greenville *et al.* 2013).

Another fire-affected habitat component that provides important resources for mammals and reptiles is unburnt patches within a fire boundary. Unburnt patches enable some individuals to survive fire events (Robinson *et al.* 2013), provide shelter and food in the post-fire landscape and act as starting points for recolonisation of burnt areas (Banks *et al.* 2011; Banks *et al.* 2017; Puig-Gironès *et al.* 2018). However, the utility of unburnt patches may depend on their size (Berry *et al.* 2015b), configuration (Brotons *et al.* 2005a), distance to surrounding unburnt vegetation (Watson *et al.* 2012a) and the movement ecology of individual species (Brotons *et al.* 2005b; Simms *et al.* 2019). There is still uncertainty around how the size and configuration of unburnt patches affects their usefulness as refuges for mammals and reptiles, and these characteristics need to be understood if unburnt patches are to be manipulated to promote animal populations.

Unburnt patches may also be important to mammals and reptiles because they provide refuge from native and introduced predators, along with other habitat components such as rocks and dead vegetation (Watson *et al.* 2012a; Robinson *et al.* 2013; McGregor *et al.* 2015; Puig-Gironès *et al.* 2018; Hradsky 2020). There is mounting evidence to suggest that the threats of fire and predation by invasive cats and foxes interact in Australian ecosystems (Leahy *et al.* 2015; McGregor *et al.* 2016a; Hradsky 2020; Greenville *et al.*

2020). Introduced predators often increase activity in recently burnt environments, potentially increase hunting efficiency (McGregor *et al.* 2016b; Hradsky *et al.* 2017; Smith 2018), and can cause mortality of native animals (Leahy *et al.* 2015; Hradsky *et al.* 2017) and local extirpation (Frank *et al.* 2014). Habitat refuges may mitigate increased post-fire predation risk for native animals, but there are still large knowledge gaps about the types and scales of refuges that benefit animals and their effectiveness in different ecosystems.

Other environmental drivers

Environmental and climatic gradients can also shape pyrodiversity-biodiversity relationships. Regional rainfall or aridity gradients are important in shaping species distributions (Trainor and Woinarski 1994; Letnic *et al.* 2004; Nimmo *et al.* 2013; Pastro *et al.* 2013; Shryock *et al.* 2015). Fire responses of the same species can vary between different vegetation types, even within the same ecosystem (Lindenmayer *et al.* 2008b; Brehme *et al.* 2011; Foster *et al.* 2017; Chergui *et al.* 2019). Moreover, responses of biodiversity to fire regimes can also interact with environmental gradients, such as climate and rainfall gradients (Kelly *et al.* 2017b; Beale *et al.* 2018).

In many semi-arid and arid ecosystems, the occurrence of rainfall drives the abundance and distribution of species (Lima *et al.* 2008; Letnic and Dickman 2010; Arthur *et al.* 2012; Greenville *et al.* 2012; Kelly *et al.* 2013). The abundance of some small mammals, particularly rodents, is very low during normal dry conditions but booms in response to large rainfall events (Dickman *et al.* 1999; Greenville *et al.* 2016b; Bennison *et al.* 2018). Growth of near-surface vegetation after rainfall events also increases the risk of very large and severe fires (Greenville *et al.* 2009; Giljohann *et al.* 2017). Rainfall drives the abundance of species in more mesic ecosystems too, and can change the rate of animal recovery after fire (Hale *et al.* 2016; Crowther *et al.* 2018). Long-term patterns in rainfall have also been linked to climate cycles, such as El Niño-Southern Oscillation and Pacific Decadal Oscillation, particularly in low rainfall ecosystems (Lima *et al.* 1999; Letnic and Dickman 2006; Harris *et al.* 2018).

Daily weather conditions and meteorological cycles such as moon phase can be important determinants of animal activity. Extreme temperatures over the course of a day can cause mass mortality of species such as flying foxes (O'Shea *et al.* 2016; Ratnayake *et al.* 2019) and arid-zone birds (Conradie *et al.* 2020), while also contributing to severe fire-weather

events (Boer *et al.* 2020). Weather conditions also influence animal behaviour, and therefore the interpretation of results from ecological surveys and estimates of detectability and population size (Bas *et al.* 2008; Spence-Bailey *et al.* 2010; Gibson 2011; Burns *et al.* 2019). For example, some native Australian rodents are less active when the moon is full, and therefore could mistakenly be recorded as absent from a location (Burns *et al.* 2019). Reptiles in semi-arid woodlands are more likely to be captured on warmer days, and in spring when they are more active due to seasonal breeding (Spence-Bailey *et al.* 2010). A better understanding of the impact of daily conditions on animal activity is needed to ensure confidence in field surveys of vertebrates and associated research and conservation outcomes.

Semi-arid mallee woodlands

This thesis focuses on the relationships between animals, pyrodiversity, and other environmental drivers such as climate and weather. These themes are explored using field data, biodiversity atlas data and experiments from mallee woodlands and shrublands of south-eastern Australia. This broad vegetation group is characterised by stands of 'tree mallee' – eucalyptus trees with a multi-stemmed growth form. Mallee vegetation is fire-prone and an excellent ecosystem in which to study the effects of fire: wildfires are typically stand-replacing and vegetation and habitat structure regenerates after large fires in a relatively predictable succession (Haslem *et al.* 2011). Additionally, mallee woodlands support rich assemblages of mammals and reptiles, and studies to date indicate that many species rely on habitat components, such as hummock grass, tree hollows and leaf litter, that are strongly linked to fire.

The study area, the Murray Mallee region (32.8–35.2°S, 139.7–144.1°E; Figure 1.1), encompasses 104 000 km^2 of Victoria, New South Wales and South Australia. This region contains large tracks of mallee vegetation, including large reserves managed primarily for biodiversity conservation (Figure 1.1). Three main types of tree mallee have been categorised and mapped, differing primarily in the composition of understorey plants (Haslem *et al.* 2010; Figure 1.2). Triodia mallee is dominated by *Triodia scariosa* (hummock grass), chenopod mallee has a sparser understorey of Chenopodiaceae shrubs and saltbushes, and heathy mallee a mixed understorey of *Leptospermum* and *Melaleuca* spp., with some *T. scariosa* present. The landscape is low elevation (<100 m a. s. l.) with an undulating dune-swale system that provides mild topographic variation.

The climate of the study region is semi-arid, with a strong regional aridity gradient from the more mesic south-east to more arid north-west and increasing aridity inland from the coast. Mean annual rainfall increases from 160 mm in the north to 360 mm in the south, however inter-annual rainfall is highly variable. Large wildfires >100 000 ha occur in the region every 10-20 years, although longer fire intervals are common for a single location and small fires occur with greater frequency (Clarke *et al.* 2010; Avitabile *et al.* 2013). Planned fire is used throughout the region to reduce fuel load (and subsequent risk of large-scale wildfire) and to meet ecological needs (Department of Environment Land Water and Planning 2019; Connell *et al.* 2019). Fire suppression efforts also target wildfires on public land to reduce scale and impact (Department of Environment Land Water and Planning 2015). However, the application of planned fire, including the area and configuration of fires, and fire suppression varies between states and land tenure.

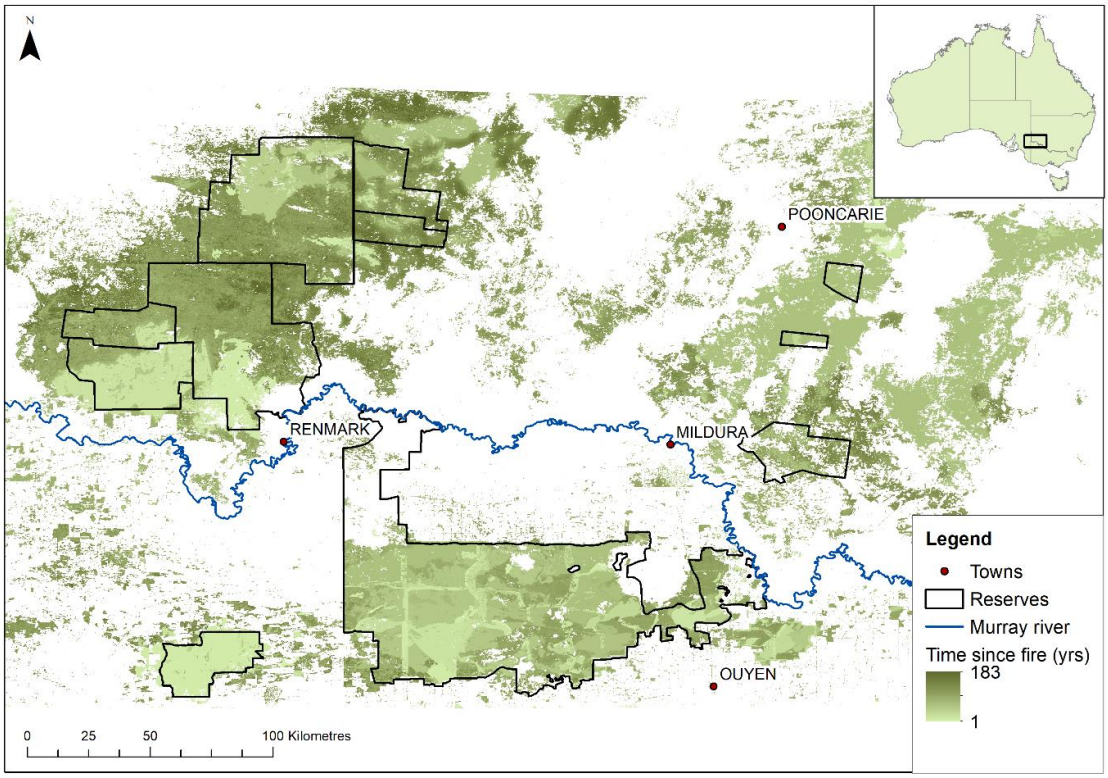

Figure 1.1: The Murray Mallee region, showing the extent of tree mallee vegetation (all coloured areas) and fire history in January 2020 (coloured gradient). Also shown are the location of major conservation reserves and towns.

The Murray Mallee region is home to a rich faunal assemblage. At least 70 reptile species are known to inhabit mallee woodlands. Many of these are dependent on fire-derived habitat components for their persistence (Nimmo *et al.* 2013) and 30 species are listed under state or federal threatened species legislation. The main taxonomic groups include skinks, geckos, dragons, elapid and blind snakes, legless lizards and varanids. The region also has a diverse mammal community, with at least 25 extant species (Bennett 2006). Population declines and extinctions (particularly among species weighing 35-5500 g) have led to the regional extinction of 10 species (Bennett *et al.* 2006), with a further 15 species currently listed on state or federal conservation legislation. Extant mammals include large macropods, microbats, the short-beaked echidna and a diverse small mammal community comprising dasyurids, burramyids and native rodents. Previous research has focused on the fire responses of common mallee small mammals (including two dasyurids, a burramyid and an introduced rodent), finding that older vegetation (>35 years post-fire) is important for small marsupials in mallee ecosystems (Kelly *et al.* 2012). However, the relationship between pyrodiversity and insectivorous bats, macropods and native rodents is largely unknown.

Figure 1.2: Three main types of mallee vegetation. From left to right: triodia mallee, chenopod mallee and heathy mallee.

The Murray Mallee region, and other mallee woodlands, have been the focus of significant research into how drivers such as fire and climate influence animal distributions and populations (Bos *et al.* 2002; Driscoll and Henderson 2008; Haslem *et al.* 2011; Kelly *et al.* 2013; Smith *et al.* 2013; Swan and Wilson 2015; Doherty *et al.* 2015a). Previous research on the relationships between long-term weather patterns and small mammal occurrence found that antecedent rainfall shapes mallee small mammal communities, but associations with rainfall are less predictable than in more arid parts of Australia (Morrant and Petit 2012; Kelly *et al.* 2013). However, the effects of short-term (i.e. daily) weather conditions, and other meteorological conditions such as moon illumination, on small mammal activity are largely unknown.

Early work in mallee woodlands and hummock grasslands in arid and semi-arid Australia helped advance understanding of how fire influences reptile and small mammal populations and habitat (Suijdendorp 1981; Caughley 1985; Caughley *et al.* 1985; Noble 1989; Friend 1993; Masters 1993; Masters 1996; Schlesinger *et al.* 1997). Extensive landscape-scale surveys were carried out across the Murray Mallee region from 2006-2008 to determine the relationships of common mallee species to time since fire and certain properties of fire mosaics, such as the spatial extent of a post-fire age class. This research showed that occurrence of some small mammals, birds and reptiles is strongly associated with time since fire, peaking between 20-55 years post-fire which coincides with peak growth of the keystone species *T. scariosa* (Kelly *et al.* 2011; Nimmo *et al.* 2012; Watson *et al.* 2012b). Additionally, long-unburnt vegetation (>35 years post-fire) provides important habitat for a wide range of mallee species, including small mammals and birds (Kelly *et al.* 2012; Taylor *et al.* 2013). More recent work has shown that retaining large amounts of long-unburnt vegetation in the landscape provides better outcomes for fire-sensitive biodiversity than a diverse mix of age classes (Kelly *et al.* 2015; Giljohann *et al.* 2015; Connell *et al.* 2017). However, it remains unknown how distributions of whole faunal assemblages, including both rare and common species, vary across the region in relation to pyrodiversity and other environmental gradients.

Only a few studies have examined the influence of burn configuration on mallee vertebrates. Berry *et al.* (2015b) examined the effect of patch size, unburnt refuges and patch isolation on birds in recently burnt mallee. Bird species richness was higher in continuous unburnt vegetation outside of the burnt area and in larger unburnt patches

(Berry *et al.* 2015b). Simms *et al.* (2019) investigated the effects of surrounding landscape context on recolonisation of burnt areas by mallee animals. They found that post-fire faunal communities were composed of species from the surrounding landscape, with degraded or fragmented surrounding habitat resulting in lower diversity in the newly burnt environment (Simms *et al.* 2019). Additionally, reptiles showed a low dispersal capability and this was further constrained by habitat structure and other patch attributes (Simms *et al.* 2019). Further research on how the size, configuration and connectivity of burnt and unburnt patches affects the ability of mammals and reptiles to persist in recently burnt environments is needed to advance understanding of desirable configurations of fire-derived habitat, and how refuges might mitigate predation by invasive cats and foxes.

Thesis outline and aims

In this thesis, I explored broad themes about variation in fire regimes and animal populations that are relevant to a variety of ecosystems worldwide. I did this by examining the drivers of small mammal and reptile distribution and abundance in semi-arid mallee woodlands of Australia.

Chapter 2 aimed to determine the influence of daily meteorological conditions on small mammal capture rates during pitfall surveys. Using a large dataset, I examined correlations between capture rates of small mammals and six meteorological variables to determine favourable conditions for trapping and improve the outcomes of field surveys.

Chapter 3 aimed to determine how different aspects of pyrodiversity, and other environmental gradients, affect whole mammal assemblages. I tested this by examining how these factors influence the distributions of mammals across the entire Murray Mallee region. I built species distribution models for 17 mammal species from a range of taxonomic groups including insectivorous bats, macropods, small marsupials, native rodents and the echidna, against a range of spatial and temporal measures of fire and other environmental covariates. The distribution of each species was mapped at 1 ha resolution to show how pyrodiversity influences mammal distributions. This chapter provided a framework for measuring and mapping pyrodiversity-biodiversity relationships in other fire-prone ecosystems and outlines how to begin to incorporate this knowledge into fire management to improve ecological outcomes.

Chapter 4 aimed to determine how different aspects of pyrodiversity, and other environmental gradients, affect whole reptile assemblages. I tested this by examining how these factors influence the distributions of reptile across the entire Murray Mallee region. I built species distribution models for 54 reptile species from different taxonomic and functional groups, against a range of spatial and temporal measures of fire and other environmental covariates. There are many more reptile species than mammals in the study region, which allowed for a greater exploration of similarities and differences of fire responses between taxonomic and functional groups. The models built in this chapter allow for the fire responses of more species to be incorporated into fire planning for more nuanced fire management that meets the needs of a whole faunal assemblage.

Chapter 5 tested five hypotheses that describe how pyrodiversity affects animal recovery after fire: the habitat status, habitat amount, habitat distance, habitat complementation and habitat refuge hypotheses. I used a field experiment structured around a planned burn to determine how small mammal, reptile and introduced predator populations and/or activity changes in relation to fine-scale configuration and patchiness of unburnt vegetation. I linked these results to broader ecological theory using the five tested hypotheses and provided recommendations for improving fire management for biodiversity outcomes.

Chapter 6 synthesised my research and highlights the scientific advances made and their application. I outlined the relevance to theoretical and practical aspects of fire ecology and conservation management and discuss future research directions.

Chapter 2 – The influence of weather and moon phase on small mammal activity

Preface

This chapter has been published in the journal *Australian Mammalogy* as the following reference:

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https://doi.org/10.1071/AM19056

Abstract

Small mammals are commonly surveyed using live trapping but the influence of weather conditions on trap success is largely unknown. This information is required to design and implement more effective field surveys and monitoring. I tested the influence of weather and moon phase on capture rates of small mammals in the Murray Mallee region of semi-arid Australia, using extensive pitfall trapping data collected at 267 sites (totalling 54,492 trap nights). I built regression models to explore the relationship between the capture rates of five species and daily meteorological conditions, and across families of mammals including dasyurids, burramyids and rodents. A relationship common to several taxa was the positive influence of high winds >20 km/h on capture rates. I also identified differences between taxa, with warmer overnight temperatures increasing capture rates of mallee ningaui but decreasing those of Bolam's mouse. This makes it difficult to determine a single set of 'optimal' meteorological conditions for surveying the entire community but points to conditions favourable to individual species and groups. I recommend that surveys undertaken in warmer months encompass a variety of meteorological conditions to increase capture rates and provide a representative sample of the small mammal community present in a landscape.

Introduction

Field surveys are vital for understanding the spatial and temporal dynamics of mammal populations, and for supporting conservation management. Effective surveys and monitoring rely on knowledge of mammal activity and how it is influenced by environmental changes at both short and long time scales (Gibson 2011; Wintle *et al.* 2012). Daily and seasonal fluctuations in meteorological conditions are some of the biggest environmental changes experienced by mammals (Bas *et al.* 2008; Buckley *et al.* 2018) but for many species the effects of these short-term dynamics are largely unknown. Quantifying the influence of weather and moon phase on mammal activity is likely to improve field survey design and understanding of mammal distributions and population dynamics (Brubaker *et al.* 2014; Burns *et al.* 2019).

Small mammals have several life history characteristics that make them responsive to changes in weather (Diete *et al.* 2017). A low surface-area to body mass ratio means that small mammals can have difficulty thermoregulating and therefore experience the effects of weather (e.g. temperature) more strongly than larger mammals (Ruf and Geiser 2015). A high metabolism means that small mammals need to feed regularly, and adverse weather can therefore have large impacts on activity, survival and mortality. Moreover, small mammals often have short life cycles and restricted breeding seasons, so adverse weather during critical life-stages can have particularly large effects (Gibson 2011; Rocha *et al.* 2017; Buckley *et al.* 2018). Consequently, short-term patterns in small mammal activity, generally measured using capture rate or foraging effort, are often associated with changes in weather (Roschlau and Scheibler 2016; Rocha *et al.* 2017; Burns *et al.* 2019).

In arid and semi-arid Australia, small mammals have evolved a range of life history strategies to deal with marked variation in weather and climate conditions (Kelly *et al.* 2013; Buckley *et al.* 2018). Most species are nocturnal, allowing them to avoid the extremely hot daytime conditions commonly experienced in these environments. Several species of burramyids and dasyurids engage in short-term hibernation, or torpor, that lowers their metabolic rate in times of stress (Ruf and Geiser 2015; Turner and Geiser 2017). By contrast, it appears that some species of native rodent in the Australian arid zone cannot control their thermoregulation through torpor (Tomlinson *et al.* 2007), and instead make use of behavioural adaptations to avoid extreme weather conditions, such

as burrowing (Dickman *et al.* 2010). Knowledge of species ecology and life history, combined with empirical studies to date, can guide predictions of how their activity will be influenced by short-term changes in meteorological conditions such as temperature, precipitation, wind speed and moon phase (see Table 2.1 and references therein).

In this study, I investigated how weather and moon phase influence the capture rate (i.e. activity) of five small mammal species in semi-arid mallee vegetation of the Murray Mallee region. The Murray Mallee contains a diverse small mammal community of rodents, burramyids and dasyurids and includes several threatened species (Bennett *et al.* 2006; Kelly *et al.* 2012). While several studies have explored how this community of mammals responds to environmental and disturbance gradients at relatively long time scales (Kelly *et al.* 2012; Kelly *et al.* 2013) little is known about short-term (daily) patterns of mammal activity. Drawing from ecological theory, the literature, and my experience in the field, I generated seven hypotheses about the relationships between small mammal capture rates and meteorological conditions on the night of trapping (Table 2.1). I tested these hypotheses using field surveys completed during warmer months (October-March) at 267 sites across an area of 104 000 km^2. Specifically, I explored the role of minimum temperature, relative humidity, wind speed, rainfall, cloud cover, moon illumination and survey night on small mammal capture rates (Table 2.1).

Table 2.1: Hypotheses developed and tested to determine the influence of meteorological conditions on daily capture rates (i.e. activity) of small mammals during warmer months of the year. Unless specified, hypotheses relate to all groups of small mammals. Examples from the literature highlight empirical studies that offer some support for each hypothesis. Abbreviations: Pred = relationship predicted; Lit. = examples of supporting literature; Neg.= negative relationship; Pos.= positive relationship; Neut.= no relationship.

Variable	Taxa	Pred.	Hypothesis and rationale	Lit.
Minimum temperature	Dasyurids Burramyids	Neg.	Dasyurid and burramyid activity will decline during low overnight temperatures because these marsupials will enter torpor to avoid cold conditions.	Read 1988; Ruf and Geiser 2015
	Rodents	Pos.	Native rodent activity will peak during low overnight temperatures, within warmer months, as native rodents appear to not use torpor and must forage to maintain their metabolisms when the temperature is low.	Vickery and Bider 1981; Wrobel and Bogdziewicz 2015; Tomlinson *et al.* 2007)
Relative humidity	All	Neg.	Small mammal activity will decrease with high relative humidity as predators have increased olfactory perception, therefore increasing predation risk.	Van Hensbergen and Martin 1993
Wind speed	Burramyids	Pos.	Burramyid activity will increase when wind speed is high as they are forced to move from the canopy to the ground.	Jacques et al. 2017
	Rodents Dasyurids	Pos.	Small mammal activity will increase when wind speed is high as the noise will mask the sound of small mammal movements and decrease predation risk.	Van Hensbergen and Martin 1993
Rainfall	All	Neg.	Small mammal activity will decrease with occurrence of precipitation. Wet conditions are adverse for small mammals that are not adapted to deal with rain (which is sparse and irregular in this system). They will avoid these conditions.	Read, 1988; Stokes et al. 2001
Cloud cover	All	Pos.	Small mammal activity will increase with higher cloud cover as predation risk is lower due to reduced visibility.	Stokes et al. 2001; Spence-Bailey et al. 2010
Moon illumination	All	Neg.	Small mammal activity will decrease with high moon illumination, due to the illumination increasing predation risk	Prugh and Golden 2014; Read et al. 2015
Survey night	All	Neg.	Small mammal activity will decrease on later trap nights in a survey because animals learn trap avoidance strategies.	Pavey et al. 2014; Greenville et al. 2016

Methods

Study area and small mammal surveys

This study was conducted in eucalypt dominated 'tree mallee' vegetation in the Murray Mallee region of semi-arid south-eastern Australia. This region covers 104 000 km^2 and contains large nature reserves managed primarily for biodiversity conservation. Mallee vegetation is dominated by multi-stemmed *Eucalyptus* trees, growing from a basal lignotuber. There are three main vegetation types: chenopod mallee, triodia mallee and heathy mallee (Haslem *et al.* 2010). Dunes and swales provide a small amount of topographic variation. Rainfall is variable and increases along a north-south gradient, from 161 mm in the north to 367 mm in the south. The mean maximum January temperature varies from 32 °C in the south to 36 °C in the north. Minimum temperatures vary from 15-19 °C in summer to 4-6 °C in winter.

Between 2006-2011, extensive small mammal surveys were undertaken in the Murray Mallee region in 27 landscapes across Victoria, New South Wales and South Australia (Kelly *et al.* 2012). Landscapes were selected to represent a variety of fire histories, with each landscape comprising ten sites (see Kelly *et al.* 2012 for more details on site selection). Within 26 of the landscapes, trapping surveys were conducted four times at each site; in late October-November 2006 (spring), January-February 2007 (summer), late October-November 2007 (spring) and January-February 2008 (summer). Additional surveys were conducted once in 2011 within two of the landscapes in Tarawi Nature Reserve in late October 2011 (spring) and twice within four landscapes in Murray Sunset National Park in Victoria in November 2011 (spring) and February-March 2012 (summer). The latter (2011-12) Murray Sunset surveys used three of the original landscapes but also established one additional study landscape, consisting of seven individual sites (Table S2.1.1, Appendix S2.1).

Each site consisted of ten 20 L pitfall buckets, spaced 5 m apart and connected by a 30 cm high flywire drift-fence. The base of buckets contained sand, cloth and PVC pipe to provide shelter for captured animals. Trapping was conducted for five continuous nights at each site, with trap checks each morning. Small mammals were identified to species level, sexed, weighed, assessed for reproductive condition, and marked by clipping a small patch of fur to identify recaptures within the same trapping session (Kelly *et al.* 2011).

Explanatory variables

Weather data was sourced from the Australian Bureau of Meteorology (Bureau of Meteorology 2018). I identified the closest permanent weather station to each mosaic using a nearest-neighbour analysis in ArcGIS v10.3.1 (ESRI 2014; Table S2.1.2, Appendix S2.1). I chose to use weather station data over mapped products because it was available at a finer temporal accuracy, including three-hourly synoptic measurements of several variables. Not every weather station recorded all weather variables. If a variable was not recorded at a weather station, I sourced it from the closest site within 120 km (Table S2.1.2). Weather station variables obtained were precipitation in the previous 24 hours (prior to 12 a.m.), minimum daily temperature, relative humidity at 9 p.m., cloud cover at 9 p.m. and wind speed at 9 p.m. I selected 9 p.m. for the synoptic weather variables because it was the first nocturnal measurement each night and likely to coincide with the time that nocturnal mammals are becoming active. Minimum temperature was used as all small mammals in the study area are nocturnal, and therefore more likely to be active during the period coinciding with minimum temperature (rather than maximum). I calculated Spearman rank correlations for each of the synoptic weather measurements (wind speed, cloud cover and relative humidity) at 9 p.m., 12 a.m. and 3 a.m. Each of the synoptic measurements on a given night were >80% correlated, indicating that 9 p.m. measurements are representative of weather conditions throughout the night. Cloud cover was measured as a fraction of the total sky covered in cloud, in eighths (Bureau of Meteorology 2018). I sourced moon illumination data from the U.S. Naval Observatory's Astronomical Applications Department (USNO 2017), specific to the study area. Moon illumination is the fraction of the Moon's visible disk that is illuminated between 1700 and 2100 and provides a quantitative measure of moon phase (USNO 2017).

I assessed correlations between predictor variables using Spearman rank correlations. All explanatory variables were <50% correlated and I interpreted this as independent enough to include in the same model. The night of trapping (one to six nights) was also included in the analysis as capture rates of small mammals tend to decrease during pitfall trap surveys, either because individuals learn trap avoidance or because these animals are highly mobile and move away from a trapping site (Pavey *et al.* 2014; Read *et al.* 2015; Greenville *et al.* 2016a).

Statistical analysis

I collated daily trapping data for each landscape mosaic (n = 27) and used this to calculate daily captures of individual species and taxonomic groups (the response variables). The response variable was therefore the daily trapping rate of a species in each landscape. Four species, *Cercartetus concinnus* (western pygmy possum), *Ningaui yvonneae* (mallee ningaui), *Sminthopsis murina* (common dunnart) and *Mus musculus* (house mouse; Table S2.2.2, Appendix S2.2) had enough trapping data to explore species-specific models across the whole study area. The distribution of *Pseudomys bolami* (Bolam's mouse) is restricted to the north-west of the study area, so for this species I restricted analysis to mosaics within the species' current range (mosaics 13-17 and 19-24). Captures of *Notomys mitchellii* (Mitchell's hopping mouse) and *Cercartetus lepidus* (little pygmy possum) were insufficient to build species-specific models but were included in family-level analysis. I combined trapping data for families (rodents, dasyurids and burramyids) to explore the influence of weather for groups of several species.

I used Generalised Linear Mixed Models (GLMMs) to quantify the influence of weather and moon on daily capture rates of small mammals. The response data included many zeros, so I explored two different types of GLMMs in R v3.6.1 (R Core Team 2019): a binomial model and a zero-inflated binomial model. First, I fitted binomial GLMMs using the package 'glmmTMB' (Brooks *et al.* 2017). The response variable was expressed as a two-column matrix with the nightly captures of a species in one column and daily trap effort (the number of open buckets in a landscape on each night of trapping) in the other column. Each model had seven fixed effects: precipitation, minimum temperature, relative humidity, cloud cover, wind speed, moon illumination and night of trapping. Each landscape was surveyed multiple times, so landscape was included as a random effect to incorporate these repeated measures. All explanatory variables were treated as continuous and standardised for analysis following Gelman and Hill (2007) by subtracting the mean from each predictor and dividing by two standard deviations. I analysed model residuals using the DHARMa package (Hartig 2017), which showed that the residuals of four of the eight models deviated significantly from what would be expected according to the binomial distribution. That is, model structure did not adequately capture the relationship between the response and predictor variables. To address this issue, I built a second round of models with a zero-inflated binomial distribution for four of the eight response variables, with the same fixed and random effects as the original round of models. Zero-

inflated models incorporate a structural zero model with a regular conditional model and help differentiate between structural zeros (i.e. the species can never be present at the site) and sampling zeroes (i.e. the species could be present but was not detected during the sampling period; Tyre *et al.* 2003; Brooks *et al.* 2017). Analysis of the simulated model residuals for the zero-inflated models indicated good model fit.

I extracted model coefficients for all predictor variables in the final binomial and zero-inflated binomial models. In the main text, I present the 'best' model for each species and family, defined as the model with the best fit from analysis of model residuals when comparing between a binomial and zero-inflated binomial model for an individual species or family. I explored the relationship between species capture rates and a predictor variable by considering the size and uncertainty (95% confidence intervals) of model coefficients.

Results

A total of 1605 small mammals were recorded, from eight species, during 54492 trap nights. Capture rates on a given night varied from 0.00% (observed on 129 occasions) to 39% (during a 'boom' in *M. musculus* activity in October 2011). Total capture rates for each species were: *N. yvonneae* 1.25% (*n* = 679 captures), *S. murina* 0.57% (n = 309), *M. musculus* 0.56% (n = 303), *C. concinnus* 0.43% (n = 233), *C. lepidus* 0.07% (n = 39), *P. bolami* 0.07% (n = 39) and *N. mitchellii* 0.006% (n = 3).

Meteorological effects on small mammal capture rates

Capture rates of *N. yvonneae* increased with warmer minimum temperature (regression coefficient = 0.49 ± 0.09 SE; Table S2.3.1, Appendix S2.3), supporting hypothesis H1 for this species (Table 2.1). Similarly, capture rates of all dasyurids combined increased with warmer temperature (coefficient = 0.38 ± 0.08 SE). By contrast, capture rates of *P. bolami*, a rodent, decreased with warmer temperature (coefficient = -1.13 ± 0.48 SE; Figure 2.1) and captures rates of all rodents combined also decreased (coefficient= -0.47 ± 0.21 SE), which supports hypothesis H1 for this group (Table 2.1). I did not detect an effect of minimum temperature for any of the other species, or the burramyid family.

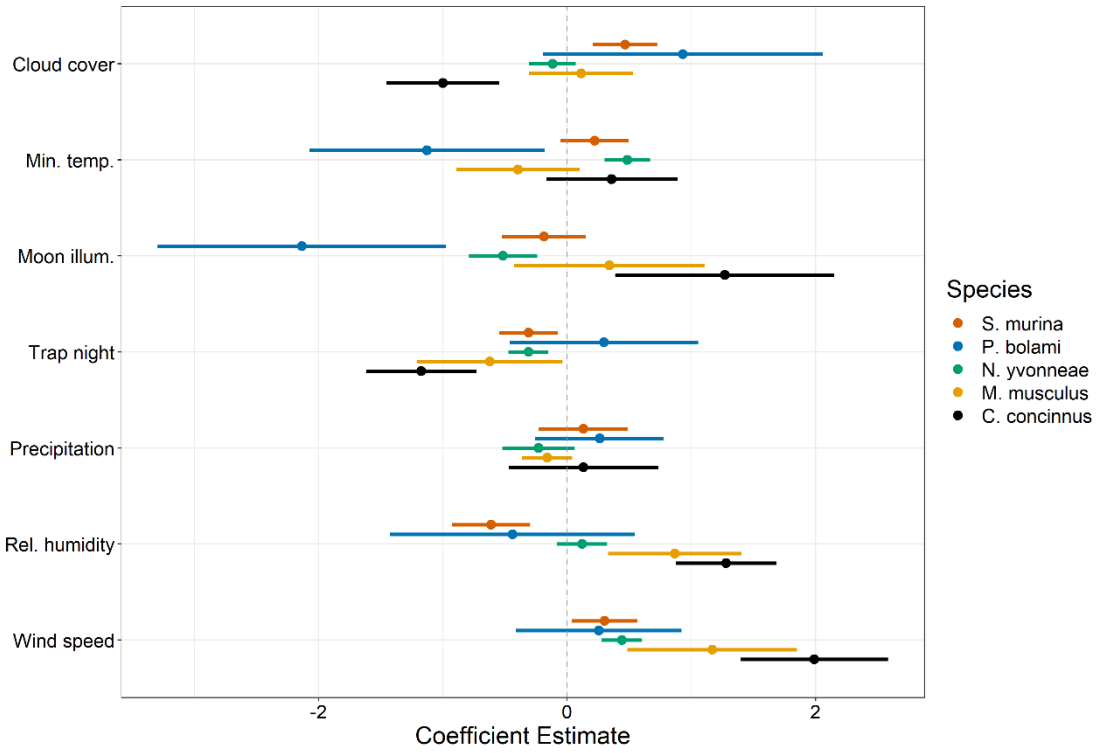

Figure 2.1: Standardised model coefficients (with 95% CIs) of the seven predictor variables in the individual species models

Capture rates of *M. musculus* and *C. concinnus* increased with higher relative humidity (coefficients = 0.87 ± 0.27 SE and 1.28 ± 0.21, respectively), contrary to hypothesis H2. Changes in relative humidity increased the capture rate of *C. concinnus* from approximately 0% at zero relative humidity to >1.5% at 100% relative humidity. Capture rates of burramyids and rodents also increased with relative humidity (Figure 2.2). I did not detect an effect of relative humidity on the dasyurid *N. yvonneae* or the dasyurid family (coefficients = 0.12 ± 0.1 SE and -0.12 ± 0.09 SE), which is contrary to the predictions in hypothesis H2. However, captures rates of the other dasyurid species *S. murina* (-0.61 ± 0.16 SE) displayed the predicted negative relationship with relative humidity. I did not detect an effect of relative humidity in the zero-inflated models for any species or families.

Wind speed was a strong explanatory variable for all species except for *P. bolami* (Figure 2.1). Capture rates of four species (*C. concinnus*, *M. musculus*, *N. yvonneae* and *S. murina* all increased with wind speed (Figure 2.1). The relationship was strongest for *C. concinnus* (coefficient= 1.99 ± 0.3 SE), with wind speeds of >20 km/h leading to a two-fold increase in capture rates of this species, relative to a wind speed of 0 km/h (Figure

S2.4.1, Appendix S2.4). Wind speed was also a strong explanatory variable for models of each family of mammals (Figure 2.2), with the positive relationship being strongest for the burramyid family model (coefficient = 1.22 ± 0.29 SE).

I was unable to detect an effect of precipitation in the 24 hours prior to trapping on the capture rate of any species (Figure 2.1) or families (Figure 2.2), contrary to the hypothesis that capture rates of small mammals would decrease with high precipitation. In part, this is because rainfall was very low during the mammal surveys, with most nights recording no precipitation (Table S2.2.1 and Figure S2.2.1, Appendix S2.2)

The relationship between cloud cover and the capture rate of most species and families was not consistent with the expectation that small mammal capture rates would increase with high cloud cover. Only the capture rate of *Sminthopsis murina* had the predicted positive relationship with cloud cover (coefficient = 0.47 ± 0.13 SE). The capture rates of burramyids and *C. concinnus* decreased with cloud cover (coefficient = -0.59 ± 0.22 SE and -1 ± 0.23 SE, respectively). I was unable to detect an effect of cloud cover on the capture rates of dasyurids or rodents (Figure 2.2), on the capture rates of the other species (Figure 2.1) or in any of the zero-inflated models (Table S2.3.1).

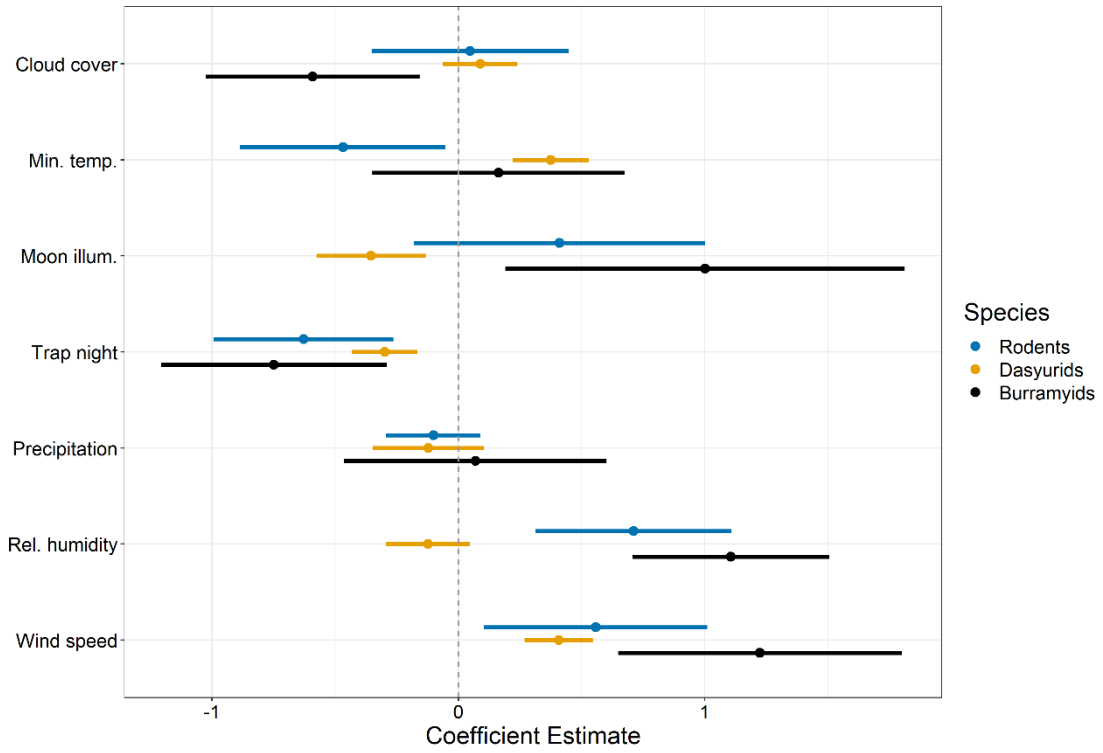

Figure 2.2: Standardised model coefficients (with 95% CIs) of the seven predictor variables in the family models

Capture rates of *P. bolami* and *N. yvonneae* decreased with moon illumination (coefficients = -2.13 ± 0.59 SE and -0.52 ± 0.14 SE; Figure 2.1). However, the capture rates of *C. concinnus* and the burramyid family showed strong positive relationships with moon illumination in both the conditional (coefficients = 1.27 ± 0.45 SE and 1 ± 0.41 SE, respectively) and zero-inflated models (coefficients = 1.52 ± 0.56 SE and 1.18 ± 0.44 SE, respectively). This indicates they are more likely to be captured, and in higher abundances, when the moon illumination is high.

As predicted, captures rates of most species (Figure 2.1) and families (Figure 2.2) had a negative relationship with trap night (4 of 5 species, and all three families), indicating that more small mammals are likely to be captured earlier in a survey period.

Discussion

Capture rates of small mammals in semi-arid Australia are strongly influenced by factors that change daily, such as weather and moon phase. Below, I discuss my results in relation to each of the hypotheses developed and tested.

Weather and small mammals

I hypothesised that rodents would be more active at low temperatures, as they must forage nightly to maintain their metabolisms and would have higher energetic costs on cold nights. The model for *P. bolami* and the rodent family model support this prediction, but the model for *M. musculus* did not show a relationship with temperature. One explanation for this is that *M. musculus* is a species that exhibits large, rapid changes in population size in response to environmental conditions, including after periods of high rainfall at time lags of ~6 months (Kelly *et al.* 2013). It is an invasive species that also occupies adjacent agricultural croplands (wheat) in our study area and population changes in the croplands can have flow-on effects in adjacent mallee landscapes. *Mus musculus* may not show a short-term relationship with minimum temperature because large changes in capture rate (from 1% - 33%) are more likely to reflect changes in population size over changes in trapping conditions. When considering the marsupials in this study, only one species, *N. yvonneae*, responded to minimum temperature with the negative relationship I predicted (due to their ability to avoid cold weather by entering torpor). The other marsupials showed no relationship with this variable, possibly because they are larger than *N. yvonneae* and may be more able to thermoregulate independently of the weather,

due to a smaller surface-area-to-volume-ratio (Buckley *et al.* 2018). Finally, minimum overnight temperature varies with season; however, surveys for this study were only completed in warmer months. There is substantial evidence to suggest that activity patterns and detectability of small mammals are seasonally dependent (Gibson 2011), so future research is needed to determine if these relationships with minimum temperature are consistent throughout the entire year.

I predicted that relative humidity would decrease the capture rates of small mammals, however rodents and burramyids had strong positive relationships with relative humidity. Previous work in South Africa indicated that capture rates of small mammals decline with high relative humidity (Van Hensbergen and Martin 1993). They speculated that high relative humidity increases the olfactory abilities of predators, leading to increased predation risk (Van Hensbergen and Martin 1993). If humid conditions enhance the olfactory perceptions of predators, the same may be true of small mammals like rodents and burramyids, enhancing their foraging success. In comparison to reducing predation risk, this may be of limited benefit in a resource-rich environment, such as south-west South Africa where the previous study took place. However, the Murray Mallee is a far drier area with fewer resources and the benefits of an increased sense of smell for foraging may outweigh predation risks, leading to increased small mammal activity.

Wind speed was the biggest influence on burramyid capture rates in the models, with high winds increasing capture rates. Burramyids in this semi-arid ecosystem are semi-arboreal, foraging in the canopies of flowering trees and shrubs but using the ground to move between trees as the woodland canopy has low connectivity (Morrant and Petit 2012). High winds could make the trees less habitable to the burramyids, forcing them to move down to the ground to avoid rough conditions in the trees and increasing the chances they will be captured in terrestrial pitfall traps. My modelling results suggest this is the case, as does the raw trapping data for the most common burramyid in our study, *C. concinnus*. The highest *C. concinnus* capture rate was 8.3% (50 individuals across six landscapes), when wind speed ranged from 16.6-18.4 km/h at the different study landscapes. The next highest *C. concinnus* capture rate was 4.5%, when wind speeds were also 18.4 km/h. These values are well above the mean and median wind speeds of 11.85 km/h and 12.10 km/h, respectively, and the mean capture rate of 0.43% for this species. High winds may also be beneficial to small mammals by influencing their predation risk, providing background noise that masks the auditory cues of predators and reduces predation risk

(Van Hensbergen and Martin 1993). Conversely, high winds also mask the ability of small mammals to detect predators and therefore net predation risk is difficult to determine, particularly as it is likely to vary between different types of predators. For example, native predators such as owls are largely silent hunters and the ability of small mammals to detect owls would not be affected by high winds, while the ability of owls to hunt would be hampered. In contrast, introduced predators such as red foxes are less silent and might have increased hunting success with high winds. In this study, wind speed had a positive influence on the capture rates of all small mammals except *P. bolami*, indicating that being active during high winds may benefit small mammals by reducing predation risk.

I was unable to detect an effect of precipitation on any of the species or families modelled, which was not expected. The variability of precipitation during these surveys was very low, with most nights recording zero precipitation. On the three wettest nights in this study (precipitation of 21.2 mm, 18.8 mm and 12.8 mm), capture rates of all small mammals were 5.83%, 21.51% and 1.37%, in each case differing from the mean capture rate across all nights of trapping of 3.25%. During the two nights with highest precipitation there was an increase in trapping success, opposite to what I expected. These two nights are from surveys conducted during a high rainfall period in 2010-2012, and it is known from previous research in semi-arid and arid ecosystems that many small mammal populations increase following periods of high rainfall that boost resources (Greenville *et al.* 2012; Kelly *et al.* 2013; Pavey *et al.* 2014). The relationship between capture rates of small mammals and nightly precipitation remains unclear. It is possible that rainfall also has a lag effect on small mammal activity, reducing activity during rainfall but increasing activity on subsequent nights due to increases in relative humidity and subsequent foraging efficiency. Investigation of the movement and behaviour of individual animals is needed to determine the immediate and short-term influence of rainfall on capture rates.

Only one species in my study, *C. concinnus*, had a relationship with cloud cover, with high cloud cover reducing capture rates of this species. I expected cloud cover to have a positive influence on capture rates of small mammals, due to high cover reducing visibility and therefore predation risk. *Cercartetus concinnus* is semi-arboreal and predation risk in trees (i.e. a relatively enclosed microhabitat) is likely to be lower than on the ground (i.e. an open microhabitat; Doherty *et al.* 2015). There may be another advantage (such as visibility for foraging) associated with low cloud cover that may

outweigh potential risks of predation for this species. It is also possible that cloud cover and moon illumination interact to influence visibility of small mammal and predators, which should be explored in future work. I note that the cloud cover data was only available from one weather station (Mildura Airport) and I used this data for the entire region. More comprehensive data on cloud cover across the whole region would likely reduce the uncertainty in modelled results relating to this variable. Nevertheless, cloud cover was an influential variable for capture rates of diurnal mallee reptiles over the same trapping period (Spence-Bailey *et al.* 2010). This indicates that the available data can pick up relationships between this variable and mammal capture rates, even if there is some uncertainty relating to local conditions.

Moon phase and small mammals

My results show that small mammals have differing responses to moon illumination, which contrasts with the prediction that high moon illumination would reduce the activity of all small mammals due to high visibility increasing predation risk. In a global meta-analysis of the effect of moon phase on small mammal activity, moonlight was shown to generally decrease the activity of small mammals (Prugh and Golden 2014). Based on this general trend of moon avoidance, I hypothesised that capture rates of mallee small mammals would be higher at the new moon. Like Prugh and Golden (2014), however, my results were not consistent across taxa. Contrary to the finding of a positive relationship by Read *et al.* (2015), the common dunnart did not have a relationship with moon illumination in the present study. Capture rates of some rodents and insectivores, specifically Bolam's mouse, mallee ningaui and the dasyurid family showed the predicted negative responses, while the capture rate of pygmy possums had a positive relationship with moon illumination. These species have differing habitat preferences, which may explain some of the differences in the results. For instance, *P. bolami* is a species that prefers relatively open vegetation in mallee environments (Kelly unpublished data). Relatively open areas within semi-arid environments have a higher predation risk for small mammals (Doherty *et al.* 2015b) and therefore there is a benefit for *P. bolami* to reduce activity on nights with high moon illumination. In contrast, as discussed above, *C. concinnus* occupies more sheltered microhabitats, with lower predation risk, and therefore foraging on bright nights is less risky for this species. Finally, predation risk may differ across the region, as dingoes (an apex predator) are primarily found in the north of the study area and may influence predation rates on small mammals by mesopredators, such

as *Vulpes vulpes* and *Felis catus*, when present in a study landscape (Geary *et al.* 2018; Hunter *et al.* 2018).

Implications for field surveys

Small mammals in the Murray Mallee are cryptic and occur at low densities, making them difficult to detect during field surveys. In addition to the influence of daily meteorological conditions, populations are influenced in the longer term by rainfall, fire history and habitat, all of which influence capture rates in trapping surveys (Kelly *et al.* 2012; Kelly *et al.* 2013). In the present study mean trap success was ~3% (ranging from 0% - 39%). Recaptures of the same individuals within each trapping session were also low, consistent with results from other arid ecosystems in Australia (Dickman *et al.* 1995; Pavey *et al.* 2014). This could be due to learned trap avoidance by individuals, but semi-arid and arid zone small mammals are also highly mobile and capable of moving distances greater than 1km in a single night (Dickman *et al.* 1995; Bos and Carthew 2007; Morrant and Petit 2012). Although overall capture rates drop with consecutive nights of pitfall trapping, continuing to trap for 4-5 nights increases the chances of capturing these highly mobile individuals and of detecting all species present in an area. My results show that inclement weather can more than halve capture rates, so consecutive trapping nights are needed to ensure the survey is representative of the small mammal community present at a trapping site. Additionally, as these surveys were only completed in spring and summer, future research could consider the influence of seasonality on small mammal activity and whether surveys can be further optimised by trapping at specific times of the year.

In conclusion, responses of mallee small mammals to meteorological conditions vary between species and across taxonomic groups. I recommend that surveys undertaken in warmer months should encompass a variety of meteorological conditions to increase capture rates of small mammals and provide a representative sample of the small mammal community present in a landscape.

Chapter 3 – Predicting mammal responses to pyrodiversity: from microbats to macropods

Preface

This chapter is currently under revision following review in *Biological Conservation*

Abstract

Fire has shaped Australia's diverse mammal fauna for millennia. However, ongoing changes to fire regimes threaten native mammal populations, and a significant conservation challenge is to understand and promote desirable forms of pyrodiversity (variation in fire regimes). A way forward is to quantify how different aspects of pyrodiversity influence whole mammal assemblages and produce dynamic maps of species distributions to inform conservation. I aimed to determine and map how spatial and temporal variation in fire regimes correlates with a diverse mammal assemblage comprising macropods, microbats, rodents, small marsupials and a monotreme. I built species distribution models for 17 species against fire, climate and environmental covariates in fire-prone woodlands of semi-arid Australia. Spatial measures of fire included the area, diversity and configuration of landscape elements, and temporal measures included time since fire and fire frequency. Native mammals showed a variety of responses to pyrodiversity. Microbats were more likely to occur as time since fire increased, whereas rodents were positively correlated with recently burned areas. Occurrence of small dasyurid marsupials increased with the area of older post-fire age classes, while western grey kangaroo occurrence was positively associated with high diversity of post-fire ages. This new approach, using predictive models to map mammal distributions in relation to spatial *and* temporal variation in fire regimes, provides outputs that managers can use to improve conservation planning. This enables the positive and negative effects of fire to be better understood and will assist in achieving desirable forms of pyrodiversity that meet the needs of whole mammal assemblages

Introduction

Fire has influenced Australia's unique and globally significant mammal fauna for millennia (Geiser *et al.* 2017). However, ongoing changes to fire regimes are one of the biggest threats to native mammals across the continent (Woinarski *et al.* 2015a). Understanding of how variation in fire regimes (pyrodiversity) shapes the distribution and abundance of mammals in Australia, and globally, is growing (Fuhlendorf *et al.* 2010; Kelly *et al.* 2012; Beale *et al.* 2018; Davies *et al.* 2018; Puig-Gironès *et al.* 2018). Yet, studies that explore the influence of pyrodiversity on whole mammal assemblages – from big to small and volant to non-volant species – remain rare. Moreover, most research to date is not readily available in forms that are easily translated into conservation planning and fire management. Using predictive models to map species distributions in response to spatial and temporal variation in fire regimes, and other environmental gradients, provides opportunities for improved understanding and management of pyrodiversity at scales directly relevant to decision-making.

Temporal variation in fire regimes shapes the distribution and abundance of mammals. For example, time since the last fire influences the availability of ground cover, which then impacts dasyurid marsupials that prefer vegetation that has remained unburnt for years to decades (Kelly *et al.* 2011; Hale *et al.* 2016; Greenville *et al.* 2016a). Other species, including some native rodents (Pereoglou *et al.* 2016) and grazing macropods (Ritchie *et al.* 2008), prefer recently burnt vegetation and benefit from a post-fire pulse of herbs and grasses. Fire frequency is also important but remains understudied. For example, frequent canopy fires reduce the availability of key resources such as tree hollows (Haslem *et al.* 2012). This impacts mammals that nest in hollows such as pygmy possums (Turner 2020), gliding possums (Lindenmayer *et al.* 2008a) and microbats (Lumsden and Bennet 1995). Detailed mapping of fires provides new opportunities to explore how fire frequency influences mammal assemblages.

Fire regimes vary spatially (Faivre *et al.* 2011), with the area, configuration and spatial diversity of fires having a strong influence on mammal distributions. Recent work shows large areas of long-unburnt vegetation are disproportionately important for the survival of small mammals in semi-arid woodlands (Kelly *et al.* 2012), temperate forests (Lindenmayer *et al.* 2013) and tropical savannas (Lawes *et al.* 2015). The configuration of fire-driven habitat is also important: for example, proximity to unburnt vegetation (i.e.

configuration) aids recolonization of burnt areas by microbats (Law *et al.* 2018). Spatial diversity of fires generates a range of habitats that promote relative abundance and diversity of tropical mammals in northern Australia (Davies *et al.* 2018). The influence of these spatial characteristics depends on the life history of mammals (Nimmo *et al.* 2019). For example, large, mobile macropods such as kangaroos have been observed moving between adjacent burnt and unburnt areas to access different food and shelter resources (Murphy and Bowman 2007). In contrast, less mobile mammals such as small rodents and marsupials rely on unburnt refuges to survive in burnt landscapes (Banks *et al.* 2011).

The concept of pyrodiversity combines variation in spatial and temporal dimensions of fire regimes (Martin and Sapsis 1991). While much work to date has framed animal relationships with pyrodiversity by asking the question 'does pyrodiversity promote biodiversity?' a more nuanced approach is emerging that recognises that there are many different forms of pyrodiversity (Parr and Andersen 2006; Kelly *et al.* 2017a; Gillson *et al.* 2019). Pyrodiversity relationships can be dependent on other environmental gradients and events, such as rainfall (Greenville *et al.* 2016a; Beale *et al.* 2018), so robust models are needed to map these relationships across landscapes. Species distribution models (SDMs) are an important tool in conservation science (Elith *et al.* 2011) and incorporating dynamic measures of pyrodiversity into SDMs is likely to improve mapping of where and when mammals occur (Coops and Catling 2002).

I explored how variation in fire regime characteristics influences mammal distributions using data from semi-arid woodlands of south-eastern Australia. The region has a diverse mammal fauna including small insectivorous marsupials (dasyurids), nectarivorous pygmy possums (burramyids), granivorous rodents (such as hopping mice), large herbivorous macropods, insectivorous microbats and the insectivorous echidna. Semi-arid woodlands have a legacy of mammal decline and extinction (Bennett *et al.* 2006) and 15 species are listed as threatened under state or federal legislation. I modelled the response of this mammal assemblage to environmental gradients and pyrodiversity to ask: 1) What is the influence of temporal fire regime characteristics, including time since fire and fire frequency, and spatial characteristics, including amount, diversity and configuration of post-fire ages on mammal occurrence? 2) How do pyrodiversity and other environmental gradients combine to influence the occurrence of each mammal species in space and time? 3) Do responses differ between taxa?

Methods

Study area

The study area was the Murray Mallee region (32.8‑35.2° S, 139.7‑144.1° E; Figure 3.1), which covers 104 000 km^2 of south-eastern Australia and encompasses large areas managed primarily for biodiversity conservation. The landscape is low elevation (<100 m a. s. l.), with an undulating dune and swale system. Mean annual rainfall increases from 160 mm in the north to 360 mm in the south. The vegetation is characterised by stands of 'tree mallee', multi-stemmed *Eucalyptus* species. Three main types of mallee vegetation have been mapped across the region: triodia mallee (TM), heathy mallee (HM) and chenopod mallee (CM; Haslem et al., 2010). Wildfires greater than 100 000 ha occur in the region every 10-20 years, although longer fire intervals are common for any single location and smaller fire events occur more frequently (Clarke *et al.* 2010). Fire is actively managed through suppression and planned burning, to achieve a variety of socio-economic and ecological goals.

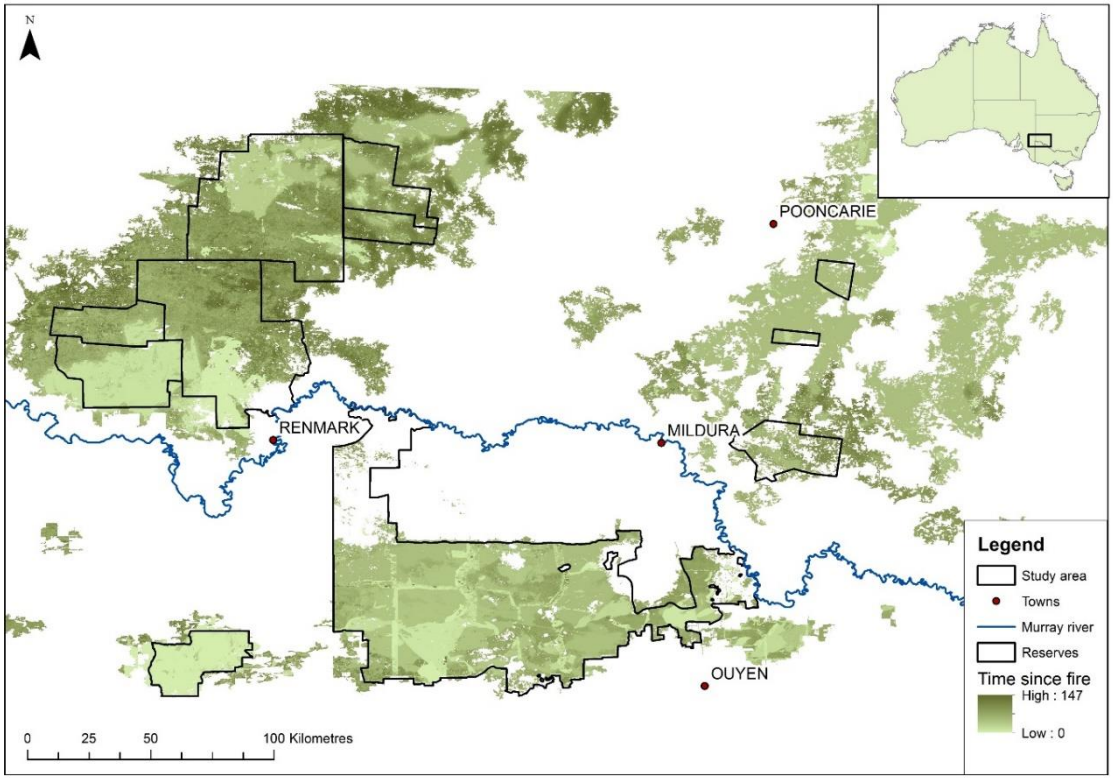

Figure 3.1: Extent of the 'tree mallee' vegetation type in the Murray Mallee region, showing the fire history to 2016. This map, and study, excludes tree mallee vegetation within reserves ≤1000 ha and continuous tracts of mallee vegetation outside of the reserve system ≤5000 ha in size.

Species occurrence data

I used two sources of occurrence data: data from biodiversity atlases and data from systematic field surveys. First, mammal occurrence data from 1975 to 2016 were sourced from the Atlas of Living Australia (ALA 2011) for all mammal species (n = 26) occurring within tree mallee vegetation in reserves ≥1000 ha and continuous tracts of mallee vegetation ≥5000 ha. To minimise inaccuracies in the atlas data I only included records with spatial accuracy of ≤100 m. Most records meeting this criterion were from systematic surveys by expert sources, including universities and government agencies. Second, I included records of small mammals from extensive surveys conducted within mallee woodlands from 2006 to 2008 and 2011 to 2012 (see Kelly *et al.* 2012; Kelly *et al.* 2013). I combined and filtered the occurrence data to retain one record per species per year per 100 x 100 m grid cell (the scale of the environmental data used in modelling).

Environmental data

I used eight dynamic fire variables (i.e. fire regime characteristics) and three static environmental variables to predict species distributions (Table 3.1). All data were converted to raster geoTIFFs at 1 ha spatial scale and all calculations were undertaken in ArcGIS v10.3.1 (ESRI 2014) unless otherwise stated. Static variables included vegetation type (Haslem *et al.* 2010), mean annual rainfall created with ANUCLIM (Houlder *et al.* 2001) applied to a 30 m Shuttle Radar Topography Mission Digital Elevation Model (SRTM DEM), and elevation derived from a 1-second SRTM DEM (GeoScience Australia, 2011). I used elevation to indicate position in the landscape. Mean annual rainfall represented a regional aridity gradient previously found to be important in structuring mallee small mammal communities (Kelly *et al.* 2012).

Fire history data were compiled from several sources to obtain fire histories for the whole region from 1900 to 2016. From 1972 to 2011 I used annual fire scars mapped from satellite images by Avitabile et al. (2013) and from 2011 to 2016 annual fire mapping provided by the NSW National Parks and Wildlife Service, the Victorian Department of Environment, Land, Water and Planning, and sourced from the South Australian Department for Environment and Water (2016). For fire history prior to 1972 I used predictive modelling by Callister et al. (2016) that was truncated at 1900. This modelling extrapolates time since the last fire across the whole region based on relationships between remotely sensed images and field survey data from sites of known fire history

(Callister *et al.* 2016). Fire data were merged to create a single, composite layer of fire history. Time since fire, fire frequency and the six spatial context variables (discussed below) were calculated from a unique fire history layer for each year from 1975-2016, using all fire history data back to 1900. For example, to calculate time since fire in 2016, areas burnt in 2016 were assigned a value of 0 and areas not burned since 1923 were assigned a value of 93. Fire frequency was only able to be calculated using fire history data from 1972 onwards (i.e. the start of satellite data) because predictive modelling only shows the most recent fire (not all previous fires). Fire frequency for each year was therefore calculated as the number of fires from 1972 to that year; areas with no fires in this period were assigned a frequency of 0.

I used three landscape metrics as spatial measures of fire: area (amount) of post-fire age classes, diversity of post-fire age classes, and configuration of post-fire age classes. These were calculated within a specific buffer distance (see below) of a mammal record to represent local landscape conditions. Configuration was measured using Li and Reynolds (1993) contagion index, which measures both how prevalent different patch types (i.e. older and younger post-fire age classes) are within a landscape and how interspersed the patch types are with one another. This index varies from 0-100 for each patch type, with 100 representing a homogenous landscape made up of only the single patch type of interest. Landscape metrics were calculated from post-fire age class rasters classified as early (0-10 years), mid (11-35 years) and old (>35 years). A buffer of 9 x 9 cells (~1 km^2, hereafter 'small buffer') and a buffer of 19 x 19 cells (~3km^2, hereafter 'large buffer') were chosen to represent the spatial context of fire at ecologically meaningful scales. Small buffers were used for small marsupials and rodents, while large buffers were used for microbats, macropods and the echidna which roam further and access a larger area of resources.

I created two area variables: the area (ha) of early post-fire ages in a large buffer and the area of mid-old triodia mallee in a small buffer. The area of mid-old triodia mallee (>10 years post-fire), which often includes high cover of *Triodia scariosa* (hummock grass), was hypothesised to be important for small marsupials and rodents. In models for microbats, macropods and the echidna, I included the area of early post-fire ages, of all vegetation types (Table 3.1). Shannon's diversity of post-fire age classes was calculated in GRASS GIS v7.2.2 (GRASS Development Team, 2017) at small and large buffer sizes. Configuration of mid-old triodia mallee in a small buffer (for small mammals) and the

configuration of early post-fire ages in a large buffer (all other taxa) was calculated in FRAGSTATS v4 (McGarigal *et al.* 2012).

Species distribution models

I developed models for mammal species with ≥19 presence records, located within tree mallee vegetation. Seventeen species met these criteria, including six small mammals, nine microbats, a macropod and the echidna. The median number of presence records was 68 (range 19-531; Table 3.2). Only four species had <35 presence records (Table 3.2), all of which have restricted ranges within the Murray Mallee region (e.g. *Notomys mitchellii* is only found south of the Murray River).

I developed SDMs using the Maxent algorithm, which uses presence-only data (i.e. atlas data) to predict species distributions. Maxent was chosen as it is robust with relatively small sample sizes and is one of the highest-performing methods for building SDMs (Elith *et al.* 2011). I ran all models in R v3.6.1 (R Core Team 2019) using the 'dismo' package (Hijmans *et al.* 2011). Maxent builds SDMs by comparing the conditions at the location of species records against conditions at random, 'background' points. Presence-only data is often prone to sampling bias as survey effort is unknown; common biases include survey distance from roads and survey year (Elith *et al.* 2011). To overcome sampling bias, I created a stratified random background of 10 000 points for each taxonomic group (small mammals, microbats etc.) that was weighted by the survey year and distance from roads (in 100 m blocks) in the presence data of each taxonomic group.

Several pairs of fire variables were highly correlated (Spearman correlation >0.8), including time since fire and fire frequency, and amount and contagion. I therefore developed three separate models for each species (models M1, M2 and M3) that incorporated all the fire variables of interest but avoided including correlated variables in the same model (Table 3.1). All models had six predictor variables, including annual rainfall, vegetation type, elevation and three fire variables (Table 3.1).

I built each model using occurrence data for each species (n = 17) against a stratified random background built for each taxonomic group (n = 4), using five-fold cross-validation without spatial segregation to estimate predictive performance on held-out data. To reduce model complexity and improve model interpretation, I excluded threshold and hinge features (Elith *et al.* 2011). I assessed models using cross-validated area under the curve (AUC), with models with AUC >0.7 showing reasonable discrimination

between presence and background points, and visual assessment of response curves and maps. I assessed the relative influence of explanatory variables by comparing cross-validated variable contribution, which is the amount each variable contributed to the gain of the model during training, converted to a percentage. I predicted each species distribution against environmental conditions in 2016, using 'cloglog' outputs to measure relative likelihood of occurrence from 0 to 1 (Phillips *et al.* 2017).

Table 3.1: Model sets built for each group of mammals. In addition to the listed variables describing fire regime characteristics, each model contained mean annual rainfall, elevation and vegetation type.

Model	Taxa	Fire variables
M1	Small marsupials and native rodents	Time since fire + diversity of post-fire age classes + area of mid-old triodia age class
	Microbats, western grey kangaroo and short-beaked echidna	Time since fire + diversity of post-fire age classes + area of early post-fire age class
M2	Small marsupials and native rodents	Time since fire + diversity of post-fire age classes + configuration of mid-old triodia age class
	Microbats, western grey kangaroo and short-beaked echidna	Time since fire + diversity of post-fire age classes + configuration of early post-fire age class
M3	Small marsupials and native rodents	Fire frequency + diversity of post-fire age classes + area of mid-old triodia age class
	Microbats, western grey kangaroo and short-beaked echidna	Fire frequency + diversity of post-fire age classes + area of early post-fire age class

Results

I compared three different model sets for each of the 17 mammal species, including six small mammals, nine microbats, *Macropus fuliginosus* (western grey kangaroo) and *Tachyglossus aculeatus* (short-beaked echidna; Table 3.2). Model discrimination was good (AUC >0.70) for 13 species, but only moderate (AUC 0.60-0.69) for three microbat species and *T. aculeatus* (Table 3.2). Response and rug plots for all models and species are shown in Figures S3.2.1 – S3.2.17 in Appendix S3.2.

Time since fire and fire frequency

In models where it was included, time since fire was the strongest contributing variable for three small mammals: *Cercartetus concinnus* (western pygmy possum), *Notomys mitchellii* (Mitchell's hopping mouse) and *Sminthopsis murina* (common dunnart; variable contributions 57.7%, 40.3% and 44.5% in M1, respectively). All three species had a negative relationship with time since fire, with relative likelihood of occurrence >0.8 for *N. mitchellii* between 0-20 years post-fire, >0.7 for *C. concinnus* between 5-40 years post-fire, and >0.7 for *S. murina* between 0-30 years post-fire. When combined with other environmental gradients, predictive discrimination of models including time since fire was useful (AUC >0.7) for all three species, and high for *N. mitchellii* (AUC = 0.93). For three microbat species, *Chalinolobus gouldii* (Gould's wattled bat), *Nyctophilus geoffroyi* (lesser long-eared bat) and *Nyctophilus corbeni* (south-eastern long-eared bat), time since fire contributed >5% of total deviance. All three species had a positive relationship with time since fire (e.g. *C. gouldii*, Figure 3.3) and useful model discrimination (AUC >0.7). Time since fire was not a strong influence on several other microbat species (Table 3.1.1 and 3.1.2; Appendix 3.1).

Fire frequency was the strongest contributing variable to M2 for *S. murina* (variable contribution = 32.6%), and a moderate contributor for *C. concinnus* (variable contribution = 23.3%). Sites burnt one or two times since 1972 had higher relative likelihood of occurrence of *C. concinnus* and *S. murina*, compared to sites with no fires during this period (Figures S3.2.1 and S3.2.6, Appendix S3.2). *Notomys mitchellii* had no relationship with fire frequency (variable contribution = 0%), despite the strong relationship of this species with time since fire. Fire frequency was moderately important for the microbats *N. geoffroyi* and *Chalinolobus morio* (chocolate wattled bat; variable contribution = 19.0% and 12.7%, respectively). The occurrence of both species declined

with fire frequency, and the models had good discrimination (AUC = 0.72 and 0.92, respectively). Six species did not respond to fire frequency (Figure 3.2).

Area, configuration and diversity of post-fire age classes

Area of mid-old triodia mallee was the most important explanatory variable for *N. yvonneae* (variable contribution = 61.5% and 61.9%, M1 and M3). For this species, the relative likelihood of occurrence increased substantially if there was more than 40 ha of mid-old triodia mallee present in the 81 ha neighbourhood (Figure 3.3). Area of mid-old triodia was correlated with configuration of mid-old triodia (Pearson correlation 0.92), to which *N. yvonneae* also had a positive relationship (variable contribution = 62.1%). Model discrimination was useful in all models for *N. yvonneae* (AUC= 0.74-0.75; Table 3.2). Area of early post-fire ages was moderately important in the models for the macropod *M. fuliginosus* and the microbats *Vespadelus baverstocki* (inland forest bat) and *N. corbeni* (variable contribution = 27.1%, 18.2% and 18.0% for M1, respectively).

Table 3.2: Results of the three model sets built for each of 17 mallee mammal species, including number of presence records and cross-validated AUC values.

Taxon. group	Scientific name	Common name	No. records	AUC		
				M1	M2	M3
Small marsupial	*Cercartetus concinnus*	Western pygmy possum	202	0.77	0.77	0.73
Small marsupial	*Cercartetus lepidus*	Little pygmy possum	45	0.90	0.90	0.90
Small marsupial	*Ningaui yvonneae*	Mallee ningaui	531	0.74	0.74	0.75
Rodent	*Notomys mitchellii*	Mitchell's hopping mouse	19	0.93	0.93	0.89
Rodent	*Pseudomys bolami*	Bolam's mouse	101	0.84	0.83	0.82
Small marsupial	*Sminthopsis murina*	Common dunnart	459	0.70	0.70	0.70
Microbat	*Chalinolobus gouldii*	Gould's wattled bat	127	0.70	0.70	0.72
Microbat	*Chalinolobus morio*	Chocolate wattled bat	25	0.91	0.91	0.92
Microbat	*Chalinolobus picatus*	Little pied bat	47	0.69	0.70	0.69
Microbat	*Ozimops petersi*	Inland free-tailed bat	31	0.68	0.68	0.67
Microbat	*Nyctophilus corbeni*	South-eastern long-eared bat	68	0.80	0.79	0.79
Microbat	*Nyctophilus geoffroyi*	Lesser long-eared bat	138	0.70	0.71	0.72
Microbat	*Scotorepens balstoni*	Inland broad-nosed bat	68	0.72	0.71	0.72
Microbat	*Vespadelus baverstocki*	Inland forest bat	62	0.61	0.61	0.63
Microbat	*Vespadelus regulus*	Southern forest bat	32	0.87	0.87	0.86
Macropod	*Macropus fuliginosus*	Western grey kangaroo	171	0.71	0.71	0.71
Monotreme	*Tachyglossus aculeatus*	Short-beaked echidna	65	0.60	0.61	0.63

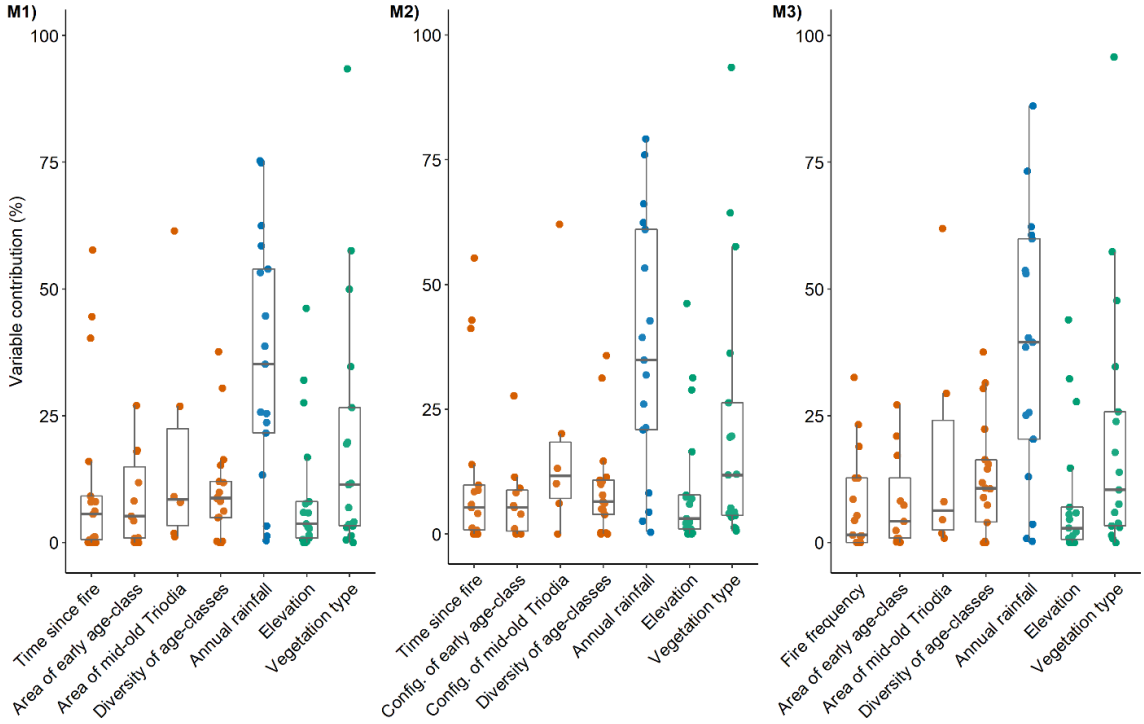

Figure 3.2: Contribution (%) of each predictor variable during training of the three model sets, for 17 mammal species. Points represent the contribution of one variable to the model of one species. Boxplots show the 25th and 75th percentiles around the median variable contribution across all 17 species, with whiskers extending to 1.5 interquartile range.

Macropus fuliginosus had a positive relationship with this variable if the area of early post-fire ages in the neighbourhood exceeded 100 ha (Figure 2), with useful model discrimination (AUC = 0.71). *Vespadelus baverstocki* and *N. corbeni* both had negative relationships with area of early post-fire ages, however model discrimination was only good for *N. corbeni* (AUC= 0.61 and 0.80, respectively).

Diversity of post-fire age classes was correlated with occurrence of most microbat species, including *S. balstoni* and *C. gouldii*, and with *M. fuliginosus* (variable contribution = 37.6%, 15.2% and 16.4%, respectively for M1). Occurrence of all microbats declined with increased diversity in post-fire age classes, except for *Vespadelus regulus* (southern forest bat) which had no relationship with this variable. Occurrence of *M. fuliginosus* increased with diversity of post-fire age classes (Figure 3.3). Interestingly, when time since fire was replaced with fire frequency, diversity contributed more to the models of several small mammals including *C. concinnus*, *N. yvonneae* and *S. murina* (variable contribution = 22.4%, 16.3%, 31.5%, respectively). Time since fire and

diversity were not correlated (Pearson correlation <0.1) and model discrimination was good (AUC >0.7; Table 3.2).

Area and configuration of early post-fire ages were correlated (Pearson correlation = 0.84), but shaped species distributions in different ways. These variables contributed similarly to the models for *M. fuliginosus* (variable contribution = 27.7%), however the species' response differed. Relative likelihood of occurrence of *M. fuliginosus* decreased between configurations of 0 and 50 and then increased when configuration was >50 (Figure S3.2.16, Appendix S3.2). For *N. geoffroyi*, the amount of early post-fire ages contributed little to the models (variable contribution = 4.3% and 2.4%, M1 and M3), however configuration of early post-fire ages was moderately important (variable contribution = 11.4%), with adequate model discrimination (AUC >0.7; Table 3.2).

Across all taxa, spatial fire variables had a median variable contribution similar to the median variable contribution of temporal fire variables (Figure 3.2).

Rainfall, topography and vegetation covariates

Mean annual rainfall was the strongest contributing predictor variable for eight species in M1, nine species in M2 and 10 species in M3, contributing 38.5% - 86.1% to models for these species (Tables S3.1.1-S3.1.3, Appendix S3.1). Mammals typically responded monotonically to rainfall and elevation (Figures S3.2.1-3.2.17, Appendix S3.2). Elevation was a strong contributing variable for four species, contributing 14.7% - 43.9% to these models. Elevation was the strongest contributing predictor variable for one species (*M. fuliginosus*) in M1 and M3 and no species in M2.

Vegetation type was the strongest contributing predictor variable for four species in all three models. These species were all microbats, *Ozimops petersi* (inland free-tailed bat), *Scotorepens balstoni* (inland broad-nosed bat), *Chalinolobus picatus* (little pied bat) and *V. baverstocki*, with variable contributions for vegetation type between 34.7% - 93.4% (Tables S3.1.1-S3.1.3). All four species had a higher likelihood of occurrence in triodia mallee. However, model discrimination was only moderate for three of the species (AUC <0.7) and only consistently good for *S. balstoni* (AUC >0.7; Table 3.2). Vegetation type was also an important contributing variable for *N. mitchellii* and *T. aculeatus* and moderately important for *C. concinnus* (M1 variable contributions = 25.8%, 13.9% and 7.6%, respectively).

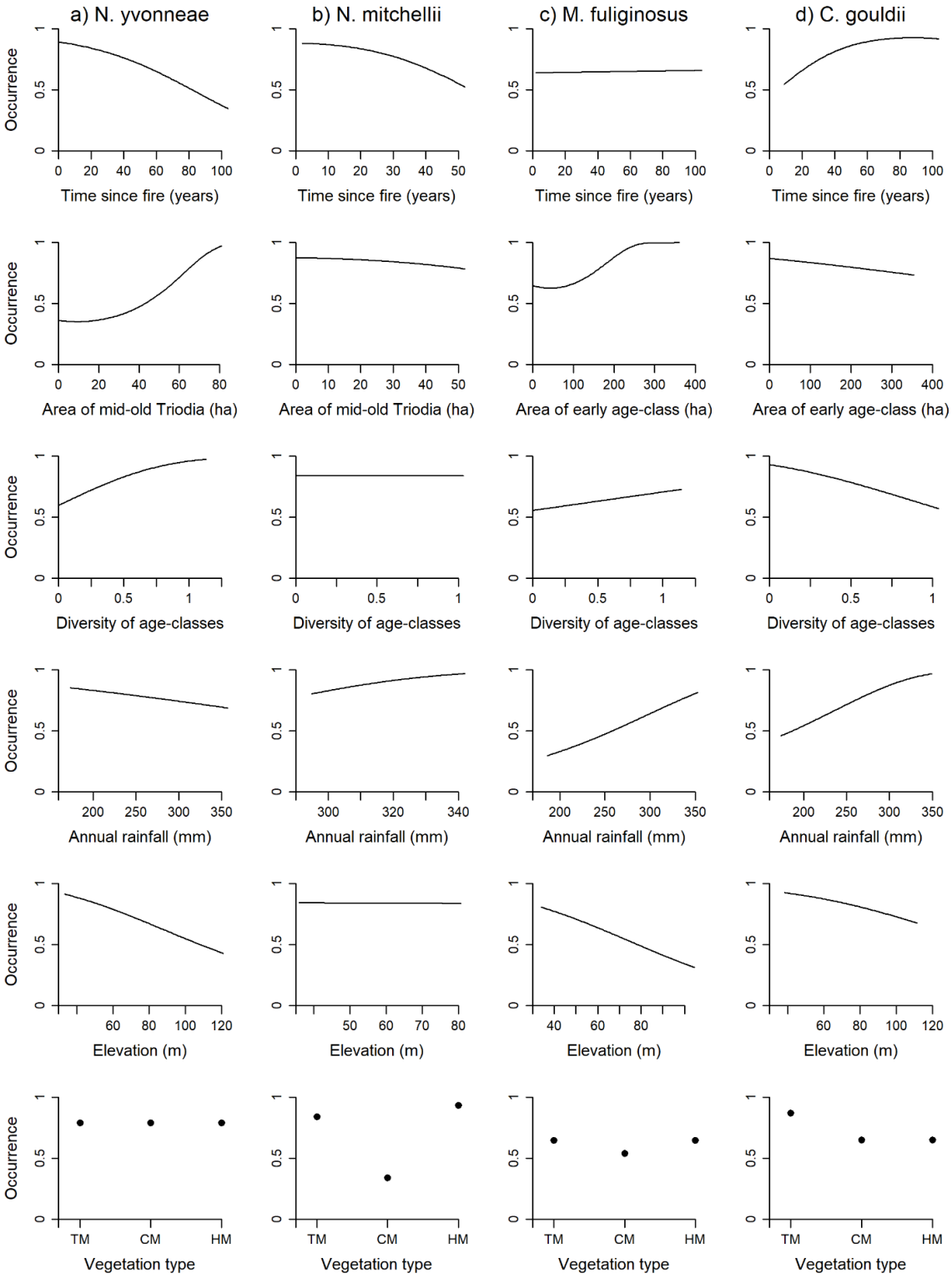

Figure 3.3: Relative likelihood of occurrence of four species in response to M1 predictor variables, with all other variables held at their median value. Vegetation types; TM = triodia mallee, CM = chenopod mallee; HM = heathy mallee.

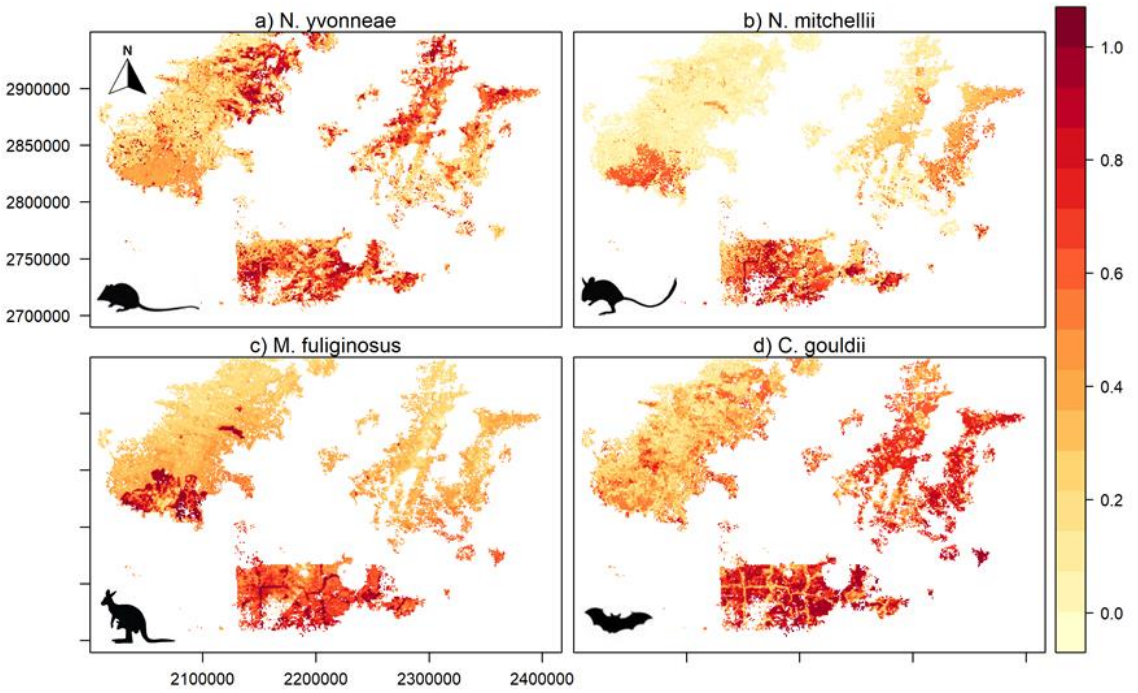

Figure 3.4: Mapping pyrodiversity relationships, specifically the relative likelihood of occurrence of four species from different taxonomic groups in relation to M1 predictor variables (time since fire, area, diversity, annual rainfall, elevation and vegetation type). Each species' relative likelihood of occurrence ranges from 0 to 1 (coloured scale bar) and is predicted to tree mallee vegetation in reserves ≥1000 ha and continuous tracts of mallee vegetation ≥5000 ha. White areas are agricultural land or other vegetation types.

Discussion

This study provides new insights into mammal distributions in fire-prone environments and how to model and map them. I have shown that several microbat species, including the threatened species *N. corbeni*, are less likely to occur in areas subject to frequent fire. In contrast, two species of native rodents were more likely to occur in recently burnt areas. Together, spatial and temporal characteristics of fire regimes shaped the distributions of mammals across other environmental gradients. I discuss the key results below.

Time since fire and fire frequency

Seven microbat species had a positive relationship with time since fire and a negative relationship with fire frequency. These relationships are likely due to the influence of fire on tree hollows. In semi-arid mallee vegetation, microbats require hollows in mallee stems for roosting (Lumsden and Bennet 1995). Live mallee stems do not begin to produce hollows until at least 40 years post-fire, and the likelihood of mallee eucalypts

containing hollows increases with age (Haslem *et al.* 2012). Repeat fires also reduce the occurrence of tree hollows, particularly in dead wood (Haslem *et al.* 2012). As hollows are a critical resource for microbats, these species are more likely to occur in older mallee vegetation and be negatively impacted by high frequency canopy fires. Microbat responses to fire could also be related to fire-induced changes in their invertebrate food source. However, previous research on mallee invertebrates suggests they are reasonably resilient to fire (Avitabile *et al.* 2015; Raap *et al.* 2015; Davis *et al.* 2019) and therefore unlikely to be a limiting factor affecting microbat fire responses.

Occurrence of the native rodents *N. mitchellii* and *P. bolami* declined with increasing time since fire. These rodents are granivorous, with rapid vegetation growth in the post-fire environment likely providing them with key resources, such as seeds. Both *N. mitchellii* and *P. bolami* are also burrowing, relying less on habitat structure for protection and shelter, and therefore able to survive in exposed, post-fire environments. A preference for recently-burnt environments has been reported for other native rodents in Australia, including members of the *Pseudomys* genus (Hale *et al.* 2016; Pereoglou *et al.* 2016; Geary *et al.* 2018). However, while occurrence of *N. mitchellii* and *P. bolami* is highest in early to mid-successional vegetation, they still occur in a range of post-fire habitats. Interestingly, *N. mitchellii* showed no response to fire frequency, being present at sites with fire frequency varying from 0-2 past fires. This is likely because key resources for this species are only available for short periods of time following fire (Kelly *et al.* 2010) and, at the frequencies associated with *N. mitchellii* records, are too infrequently present in the landscape for this variable to shape *N. mitchellii* distributions.

Small marsupials showed a variety of responses to fire frequency and time since fire, with the two pygmy possum species occurring in a range of post-fire environments. Occurrence of *C. lepidus* was not strongly linked to fire, but the likelihood of *C. concinnus* occurrence was highest at sites <40 years post-fire. *Cercartetus concinnus* is associated with cover of hummock grass and floristic diversity (Kelly et al. 2013) which may explain the high likelihood of occurrence at sites 5-40 years post-fire. The dasyurid *S. murina* had a high likelihood of occurrence in the first 30 years post-fire, which contradicts previous work in mallee and temperate woodlands that show no relationship with time since fire (Kelly *et al.* 2012; Hale *et al.* 2016). Including additional data sources (e.g. atlas data) in this modelling may have helped disentangle the relationships between *S. murina* and time since fire. Surprisingly, after including the spatial area measure, the

hummock grass specialist *N. yvonneae* was negatively associated with time since fire (discussed below).

Within the study area, the monotreme *T. aculeatus* occurs in all three vegetation types and a range of post-fire habitats. Because *T. aculeatus* is such a widespread species, my models did not discriminate preferred habitat for this species. One way that echidnas can survive in the immediate post-fire environment is by reducing their metabolic needs through torpor (Nowack *et al.* 2016). Previous work in temperature forests also indicates that echidna presence is poorly correlated with habitat and fuel measures that change with fire (Rochelmeyer *et al.* 2019).

Area, diversity and configuration of fires

Spatial variables such as area of a post-fire age class consider the neighbourhood context around a species occurrence record, to better represent a species' habitat requirements in fire-prone landscapes. These measures may allow threshold levels of habitat to be identified. For example, the small mammal *N. yvonneae* is strongly associated with *T. scariosa*, and previous research has shown that populations peak in triodia mallee between ~20-50 years post fire (i.e. mid-old post-fire ages; Kelly *et al.* 2011). My modelling took this relationship further, by showing that once half (or more) of the neighbourhood consists of mid-old aged triodia mallee the chances of *N. yvonneae* occurring there increase substantially. This is a new method of quantifying a species' relationship with fire and has the potential to improve how disturbance processes are incorporated into conservation planning and management.

The two dasyurids in the study area, *N. yvonneae* and *S. murina*, had moderate, positive relationships with neighbourhood diversity. This seems counter-intuitive to their preference for mid-old triodia mallee. However, the triodia mallee fire variable is a composite of mid and old post-fire age classes from one vegetation type, while variables such as diversity include two other vegetation types. It is likely that the dasyurids' preference for triodia mallee drives their relationships with other variables, including diversity and TSF. Additionally, diverse landscapes may allow a fire-sensitive species such as *N. yvonneae*, which is virtually absent immediately after fire, to utilise recently burnt areas if there is sufficient older vegetation in the neighbourhood.

Configuration of habitat was important for the microbats *N. geoffroyi* and *C. morio* and both species declined in occurrence with more aggregated early post-fire ages. However,

the likelihood of occurrence of both species remained reasonably high (>0.7) up to intermediate levels of neighbourhood configuration that reflect a mix of interspersed post-fire ages. Microbats are quite mobile, and previous research has suggested that proximity to unburnt vegetation can be an important factor allowing microbats to utilise burnt areas (Law *et al.* 2018). My results show that some microbats may be able to utilise burnt environments if burnt areas are interspersed with older vegetation.

Species that had strong relationships with temporal measures of fire did not always respond to spatial measures of fire, and vice-versa. For example, the grazing macropod *M. fuliginosus* was strongly influenced by fire, but only by spatial measures. *Macropus fuliginosus* occurrence was correlated with the area and configuration of young vegetation. This contrasts with results from Caughley *et al.* (1985), who found that *M. fuliginosus* density was similar in recently burnt and older vegetation and attributed this to high rainfall providing grazing resources in both areas. However, positive correlations with recently-burnt areas have been found in northern Australia for congeneric grazing macropods, which likely prefer recently burnt landscapes because a flush of new growth post-fire provides a key resource for grazing herbivores (Ritchie *et al.* 2008).

Diversity in post-fire ages was important for *M. fuliginosus*; it is highly mobile, and previous research has shown that macropods use different age classes in a landscape for specific resources (e.g. shelter vs foraging; Murphy and Bowman, 2007). *Macropus fuliginosus* is known to make daily movements between different habitat types in mallee ecosystems (Coulson 1993; Law and Dickman 1998) and I have observed individuals sheltering from the sun in patches of older mallee vegetation and grazing in recently burnt areas. If spatial measures of fire regime were not included in models for *M. fuliginosus*, this species may not be considered to have specific pyrodiversity requirements.

Desirable forms of pyrodiversity for conserving whole mammal assemblages

This work helps to clearly define the forms of pyrodiversity that can promote the conservation of whole mammal assemblages. For several species, including most microbats and *N. yvonneae*, the presence of mid to long-unburnt vegetation at a site or landscape is important. Other species, such as *N. mitchellii* and *M. fuliginosus*, show preferences for sites and landscapes comprising recently burnt vegetation. Previous work has emphasised the important of long-unburnt vegetation in achieving an appropriate mix of fire histories (Kelly *et al.* 2015). Retaining long-unburnt vegetation in the landscape

remains crucial to biodiversity conservation in mallee woodlands, yet by examining the requirements of the whole mammal assemblage I also highlight an important role for early successional habitats.

Management strategies for protecting long-unburnt vegetation in mallee woodlands currently focus on planned burning and fire suppression, with the aim of reducing the likelihood of wildfires burning large tracts of vegetation. There are important trade-offs involved in applying this approach, notably that fuel reduction treatments may burn older vegetation, removing suitable habitat in the short-medium term (Holland *et al.* 2017; Connell *et al.* 2019; Hiers *et al.* 2020). Predictive mapping shows that species dependent on older vegetation, such as *C. gouldii*, have low occurrence in planned burn fire scars which appear as long, linear strips in the distribution map for this species (Figure 3.4). By contrast, *N. mitchellii* and *M. fuliginosus* show strong preference for burn scars (Figure 3.4) and current planned burning operations appear to benefit these species. Planned burning can also affect mammals in ways beyond loss of habitat. For example, small mammals in recently burnt environments are often at greater risk of predation by invasive cats and foxes (Leahy *et al.* 2015; McGregor *et al.* 2016b; Hradsky 2020) which can impact their responses to fire. The invasive predator *Vulpes vulpes* (red fox) is widespread across the Murray Mallee region (Payne *et al.* 2014) and further research is needed to determine how the presence of foxes interacts with fire history to affect small mammal populations.

By mapping the distributions of mammals in relation to spatial and temporal measures of fire regimes and environmental gradients, I provide outputs that managers can use to incorporate biodiversity into fire planning. This new approach, underpinned by field surveys and atlas data, helps to translate ecological understanding into conservation actions in several key ways. First, maps of species distributions allow for identification of critical habitat and biodiversity hot spots, which can guide conservation planning and monitoring. Second, predictive mapping of biodiversity can guide real-time responses to wildfire, through identifying the location of biodiversity assets for targeted suppression and guide longer-term post-fire recovery efforts. Third, spatial information can be used to identify areas that may require more fire, to target with planned burning. Finally, this new approach paves the way for improved strategic planning. The next step in this research is to link these models and maps to simulations of alternative fire management strategies and climate scenarios to forecast future changes in mammal populations (Kelly

et al. 2015). This will also allow land managers to weigh up the benefits and risks of different fire management scenarios for different groups of mammals, across short-term and decadal time periods, to make informed decisions about how fire management will impact mammal conservation.

Inappropriate fire regimes threaten mammals in a range of ecosystems around the world, including in arid, temperate, tropical and even tundra ecosystems (Woinarski *et al.* 2015a; Beale *et al.* 2018; Davies *et al.* 2018; Kelly *et al.* 2020). Through quantitative assessment and mapping of mammal-fire associations, I provide valuable tools that managers can use to improve conservation actions. This novel approach was developed using species with a diverse range of life histories and thus is likely to be applicable to fire-prone regions across the globe. Incorporating multiple aspects of fire regimes into predictive models enables the positive and negative effects of fire to be better understood and will assist in achieving desirable forms of pyrodiversity that benefit whole mammal assemblages.

Chapter 4: Mapping desirable forms of pyrodiversity for snakes and lizards

Abstract

Ongoing changes to fire regimes represent a significant threat to the conservation of snakes and lizards. Despite fast-growing knowledge of reptile associations with fire regimes, how variation in fire regimes (pyrodiversity) affects whole reptile assemblages across large-scale environmental gradients remains uncertain. I aimed to quantify and map how different types of pyrodiversity shape terrestrial reptile occurrence in semi-arid woodlands of south-eastern Australia. I built species distribution models for 54 reptile species – ranging from blind-snakes, dragons and geckos, to legless lizards, skinks, snakes and a goanna – against fire and environmental covariates. Landscape measures of fire included the area, diversity and configuration of post-fire age classes, and site-based measures included time since fire and fire frequency. Of species that were associated with time since the last fire, most had high occurrence in mallee environments between 0-50 years post-fire. Spatial measures of fire such as area and configuration of age classes helped to explain and predict the distributions of 32 and 35 reptile species, respectively. For example, relative likelihood of occurrence of *Ctenotus brachyonyx* (short-clawed ctenotus) sharply increased once older age classes of vegetation were present in larger, aggregated patches in the local (<100 ha) landscape. Many snake and lizard species can occupy recently burnt environments, provided that landscapes include appropriately configured patches of longer unburnt vegetation. Predictive maps of snake and lizard distributions open new opportunities for incorporating the needs of reptiles into fire planning. This includes identifying areas of suitable configurations of different post-fire age classes that are critical to retain to maintain high prevalence of many reptile species. My predictive maps demonstrate that some mallee reptiles inhabit areas subject to planned burning and that the assemblage on a whole is relatively resilient to the kinds of burning undertaken to date. Understanding the pyrodiversity needs of whole reptile assemblages, and incorporating these into management, can assist in achieving desirable forms of pyrodiversity for reptile conservation.

Introduction

Fire is a key disturbance process driving patterns of biodiversity in many ecosystems around the world (He *et al.* 2019; Kelly *et al.* 2020). Many Australian ecosystems are fire-prone and have characteristic fire regimes defined by the time between fires, and fire frequency, severity, patchiness and season (Bradstock *et al.* 2012). Reptile species occupying these environments have evolved with these fire regimes, over long time scales, and benefit from fires that promote suitable habitat and resources (Smith *et al.* 2013; Dixon *et al.* 2018; Scroggie *et al.* 2019). However, changes in fire regimes associated with new land uses, climate patterns and mixes of species mean that inappropriate fire now represents a considerable threat to biodiversity. Globally, inappropriate fire regimes threaten 14% of terrestrial and freshwater reptiles listed as Vulnerable, Endangered and Critically Endangered under the International Union for Conservation of Nature (IUCN) Red List (Kelly *et al.* 2020). In Australia, fire is a key threat for ~10% of Australia's reptiles including 35% of those species listed as Vulnerable, Endangered and Critically Endangered under the IUCN Red List (Tingley *et al.* 2019; Geyle *et al.* 2020). Importantly, there remain large knowledge gaps around how different aspects of fire regimes affect whole reptile assemblages, particularly those comprised of large numbers of species and functional groups, and how spatial and temporal variation in fire regimes can best be managed to support diverse assemblages of reptiles.

Fire management often aims to create pyrodiversity – variation in spatial and temporal patterns of fire (Martin and Sapsis 1991) – under the assumption this will benefit biodiversity, including reptiles (Pianka and Goodyear 2012; Farnsworth *et al.* 2014). However, there are many different types of pyrodiversity and not all fire-driven variation benefits reptile diversity (Valentine *et al.* 2012; Nimmo *et al.* 2013; Santos and Cheylan 2013). For example, in Australian woodland and forest ecosystems, reptile assemblages benefit from increasing amounts of mid or long-unburnt vegetation over diversity in fire regimes (Smith *et al.* 2013; Kelly *et al.* 2015; Giljohann *et al.* 2015; Dixon *et al.* 2018). Fire does not act alone and, in combination with other environmental factors such as topography, vegetation type and rainfall, it modifies habitats across large-scale environmental gradients (Pastro *et al.* 2013; Nimmo *et al.* 2014; Ferreira *et al.* 2016; Chergui *et al.* 2019). Identifying specific aspects of fire regimes, across environmental

gradients, that benefit multiple species and functional groups will provide a more complete understanding of how pyrodiversity influences reptiles.

Time since fire is commonly used to quantify the effects of fire on reptiles, with the relative abundance of species often associated with different age classes (or successional stages) that provide suitable food and shelter (Letnic *et al.* 2004; Santos and Poquet 2010; Smith *et al.* 2013; Doherty *et al.* 2015a). These relationships are also classified by the shape of the response curve a species displays to time since fire, such as irruptive, unimodal or plateau (Caughley 1985; Driscoll and Henderson 2008; Nimmo *et al.* 2012). However, other characteristics of fire regimes, such as fire frequency, also impact reptile habitat and can therefore be important in shaping reptile species distributions (Lindenmayer *et al.* 2008c; Santos and Cheylan 2013; Grundel *et al.* 2015).

Spatial characteristics of fire, including area, configuration and diversity, can also influence reptile distributions and community composition (Nimmo *et al.* 2013; Kelly *et al.* 2015; Santos *et al.* 2016). For example, large areas of mid-late successional vegetation are required for the persistence of fire sensitive species in semi-arid woodlands as these age classes provide key shelter resources for reptiles (Nimmo *et al.* 2013; Giljohann *et al.* 2015). Planned fire improves connectivity between isolated habitat patches for *Crotaphytus collaris collaris* (eastern collared lizard) and increases the stability of their metapopulations (Templeton *et al.* 2011). In savannas and deserts, reptile communities are more diverse in landscapes composed of a patchy mosaic of different post-fire ages (Price *et al.* 2010; Pastro *et al.* 2011). However, most of these examples are from studies that are restricted to more common reptile species or only incorporate a few fire regime attributes. Studies that explore the influence of many different fire regime attributes on entire reptile assemblages are rare.

Functional groups – species with similar life-histories, traits and habitat preferences – provide another approach to categorise the responses of reptile assemblages to fire (Driscoll and Henderson 2008). Consideration of reptiles at this scale helps to synthesise fire responses of large groups of species and allows information about common species to inform the responses of rarer species within functional groups. For example, semi-arid mallee woodlands are home to a diverse reptile assemblage, including dragons, elapid and blind snakes, geckos, goannas, legless lizards and skinks. Previous work has categorized reptile functional groups in mallee ecosystems according to different life

history traits such as: habitat preferences (e.g. triodia or litter specialists, burrowing, semi-arboreal; Driscoll and Henderson, 2008; Nimmo et al., 2012), activity patterns (e.g. diurnal vs nocturnal; Smith et al., 2013), diet (Friend 1993) and social structure (Smith *et al.* 2014). However, empirical support for a functional group approach to quantify the responses of mallee reptiles to fire is mixed, with some species within the same functional group showing different responses to fire (Driscoll and Henderson 2008; Nimmo *et al.* 2012; Smith *et al.* 2013). A next step is to consider whether functional or taxonomic group approaches are useful for categorising the fire responses and pyrodiversity requirements of entire assemblages of reptiles, or if species-specific information is required for fire management.

Here, I explore how temporal and spatial variation in fire regimes influence 54 of the at least 70 reptile species that occur in semi-arid, tree mallee vegetation in south-eastern Australia (ALA 2011). These species are from seven different taxonomic groups, which I used to cluster the species and examine differences between taxonomic groups. The fire responses of 27 of these species have previously been studied in relation to some types of fire-driven variation, such as time since fire and the extent of post-fire age classes (Caughley 1985; Driscoll and Henderson 2008; Nimmo *et al.* 2012; Nimmo *et al.* 2013; Smith *et al.* 2013). However, the remaining species have not been modelled in relation to fire, mostly because of a lack of data. Additionally, important aspects of fire regimes, such as fire frequency and configuration, have not been studied before in relation to this reptile assemblage. Tree mallee vegetation is fire-prone, and several key habitat components for reptile – such as hummock grass – are strongly linked to fire (Haslem *et al.* 2011; Nimmo *et al.* 2013). I modelled the response of mallee reptiles to pyrodiversity and environmental gradients to ask: 1) What is the influence of temporal and spatial measures of fire on reptile occurrence? 2) How do pyrodiversity and other environmental gradients combine to influence the occurrence of reptiles in space and time? 3) Do relationships between pyrodiversity and other environmental gradients differ within and/or between functional groups?

Methods

Study area

The Murray Mallee region (32.8-35.2°S, 139.7-144.1°E; Figure 4.1) covers 104 000 km^2 of south-eastern Australia and comprises large reserves managed primarily for biodiversity conservation. The climate is semi-arid, with mean annual rainfall increasing from 160 mm in the north to 360 mm along a regional aridity gradient and mean annual temperature and evapotranspiration lower in the south. The landscape is low elevation (<100 m a. s. l.) and an undulating dune-swale system provides mild topographic variation. The vegetation is dominated by woodlands of 'tree mallee' – *Eucalyptus* species with a multi-stemmed growth form that re-sprouts after fire. Haslem et al. (2010) classified three main types of tree mallee vegetation, differing mainly in understorey composition: triodia mallee (TM), heathy mallee (HM) and chenopod mallee (CM). The region is fire-prone, with large wildfires > 100 000 ha occurring every 10-20 years, small fire events occurring more frequently and variability in fire resulting in longer intervals at any single location within the landscape (Clarke *et al.* 2010; Avitabile *et al.* 2013). Fire is actively managed throughout the region through suppression and planned burning to reduce the risk of large-scale wildfires. Planned burning is also used to meet the ecological needs of species that benefit from fire.

Species occurrence data

I sourced reptile occurrence data from the Atlas of Living Australia (ALA 2011) from between 1975 to 2016 for all reptile species occurring within tree mallee vegetation in reserves ≥1000 ha and continuous tracts of mallee vegetation ≥5000 ha (i.e. the study area: Figure 4.1). Several approaches were used to minimise the risk of inaccuracies in the atlas data (a potential problem with atlas data), including filtering records to retain only those with spatial accuracy of ≤100 m and with taxon ID marked as "NOISSUE". Visual examination of records meeting this criteria indicated most were from systematic surveys by reputable, expert sources such as universities and government agencies. Screened data was then spatially filtered to retain only one record per species per year per 100 x 100 m grid cell (the scale of the environmental data used in modelling). This resulted in 70 species, however I only modelled species with ≥15 filtered presence records which reduced the number down to 54 species from seven taxonomic groups (Table S4.1.1, Appendix S4.1).

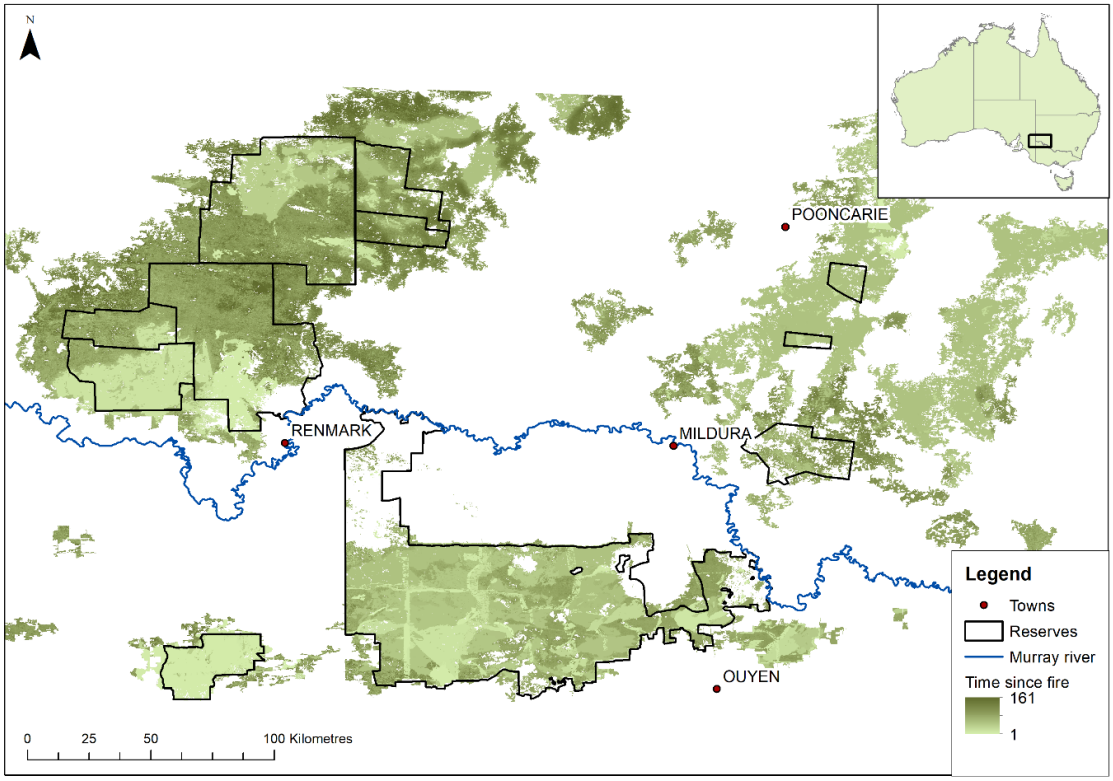

Figure 4.1: Map of tree mallee vegetation across the Murray Mallee region in continuous tracts of mallee vegetation ≥5000 ha and reserves ≥1000 ha. Fire history (time since fire and spatial extent) of the most recent fire is shown as of January 2020.

Environmental data

Values for the environmental data used in modelling were spatially extracted from raster geoTIFFs (1 ha resolution) that either represented conditions in the year of the occurrence record (fire variables) or represented static conditions through time. The three static variables were vegetation type as defined by Haslem *et al.* (2010), mean annual rainfall created with ANUCLIM (Houlder *et al.* 2001) applied to a 30 m Shuttle Radar Topography Mission Digital Elevation Model (SRTM DEM), and elevation derived from a 1-second SRTM DEM (GeoScience Australia, 2011). I used elevation to indicate position in the landscape (i.e. dune vs swale), and mean annual rainfall to represent the regional north-south aridity gradient which has previously been shown to influence the structure of mallee reptile communities (Nimmo *et al.* 2012; Nimmo *et al.* 2014). Preliminary analyses showed that mean annual rainfall is strongly correlated with other measures of aridity, such as mean temperature and evapotranspiration. Alternative models

run with these variables showed similar responses to models with annual rainfall, so I considered it appropriate to use as a representative variable for aridity.

Fire history data were compiled from several sources to obtain yearly fire histories for the whole region from 1900 to 2020. From 1972 to 2011 I used annual fire scars mapped from satellite images by Avitabile et al. (2013) and from 2011 to 2020 I used annual fire mapping from the NSW National Parks and Wildlife Service, the Victorian Department of Environment, Land, Water and Planning, and sourced from the South Australian Department of Environment and Water (2020). For fire history from 1900 to 1972 I used predictive modelling by Callister et al. (2016) that extrapolates time since the last fire across the whole region based on relationships between remotely sensed images and field survey data from sites of known fire history.. These sources of fire data were merged to create layers representing yearly fire histories from 1900 to 2020. To create fire variables for modelling, all calculations were undertaken in ArcGIS v10.3.1 (ESRI 2014), unless otherwise stated. Time since fire, fire frequency and the six spatial fire variables (discussed below) were calculated for each year from 1975 to 2020. However, fire frequency data was only able to be calculated from 1972 onwards (i.e. the start of satellite data) because predictive modelling only shows the most recent fire and not all previous fires at a location. Fire frequency for each year was therefore calculated as the number of fires from 1972 to that year, which potentially. underestimates the number of past fires early on in this period. However, exploratory plots and analysis of models developed with all fire frequency data and only data from later years showed similar results for reptiles, indicating that this is not a large source of bias.

I used three landscape metrics to represent the spatial distribution of fire elements: area (amount in hectares) of post-fire age classes, diversity of post-fire age classes, and configuration of post-fire age classes. These landscape metrics were calculated within a square buffer (~1 km^2) from post-fire age class rasters classified as early (0-10 years), mid (11-35 years) and old (>35 years). The buffer size was chosen to represent the local spatial context of fire at an ecologically meaningful scale for semi-arid reptiles. Configuration was measured using Li and Reynolds (1993) contagion index and indicates the degree (0-100) of aggregation and prominence of a specific post-fire age class. Values close to 0 indicate that the landscape is comprise of interspersed patches of different types in equal quantities, while 100 represents a homogenous landscape completely made up of the single post-fire age class of interest. Configuration was calculated in FRAGSTATS

v4 (McGarigal *et al.* 2012). Diversity (Shannon-Wiener diversity index) of post-fire age classes was calculated in GRASS GIS v7.2.2 (GRASS Development Team, 2017).

Area and configuration variables were calculated using the combined area of triodia mallee in the mid and old age classes (>11 years post-fire). This is a key habitat component, as many mallee reptiles are associated with this vegetation type, and growth of *Triodia scariosa* (hummock grass) peaks in these age classes (Haslem *et al.* 2011; Nimmo *et al.* 2013; Kelly *et al.* 2015). I also explored alternative area and configuration variables based on the early-post fire age class of all vegetation types. These variables generally contributed less to models of mallee reptile occurrence than variables based on the mid-old triodia mallee age class (Tables S4.2.1 – S4.2.7, Appendix S4.2).

1.2.4: Species distribution models

I used Maxent to build species distribution models (SDMs) using the 'dismo' package (Hijmans *et al.* 2011) to run these models in RV3.6.1 (R Core Team 2019). Maxent was selected because: it uses presence-only data (i.e. atlas data) to predict species relationships with environmental variables and subsequent distributions; is robust with relatively small sample sizes; and it is one of the highest-performing methods for developing SDMs (Elith *et al.* 2011). Presence-only methods such as Maxent build SDMs by comparing environmental conditions at the locations of species records to a representative sample of conditions at random, background points across the study area (Phillips *et al.* 2006; Elith *et al.* 2011). However, as sampling effort and species absences are unknown, methods using presence-only data are often prone to sampling bias that can affect model results (Phillips *et al.* 2009; Elith *et al.* 2011). To overcome common sources of sampling bias, including survey distance from roads and uneven survey effort across years, I created 10000 background points in a stratified, random design that was weighted by survey year (in the reptile occurrence data) and distance from roads.

Several pairs of fire variables were highly correlated (Spearman correlation >0.8), including TSF and fire frequency, and area and configuration. Despite these correlations, I expected these variables to contribute differently to models of mallee reptile distributions, and therefore developed three sets of models (M1, M2 and M3) for each species that incorporated all the fire variables of interest but avoided including correlated variables in the same model (Table 4.1). All models had six predictor variables, including mean annual rainfall, vegetation type, elevation, diversity of post-fire ages and two

additional fire variables. M1 included time since fire and area, M2 time since fire and configuration and M3 fire frequency and area.

Models for each species (n =54) were built using the individual species' occurrence data against the same stratified random background developed using input data from all species modelled. Five-fold cross-validation was used to estimate predictive performance on held-out data and model complexity was reduced by excluding threshold and hinge features (Elith *et al.* 2011). Models were assessed in several ways: through visual assessment of response curves and maps: diagnostic plots of omission rate and predicted area and receiver operating characteristics (ROC); and cross-validated area under the curve (AUC). Models with AUC >0.7 are indicative of good discrimination between presence and background points, with AUC values of 0.6-0.7 indicative of moderate discriminatory ability. I present results for species where AUC >0.6 in a least one of the three models. I assessed the relative influence of explanatory variables by comparing cross-validated variable contribution (the amount each variable contributed to the gain of the model during training, converted to a percentage). I defined variable influence as strong when variable contribution ≥25% and moderate when variable contribution ≥10%. I predicted each species distribution against environmental conditions in 2020, using 'cloglog' outputs to measure relative likelihood of occurrence from 0 to 1 (Phillips *et al.* 2017).

Table 4.1: Model sets for mallee reptiles. In addition to the fire variables listed, each model contained annual rainfall, elevation and vegetation type.

Model set	**Fire variables**
M1	Time since fire + diversity of post-fire age classes + area of mid-old triodia age class
M2	Time since fire + diversity of post-fire age classes + configuration of mid-old triodia age class
M3	Fire frequency + diversity of post-fire age classes + area of mid-old triodia age class

Results

I modelled 54 species of squamate reptiles from seven taxonomic groups (Table S4.1.1). The median number of presence records for a species was 144 (range 15 to 994; Table 4.2). Models of 47 species (87%) discriminated against background better than chance (AUC ≥0.6), and model discrimination was good (AUC ≥0.7) for at least one model for 41 species (76%; Table 4.2). On average, models built with variables based on the mid-old triodia mallee age class had higher AUC than models built with variables based on the early post-fire age class (Table S4.2.4, Appendix S4.2). Only six species did not have a moderate-strong contribution of at least one fire variable to one of the three models (Table S4.2.1 – S4.2.3, Appendix S4.2).

Time since fire and fire frequency

Time since fire of sampled sites ranged from 0 to 129 years. This measure had a strong influence (variable contribution >25%) in M1 for 14 species, and a moderate influence (variable contribution >10%) in M1 for 18 species (Figure 4.2). The most common response among these 32 species, shown for example by *Strophurus elderi* (jewelled gecko) and *Diporiphora nobbi* (nobbi dragon) in Figure 4.4, was a high relative likelihood of occurrence between 0-50 years post-fire followed by steeper decline >50 years post-fire. Other common responses include a linear, negative relationship with time since fire or occurrence peaking between 20-50 years post-fire, the latter exemplified by *Delma australis* (marble-faced delma) in Figure 4.3. Mean variable contribution of time since fire in M1 was higher for dragons and legless lizards than other groups, but responses varied substantially even within taxonomic groups (Figure 4.3).

Fire frequency varied from 0-3 fires at a site. Fire frequency was a strong contributing predictor variable for five species and a moderate contributor for 14 species (Figure 4.2). Of these 19 species, 16 had a positive relationship with fire frequency and three species had a negative relationship. Interestingly, species that responded strongly to time since fire did not necessarily respond strongly to fire frequency, despite correlations between the two predictor variables. For example, fire frequency contributed 96% to M3 for the snake *Pseudonaja textilis* (eastern brown snake), with species occurrence highest in areas with low fire frequency, but time since fire was not a strong contributing variable. When fire frequency was included in the model for *P. textilis*, model discrimination improved substantially, from AUC = 0.49-0.52 in M1 and M2, to AUC = 0.77 in M3.

Table 4.2: Model results, including number of presence records and AUC for the three different model sets, for species where AUC > 0.6 for at least one model.

Taxon. group	Scientific name	Common name	No. of records	AUC		
				M1	M2	M3
Blind snakes	*Anilios bicolor*	Dark-spined blind snake	108	0.70	0.69	0.69
	Anilios bituberculatus	Prong-snouted blind snake	129	0.81	0.82	0.78
Dragons	*Ctenophorus fordi*	Mallee dragon	864	0.76	0.76	0.75
	Ctenophorus pictus	Painted dragon	261	0.77	0.77	0.75
	Diporiphora nobbi	Nobbi dragon	642	0.77	0.77	0.76
	Pogona vitticeps	Central bearded dragon	447	0.75	0.75	0.74
Elapid snakes	*Brachyurophis australis*	Coral snake	292	0.76	0.76	0.73
	Parasuta nigriceps	Short-tailed snake	142	0.78	0.80	0.76
	Pseudonaja modesta	Ringed brown snake	52	0.87	0.87	0.85
	Pseudonaja textilis	Eastern brown snake	16	0.52	0.49	0.77
	Vermicella annulata	Bandy-bandy	47	0.58	0.60	0.55
Geckos	*Diplodactylus vittatus*	Wood gecko	340	0.78	0.79	0.77
	Gehyra variegata	Tree Dtella	183	0.67	0.66	0.66
	Nephrurus levis	Smooth knob-tailed gecko	144	0.72	0.72	0.72
	Lucasium damaeum	Beaded gecko	994	0.90	0.90	0.88
	Rhynchoedura ormsbyi	Eastern beaked gecko	212	0.77	0.78	0.78
	Rhynchoedura ornata	Western beaked gecko	51	0.68	0.72	0.68
	Strophurus elderi	Jewelled gecko	76	0.84	0.83	0.84
	Strophurus intermedius	Southern spiny-tailed gecko	107	0.75	0.75	0.74
	Strophurus williamsi	Eastern spiny-tailed gecko	38	0.83	0.81	0.83
	Underwoodisaurus milii	Thick-tailed gecko	24	0.76	0.77	0.75
Goanna	*Varanus gouldii*	Sand goanna	210	0.69	0.70	0.68
Legless lizards	*Aprasia inaurita*	Mallee worm lizard	64	0.71	0.74	0.71
	Delma australis	Marble-faced delma	200	0.80	0.81	0.79
	Delma butleri	Unbanded delma	175	0.81	0.80	0.81
	Lialis burtonis	Burton's snake-lizard	191	0.74	0.75	0.73
	Pygopus lepidopodus	Common scaly-foot	56	0.77	0.78	0.73
Skinks	*Ctenotus atlas*	Southern mallee ctenotus	451	0.75	0.75	0.74
	Ctenotus brachyonyx	Short-clawed ctenotus	500	0.80	0.80	0.80
	Ctenotus brooksi iridis	Wedgesnout ctenotus	24	0.86	0.86	0.77
	Ctenotus orientalis	Eastern ctenotus	46	0.91	0.91	0.90
	Ctenotus regius	Pale-rumped ctenotus	511	0.70	0.69	0.70
	Ctenotus schomburgkii	Barred wedgesnout ctenotus	482	0.70	0.71	0.71
	Cyclodomorphus melanops	Spinifex slender bluetongue	20	0.80	0.76	0.73
	Eremiascincus richardsonii	Broad-banded sand-swimmer	65	0.81	0.81	0.58
	Lerista aericeps	Desert plain slider	26	0.85	0.87	0.78
	Lerista bougainvillii	South-eastern slider	168	0.89	0.89	0.89
	Lerista labialis	Southern sandslider	104	0.93	0.92	0.93
	Lerista muelleri	Wood mulch-slider	131	0.82	0.82	0.81
	Lerista punctatovittata	Eastern robust slider	690	0.69	0.68	0.69
	Lerista xanthura	Yellow-tailed plain slider	19	0.65	0.62	0.65
	Liopholis inornata	Desert skink	460	0.78	0.78	0.76
	Menetia greyii	Common dwarf skink	536	0.69	0.69	0.67
	Morethia boulengeri	South-eastern morethia skink	298	0.65	0.65	0.62
	Morethia obscura	Shrubland morethia skink	168	0.76	0.76	0.72
	Tiliqua occipitalis	Western blue-tongued lizard	38	0.76	0.78	0.72
	Tiliqua rugosa	Shingle-back	119	0.75	0.73	0.72

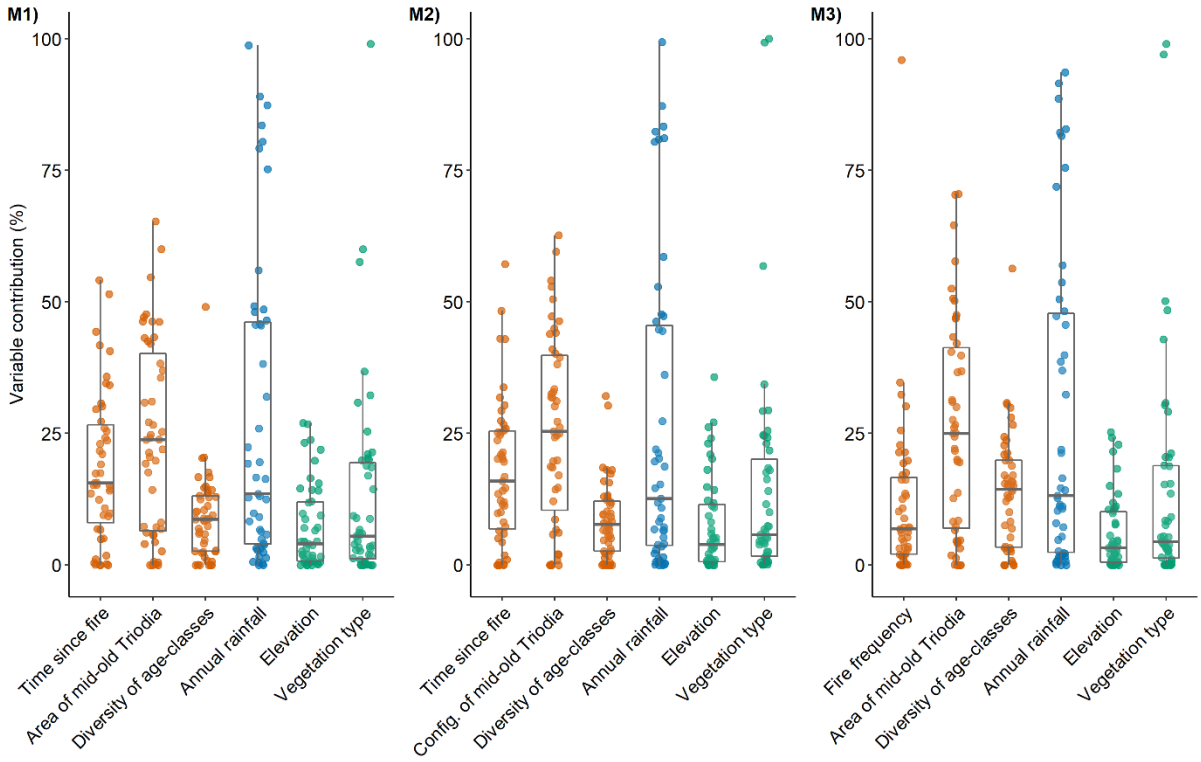

Figure 4.2: Contribution (%) of each predictor variable during training of the three model sets, for 47 species. Points represent the cross-validated contribution of one variable to the model of one species. Boxplots show the 25th and 75th percentiles around the median variable contribution across all 47 species, with whiskers extending to 1.5 interquartile range.

Amount, configuration and diversity of post-fire age classes

Area of mid-old triodia varied from 0 to 81 ha in the neighbourhood around each reptile presence record. Area of mid-old triodia was a strong contributing variable for 20-23 species and a moderate contributing variable for 9-14 species in M1 and M3 (Tables S4.2.1 and S4.2.3). These 32 species all had a positive relationship with the area of mid-old triodia, except for the dragon *Ctenophorus pictus* (painted dragon) and skink *Lerista aericeps* (desert plain slider) which showed a negative relationship with this variable. Importance of the area of mid-old triodia varied within taxonomic groups, although it was a particularly important contributing variable for blind snakes and legless lizards and less important for elapid snakes (Figure 4.3).

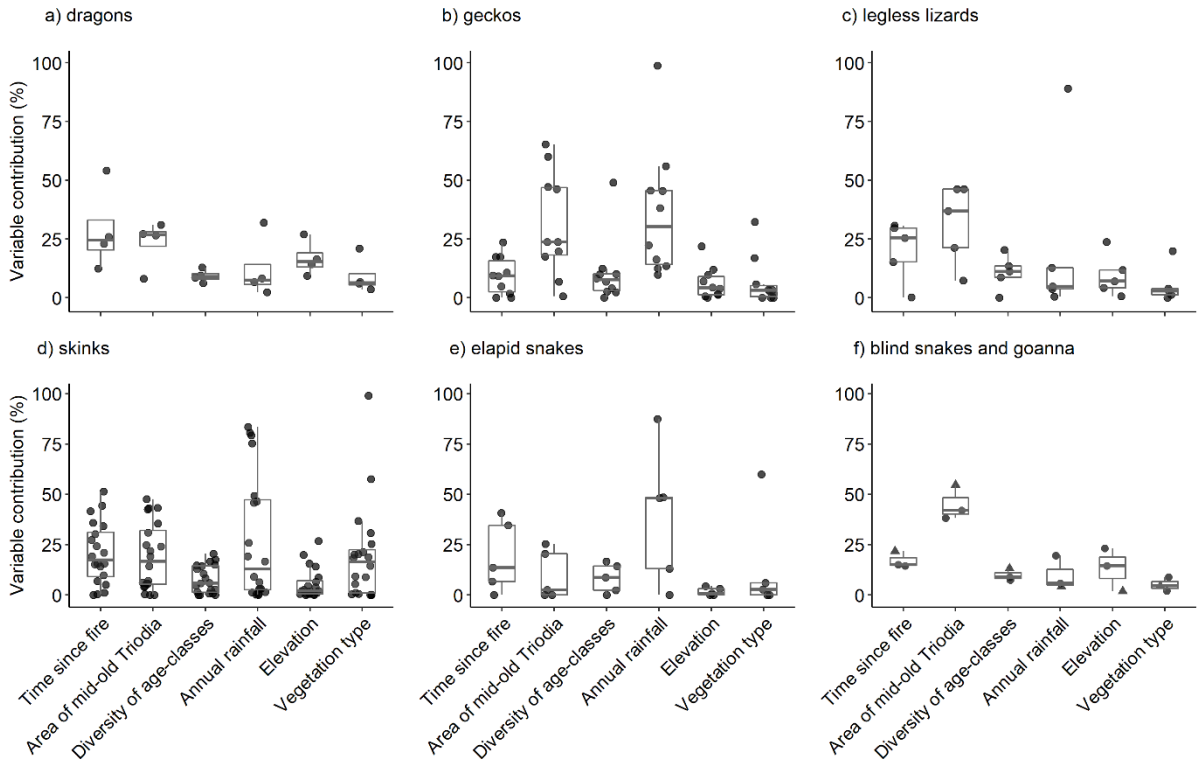

Figure 4.3: Contribution (%) of each predictor variable during training of M1, for seven taxonomic groups of mallee reptile. Points represent the cross-validated contribution of one variable to the model of one species. Triangular points represent variable contributions to the model for *Varanus gouldii* (sand goanna). Boxplots show the 25th and 75th percentiles around the median variable contribution across all 17 species, with whiskers extending to 1.5 interquartile range.

Configuration of mid-old triodia in a local (<100 ha) neighbourhood varied from 0 (i.e. no mid-old triodia mallee) to 90.9 (i.e. high amounts of aggregated mid-old triodia mallee). Configuration of early post-fire ages in a local neighbourhood also varied from 0 (i.e. no early post-fire vegetation present) to 90.9 (i.e. high amounts of aggregated early post-fire vegetation). Configuration metrics were, on average, more important to mallee reptiles than area metrics (Figure 4.2). However, this trend was stronger for models that included the area and configuration of early post-fire ages rather than the area and configuration of mid-old triodia (Tables S4.2.4 - S4.2.7). Configuration of mid-old triodia was a strong and moderate contributing variable for 24 and 11 species, respectively. Eighteen of these species responded positively to configuration and 16 species decreased in relative likelihood of occurrence up to mid-levels of configuration then increased in occurrence at higher levels of configuration. One species, *L. aericeps*, had a negative relationship with configuration (Figure S4.3.36, Appendix S4.3).

Diversity of post-fire ages classes was the only fire variable included in all three model sets and varied from 0 to 1.28 on the Shannon's Diversity Index. Responses to diversity were reasonably consistent among models including time since fire, with diversity a strong and moderate contributing variable for 1-2 and 14-18 species, respectively (Figure 4.2). However, when fire frequency was included in the model instead of time since fire, diversity was a strong contributing variable for six species and a moderate contributor for 24 species. Most species responded positively to the diversity of post-fire age classes, either linearly or with a plateaued relative likelihood of occurrence from intermediate-high levels of diversity, as shown by *S. elderi* in Figure 4.4.

Other environmental gradients

Mean annual rainfall, elevation and vegetation type were also important in shaping reptile distributions. Mean annual rainfall varied from 173-349 mm and was a strong contributing predictor variable for 16-18 species and a moderate contributing predictor variable for 9-12 species across the different models. Mean annual rainfall contributed >75% to models for seven species and was particularly important for geckos and elapid snakes (Figure 4.3). Reptiles typically responded monotonically to rainfall along the aridity gradient, which was reflected in their mapped distributions. For example, the dragon *D. nobbi* had a strong positive relationship with annual rainfall (Figure 4.4) and had higher relative likelihood of occurrence in the south where annual rainfall is higher (Figure 4.5). In contrast, the gecko *S. elderi* had a negative relationship with annual rainfall and was more likely to occur further north (Figure 4.5).

Vegetation type was a strong contributing predictor variable for 7-8 species across the three models, and a moderate contributor for 8-11 species (Figure 4.2). Among these species, the most common preference was for triodia mallee, or a combination of triodia mallee and heathy mallee (the latter which also has an understorey including *T. scariosa*). Only one species with a strong or moderate variable contribution of vegetation type showed a clear preference for chenopod mallee alone: the skink *Morethia boulengeri* (south-eastern morethia skink).

Elevation varied from 29-121 m and was a strong contributing predictor variable for 1-3 species and a moderate contributor for 11 species, depending on the model. Of these species, most had a negative relationship with elevation (Figures S4.3.1-S4.3.47, Appendix S4.3). Only two species with a strong or moderate variable contribution of

elevation had a positive relationship with elevation: the slider *Lerista muelleri* (wood mulch-slider) and gecko *Rhynchoedura ormsbyi* (eastern beaked gecko).

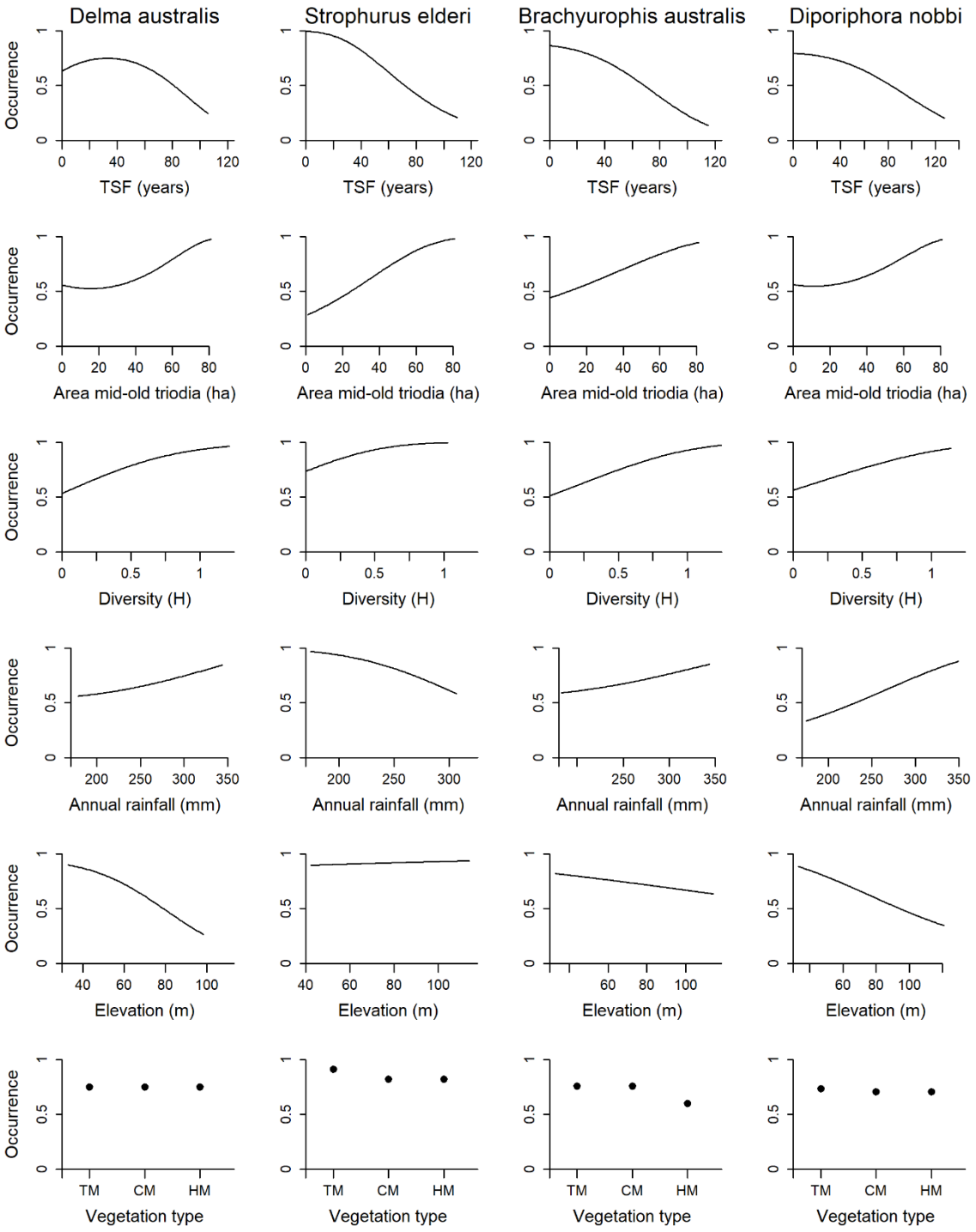

Figure 4.4: Relative likelihood of occurrence of four species in response to M1 predictor variables, with all other variables held at their median value. Vegetation types; TM = triodia mallee, CM = chenopod mallee; HM = heathy mallee.

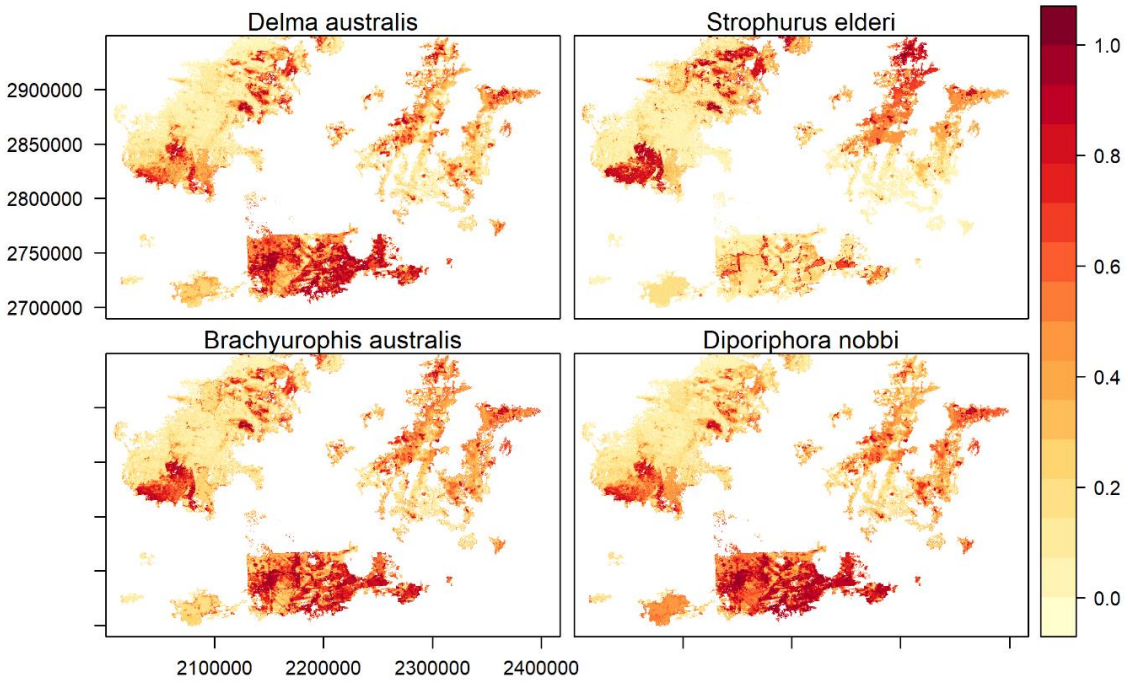

Figure 4.5: Mapped pyrodiversity relationships, specifically the relative likelihood of occurrence of four species from different taxonomic groups in relation to M1 predictor variables (time since fire, area, diversity, annual rainfall, elevation and vegetation type). Each species' relative likelihood of occurrence ranges from 0 to 1 (coloured scale bar) and is predicted to tree mallee vegetation in reserves ≥1000 ha and continuous tracts of mallee vegetation ≥5000 ha. White areas are agricultural land or other vegetation types.

Discussion

This study provides new insights into the influence of pyrodiversity on reptile distributions. For the first time, I modelled the response of most mallee snake and lizard species to pyrodiversity, including 19 species that have not been modelled in previous work. Of the 54 species included in this study, 12 of are listed as threatened under state conservation legislation. Fire was a strong influence on reptile distributions, with 87% of species having a moderate-strong relationship with at least one fire variable. Relative likelihood of occurrence of most species was highest in the first 50 years after fire. However, the importance of mid-old triodia mallee to many species shows that older vegetation still provides significant resources for mallee reptiles. My results show that many species can persist in recently burnt environments, provided the landscape is configured with sufficient quantities of mid-late post-fire age classes to meet their key habitat requirements. While reptile preferences for specific post-fire age classes were reasonably consistent, there is still substantial variation within taxonomic and functional

groups regarding the relative importance of different fire characteristics. This reinforces the need for species-specific modelling into the relationships between reptiles and pyrodiversity, and how individual maps of species distributions in relation to fire are a useful tool to incorporate the needs of many different species into fire management.

Time since fire and fire frequency

The most common response to time since fire was a high relative likelihood of occurrence between 0-50 years post-fire, or 20-50 years, then declining occurrence >50 years. Reptiles responding to time since fire in this way include two key groups: early successional specialists and *T. scariosa* specialists. Early successional species include those less dependent on vegetation for shelter, such as burrowing snakes and lizards like *Lerista punctatovittata* (eastern robust slider) and *Brachyurophis australis* (coral snake). However, not all burrowing species had a strong relationship with time since fire, or highest relative likelihood of occurrence in early post-fire ages. This result is consistent with previous work in a variety of ecosystems, including mallee woodlands, showing the difficulty of extrapolating species fire responses based on estimates of habitat use (Lindenmayer *et al.* 2008b; Santos and Poquet 2010; Nimmo *et al.* 2012; Smith *et al.* 2013; Chergui *et al.* 2020).

The keystone species *T. scariosa* reaches maximum height and density between 20-50 years post-fire (Haslem *et al.* 2011), and provides high-quality shelter for reptiles in this time period. Counterintuitively, many species considered *T. scariosa* specialists also had high relative likelihood of occurrence in early post-fire ages (0-10 years), such as the skink *Ctenotus brachyonyx* (short-clawed ctenotus). However, most of these species had a stronger relationship with the amount or configuration of mid-old triodia mallee than time since fire, which affected their occurrence at a site. For example, relative likelihood of occurrence of *C. brachyonyx* sharply increased once mid-old triodia mallee was present in larger, aggregated patches in the local (<100 ha) landscape (Figure S4.3.29). Overall, this indicates that *T. scarisa* specialists can persist in recently burnt areas, provided that a suitable amount and configuration of older triodia mallee is present in the local landscape.

The relative likelihood of occurrence of several species increased with more frequent fire. This is likely associated with the preference of many species for early to mid-successional vegetation shown in the present study, as well as in previous work (Nimmo *et al.* 2012;

Giljohann *et al.* 2015). In the present study sites burnt most frequently experienced two or, rarely, three fires within an approximately 50 year period. Reptile responses to higher fire frequencies, that were not sampled in the present study, are currently rare in mallee ecosystems and remain unknown.

Amount, configuration and diversity of post-fire age classes

Spatial measures of fire, including the area, configuration and diversity of post-fire age classes, were key to interpreting reptile responses to time since fire. For example, large areas of mid-old triodia mallee in the local (<100 ha) landscape allow species such as the threatened gecko *S. elderi* to occupy recently burnt environments. The response of many mallee reptiles to fire appears more complex than categorisations as early, mid or late successional specialists, and highlights the need for research and management to include different aspects of fire for a comprehensive understanding of pyrodiversity.

On average, configuration variables were more important than area variables to models of mallee reptile occurrence (Figure 4.2). This is consistent with the habitat complementation hypothesis that species occurrence in heterogenous landscapes is not driven by area alone, but also the configuration of key resources (Brotons *et al.* 2005a). Mallee reptiles are quite dispersal-limited, with some species moving an average of 20 m per day (Smith *et al.* 2016) and gaps >1 km between patches of habitat being a barrier to metapopulation persistence (Driscoll *et al.* 2012). This may explain why they are strongly influenced by the configuration of habitat in the local (<100 ha) neighbourhood. Eleven species with a moderate-strong relationship with configuration showed low prevalence (0.2-0.4 relative likelihood of occurrence) in certain configurations that are present in the current landscape. For example, the gecko *R. ormsbyi* had low occurrence (<0.3 relative likelihood of occurrence) in areas where mid-old triodia was poorly configured (configuration <40) in the local (<100 ha) landscape. Configuration of post-fire age classes clearly has a strong influence on mallee reptile occurrence and should be more carefully considered and incorporated into fire management.

Spatial diversity of fire regimes have seldom been included in studies of reptile responses to fire (but see Price et al., 2010; Pastro et al., 2011; Nimmo et al., 2013). I found that mallee reptiles with a moderate-strong relationship with diversity generally responded positively to higher diversity in post-fire age classes. However, approximately half of these species exhibited a plateaued response to diversity and were equally as likely to

occur in areas with intermediate levels of diversity in post-fire age classes as areas with high diversity. These preferences are consistent with the ability of some species to persist in heterogenous landscapes, discussed above. These results offer new insights into how diversity in post-fire age classes can influence reptile assemblages. Information about the positive responses of individual reptile species to neighbourhood diversity in post-fire age classes supports previous work showing that tailored mixes of age classes promote reptile diversity (Giljohann *et al.* 2015; Kelly *et al.* 2015; Giljohann *et al.* 2018).

Influence of other environmental gradients

Other environmental gradients had a strong influence on reptile distributions. The Murray Mallee region has a strong south-north aridity gradient has previously been shown to influence reptile distributions (Nimmo *et al.* 2014). Represented in the models by mean annual rainfall, this aridity gradient was a moderate-strong predictor variable for approximately half of the species modelled and influenced the distributions of species such as *D. nobbi* (Figure 4.4). Many mallee reptiles have limited ranges, being adapted to more arid or mesic conditions within the wider region, or limited in dispersal capacity by the Murray River which dissects the region (Nimmo *et al.* 2012). However, mean annual rainfall was not the strongest contributing predictor variable for all species that are restricted in range within the region. This lends support for the ability of the models to identify robust relationships between reptiles and measures of pyrodiversity, even across other environmental gradients. Reptile fire responses are influenced by environmental gradients such as topography, rainfall and vegetation type in other fire-prone ecosystems (Pastro *et al.* 2013; Grundel *et al.* 2015; Ferreira *et al.* 2016; Greenville *et al.* 2016b) and my approach shows how these can be accounted for when predicting how pyrodiversity affects reptile assemblages.

Vegetation type was also a strong contributing predictor variable for approximately half of the species I modelled. Interestingly, almost all these species showed a preference for triodia and/or heathy mallee over chenopod mallee. Both these vegetation types have an understorey of *T. scariosa* which may provide better shelter for reptiles than the more open understorey of chenopod mallee. In contrast, most species with a moderate-strong relationship with elevation preferred lower elevations, which are generally associated with chenopod mallee as it grows in the inter-dune swales. These variables were not strongly correlated, likely because of regional differences in the relative elevation of the

dune-swale system. However, it is possible that elevation (and therefore dune position) is accounting for some of the variation in the responses of chenopod-associated species.

Management implications

My work helps to clearly define the forms of pyrodiversity that benefit diverse reptile assemblages. I showed that most mallee reptiles prefer landscapes between 0-50 years post-fire. Many species also have specific requirements for certain amounts and configurations of vegetation, such as large amounts of mid-old triodia mallee, that increase their likelihood of occurrence when these requirements are met. The ability of reptiles to occupy recently burnt landscapes, dependent on the local availability of older vegetation, indicates that mallee landscapes that include recently burnt vegetation can still support diverse reptile assemblages. Identifying ways of incorporating recently burnt vegetation into the landscape is important as fuel reduction burning routinely occurs across the Murray Mallee region to reduce the risk of large-scale wildfire. Maps of reptile distributions in relation to pyrodiversity provide a key tool to aid land managers in incorporating reptile needs into fire planning. For example, maps could be used identifying areas of suitable configurations of different post-fire age classes that are important to retain for reptiles. These areas could then be the target of fire suppression efforts or adjacent planned burning to reduce the risk of large-scale wildfire.

My research highlights ways that reptiles can be resilient to management actions such as planned burning, and the important role of early successional habitat for some species such as *L. punctatovittata* and *B. australis*. However, as other taxa such as birds and small mammals are associated with long-unburnt vegetation (Kelly *et al.* 2015; Giljohann *et al.* 2015), the needs of other faunal groups must also be incorporated into fire management. My results show that the configuration of age classes is important to many reptile species. Management actions that manipulate the distribution of post-fire age classes (such as planned burning) should consider the connectivity and aggregation of key age classes (such as mid-old triodia mallee) and not just the amount of each age class in a landscape.

Through quantitative assessment and mapping of reptile fire associations, I provide valuable tools that managers can use to improve fire management for positive ecological outcomes. This new approach, underpinned by field surveys and atlas data, was developed using species from a variety of taxonomic and functional groups, and is therefore applicable to other reptile assemblages in fire-prone regions around the world.

Incorporating spatial and temporal aspects of fire regimes into predictive models, across environmental gradients, better enables the impacts of fire on reptiles to be understood. This will assist in achieving desirable patterns of landscape and regional scale pyrodiversity that benefit diverse reptile assemblages.

Chapter 5 – A field test of mechanisms underpinning terrestrial animal occurrence in recently burnt landscapes

Abstract

Planned burning can generate different types of pyrodiversity, however, experimental tests of how alternative spatial patterns of burning influence animal communities remain rare. Field tests are needed to understand the mechanisms through which spatial variation in planned fire affects fauna, and how fire can be applied to promote the persistence of animal populations. I developed and tested five hypotheses for how spatial variation in fire affects fauna: the habitat status, habitat amount, habitat complementation, habitat connectivity and habitat refuge hypotheses. Small mammal, reptile and invasive predator activity were monitored in the year following a large planned burn in semi-arid woodlands. Monitoring occurred at 12 burnt and eight unburnt sites that varied in the spatial arrangement of planned fire. I explored the following predictor variables: burn status (habitat status); amount of unburnt vegetation (habitat amount); interspersion of burnt and unburnt patches (habitat complementation); distance to unburnt vegetation (habitat connectivity); largest unburnt patch size (habitat refuge), and cover of *Triodia scariosa* (habitat refuge). Generalized linear models were used to test the influence of each variable on capture rate of three small mammal and 11 reptile species, on activity of *Vulpes vulpes* (red fox), and on species richness of native animals. I found partial support for all five hypotheses. Reptile assemblages varied between burnt and unburnt sites, and strong relationships were identified between the relative abundance of individual reptile species and each predictor variable. Reptile species richness increased at sites with higher amounts of unburnt vegetation, and sites that were less connected to unburnt vegetation had fewer species. Mammals, including the common dunnart, little pygmy possum and the invasive red fox, did not have clear relationships with burn status and spatial patterns of planned fire at a given site. Interestingly, fox activity increased after the fire at all sites. Planned fire that retains well-connected unburnt refuges is important for maintaining reptile diversity. The small mammals recorded appear resilient to the amount and configuration of planned fire experienced in this study, despite increased activity of introduced predators. Overall, animal diversity can remain relatively high in areas subject to planned burning, provided that suitable refuges are retained for reptiles and small mammals.

Introduction

A central theme in fire ecology is that variation in the timing, frequency, intensity and patchiness of fire promotes animal diversity. However, variation in fires, sometimes called pyrodiversity, shapes animal communities through a variety of mechanisms (Martin and Sapsis 1991; Griffiths *et al.* 2015b; Kelly *et al.* 2017a; Bliege Bird *et al.* 2018) that are rarely disentangled using manipulative experiments. These include mechanisms associated with spatial variation in the amount, diversity and configuration of fire-generated patches (Kelly *et al.* 2017a; Davies *et al.* 2018; Stillman *et al.* 2019). Planned burning is widely applied to achieve multiple goals in fire-prone landscapes and provides an excellent opportunity to explore the alternative mechanisms through which fire shapes biodiversity (Letnic 2003; Pastro *et al.* 2011; Valentine *et al.* 2012; Holland *et al.* 2017). The application of fire is also currently the focus of substantial debate about how to plan for and achieve desirable fire regimes that aid population persistence (Williams *et al.* 2017; Morgan *et al.* 2020; Berlinck and Batista 2020). Yet, uncertainty about how species respond to fire management actions, including planned burning, is a major impediment to conservation management (Nicol *et al.* 2019).

The amount (or extent) of fire-generated elements is a key landscape feature that can influence animal populations (Berry *et al.* 2015b; Lawes *et al.* 2015; Kelly *et al.* 2017a). The habitat amount hypothesis proposes that species richness and population size increases in landscapes with more habitat, regardless of patch size and connectivity (Fahrig 2013). In the context of fire, this can be interpreted to mean that sites with larger amounts of preferred fire-driven habitat (e.g. specific age classes or fire severity types) in the surrounding area will have higher species richness and higher relative abundance of individual species. There is some support for the habitat amount hypothesis at broader scales (e.g. >1000 ha), with increased extent of long-unburnt vegetation correlated with populations of small mammals and reptiles in eucalypt forests and woodlands in Australia (Kelly *et al.* 2012; Lindenmayer *et al.* 2013; Lawes *et al.* 2015; Dixon *et al.* 2018). In other cases, populations of some rodents benefit from large amounts of recently burnt vegetation (Pereoglou *et al.* 2016). While many studies have explored how burn status of planned fire (burnt vs. unburnt; hereafter referred to as "habitat status hypothesis") shapes small mammal and reptile populations (Letnic 2003; Pastro *et al.* 2011; Leahy *et al.* 2015; Hradsky *et al.* 2017; Flanagan-Moodie *et al.* 2018; McHugh *et al.* 2020), few

experimental studies have examined how the amount of burnt and unburnt vegetation influences animals at the fine scales associated with planned burning.

The spatial configuration of habitat – including the aggregation, interspersion and juxtaposition of different fire histories – is also an important element directly influenced by planned burning. For example, interspersed burnt and unburnt patches have been shown to provide a species with different resources which can support higher populations in the post-fire environment (Letnic 2001; Pereoglou *et al.* 2016; Stillman *et al.* 2019). This observation underpins the habitat complementation hypothesis, which proposes that landscapes with a mix of contrasting fire histories, juxtaposed at fine scales, complement the needs of more species and individuals (Brotons *et al.* 2005a). Few manipulative experiments have tested the habitat complementation hypothesis in relation to planned burning, therefore, little is known about how planned fire shapes the juxtaposition of small mammal and reptile habitat.

Another important aspect of configuration is the connectivity of different landscape elements, including the distance to unburnt patches or other preferred habitats (Simms *et al.* 2019). Although individual animals often survive a fire event *in situ*, animals dispersing from unburnt areas outside of a fire perimeter are also an important source of recolonisation (Banks *et al.* 2017; Puig-Gironès *et al.* 2018). Burnt areas may provide a barrier to animal movement, but unburnt patches, common after low-severity fires, may mitigate this by providing stepping-stones for animals to use when re-colonising an area (Banks *et al.* 2017; Nimmo *et al.* 2019). In the present study, I refer to the expectation that sites will vary in species richness and the relative abundance of individual species based on the distance between unburnt patches, as the habitat connectivity hypothesis (Brotons *et al.* 2005a). Specifically, sites furthest from unburnt habitat are predicted to have lower abundances of surviving animals and lower species richness in the short-term following fire due to dispersal constraints (also see Puig-Gironès et al., 2018).

Variation in fire elements within a landscape also shapes the availability of habitat refuges used by mammals and reptiles including vegetation cover, logs or tree hollows (Smith *et al.* 2013; Robinson *et al.* 2013; Berry *et al.* 2015a; McDonald *et al.* 2016; Flanagan-Moodie *et al.* 2018). Habitat refuges may mitigate predation after fire and the idea that increased availability of habitat refuges decreases predation risk in recently burnt environments underpins the habitat refuge hypothesis (Kelly *et al.* 2017a). Recent

research has shown that predation by invasive predators, such as *Vulpes vulpes* (red fox) and *Felis catus* (feral cat) increases after fire (McGregor *et al.* 2014; Leahy *et al.* 2015; Hradsky *et al.* 2017). Native predators, such as raptors, also actively seek fires to aid their hunting, which can increase local predation pressure on small vertebrates (Hovick *et al.* 2017). Increased predation pressure on native animals after fire can affect animal recovery in some ecosystems, but the impact is still unknown for many fire-prone areas, as is the size and type of remnant patches that might provide refuge from predation.

In this study, I completed a field test to investigate five key hypotheses that could explain how spatial variation in fire elements shapes the relative abundance and species richness of small mammals and reptiles. I sampled biodiversity within and outside of a large planned burn – designed to produce spatial variation across a treated area of 1400 hectares – in semi-arid woodlands of south-eastern Australia. This ecosystem is home to a number of small mammals and reptiles whose broad scale habitat preferences are strongly shaped by fire (Kelly *et al.* 2012; Nimmo *et al.* 2013; Smith *et al.* 2013). Planned burns are used in mallee to create barriers with low fuels to reduce the risk and extent of wildfires, and to meet the ecological needs of biota that benefit from fire. However, the influence of fine-scale variation in fire on mallee biota is largely unknown. Specific hypotheses I developed and tested were:

1. Species richness will be higher at unburnt sites compared to burnt sites. Abundance of individual species will vary between burnt and unburnt sites, with some species being more abundant at one type of site (‘habitat status hypothesis’)

2. Species richness, and the abundance of individual species, will be higher at sites with larger amounts of unburnt vegetation in the surrounding area (‘habitat amount hypothesis’).

3. Species richness, and the abundance of individual species, will be higher at sites where unburnt and burnt patches are interspersed than at sites where a single patch type is more aggregated (‘habitat complementation hypothesis’).

4. Sites further from unburnt vegetation will have lower species richness and species abundances compared to sites closer to unburnt vegetation (‘habitat connectivity hypothesis’).

5. Sites with larger unburnt patches, and higher plant cover that was unburnt during the fire, will have higher abundances of small mammals and reptiles as these elements provide protection from predation ('habitat refuge hypothesis').

Methods

Study area and design

The Murray Mallee region in south-eastern Australia comprises large areas managed primarily for biodiversity conservation. This study was conducted in Murray Sunset National Park, which is one of the largest continuous areas of tree mallee vegetation, encompassing 6330 km^2 of north-west Victoria. The climate is semi-arid, with high inter-annual rainfall variability and mean annual rainfall of 320 mm. A dune-swale system provides mild topographic variation. The vegetation is characterised by woodland stands of 'tree mallee', *Eucalyptus* species with a multi-stemmed growth form. Triodia mallee – mallee vegetation dominated by an understorey of *Triodia scariosa* (hummock grass) – is the most common of three main vegetation types (Haslem et al., 2010).

Mallee vegetation is fire-prone, and Murray Sunset National Park (hereafter, the Park) is the focus of significant fire management, with planned burning undertaken in spring and autumn (when fire risk is lower than summer, but fuel is dry and can still carry fire) to form long, linear strips ~1 km wide. This burning has the joint aims of creating barrier areas with low fuel loads, to reduce the risk and extent of wildfires, and of meeting ecological needs of biota that benefit from fire. While wildfires in mallee vegetation typically consume most vegetation, planned burns are less severe and typically retain unburnt patches of vegetation. This makes carefully designed planned burns a useful way to explore the influence of fire-driven variation on animal diversity and abundance.

The study was designed to take advantage of a large (1 km by 14 km perimeter) planned burn in the Park (Figure 5.1), implemented at a low severity by land managers in May 2018 via aerial ignition. This typical, autumn burn was selected for study through collaboration with land managers, who tailor their application of fire to generate spatial variation in mallee vegetation with a *T. scariosa* understorey. This vegetation type is important for several small mammal (Kelly *et al.* 2012) and reptile species (Nimmo *et al.* 2013), and *T. scariosa* hummocks provide food and shelter resources. The area surrounding the planned burn had experienced a similar recent fire history, with the last

fire occurring in 1977 (time since fire in May 2018 = 41 years). Prior to the planned burn, I established 12 'impact sites' within the proposed burn perimeter, stratified at distances of 100 m (n = 4), 300 m (n = 4) and 500 m (n = 4) from the expected edge of the burn (Figure 1) with the aim of exploring how connectivity to unburnt vegetation outside the burn area impacts fauna. All impact sites were located at least 500 m from the road to ensure connectivity effects were due to fire edges and not road edges. I also established 8 'control' sites outside of the expected burn area but within the same vegetation type and fire history (Figure 5.1). All sites were located on sandy dune slopes and had between 25-50% cover of *T. scariosa* prior to the planned burn.

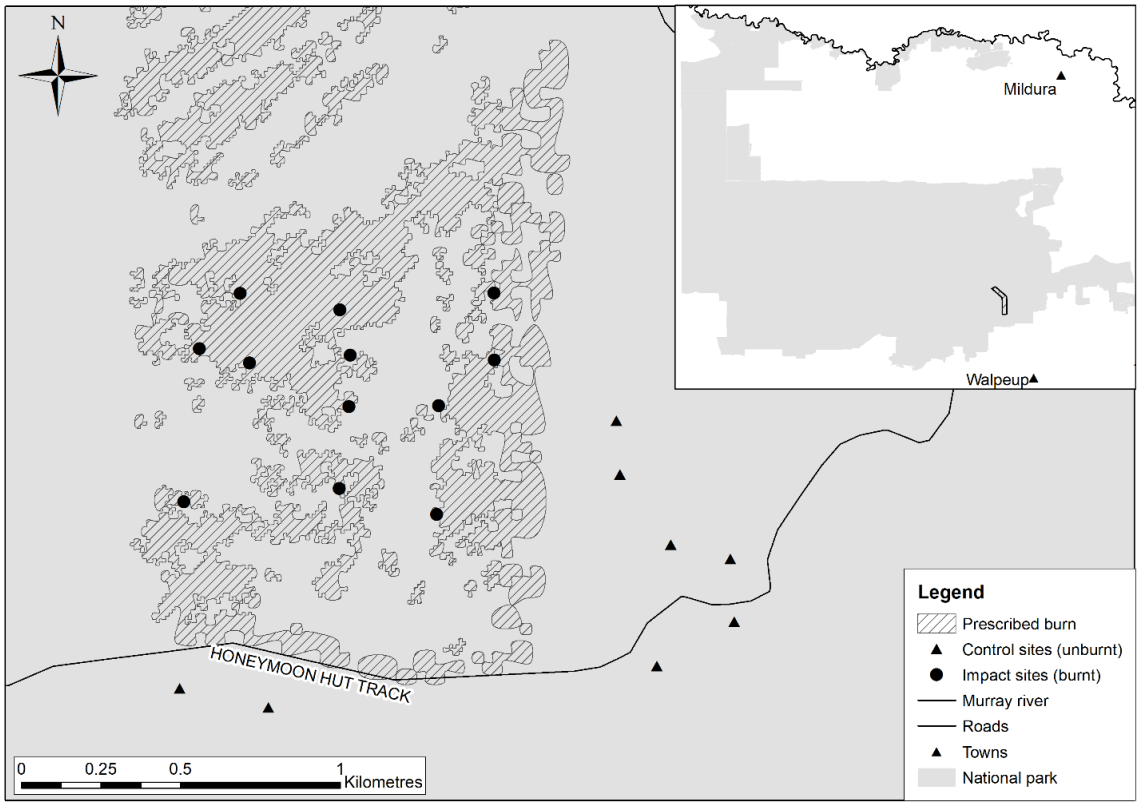

Figure 5.1: Map of the study area showing the location of 12 impact sites within the planned burn perimeter (hashed polygons) and 8 control sites in adjacent unburned areas.

Field surveys

At each site I established a 50 m pitfall trap line, consisting of 10 20 L buckets spaced five metres apart and connected by a 30 cm high flywire drift-fence. Pitfall traps were open for three to five consecutive nights during each survey period, with inclement weather (high rainfall or temperatures creating animal welfare concerns) forcing some

surveys to conclude early after only three or four nights of trapping. Each trap contained shelter for captured animals, consisting of a PVC pipe and cloth, and a polystyrene float in case of high rainfall. I sprayed the sand around each bucket with insecticide surface spray to deter ants and other invertebrates that might harm trapped animals. Traps were checked each morning soon after dawn and captured animals were identified to species-level. Reptiles were marked ventrally with a non-toxic permanent marker to identify recaptures within the same trapping session. Small mammals were marked by clipping small notches out of one or both ears to create unique patterns to identify individuals over the course of all field surveys.

A parallel line of Elliott aluminium box-traps (9 cm x 10 cm x 33 cm) was placed 10 m up-dune of the pitfall line, consisting of five traps placed 10 m apart. Elliott traps were baited with a mix of oats, peanut butter, vanilla extract, golden syrup and linseed oil, and baits were replaced halfway through each survey period to keep the scent fresh. Each Elliott trap contained leaf litter for shelter, was insulated by a plastic bag for warmth and, where possible, was placed in a sheltered position under or next to vegetation. I checked traps for small mammals every morning and processed captured individuals using the same methods as for animals captured using pitfall trapping.

I also installed a remote-sensing camera five metres from the start of the pitfall line at each site. Cameras were used primarily to target invasive predators, such as *F. catus* and *V. vulpes*, as well as small mammals that are not easily captured with live traps such as *Notomys mitchellii* (Mitchell's hopping mouse). Cameras were mounted 50 cm high on a star picket and angled 45° towards a lure on the ground 1 m away from the base of the star picket. Lures were made from super-absorbent cladding soaked in a mix of peanut butter, tuna and linseed oil, encased in a 10 cm long PVC pipe, capped at one end and pegged to the ground with the exposed end upright. Fresh lures were placed at each camera after each trapping survey and left in place for at least 14 nights.

A total of six field surveys, using both live and camera traps, were completed to measure changes in animal populations at each site. Surveys were conducted in May 2018 (before the planned fire), May-June 2018 (immediately after the planned fire), October 2018, December 2018, March 2019 and May 2019 (one-year post-fire). Three surveys were specifically clustered in warmer months (October-March) to capture reptiles, which are less active in cool weather.

Data analysis

Live captures of small mammals and reptiles were pooled across the five post-fire survey periods for statistical analyses. Capture rates of both groups of animals were low in the cool conditions before the planned burn (which was implemented in late May 2018 at short notice when local fire management resources aligned with appropriate weather conditions). Therefore, while I could model the effects of spatial variation in burnt and unburnt patches in the post-fire environment, I was unable to model changes before and after fire. Capture rate of native mammals and reptiles was expressed using a two-column matrix with the number of captures of a species in one column and pitfall trap effort in the other column. The introduced species *Mus musculus* (house mouse) was the only species captured in Elliott traps and the capture rate of that species included the combined survey effort from pitfall and Elliott traps. Reptile and native mammal species richness was also calculated for each site.

Camera traps recorded *V. vulpes* at all sites post-fire, so reporting rate was used as an index of fox activity for statistical analysis, rather than presence/absence. Records of *V. vulpes* were not temporally independent, so I only included the first record of a fox at a site within a 24 hr period from 10 a.m. to 10 a.m. and excluded additional records with each time period. Fox records for each separate 24 hr period were then summed for each site and divided by camera trap effort, which varied from 78 to 84 days, to create a reporting rate of fox activity for each site.

To measure the influence of fire on vegetation, and therefore habitat, vegetation surveys were completed each site. I used a 50 m line transect and measured the number of times *T. scariosa* was present (defined as ≥1 touch on a structure pole) at each metre point. This measurement was used as an explanatory variable (amount of *T. scariosa*) to represent fine-scale habitat structure in analysis.

I used fine-scale aerial photography to calculate site-based measures of the amount and configuration of unburnt vegetation at a finer resolution than what was available from burn scar mapping by land managers. I used a DJI Mavic Air drone to take images at 0.02 m grid resolution. Images were taken with the drone travelling at 50 m altitude, with 85% frontal and 70% side image overlap, in a 1 ha area surrounding the centre of each pitfall trapping line. These photos were knitted into a single orthomosaic for each site at 0.02 m grid resolution, using Pix4D cloud mapping software (Pix4D 2019). Object-based image

analysis in ArcGIS v10.3.1 (ESRI 2014) was then used to classify burnt and unburnt patches present in each orthomosaic. First, each image was 'segmented' to group pixels of similar colour and texture, using a segment mean shift with spectral detail of 15.5, spatial detail of 15 and minimum segment size of 20. Second, I created training sets for burnt and unburnt vegetation classes, ensuring samples came from all 20 site images and used the 'train support vector machine classifier' to classify vegetation as burnt or unburnt at all 20 sites. During classification, only green, living vegetation was defined as unburnt. Visual assessment of the classified images and aerial photos indicated that the presence of green vegetation was consistently associated with the presence of unburnt vegetation and therefore appropriate to use as a surrogate for unburnt vegetation.

The classified drone images were then used to calculate three variables in Fragstats v4 (McGarigal *et al.* 2012) that represented the configuration of the planned burn. Two of these were 'class-level' metrics: the amount of unburnt vegetation surrounding each site (m^2) and the percentage of the landscape comprised of the largest patch of unburnt vegetation (LPI or largest patch index). I also calculated a 'landscape-level' measure of the interspersion of burnt and unburnt vegetation at each site: a contagion index which calculates the interspersion and dispersion of different vegetation classes on a percentage scale from 0 to 100% (Li and Reynolds 1993).Values of this index closer to zero indicate that unburnt patches are highly interspersed with burnt patches (or bare ground), whereas values closer to 100 indicate that single patch types are more aggregated. Therefore, a negative relationship with interspersion would support the habitat complementation hypothesis.

Several fire variables were correlated (Pearson correlation coefficient >0.80) so could not be included in the same model. I therefore built univariate Generalized Linear Models (GLMs) for each species against predictor variables of burn status (i.e. burnt vs unburnt); amount of unburnt vegetation (habitat amount hypothesis); interspersion of burnt and unburnt patches (habitat complementation hypothesis); distance from external unburnt vegetation (i.e. external vegetation surrounding the planned burn boundary; habitat connectivity hypothesis); amount of *T. scariosa* (habitat refuge hypothesis); and largest patch index (LPI) of unburnt vegetation (habitat refuge hypothesis). Overall, I built models for capture rates of three small mammal and 11 reptile species, *V. vulpes* reporting rate, native small mammal species richness and reptile species richness (Table S5.1.1, Appendix S5.1). Continuous predictor variables were standardized by subtracting the

mean from each predictor and dividing by two standard deviations (Gelman and Hill 2007). Binomial GLMs were used for the individual species models, and Poisson GLMs for models of reptile and native mammal species richness. I explored the relationship between species capture rates and each predictor variable by considering the size and uncertainty (95% confidence intervals) of model coefficients. Models where the 95% CI around a model estimate did not overlap zero were considered indicative of a clear, unambiguous relationship between a response and predictor variable. Additionally, the response of small mammals and their predator *V. vulpes* to time since the planned fire was plotted to visualize and explore temporal variation in activity of this invasive species.

Results

I recorded 19 species of reptile, 3 species of native small mammals, one species of introduced rodent and two species of introduced predators, from 10 families (Table S5.1.1; Appendix S5.1). In total, surveys completed in warmer months recorded 308 captures of reptiles (2760 pitfall trap nights; Table S5.1.1). Capture rates of small mammals were low, and trapping (1406 Elliott and 4625 pitfall trap nights) resulted in 55 captures. Average small mammal capture rates were 0.3% for *Sminthopsis murina* (common dunnart), 0.13% for *Cercartetus lepidus* (little pygmy possum) and 0.04% for *Notomys mitchellii* (Mitchell's hopping mouse). The average capture rate of the introduced rodent *M. musculus* was higher at 0.86%, and varied substantially across the study period, from 3.77% in May 2018 to 0% in October and December 2018. Detailed results are presented below in relation to each of the hypotheses.

Habitat status hypothesis

Overall, three reptile species were associated with unburnt sites, one species was associated with burnt sites and seven showed no clear relationship with burn status (Figure 5.2). *Brachyurophis australi*s (coral snake) and *Strophurus intermedius* (southern spiny-tailed gecko) were more abundant in unburnt sites compared to burnt sites (model coefficients = 0.70 ± 0.33 SE and 1.54 ± 0.51 SE, respectively). Complete separation of the predictor variables occurred in the models for *Liopholis inornata* (desert skink) and *Menetia greyii* (common dwarf skink) and model coefficients could not be estimated. These species had strong relationships with burn status, with *L. inornata* only found in unburnt sites (found in 4 of 8 unburnt sites; Figure S5.3.8, Appendix S5.3) and *M. greyii*

only found in burnt sites (found in 8 of 12 burnt sites; Figure S5.3.10). Reptile species richness was higher in unburnt (control) sites than burnt (impact) sites (model coefficient = 0.51 ± 0.17 SE; Figure 5.3). Small mammals and *V. vulpes* did not have clear relationships with burn status (Figure 5.4).

Habitat amount hypothesis

Three reptile species had a positive relationship with the amount of unburnt vegetation and eight showed no clear relationship with this variable (Figure 5.2). Capture rates of *Diporiphora nobbi* (nobbi dragon), *L. inornata* and *S. intermedius* increased with more unburnt vegetation at a site (model coefficients = 1.37 ± 0.57 SE, 2.13 ± 0.91 SE and 1.01 ± 0.45 SE, respectively). Reptile species richness also showed a positive relationship with the amount of unburnt vegetation at a site (model coefficient = 0.47 ± 0.17 SE; Figure 5.3). Amount of unburnt vegetation did not have a clear relationship with capture rates of the three small mammal species modelled, native mammal species richness or reporting rate of *V. vulpes* (Figure 5.4).

Habitat complementation hypothesis

One reptile species had a positive relationship with the interspersion of burnt and unburnt patches, one species had a negative relationship with interspersion and nine species had no clear relationship with this variable (Figure 5.2). Capture rates of *M. greyii* increased at sites where burnt and unburnt patches were more interspersed (model coefficient = -1.62 ± 0.78 SE). In contrast, captures rates of *D. nobbi* increased at sites where a single patch type was more aggregated and dominant (model coefficient = 1.41 ± 0.66 SE). Reptile species richness did not show a clear relationship with the interspersion of burnt and unburnt patches (Figure 5.3). Interspersion of burnt and unburnt patches did not have a clear relationship with any of the mammal species modelled, or native mammal species richness (Figure 5.4.).

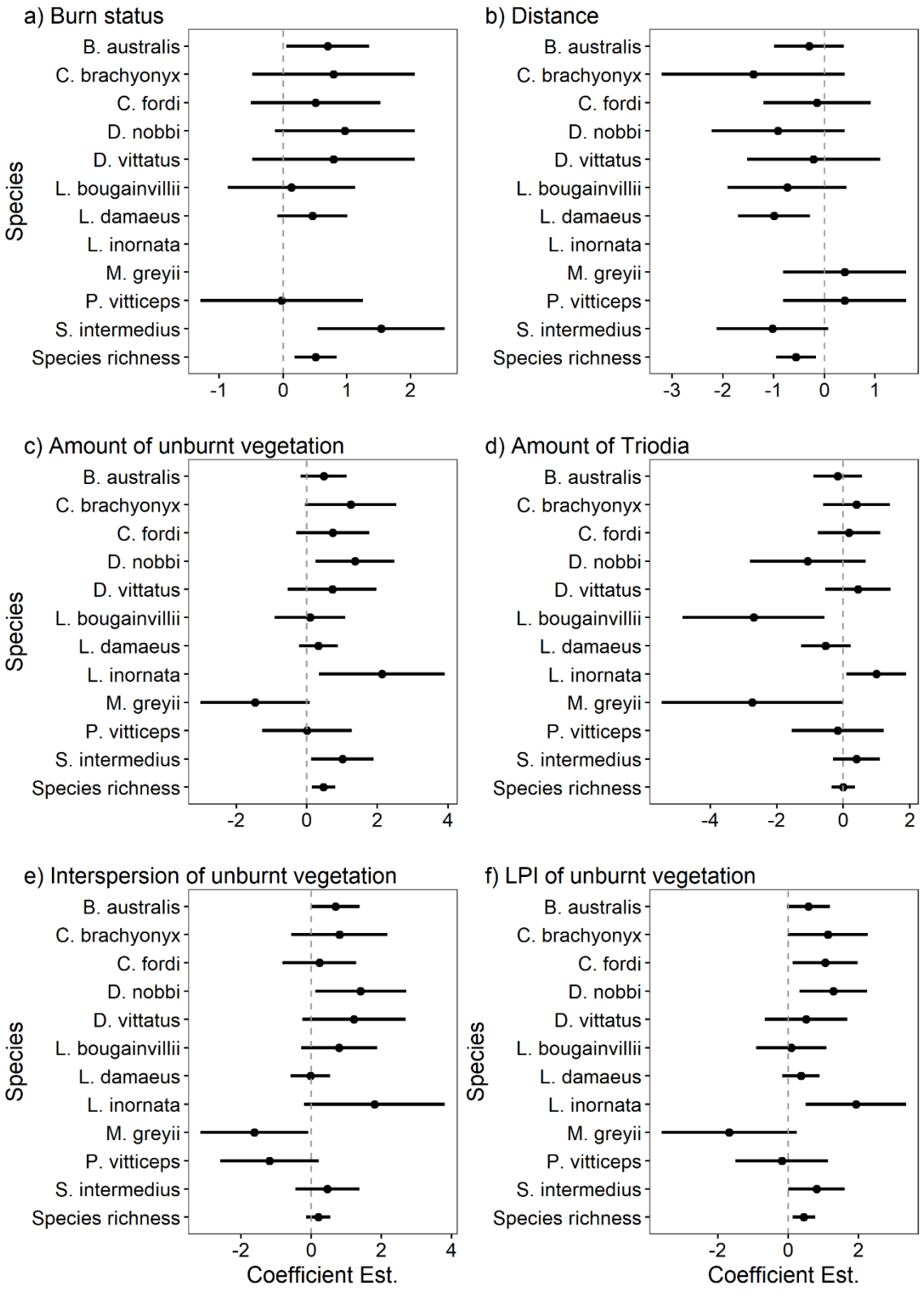

Figure 5.2: Results from univariate GLMs testing the influence of fire variables on the capture rate of reptile species and reptile species richness. Dots represent the coefficient estimate for a predictor variable and lines represent the 95% confidence intervals around that estimate. Plots where the coefficient estimate and confidence intervals remain above zero (positive relationship) or below zero (negative relationship) are indicative of a clear relationship between predictor and response variable.

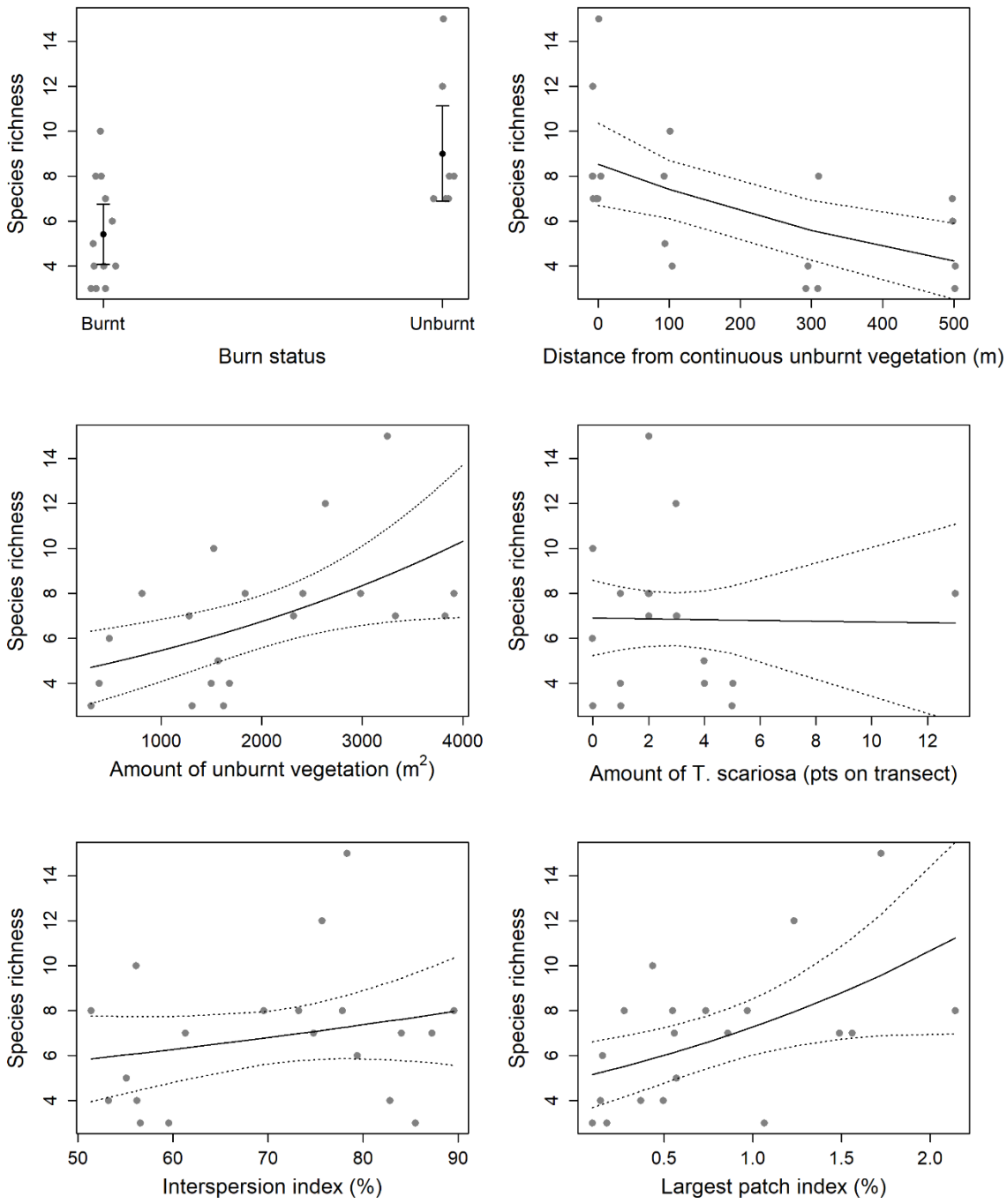

Figure 5.3: Predicted response of reptile species richness to each fire variable. Solid lines (or dots, for the categorical variable of burn status) show the predicted response from each model and dotted lines (or error bars, for burn status) represent 95% confidence intervals around those prediction. Grey dots are the raw data of reptile species richness against each fire variable from 20 survey sites.

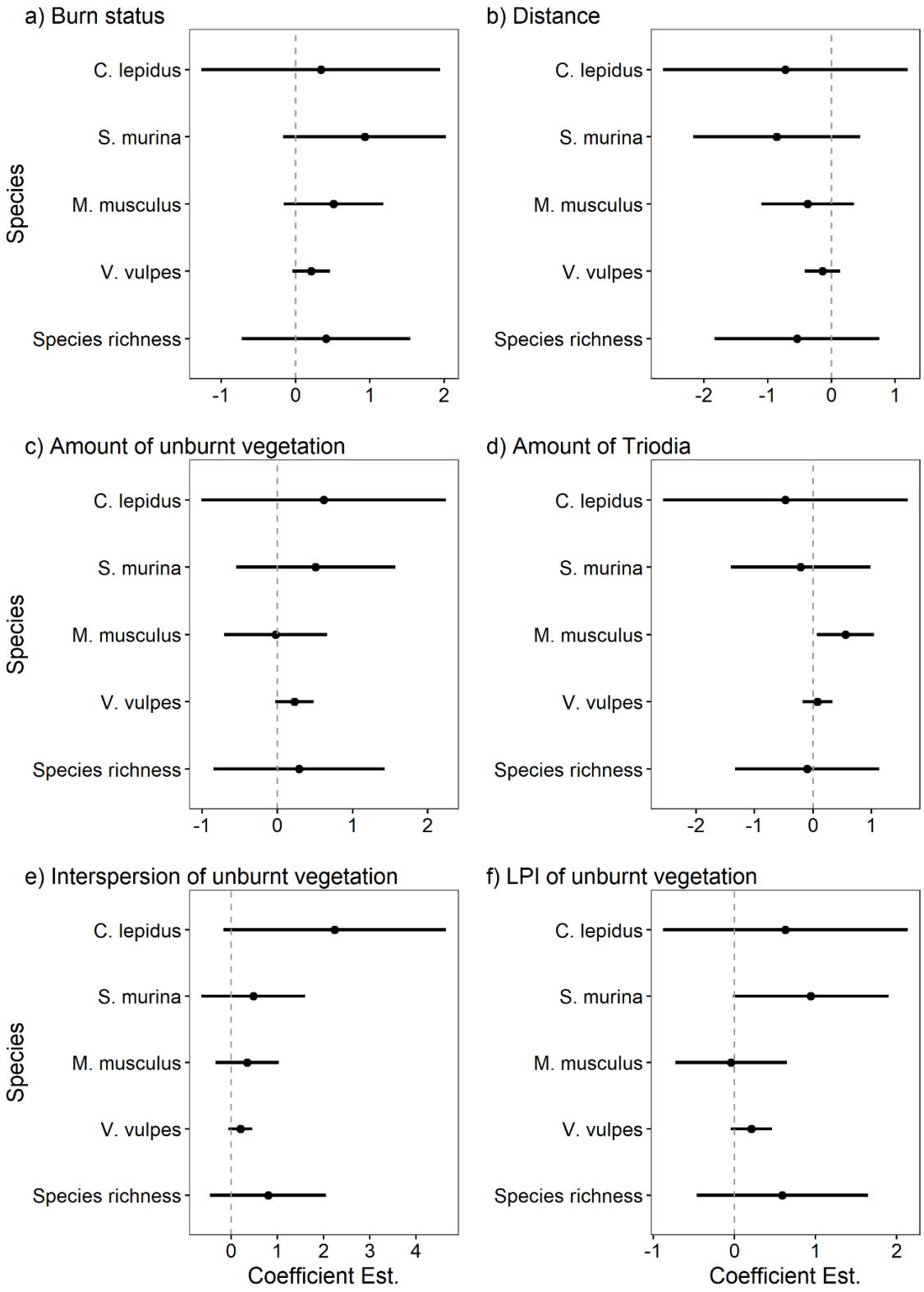

Figure 5.4: Results from univariate GLMs testing the influence of spatial characteristics of fire on individual small mammal species capture rate, small mammal species richness or *V. vulpes* reporting rate. Dots represent the coefficient estimate for a predictor variable and lines represent the 95% confidence intervals around that estimate. Plots where the coefficient estimate and confidence intervals remain above zero (positive relationship) or below zero (negative relationship) are indicative of a clear relationship between predictor and response variable.

Habitat connectivity hypothesis

Two reptile species had a negative relationship with distance from external unburnt vegetation and nine species had no clear relationship with distance (Figure 5.3). *Lucasium damaeum* (beaded gecko) abundance decreased towards the interior of burned areas (model coefficient = -0.99 ± 0.36 SE). Complete separation of the predictor variable occurred in the model for *L. inornata* as this species was only found at the unburnt control sites (at 4 of 8 sites; Figure S5.3.8). Reptiles species richness showed a negative relationship with distance from external unburnt vegetation, decreasing towards the interior of the burned area (model coefficient = -0.56 ± 0.20 SE; Figure 5.3). Distance to external unburnt vegetation did not have a clear relationship with abundance of the three small mammal species modelled, reporting rate of *V. vulpes* or native mammal species richness.

Habitat refuge hypothesis

Three reptile species had a positive relationship with largest patch index and eight species did not have a clear relationship with this variable (Figure 5.2). Capture rates of three reptile species, *Ctenophorus fordi* (mallee dragon), *D. nobbi* and *L. inornata* increased with higher values of LPI (i.e. larger unburnt patches; model coefficients = 1.06 ± 0.47 SE, 1.29 ± 0.49 SE and 1.93 ± 0.73 SE, respectively). Reptile species richness also increased with larger unburnt patches (model coefficient = 0.45 ± 0.16 SE; Figure 5.3). Largest patch index did not have a clear relationship with any of the mammal species modelled, or with native mammal species richness (Figure 5.4).

One reptile species was positively correlated with the amount of *T. scariosa* at a site, one species was negatively correlated with amount of *T. scariosa* and nine species showed no clear relationship with this variable (Figure 5.2). Capture rates of *Lerista bougainvillii* (south-eastern slider) decreased as *T. scariosa* became more prevalent (model coefficient = -2.70 ± 1.09 SE). Conversely, capture rates of *L. inornata* increased with higher amounts of *T. scariosa* (model coefficient = 0.99 ± 0.46 SE). Amount of *T. scariosa* did not have a clear relationship with reptile species richness (Figure 5.3). One mammal had a positive relationship with the amount of *T. scariosa*, and three species did not have a clear relationship with this variable (Figure 5.4). *Mus musculus* was more abundant at sites with more *T. scariosa* (model coefficient = 0.56 ± 0.25 SE). The amount of *T.*

scariosa did not have a clear relationship with native mammal species richness (Figure 5.4).

Two introduced predators were recorded in the study area: *V. vulpes* and *F. catus*. *Vulpes vulpes* was present at every site and reporting rate varied between survey periods (0% to 37.5%). *Felis catus* was present at <50% of sites but only detected from the fifth survey period. This species had a low overall reporting rate for each site (0% to 1.2%) and was not included in analysis. *Vulpes vulpes* was almost entirely absent from the study area before the fire but was present at both burnt and unburnt sites within months of the fire and maintained high activity throughout the subsequent year of monitoring (Figure 5.5).

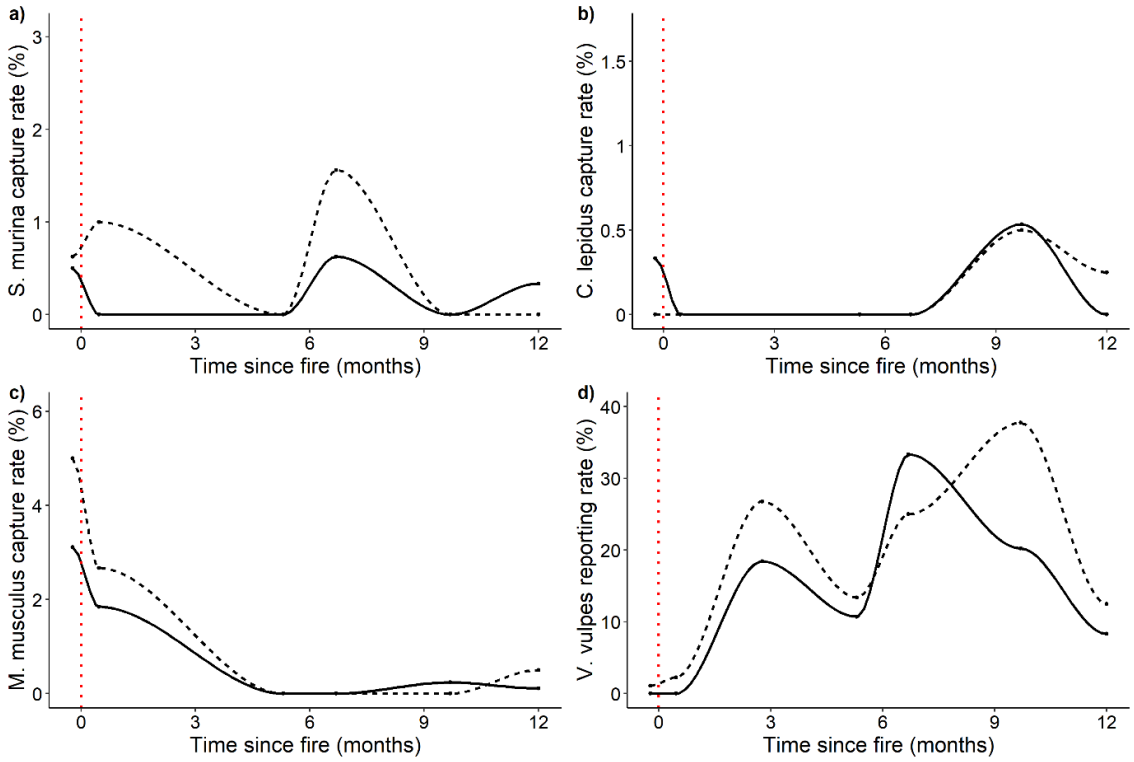

Figure 5.5: Activity (capture or reporting rate) of mammals before and in the 12 months following the planned fire. Solid black lines show activity at burnt sites, dashed black lines show activity at control sites. The dashed red, vertical line at 0 years shows the fire event. Species are; a) *Sminthopsis murina*; b) *Cercartetus lepidus*; c) *Mus musculus*; d) *Vulpes vulpes*.

Discussion

I found partial support for all five hypotheses about how the spatial heterogeneity of fire elements affects small mammal, reptile and predator occurrence in a recently burnt landscape. Surprisingly, capture rates of small mammals and reporting rate of the introduced predator *V. vulpes* were not correlated with the six spatial fire variables and these models did not support the hypotheses. Below, I discuss the results in the context of each hypothesis and broader implications for management of terrestrial animals in fire-prone landscapes.

Habitat status hypothesis

Many studies of the impact of planned fire on mammals and reptiles focus on differences between burnt and unburnt areas. In the present study, I found support for the idea that species richness would be higher in unburnt areas, but overall composition of reptile communities in burnt and unburnt areas would vary in relation to the habitat preferences of individual species. *Liopholis inornata* was only found in unburnt sites and had a positive relationship with the amount of triodia. Although *Liopholis inornata* is a burrowing species and is likely to persist in recently burned environments because its primary shelter is not affected by fire (Driscoll and Henderson 2008; Nimmo *et al.* 2012), differing fire responses have been recorded for this species. Smith *et al*. (2013) found low abundances of *L. inornata* in recently burnt sites, while (Letnic *et al.* 2004) recorded the species at sites <3 years post-fire and its fire response varies geographically across the Murray Mallee region (Nimmo *et al*. 2014). *Liopholis inornata* is commonly associated with hummock grass (Cogger 2014), and may be responding to availability of this specific habitat component in the present study, as well as to sandy soils that it burrows in. Abundances of other burrowing species, such as *M. greyii* and *L. bougainvilli*, were higher in burnt than unburnt sites, consistent with assumptions that burrowing species are able to occupy and exploit recently burned environments (Caughley 1985; Driscoll and Henderson 2008).

Habitat amount hypothesis

While I found that reptile species richness and assemblage composition varied between burnt and unburnt sites, the amount of unburnt vegetation was also important for several reptile species and for reptile species richness. This lends support to the hypothesis that

amount of preferred fire-driven habitat is a key constraint influencing reptile communities in burnt environments (Fahrig 2013). Species that had strong relationships with the amount of unburnt vegetation did not necessarily have a strong relationship with burn status, indicating that these variables may affect mallee reptiles in different ways. For example, capture rates of the dragon *D. nobbi* increased at sites with more unburnt vegetation, however this species did not show clear differences in relative abundance between burnt and unburnt sites. Previous site-scale studies have shown no clear relationships between *D. nobbi* and either time since fire (Nimmo *et al.* 2012) or structural habitat components that change with fire (Nimmo *et al.* 2013). *Diporiphora nobbi* is semi-arboreal and requires trees or logs to be present in the environment for perching (Cogger 2014). By considering neighbourhood measures, my results indicate that *D. nobbi* is resilient to low severity fire when the resultant mosaic comprises at least 15% unburnt vegetation. Furthermore, *D. nobbi* may only require trees or logs to be present at relatively low densities to persist in a landscape. Native small mammals did not have a clear relationship with the amount of remnant vegetation. There are several potential explanations for this. Semi-arid and arid zone small mammals can travel large distances over a short period of time, and possibly shift their home ranges due to resource availability (Dickman *et al.* 1995; Letnic 2001; Bos and Carthew 2007). The ability to move out of burnt areas could be a reason that small mammals did not show a clear response to any of the fire variables. Another is the ability of small mammals to use torpor to adjust their metabolic rate and cope with changes in resource ability caused by fire (Stawski *et al.* 2015). This may allow individual mammals to persist in a post-fire environment with limited resources. My results provide some support for this explanation: there was no difference in small mammal capture rates between burnt and unburnt sites, and between sites with more remnant vegetation and sites burnt uniformly.

Habitat complementation and connectivity hypotheses

This study tested two hypotheses for how the configuration of habitat might affect mammal and reptile populations post-fire: the habitat complementation and habitat connectivity hypotheses. I found limited support for the habitat complementation hypothesis that fine-scale juxtaposition of different patch types would support higher abundances and diversity of species. Instead, measures of native reptile and mammal species richness showed no relationship with the interspersion of burnt and unburnt patches and only one species, *M. greyii*, preferred sites with more interspersed patches.

Individual species and native species richness responded more strongly to the amount of unburnt habitat rather than the availability of complementary patches. A possible explanation for this is that the burnt areas provided little food or shelter for animals during the year following the planned burn, remaining bare and with little regrowth. Rainfall during this period was very low, and growth of annual grasses and forbs that commonly occurs in mallee landscapes after fire (Kelly *et al.* 2010) was noticeably absent. Over time these burnt areas will begin to provide additional resources for mallee animals and burnt areas should continue to contrast strongly with unburnt patches for at least a decade (Haslem *et al.* 2011). Habitat complementation could potentially be a more important mechanism in later stages of fire recovery if the resources provided by each class of fire history, or the degree of contrast, changes. Alternatively, habitat complementation may operate at larger scales than explored in this study, such as >50 ha landscapes (Pereoglou *et al.* 2016), or affect alternative measures of species diversity – such as beta or gamma diversity – in different ways (Farnsworth *et al.* 2014). Future work could focus on how well species responses to fine-scale heterogeneity extrapolate to responses across larger landscapes or affects long-term population persistence.

The habitat connectivity hypothesis proposes that dispersal is a key constraint to re-colonisation of burnt area, with differences in species composition based on distance to unburnt vegetation (i.e. source populations). I found that reptile species richness decreased with distance from external unburnt vegetation surrounding the planned burn. This lends some support to the idea that distance and landscape context is a constraining factor in reptile dispersal and recolonisation of burnt landscapes.

Habitat refuge hypothesis

The activity of *V. vulpes* was consistently high at all sites post-fire, regardless of burn status and the configuration or amount of unburnt vegetation. However, *V. vulpes* activity varied temporally, with very low activity recorded prior to the planned burn and activity increasing sharply afterwards. Introduced and native predators often increase their activity in recently-burnt landscapes as these open environments are easy to hunt in, and in some cases are drawn there from substantial distances (Leahy *et al.* 2015; McGregor *et al.* 2016b; Hradsky 2020). My results support the idea that introduced predators increase their activity in burnt landscapes. However, foxes also increased their activity in adjacent, unburnt areas. This suggests that the fire might have increased fox activity at a

landscape level. *Felis catus* also increased activity in the study area after the fire at both burnt and unburnt sites. However, cats were only detected in the study area from March 2019 onwards (10 months post-fire) and were present at too few sites to be included in analysis. Detecting feral cats can be difficult, so I cannot be certain in the recorded absences. However, it is likely that cats are truly present in low numbers in the study area, as other studies in mallee environments also recorded infrequent detections of cats (Payne *et al.* 2014; Geary *et al.* 2018).

Under the habitat refuge hypothesis, I predicted that prey species will be more abundant in sites with high ground cover that provides shelter from predation (Kelly *et al.* 2017a). In mallee ecosystems, both small mammals and reptiles can be preyed upon by foxes and cats. I found no clear relationships between native small mammal capture rates and patch size or the amount of *T. scariosa*. This is possibly because native small mammals were present in very low abundances throughout the entire study, resulting in low statistical power. A possible cause of these low numbers is competition from the introduced species *M. musculus*. This species was very abundant at all sites in the first few months of monitoring, and its abundance was high across much of the region including in adjacent agricultural land where it was at plague levels. Abundances of native small mammals did not vary substantially through time, either before or after the fire or during and after the mouse plague (Figure 5.4).

Patch size of unburnt vegetation had a positive influence on reptile species richness and on the abundances of several reptile species. Unburnt patches provided important microhabitat for reptiles, ranging from 10 to 200 m^2, and allowed species such as *C. fordi* to persist within the burnt area. Interestingly, capture rate of *C. fordi* showed a clear relationship with only one variable – unburnt patch size. This species had higher relative abundances in sites with larger unburnt patches. In mallee ecosystems, *C. fordi* is considered dependent on hummock grass for shelter and thermoregulation (Cogger 1974; Nimmo *et al.* 2013) and larger unburnt patches likely have larger amounts of hummock grass for this species to use. Unburnt patches may have also mitigated predation pressure by *V. vulpes* on reptiles. Nevertheless, it is difficult to disentangle the influence remnant vegetation plays in providing refuge from predation versus providing other necessary resources for reptiles. Future research will help disentangle this relationship further, for example by manipulating the amount and patch size of unburnt vegetation in the presence and absence of introduced predators.

Conservation and management implications

Balancing the needs of biodiversity and fuel reduction in fire-prone landscapes is challenging. Planned burning occurs across much of Australia and there are calls to increase the amount of planned burning in eastern Australia following the unprecedented 2019/20 bushfire season. My study has three key outcomes that could help improve biodiversity outcomes from planned burning. First, my research shows that remnant, unburnt vegetation within the burn area provides better outcomes for reptile diversity than more uniform burns. Connectivity is also important, as the reptile community was more diverse at sites within 100-300 m of unburnt vegetation than 500m away from unburnt vegetation. Unburnt patches of greater than 150 m^2 within the burnt area provide more resources for reptiles than small patches of less than 50 m^2. Taking all of this into account, planned burns that wish to retain reptile diversity should aim to be patchy, leaving larger remnant patches (>150 m^2) and have maximum widths of 500-600m to maintain connectivity with surrounding unburnt vegetation. While these small-scale patterns may seem difficult to incorporate into management plans, these conditions were already present (to varying degrees) within the planned burn studied. This burn was typical of autumn planned burns in Victorian mallee reserves and met requirements for fuel reduction in subsequent surveys by land managers. This highlights that management actions planned across large areas (1400 ha) and with coarse ignition techniques (aerial ignition via incendiaries) can retain unburnt vegetation in patterns that favour wildlife, while still meeting fuel reduction goals. Second, in contrast to reptiles, small mammals appear more resistant to planned fire and planned burning may help protect these species from damaging wildfires. Third, planned fire might also increase activity of introduced predators at the landscape scale. Incorporating introduced predator monitoring, and if necessary, control actions, into fire planning could improve outcomes for native biodiversity. In addition, managers can add refuge structures as supplementary shelter for native prey species, potentially mitigating increased predator activity (Bleicher and Dickman, 2020).

Small mammals, reptiles and invasive predators showed a wide range of responses to fire. The responses I observed did not fit one single hypothesis for how pyrodiversity affects animal persistence and recovery after fire but helped to learn about a variety of fire-driven mechanisms in play. Incorporating different kinds of spatial patterns in fire planning –

including patch size and connectivity - will improve management of pyrodiversity and lead to better outcomes for biodiversity conservation.

Chapter 6 – Synthesis

This thesis provides insight into how drivers such as pyrodiversity and climate influence diverse groups of mammals and reptiles at different scales. I used a combination of field experiments and statistical modelling to examine the drivers of small mammal and reptile populations and distributions in semi-arid mallee woodlands. I explored these drivers across different scales, from fine-scale patterns of burnt and unburnt vegetation created by planned fire, to regional-scale patterns of pyrodiversity shaped by recurrent wildfires. In this final chapter, I synthesis the insights arising from this new work and discuss the implications for fire science and future fire and conservation management. I conclude by outlining opportunities and priorities for future research.

Small mammals, meteorological conditions and biodiversity surveys

Chapter 2 aimed to determine the influence of daily meteorological conditions on small mammal activity, to improve field survey methods and biodiversity monitoring. Based on the current literature, ecological theory and my experience in the field with these taxa, I developed nine hypotheses for how different meteorological variables would affect activity (i.e. capture rate) of small mammals. To the best of my knowledge, this study is the first to explore the influence of daily meteorological conditions on small mammals in this region and was underpinned by a large dataset (>54 000 trap nights) across a >100 000 km^2 region.

I found that metrological conditions influence the capture rates of small mammals, which can help understand how small mammal activity varies under different conditions. A key result was the positive influence of wind speeds >20 km/h on capture rates of all families and species except the native rodent *Pseudomys bolami* (Bolam's mouse). Responses to all other variables varied between taxa: for example, warmer overnight temperatures increased captured rates of *Ningaui yvonneae* (mallee ningaui) but decreased capture rates of *P. bolami*. While these diverse responses make it difficult to determine a single set of optimal meteorological conditions for surveying the entire community, the results point to conditions favourable to individual species and groups. Surveys (undertaken in warmer months) targeting the whole small mammal community should encompass a variety of metrological conditions to ensure a representative sample of the local community.

Although responses to meteorological conditions varied between species, all native small mammals had a clear relationship with at least one meteorological variable. This makes the results useful for determining conditions favourable for individual species that could improve capture rates in targeted surveys. Knowledge of how meteorological conditions affect small mammal activity could also be used to calculate detection probabilities for new or existing datasets. For example, capture rates of the threatened dasyurid *N. yvonneae* more than double when relative humidity increases from 40% to >80% and are very low in dry conditions. These applications are especially useful as several of these species have wide-ranging distributions and can be detected in ecological surveys across much of Australia – for example, the dasyurid *Sminthopsis murina* (common dunnart) occurs in different vegetation types across eastern Australia. Results from this study can be used to improve detectability of small mammals in field surveys and biodiversity monitoring.

Animal diversity and pyrodiversity at broad scales

Chapters 3 and 4 explored the role of pyrodiversity as a driver of mammal and reptile distributions across a large region. Together, these chapters aimed to determine the regional and landscape-scale influence of pyrodiversity on different species and groups of mammals and reptiles, while also quantifying the influence of other environmental drivers. In total, I used species distribution modelling to explore the response and distributions of 17 mammal and 54 reptile species against different spatial and temporal measures of fire. A particularly novel aspect of this work was mapping the predicted distribution of each species across the entire region in relation to pyrodiversity and other environmental gradients. These maps can be used to better incorporate the needs of mammals and reptiles into conservation planning and fire management.

Native mammals showed a variety of responses to variation in fire regimes. Microbat occurrence increased with time since fire, likely due to their dependency on tree hollows which take decades to develop in mallee trees after fire (Haslem *et al.* 2012). Native rodents and the macropod *Macropus fuliginosus* (western grey kangaroo) preferred recently burnt environments that provide food resources such as grass and seeds. Small marsupials showed highest relative likelihood of occurrence between 0-50 years post-fire, coinciding with peaks in the keystone species *Triodia scariosa* (hummock grass). However, some species, such as the dasyurid *N. yvonneae,* showed a stronger preference

for large amounts (and high configurations) of triodia mallee vegetation >11 years old which is an important new finding that clearly demonstrates how spatial context is integral to understanding time since fire responses. My results indicate that species with habitat preferences for mid-late successional vegetation could occupy recently burnt environments, provided that the local landscape is at least half made up of well-connected patches of mid-old triodia mallee. This work modelled the fire responses of 13 species of mammal in mallee woodlands for the first time. Of these 13 species, seven had a moderate-strong relationship with at least one fire variable and different measures of fire were important for different species. These results show the importance of both spatial and temporal fire patterns in shaping mammal assemblages.

Reptile responses to pyrodiversity were more consistent, however there was still substantial variation within taxonomic and functional groups regarding the importance of different fire elements. This highlights the need to integrate species-specific information into fire management and planning, which can be aided by maps of species distributions in relation to fire. Many reptiles occurred most frequently between 0-50 years post-fire, coinciding with peaks in the hummock grass *T. scariosa*. However, the configuration of this key habitat component was more important to reptiles than mammals, and planning for appropriate configurations of fire-derived habitat should be incorporated into fire management for reptile conservation. Of the 19 reptile species modelled for the first time, 14 had a moderate-strong relationship with at least one fire variable. Spatial mapping and predictions in relation to a wide range of pyrodiversity measures also furthered our understanding of the fire responses and distributions of entire reptile assemblages.

Many species of mammals and reptiles were modelled for the first time in relation to fire history, which was only possible through innovative use of biodiversity atlas data and field data. Of course, there are potential issues with collating and using biodiversity atlas data in species distribution modelling – chiefly, that survey effort is unknown and records are prone to location, year and taxonomic biases (Phillips *et al.* 2009; Elith *et al.* 2011; Bean *et al.* 2012). Several steps were taken to reduce and overcome biases in the biodiversity atlas data I synthesised in chapters 3 and 4. First, the atlas records obtained for modelling in chapters 3 and 4 are largely from systematic surveys of remote areas undertaken by experts (e.g., government agencies, universities, and other experts). Therefore, we can have very high confidence in the accuracy of species identification. Second, the method used to create background points to contrast against the presence

records was specifically designed to overcome known biases in survey location (more records closer to roads) and year (more records in later years of the study period). Sampling across all years recorded in the species presence records also ensures a representative sample of the environmental conditions that were available to each species. Third, I conducted extensive checks of models and maps for species with known fire responses from previous work, such as *N. yvonneae* (see Kelly *et al.* 2012; Kelly *et al.* 2015). These were done in conjunction with experts in each taxonomic group, through workshops with government agencies and other academics. These checks determined that the models and maps were ecologically reasonable. Responses of several species to time since fire (including *N. yvonneae*) differed from previous work, however we determined this was due to the inclusion of new variables that explained large amounts of variation in the data, rather than inaccuracies with the presence-only data used in modelling. This was also supported by through the maps of species distributions, which reflected expert opinion on species distributions and indicated the new models were reasonable.

The approach developed in these chapters – collating field surveys and biodiversity atlas data, modelling entire taxonomic assemblages and spatial mapping of species distributions in relation to fire – are a step forward in understanding the fire ecology of mammals and reptiles. This work is one of very few studies (but see Beale et al., 2018) that explores and models the influence of pyrodiversity on entire taxonomic assemblages. Spatial mapping of species distributions in relation to pyrodiversity allows for fire management based on the needs of individual species, including those listed as threatened with extinction that are usually excluded from analyses because data availability is low. I anticipate that the approach developed in these chapters could be applied to other ecosystems and taxonomic groups to improve understanding of pyrodiversity-biodiversity relationships.

Importantly, there are other groups of taxa in mallee ecosystems that are affected by fire, including birds (Taylor *et al.* 2013; Keane *et al.* 2013; Connell *et al.* 2017) and plants (Clarke *et al.* 2010; Giljohann *et al.* 2017; Pollock *et al.* 2018). To ensure that the needs of most fire-sensitive mallee taxa are incorporated into fire planning, the approach outlined in chapters 3 and 4 could be applied to these additional taxonomic groups to generate mapped distributions of these species in relation to pyrodiversity. Summarising this large amount of information could be aided by first excluding those species that do not respond to fire and are not listed under threatened species legislation. It is next

important to differentiate between species that prefer a particular age class, but also have a reasonably high likelihood of occurrence in other age classes (such as the native rodents *N. mitchellii* and *P. bolami*), and species that are reliant on a particular age class (such as many of the microbats modelled) to persist in an area (Nimmo *et al.* 2012). Landscape occupancy in areas being considered for fire management could be considered for individual fire-reliant or threatened species. Alternatively, biodiversity indices could be used to summarise this information and incorporate the needs of many different species in fire management (see Giljohann *et al.* 2015; Kelly *et al.* 2015), with weighting applied for fire-reliant or threatened species (Giljohann *et al.* 2018). Models can also be used to directly predict how individual species or assemblages will respond under future fire management or climate scenarios, and how these affect their mapped distributions (Connell *et al.* 2019).

My work also adds to literature showing that not all species in fire-prone environments – particularly in semi-arid and arid ecosystems – are strongly affected by fire (Letnic *et al.* 2004; Langlands *et al.* 2006; Kelly *et al.* 2012; Pastro *et al.* 2013; Davis *et al.* 2019). For example, the distributions of two mammals, the bat *Vespadelus regulus* (southern forest bat) and burramyid *Cercartetus lepidus* (little pygmy possum) were primarily driven by other environmental gradients, indicating they may be largely resilient to fire at the spatial scales investigated.

Fine-scale relationships between animals and pyrodiversity

Chapter 5 explored the role of fine-scale patterns of planned fire in shaping animal distributions. I used a field experiment to test five mechanisms relating to how spatial patterns in burnt and unburnt vegetation might influence the activity of small mammals, reptiles and introduced predators in recently burnt environments. These were: 1) the availability of burnt and unburnt habitat, 2) the amount of remnant (i.e. unburnt habitat), 3) the configuration of complementary fire elements (e.g. burnt and unburnt vegetation), 4) the distance to surrounding unburnt vegetation and 5) the presence of unburnt refuges in the burn area.

Capture rates of most reptile species had a clear relationship with at least one fire-related variable. Reptile species richness was correlated with increasing amount and largest patch size of unburnt vegetation and decreasing distance from unburnt vegetation. This provides partial support for all five hypotheses and show that multiple mechanisms affect the

ability of reptiles to survive in burnt landscapes. Surprisingly, I found almost no clear relationships between the abundance of three small mammal species (*S. murina*, *C. lepidus* and the introduced *M. musculus*) and spatial patterns of fire history. Nor did I detect a relationship between the invasive predator *Vulpes vulpes* (red fox) and patterns of fire. This is likely because the low severity of the fire retained substantial amounts of unburnt vegetation that provided resources for small mammals (dunnarts and pygmy-possums) capable of feeding on a wide range of dietary items and sheltering in many habitat components (Kelly *et al.* 2012; Kelly *et al.* 2013; Turner 2020). Small mammals with similar life-history traits show no response to patchy planned fire in arid hummock grasslands (Pastro *et al.* 2011) and often respond to other environment triggers, such as rainfall, instead of fire (Southgate and Masters 1996; Letnic 2003; Letnic *et al.* 2004).

In contrast to small mammals, reptiles are ectotherms and are likely to be more restricted by their microhabitat requirements for shelter and thermoregulation (Caughley 1985; Letnic *et al.* 2004; Pastro *et al.* 2011; Nimmo *et al.* 2012). Three of the 11 reptile species modelled did not have clear relationships with any of the variables explaining spatial patterns of fire: *Ctenotus brachyonyx* (short-clawed ctenotus), *Diplodactylus vittatus* (wood gecko) and *Pogona vitticeps* (central bearded dragon). Previous modelling of these species' relationships to fire across the Murray Mallee region found mixed responses (Nimmo *et al.* 2012; Nimmo *et al.* 2013; Kelly *et al.* 2015). *Pogona vitticeps* did not have strong relationships with time since fire or extent of post-fire age classes at larger scales (2 km diameter landscapes (Nimmo *et al.* 2012; Nimmo *et al.* 2013; Kelly *et al.* 2015). However, *C. brachyonyx* was associated with older times since fire (Nimmo *et al.* 2012; Kelly *et al.* 2015) and *D. vittatus* was negatively associated with the proportional extent of recently burnt vegetation (<10 years post-fire) in each 2 km landscape (Nimmo *et al.* 2013). The majority of variation in fire history across the region covered by these studies was generated by large wildfires (Avitabile *et al.* 2013), which are more severe than planned burns and typically remove all above-ground vegetation. *Ctenotus brachyonyx* and *D. vittatus* appear more resilient to the effects of lower severity fire (i.e. planned burning) at smaller scales (1400 ha) than wildfire, which typically burns >10 000 ha (Avitabile *et al.* 2013)".

I expected to find that small mammal populations were higher at unburnt sites and sites with larger unburnt patches, in line with research suggesting that introduced predators increase activity or hunting efficiency in open, burnt environments (Frank *et al.* 2014;

Leahy *et al.* 2015; McGregor *et al.* 2015; Hradsky *et al.* 2017; Hradsky 2020). However, activity of the introduced predator *V. vulpes* also showed no relationship with any of the fire variables and instead increased at all sites (including unburnt controls) after the planned fire. At a similar scale, a recent study on the effects of planned burning on small macropods and their predators also found that small, patchy planned burns did not increase predation of native mammals (McHugh *et al.* 2020). The authors attributed this to the presence of unburnt refuges in the burnt landscape (McHugh *et al.* 2020). Small mammals in the mallee appear resilient to the effects of patchy planned burning at the scale tested in my study, regardless of the presence of unburnt refuges.

Mallee reptiles appear more sensitive to the short term (up to one-year post-fire) effects of planned burning than small mammals. The number of species, and the relative abundance of several species, declined at sites burnt more severely. The stronger response of reptiles than mammals to fire may be due to their limited ability to move between unburnt patches within the burnt area. For example, the gecko *Nephurus stellatus* (starred knob-tailed gecko) moves an average of 20 m per day (maximum 120 m) in mallee environments of South Australia (Smith *et al.* 2016) while the dasyurid *N. yvonneae* moves an average of 80 m per day (maximum 400 m) in the same vegetation type (Bos and Carthew 2007). The limited capacity of reptiles to move between isolated patches of preferred habitat is consistent with their preference for connected mallee vegetation >10 years post-fire at broader scales. The ability of many mallee reptiles to occupy recently burnt environments is dependent on fine-scale availability of unburnt patches and the availability of significant amounts of older vegetation in the local (<100 ha) landscape.

What types of fire are desirable for mammals and reptiles?

Under current fire regimes, large wildfires >100 000 ha occur in the Murray Mallee region every 10-20 years and smaller fires with greater frequency (Clarke *et al.* 2010; Avitabile *et al.* 2013). However, longer fire intervals are common for a single location and some areas included in Chapters 3 and 4 of this thesis had not been burnt for >100 years. Key habitat elements for mammals and reptiles, including hummock grass, leaf litter and tree hollows, take different amounts of time to develop after fire (Haslem *et al.* 2011; Haslem *et al.* 2012). Changes in the availability of habitat elements with the spatial and temporal arrangement of fire results in many different animal requirements for pyrodiversity to juggle when incorporating these needs into fire management.

My research showed that many reptiles prefer mallee vegetation aged 0-50 years post-fire, with relative likelihood of occurrence of many species highest in large areas of mid-old successional vegetation (>11 years post-fire). Microbats showed negative associations with frequent fire and preferred homogenous areas of long-unburnt vegetation (most species declined with increasing diversity in post-fire age classes). Effort should be made to retain large areas of long-unburnt vegetation (>35 years post-fire) for microbats. However, other taxonomic groups with high occurrence in older ages classes (>10 years post-fire) are less dependent on large areas of these age classes. Spatial extent and configuration of key age classes was important for dasyurids and highlights how including these measures in modelling can allow important threshold levels of fire-derived habitat to be determined. For example, the dasyurid *N. yvonneae* was much more likely to occur at sites where the local (<100 ha) landscape was comprised of >50% older (>11 years post-fire) habitat of a key vegetation type. This allowed them to occupy recently burnt areas where these age classes were present in the local (<100 ha) landscape. Combined with the apparent resilience of several small mammal species to patchy planned burns, strategic planned burning may protect small mammals from damaging wildfires with few negative effects on their populations.

Mapping the distributions of entire taxonomic assemblages in relation to pyrodiversity is a novel outcome of my research. The distributions of many of these species had not been mapped at fine scales (100 ha) before and many existing maps are derived from expert assumptions about where species will occur and not empirical modelling. These outputs can be used to better incorporate the needs of reptiles and mammals into fire management in several ways. Maps can guide firefighting during wildfires by allowing for the identification of key biodiversity assets – such as areas of long-unburnt vegetation with high occurrence of many species – that should be the target of active suppression efforts. Suppression is a key tool used in fire management and should continue to be used to protect key biodiversity assets in mallee woodlands. It can also prevent large-scale wildfires from creating large, homogenised areas of recently burnt vegetation that do not support diverse mammal and reptile assemblages.

Planned burning can be used to break up the continuity of fuel in large reserves and lower the risk of large-scale wildfire (Department of Environment Land Water and Planning 2015; Morgan *et al.* 2020). It can also be used to create buffer zones of low fire risk around biodiversity assets and to reinvigorate areas of senescent vegetation. My results

show that a number of taxa are resilient to patchy planned fire and the presence of moderate amounts of recently burnt vegetation in the landscape. Some taxa, such as native rodents and macropods, benefit from planned burning. For example, planned burning appears to create the preferred habitat of the native rodent *Notomys mitchellii* (Mitchell's hopping mouse). This suggest that appropriate planned fire – namely planned burned that retains unburnt patches of vegetation and well connected mid and late successional vegetation in the surrounding landscape – can both reduce fuel loads and provide habitat for diverse animal assemblages. Large, uniform areas of recently burnt vegetation are likely to provide limited resources for species at risk of extinction in mallee landscapes (Giljohann *et al.* 2015; Kelly *et al.* 2015; Connell *et al.* 2017).

Taking all this into account, below are actions that land managers could take to begin to incorporate the results from this thesis into on-ground actions for species conservation in mallee woodlands.

1. Consideration the composition of adjacent (unburnt) areas in fire management

Fire management actions, such as planned burning, currently occur at scales of 10 – 2000 ha depending on land status (e.g. public vs private land) and state jurisdiction. Importantly, most burns are long and thin to maximise their use as firebreaks and therefore distances from adjacent areas (e.g. older post-fire age classes) range from 100-1000 m. These distances are well within the scale used to measure local landscape conditions in chapters 3 and 4, and therefore recommendations for landscape composition can be directly made using the results of these chapters. Those wishing to undertake planned burns should consider the composition of adjacent areas, particularly the post-fire age class. Ideally, burns should be located next to vegetation in the old (>35 years post-fire) age class to provide essential resources for species that require this age class to be present in the local landscape. This will also have the added benefit of reducing fire risk in these older areas.

2. Identification and retention of older vegetation for bats

Efforts should be made to identify and retain areas of older vegetation, in patches of 300 ha or greater, that provide important resources for microbats. Maps of microbat distributions presented in this thesis, in conjunction with fire history maps, can be used to identify suitable areas. Adjacent areas could also be subject to strategic used of planned fire to reduce fire risk.

3. Retention of >50% older vegetation in local landscapes (100 ha) in areas of high occurrence of dasyurids

Species maps can be used to identify areas of high likelihood of occurrence for the dasyurids *N. yvonneae* (listed as Near Threatened in Victoria and Vulnerable in NSW) and *S. murina* (listed as Vulnerable in Victoria). If planned burns are done in these areas, targets should be set to ensure that the local (100ha) landscape is made up of at least 50% older vegetation (mid-old post-fire age classes) to provide essential resources for these threatened dasyurids.

4. Consideration of habitat configuration at different scales for reptile conservation

Configuration of post-fire age classes and unburnt vegetation was important for many of the reptiles modelled, at both fine (1 ha) and broader (100 ha) scales. At fine scales, planned burns should aim to reduce the overall width (where possible) to <500m and retain larger unburnt patches >100 m^2. At broader scales, connectivity of mid-old triodia mallee is important for many species. Managers should aim to either retain large (>100 ha) patches of this age class or ensure that mid-old triodia mallee is aggregated (i.e. not in many small, disjointed patches) if more recently burned vegetation is also present in the local landscape.

5. Invasive species

My results suggest that invasive red foxes increase activity in recently burnt areas, but also increase their activity in adjacent, unburned areas. Areas with high occurrence of threatened mammals and reptiles (identifiable using maps presented in chapters 3 and 4) that are subject to planned burns or wildfire could benefit from monitoring of invasive predators and control actions where necessary.

Future directions

There are exciting opportunities for research on pyrodiversity, including opportunities building on work that I completed in this thesis. Other opportunities involve entirely new studies and different perspectives, such as:

1. The effects of planned burning and alternative fire management strategies on fauna at decadal and century long time scales

Spatial mapping of species distributions in relation to pyrodiversity allows for the effects of alternative fire management strategies, such as different types of planned burning, on fauna and flora to be evaluated (Groeneveld *et al.* 2008; Kelly *et al.* 2015). Models

presented here, combined with models of other fire-affected taxa in the region, could be linked to simulations of alternative, real-world fire management and climate change scenarios to directly assess changes in biodiversity distributions under alternative scenarios.

2. How are fire regimes changing?

Fire regimes are changing (and have the potential to continue to change in the future) due to shifting land uses, global climate change and the redistribution of species (Brotons *et al.* 2013; Pausas and Keeley 2014; Probert *et al.* 2019; Kelly *et al.* 2020; Xu *et al.* 2020; Schmidt and Eloy 2020). Future research could focus on how predicted land use changes in semi-arid ecosystems (e.g. agricultural abandonment, revegetation and restoration, habitat fragmentation, new types of crops) could impact fire regimes and animal assemblages, following work in more temperature environments (Syphard *et al.* 2009; Fontaine and Kennedy 2012; Regos *et al.* 2015; Driscoll *et al.* 2016; Tulloch *et al.* 2016; Vanbianchi *et al.* 2017; Braun de Torrez *et al.* 2018). Invasive plants, such as *Cenchrus ciliaris* (buffel grass), are also changing fire regimes (Fusco *et al.* 2019), with largely unknown impacts on the distributions of Australian semi-arid and arid zone mammals and reptiles.

3. Evaluating the role of different types of burning in creating diverse and resilient animal communities

While this thesis explored spatial and temporal variation in fire, other patterns of human-created fire are possible. In eastern Australia, reinvigoration of cultural burning practices provides opportunities for co-production of knowledge with Indigenous communities to facilitate cultural fire practices and study beneficial patterns of burning for animals. Examples of studies into how cultural fire influences pyrodiversity and animal populations are available from western (Bird *et al.* 2013; Codding *et al.* 2014; Bliege Bird *et al.* 2018) and northern Australia (Murphy and Bowman 2007).

4. Relationships between predator activity, predation risk and small mammal activity in relation to fire history

Invasive predators, such as *V. vulpes*, showed no relationship with fine-scale patterns of fire in my research. Previous research has also found no relationships at landscape and regional scales, but that foxes are present across the Murray Mallee region (Payne *et al.*

2014) and in adjoining mallee environments (Geary *et al.* 2018). Future work could quantify whether small mammals perceive predation risk differently in different types of fire mosaics (for example, by using giving up density trials; Doherty *et al.* 2015b; Bleicher *et al.* 2020), and whether their activity patterns and habitat use changes accordingly. This would help to understand whether interactions between invasive predators and fire occur in mallee landscapes and if this is a potential threat to mammal and reptile conservation.

5. Further investigation into the mechanisms underpinning pyrodiversity-biodiversity relationships

Studies into the mechanisms underpinning the specific relationships small mammals and reptiles have with pyrodiversity remain rare. Research could focus on the movement and habitat use of small mammals and reptiles in relation to fire history to understand how constraints such as dispersal, foraging effort and microhabitat availability affect fire recovery. Another option is to take genetic samples of threatened species to determine if different fire patterns influence recolonisation of burnt areas and identify patterns of fire that promote long-term persistence of animal populations.

6. Studies that explore demographic differences in fire responses

Correlative studies (including those presented in this thesis) have provided valuable insight into how fire affects the distribution, populations and composition of animal communities. A useful next step would be to explore how fire relates to population parameters and vital rates of animals. Within a population, individuals can use habitat differently based on age, sex or other demographic parameters. Studies that explore how different demographic groups use different types of fire elements, or respond to different measures of pyrodiversity, could improve our understanding of how fire influences animals. Mechanistic models would also offer opportunities to forecast changes under emerging environmental conditions (Briscoe *et al.* 2019).

7. Applying spatial mapping of pyrodiversity relationships to different ecosystems and taxa

There are many fire-prone ecosystems around the world where we lack a thorough understanding of how pyrodiversity influences fauna, and how fire can best be managed to meet ecological needs. The methods developed in chapters 3 and 4 – namely mapping

the responses of entire fauna assemblages to different measures of pyrodiversity – could help to answer questions regarding how animals respond to different measure of pyrodiversity for a range of taxonomic groups and ecosystems.

Conclusion

This thesis examined the drivers of small mammal and reptile populations and distributions in semi-arid mallee woodlands to answer ecological questions that are relevant to fire-prone ecosystems worldwide. A key finding of my work was that the spatial context of fire is critical to understanding animal responses to fire. For example, I found that several mammal and reptile species were more affected by the amount and configuration of their preferred fire-driven habitat than time since fire, and can even occupy recently burnt environments provided that at least half of the local landscape (<100 ha) is comprised of older vegetation. I also found that fine-scale (<1 ha) amounts and configurations of unburnt patches are important to maintaining reptile diversity in areas subjected to planned burning. Nevertheless, animal responses to fire showed substantial variation, even within taxonomic and functional groups This highlights the importance of species-specific knowledge for informing conservation planning and fire management. Research and management based on a single measure of fire is likely to miss important nuance in pyrodiversity-biodiversity relationships. Instead, different measures of fire need to be incorporated into fire management for more effective conservation outcomes for different groups of animals.

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Appendices – Chapter 2

Appendix S2.1 – Survey dates, effort and weather stations used for each landscape

Table S2.1.1: Survey dates and effort for each landscape used in this study.
*Survey effort per night is presented as either one consistent value for the survey or presented consecutively for each night during the survey. Survey effort varies due to delays in opening some sites and the need to close individual buckets to prevent depredation by ants.

Landscape	Survey number for landscape	Year	Month(s)	Dates	Nights	Survey effort per night*
M01	1	2006	November	26th - 30th	5	100
	2	2007	March	15th - 18th	4	99
	3	2007	October	24th - 28th	5	100
	4	2008	January - February	30th – 4th	6	40, 100, 100, 100, 100, 60
	5	2011	November	22nd – 26th	5	50
	6	2012	March	5th – 9th	5	48, 47, 44, 48, 48
M02	1	2006	November	26th – 30th	5	100
	2	2007	March	14th – 19th	6	50, 100, 100, 100, 100, 50
	3	2007	October	24th – 28th	5	100
	4	2008	January - February	30th – 3rd	5	100
M03	1	2006	December	2nd – 6th	5	100, 98, 100, 100, 100
	2	2007	March	21st – 25th	5	100
	3	2007	October - November	30th – 3rd	5	100
	4	2008	February	7th – 11th	5	100
M04	1	2006	December	2nd – 6th	5	100
	2	2007	March	22nd – 26th	5	100
	3	2007	October - November	30th – 3rd	5	100
	4	2008	February	7th – 11th	5	100
M05	1	2006	October	24th – 28th	5	100
	2	2007	February	21st – 25th	5	100

Landscape	Survey number for landscape	Year	Month(s)	Dates	Nights	Survey effort per night*
	3	2008	March	7th – 11th	5	100
M06	1	2006	October	24th – 28th	5	100
	2	2007	February	21st – 25th	5	100
	3	2007	December	3rd – 7th	5	100
	4	2008	March	7th – 11th	5	100
M07	1	2006	October	24th – 28th	5	100
	2	2007	February	21st – 25th	5	100
	3	2007	December	3rd – 7th	5	100
	4	2008	March	7th – 11th	5	100
M08	1	2006	October	24th – 28th	5	100
	2	2007	February	21st – 25th	5	100
	3	2008	March	7th – 11th	5	100
M09	1	2006	October	24th – 28th	5	100
	2	2007	February	21st – 25th	5	100
	3	2007	December	3rd – 7th	5	100, 95, 100, 100, 100
	4	2008	March	7th – 11th	5	100
M10	1	2006	October	24th – 28th	5	100, 100, 99, 100, 100
	2	2007	February	21st – 25th	5	100
	3	2007	December	3rd – 7th	5	100
	4	2008	March	7th – 11th	5	100
M11	1	2006	November	26th – 30th	5	100
	2	2007	March	22nd – 26th	5	100
	3	2007	October - November	31st – 4th	5	100
	4	2008	January	23rd – 27th	5	100
	5	2011	November	8th – 12th	5	80, 78, 80, 80, 80
	6	2012	February	20th – 24th	5	57, 74, 71, 73, 77
M12	1	2006	November	26th – 30th	5	100
	2	2007	March	22nd – 26th	5	100
	3	2007	October - November	31st – 4th	5	100
	4	2008	January	23rd – 27th	5	99, 100, 100, 99, 100

Landscape	Survey number for landscape	Year	Month(s)	Dates	Nights	Survey effort per night*
	5	2011	November	8th – 12th	5	50, 49, 50, 50, 50
	6	2012	February	20th -24th	5	50, 49, 40, 49, 49
M13	1	2006	November	20th – 24th	5	100
	2	2007	March	21st – 25th	5	100
	3	2007	October	18th – 22nd	5	100
	4	2007	December	3rd – 7th	5	100
	5	2008	January - February	30th – 3rd	5	100
M14	1	2007	November	20th – 24th	5	100
	2	2007	March	21st – 25th	5	100
	3	2007	October	19th – 23nd	5	100
	4	2007	December	3rd – 7th	5	100
	5	2008	January - February	30th – 3rd	5	100
M15	1	2006	November	26th – 28th	3	100
	2	2007	March	14th – 18th	5	100
	3	2007	October	17th – 21st	5	100
	4	2007	November	26th – 30th	5	100
	5	2008	February	5th – 9th	5	100
M16	1	2006	November	26th – 30th	5	100
	2	2007	March	14th – 18th	5	100
	3	2007	November	26th – 30th	5	100
	4	2008	February	5th – 9th	5	100
M19	1	2006	October	17th – 21st	5	100
	2	2007	February	14th – 18th	5	100, 100, 99, 100, 100
	3	2007	November	26th -30th	5	100
	4	2008	February - March	29th – 4th	5	100
M20	1	2006	October	17th – 21st	5	100
	2	2007	February	14th – 18th	5	100
	3	2007	November	26th – 30th	5	100
	4	2008	February - March	29th – 4th	5	100
M21	1	2006	October	17th – 21st	5	100

Landscape	Survey number for landscape	Year	Month(s)	Dates	Nights	Survey effort per night*
	2	2007	February	14th – 19th	6	50, 100, 95, 94, 99, 48
	3	2007	November	26th – 30th	5	100, 100, 98, 94, 96
	4	2008	February - March	29th – 4th	5	100
M22	1	2006	October	17th – 21st	5	100
	2	2007	February	14th – 18th	5	100
	3	2007	November	26th – 30th	5	100
	4	2008	February - March	29th – 4th	5	99, 100, 100, 100, 98
M23	1	2006	October	17th – 21st	5	100
	2	2007	February	14th – 18th	5	100
	3	2007	November	26th – 30th	5	100
	4	2008	February – March	29th – 4th	5	100
	5	2011	October	28th – 31st	4	48, 45, 44, 44
M24	1	2006	October	17th – 21st	5	100
	2	2007	February	14th – 18th	5	100, 100, 96, 96, 100
	3	2007	November	26th – 30th	5	100
	4	2008	February – March	29th – 4th	5	100
	5	2011	October	28th – 31st	4	49, 48, 48, 48
M25	1	2006	November	26th – 30th	5	100
	2	2007	March	14th – 18th	5	100
	3	2007	October - November	30th – 3rd	5	100
	4	2008	January	23rd – 27th	5	100
M26	1	2006	November	26th – 30th	5	100
	2	2007	March	14th – 18th	5	100
	3	2007	October- November	30th – 3rd	5	100
	4	2008	January	23rd – 27th	5	100, 99, 99, 99, 99
M27	1	2006	November	19th – 23rd	5	100
	2	2007	March	21st – 25th	5	100
	3	2007	October	24th – 28th	5	100
	4	2008	January - February	30th – 3rd	5	100
M28	1	2006	November	19th – 23rd	5	100

Landscape	Survey number for landscape	Year	Month(s)	Dates	Nights	Survey effort per night*
	2	2007	March	21st – 25th	5	100
	3	2007	October	24th – 28th	5	100
	4	2008	January - February	30th – 3rd	5	100
MH01	1	2011	November	22nd – 26th	5	70, 69, 70, 70, 70
	2	2012	March	5th – 9th	5	70, 69, 66, 70, 70

Table S2.1.2: Weather stations where data was sourced for the five weather variables for each of the 27 study landscapes

Landscape	Variable source (weather station)				
	Min. temperature	**Precipitation**	**Relative humidity**	**Wind speed**	**Cloud cover**
M01	Mildura Airport	Mildura Airport	Mildura Airport	Mildura Airport	Mildura Airport
M02	Ouyen	Ouyen	Walpeup	Walpeup	Mildura Airport
M03	Walpeup	Walpeup	Walpeup	Walpeup	Mildura Airport
M04	Walpeup	Walpeup	Walpeup	Walpeup	Mildura Airport
M05	Mildura Airport	Mildura Airport	Mildura Airport	Mildura Airport	Mildura Airport
M06	Walpeup	Walpeup	Walpeup	Walpeup	Mildura Airport
M07	Walpeup	Walpeup	Walpeup	Walpeup	Mildura Airport
M08	Mildura Airport	Mildura Airport	Mildura Airport	Mildura Airport	Mildura Airport
M09	Mildura Airport	Mildura Airport	Mildura Airport	Mildura Airport	Mildura Airport
M10	Walpeup	Walpeup	Walpeup	Walpeup	Mildura Airport
M11	Loxton	Loxton	Loxton	Loxton	Mildura Airport
M12	Loxton	Loxton	Loxton	Loxton	Mildura Airport
M13	Gluepot	Gluepot	Renmark	Renmark	Mildura Airport
M14	Gluepot	Gluepot	Renmark	Renmark	Mildura Airport
M15	Gluepot	Gluepot	Renmark	Renmark	Mildura Airport
M16	Gluepot	Gluepot	Renmark	Renmark	Mildura Airport
M19	Gluepot	Gluepot	Renmark	Renmark	Mildura Airport
M20	Gluepot	Gluepot	Broken Hill	Broken Hill	Mildura Airport
M21	Gluepot	Gluepot	Broken Hill	Broken Hill	Mildura Airport
M22	Gluepot	Gluepot	Renmark	Renmark	Mildura Airport
M23	Gluepot	Gluepot	Renmark	Renmark	Mildura Airport
M24	Gluepot	Gluepot	Renmark	Renmark	Mildura Airport
M25	Mildura Airport	Mildura Airport	Mildura Airport	Mildura Airport	Mildura Airport
M26	Mildura Airport	Mildura Airport	Mildura Airport	Mildura Airport	Mildura Airport
M27	Pooncarie	Pooncarie	Mildura Airport	Mildura Airport	Mildura Airport
M28	Pooncarie	Pooncarie	Mildura Airport	Mildura Airport	Mildura Airport
MH01	Ouyen	Ouyen	Walpeup	Walpeup	Mildura Airport

Appendix S2.2 – Explanatory variable distributions

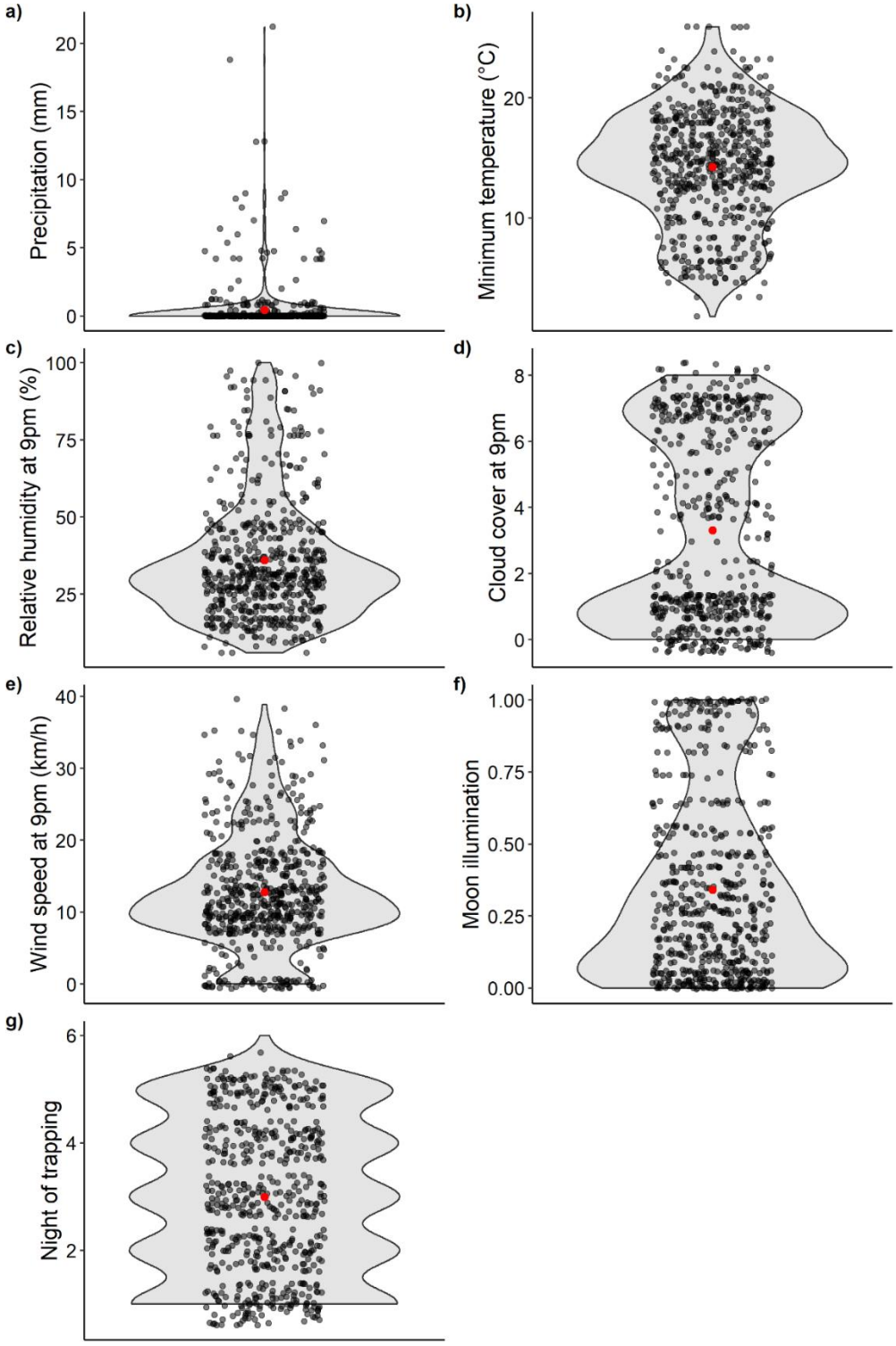


Figure S2.2.1: Violin plots of the seven explanatory variables used in our analysis. Variables are; a) Precipitation in the 24hrs prior to 12 am on the night of trapping (mm); b) Minimum temperature on the night of trapping (°C); c) Relative humidity at 9 pm (%); d) Cloud cover at 9 pm; e) Wind speed at 9 pm (km/h); f) Moon illumination; g) Night of trapping

Table S2.2.1: Summary statistics of the seven explanatory variables used in the analysis

Variable	Range	Median	Mean
Precipitation 24 hours prior to 9 pm (mm)	0.00 - 21.20	0.00	0.45
Minimum temperature (°C)	1.80 - 25.90	12.10	11.85
Relative humidity at 9pm (%)	6.00 - 100.00	31.10	36.03
Cloud cover at 9pm	0.00 - 8.00	2.00	3.31
Wind speed at 9pm (km/h)	0.00 - 38.90	11.20	12.78
Moon illumination (%)	0.00 - 100.00	26.00	34.00
Night of trapping	1.00 - 6.00	N/A	N/A

Table S2.2.2: Species included in each individual and family model used in our analysis.

Model type	Species	Family
Individual	*Cercartetus concinnus* (western pygmy possum)	Burraymid
Individual	*Mus musculus* (house mouse)	Rodent
Individual	*Pseudomys bolami* (Bolam's mouse)	Rodent
Individual	*Ningaui yvonneae* (mallee ningaui)	Dasyurid
Individual	*Sminthopsis murina* (common dunnart)	Dasyurid
Family	*Cercartetus concinnus* (western pygmy possum), *Cercartetus lepidus* (little pygmy possum)	Burraymid
Family	*Mus musculus* (house mouse), *Notomys mitchellii* (Mitchell's hopping mouse), *Pseudomys bolami* (Bolam's mouse)	Rodent
Family	*Ningaui yvonneae* (mallee ningaui), *Sminthopsis murina* (common dunnart)	Dasyurid

Appendix S2.3 – Model coefficients (from GLMMs) for conditional and zero-inflated models

Table S2.3.1: Results of GLMMs testing the influence of weather and moon phase on capture rates of small mammals. Standardised model coefficients (± SE) are reported (following Gelman and Hill 2007) for the conditional (negative-binomial) and zero-inflated models (where fitted). Results in bold indicate those where the 95% confidence intervals do not overlap zero.

Response variable	Precip.	Min. temp.	Rel. humid.	Cloud cover	Wind speed	Night	Moon illum.
	Conditional model coefficient (± SE) of predictor variable						
C. concinnus	0.13 ± 0.31	0.36 ± 0.27	**1.28 ± 0.21**	**-1 ± 0.23**	**1.99 ± 0.3**	**-1.17 ± 0.23**	**1.27 ± 0.45**
M. musculus	-0.16 ± 0.1	-0.39 ± 0.25	**0.87 ± 0.27**	0.11 ± 0.21	**1.17 ± 0.35**	**-0.62 ± 0.3**	0.34 ± 0.39
N. yvonneae	-0.23 ± 0.15	**0.49 ± 0.09**	0.12 ± 0.1	-0.12 ±0.095	**0.44 ± 0.08**	**-0.31 ± 0.08**	**-0.52 ± 0.14**
P. bolami	0.26 ± 0.26	**-1.13 ± 0.48**	-0.44 ± 0.5	0.93 ± 0.57	0.25 ± 0.34	0.3 ± 0.39	**-2.13 ± 0.59**
S. murina	0.13 ± 0.18	0.22 ± 0.14	**-0.61 ± 0.16**	**0.47 ± 0.13**	**0.3 ± 0.13**	**-0.31 ± 0.12**	-0.19 ± 0.17
Burramyids	0.07 ± 0.27	0.16 ± 0.26	**1.11 ± 0.2**	**-0.59 ± 0.22**	**1.22 ± 0.29**	**-0.75 ± 0.23**	**1 ± 0.41**
Dasyurids	-0.12 ± 0.12	**0.38 ± 0.08**	-0.12 ± 0.09	0.09 ± 0.08	**0.41 ± 0.07**	**-0.3 ± 0.07**	**-0.35 ± 0.11**
Rodents	-0.1 ± 0.1	**-0.47 ± 0.21**	**0.71 ± 0.2**	0.05 ± 0.2	**0.56 ± 0.23**	**-0.63 ± 0.19**	0.41 ± 0.3
	Zero-inflated model coefficient (± SE) of predictor variable						
C. concinnus	0.47 ± 0.55	-1.25 ± 0.67	0.6 ± 0.58	-0.82 ± 0.6	**1.58 ± 0.69**	**-2.31 ± 0.67**	**1.52 ± 0.56**
M. musculus	-0.15 ± 0.28	0.13 ± 0.47	0.3 ± 0.49	0.34 ± 0.4	**1.31 ± 0.63**	-0.68 ± 0.5	0.09 ± 0.42
Burramyids	-0.09 ± 0.55	-0.37 ± 0.57	0.52 ± 0.47	-0.4 ± 0.49	0.45 ± 0.49	**-1.19 ± 0.53**	**1.18 ± 0.44**
Rodents	-0.15 ± 0.28	0.02 ± 0.4	0.29 ± 0.38	0.27 ± 0.4	0.43 ± 0.4	**-0.76 ± 0.37**	0.11 ± 0.4

Appendix S2.4 – Response plots of each species and family to meteorological variables

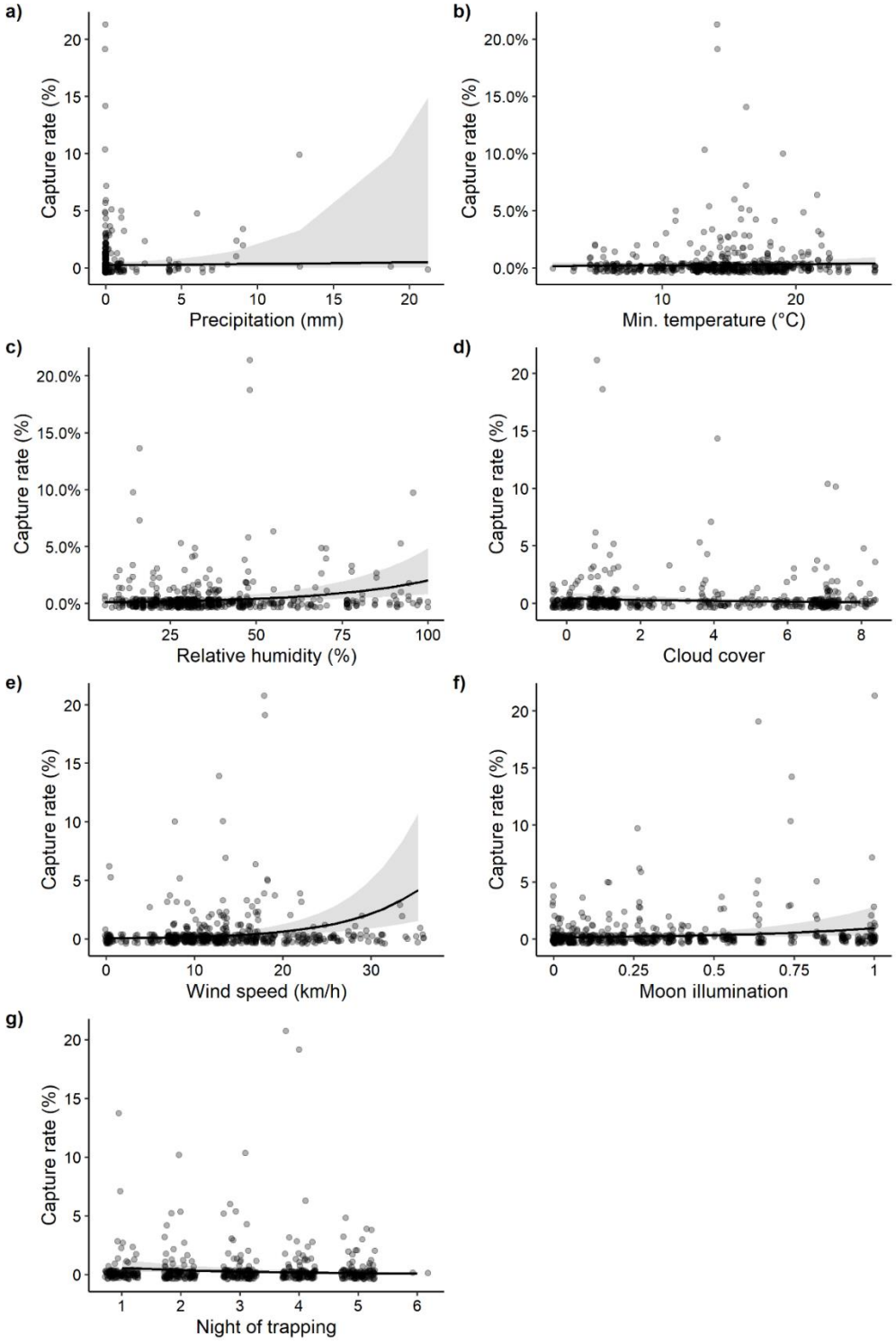


Figure S2.4.1: Response of capture rates of *Cercartetus concinnus* to; a) precipitation in the 24 hours prior to 00:00; b) minimum overnight temperature; c) relative humidity at 21:00; d) cloud cover at 21:00; e) wind speed at 21:00; f) moon illumination between 1700 and 2100; g) night of trapping within a single consecutive period

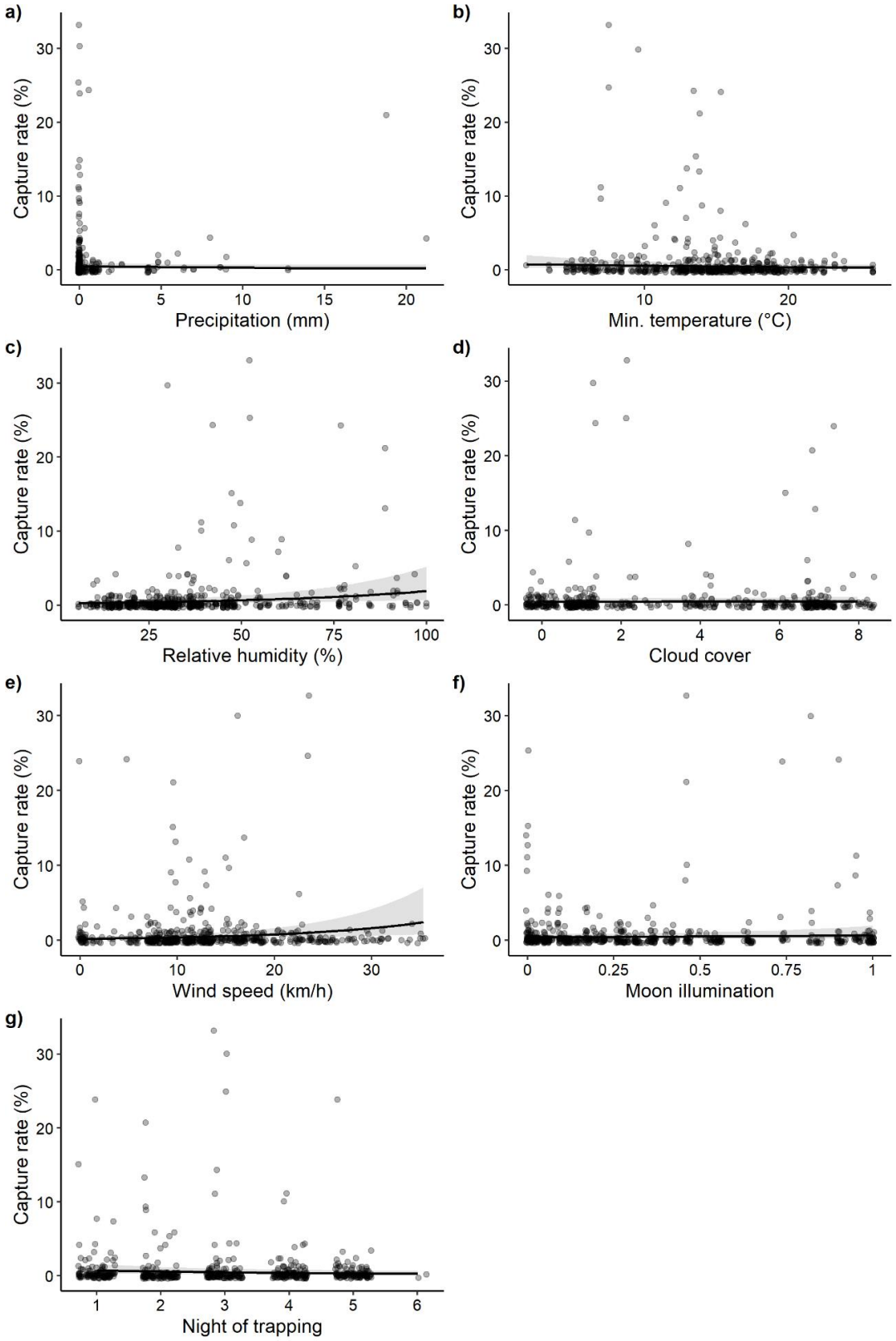


Figure S2.4.2: Response of capture rates of *Mus musculus* to; a) precipitation in the 24 hours prior to 00:00; b) minimum overnight temperature; c) relative humidity at 21:00; d) cloud cover at 21:00; e) wind speed at 21:00; f) moon illumination between 1700 and 2100; g) night of trapping within a single consecutive period

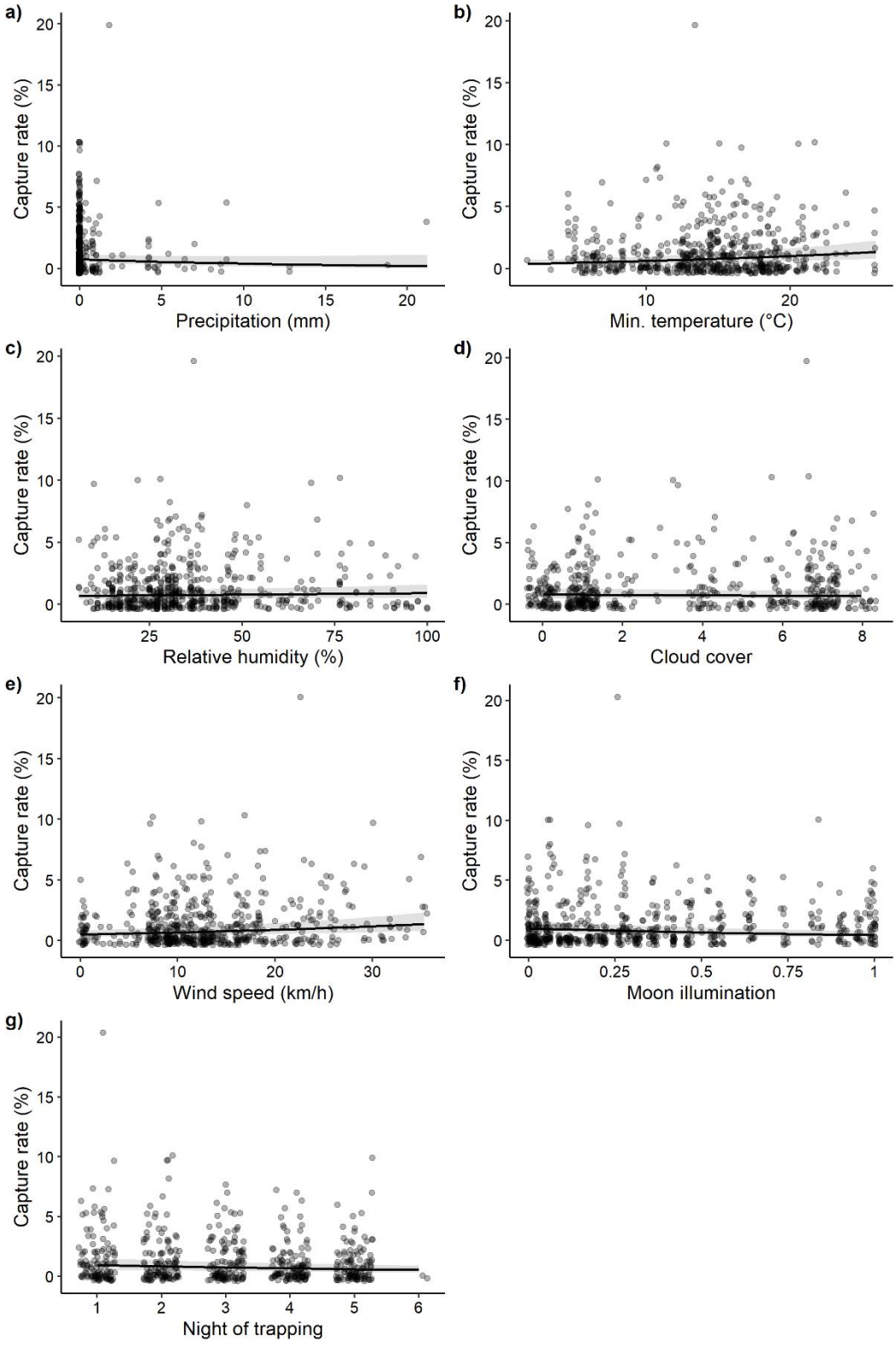


Figure S2.4.3: Response of capture rates of *Ningaui yvonneae* to; a) precipitation in the 24 hours prior to 00:00; b) minimum overnight temperature; c) relative humidity at 21:00; d) cloud cover at 21:00; e) wind speed at 21:00; f) moon illumination between 1700 and 2100; g) night of trapping within a single consecutive period

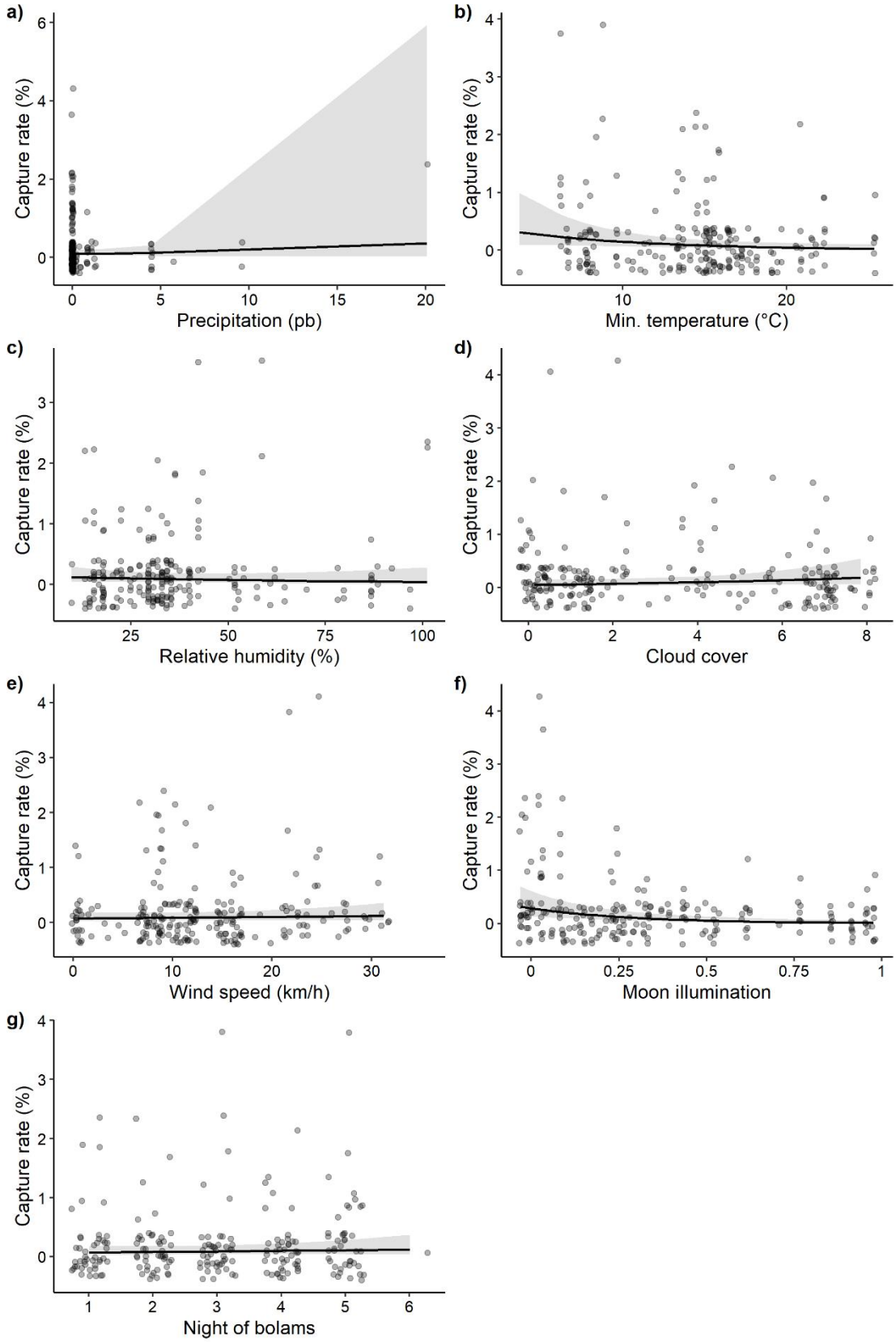


Figure S2.4.4: Response of capture rates of *Pseudomys bolami* to; a) precipitation in the 24 hours prior to 00:00; b) minimum overnight temperature; c) relative humidity at 21:00; d) cloud cover at 21:00; e) wind speed at 21:00; f) moon illumination between 1700 and 2100; g) night of trapping within a single consecutive period

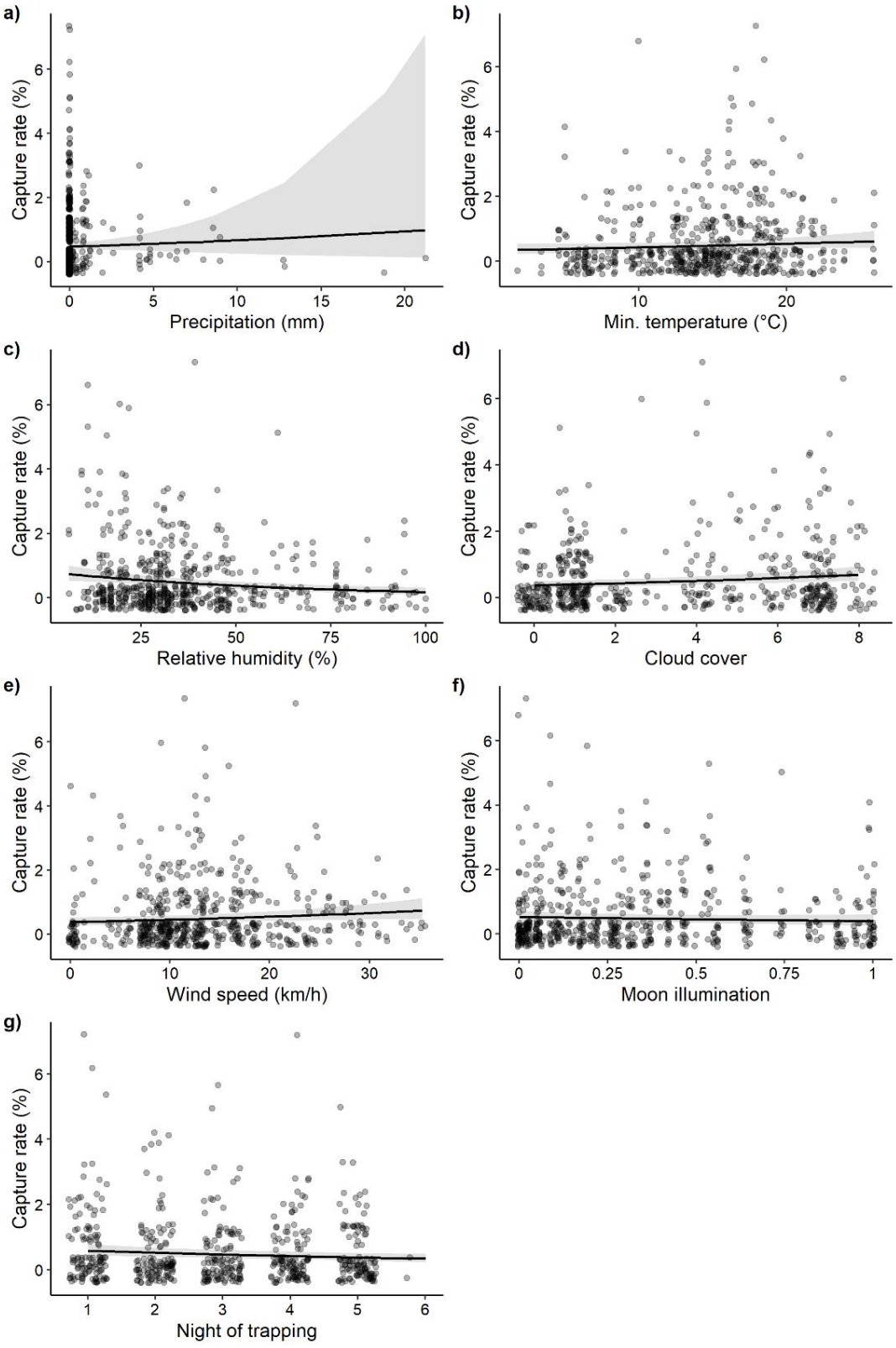


Figure S2.4.5: Response of capture rates of *Sminthopsis murina* to; a) precipitation in the 24 hours prior to 00:00; b) minimum overnight temperature; c) relative humidity at 21:00; d) cloud cover at 21:00; e) wind speed at 21:00; f) moon illumination between 1700 and 2100; g) night of trapping within a single consecutive period

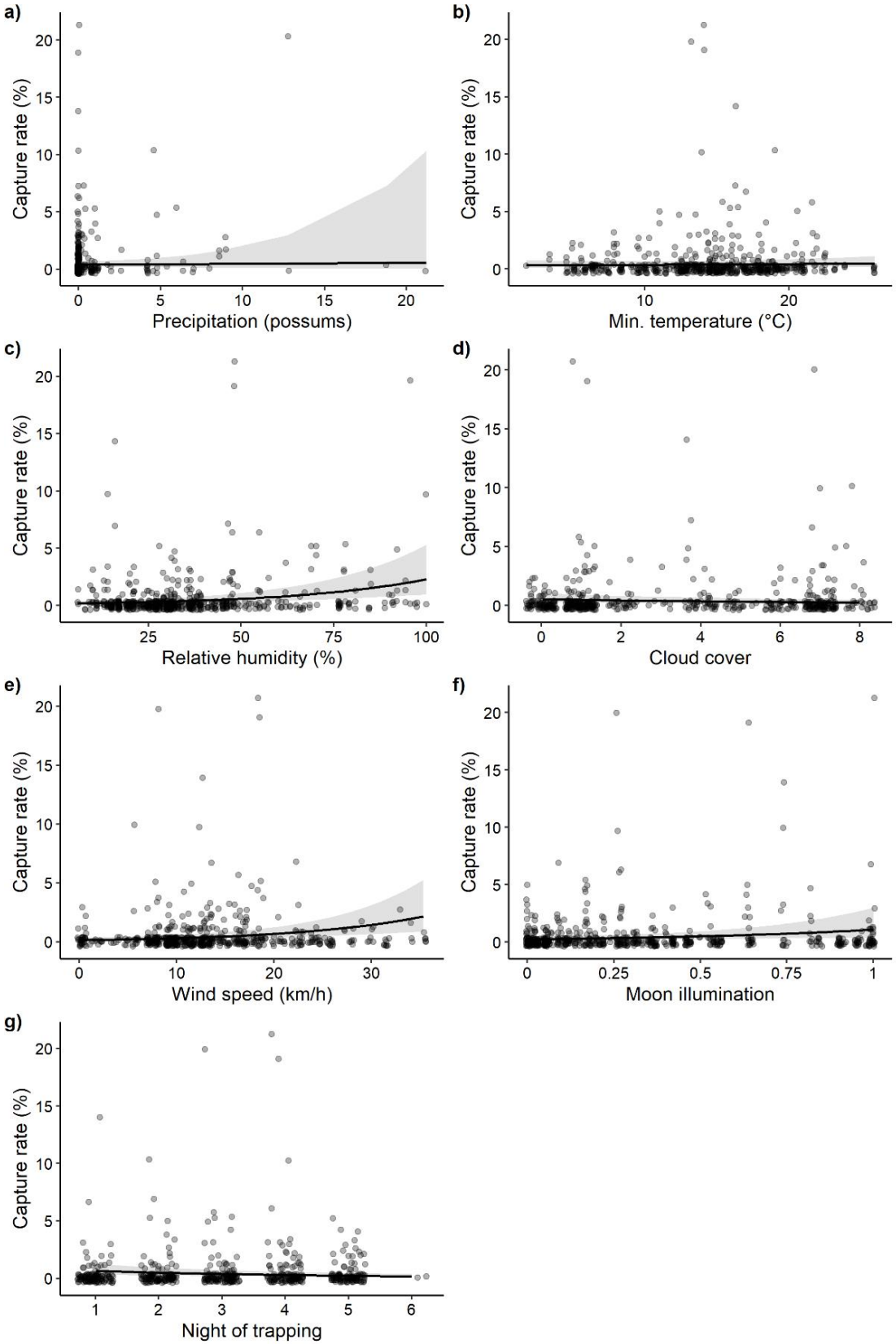


Figure S2.4.6: Response of capture rates of the burramyid family to; a) precipitation in the 24 hours prior to 00:00; b) minimum overnight temperature; c) relative humidity at 21:00; d) cloud cover at 21:00; e) wind speed at 21:00; f) moon illumination between 1700 and 2100; g) night of trapping within a single consecutive period

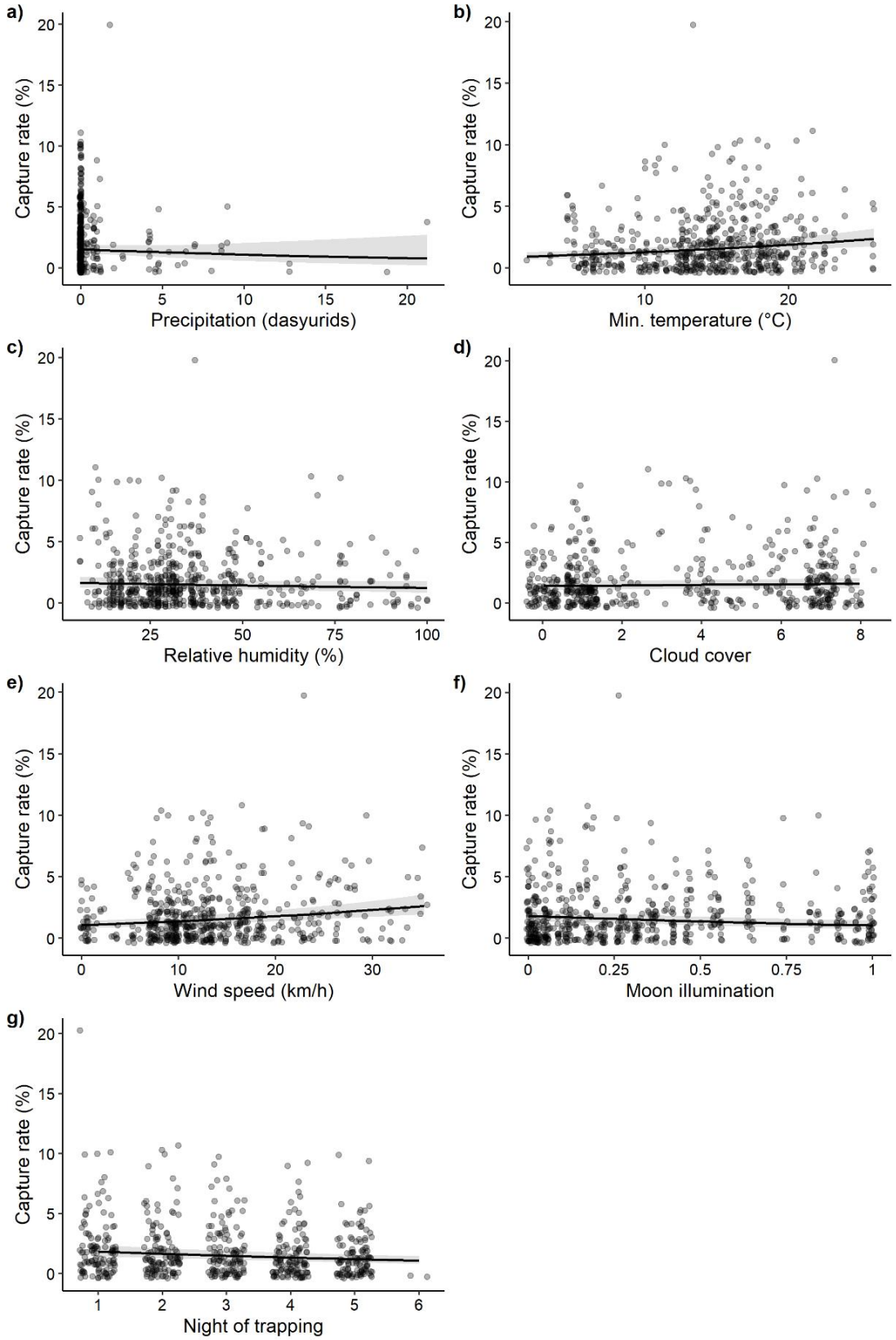


Figure S2.4.7: Response of capture rates of the dasyurid family to; a) precipitation in the 24 hours prior to 00:00; b) minimum overnight temperature; c) relative humidity at 21:00; d) cloud cover at 21:00; e) wind speed at 21:00; f) moon illumination between 1700 and 2100; g) night of trapping within a single consecutive period

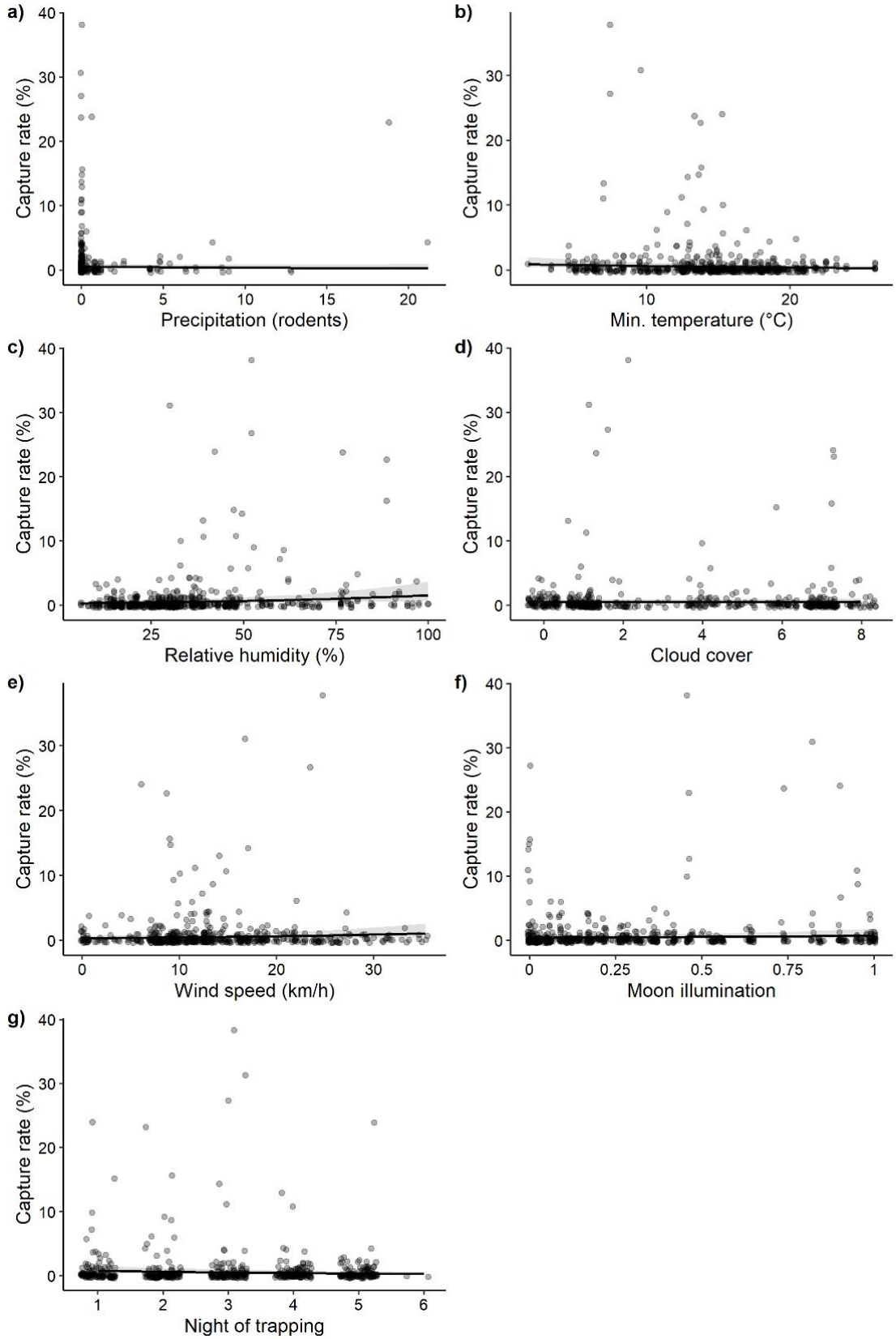


Figure S2.4.8: Response of capture rates of the rodent family to; a) precipitation in the 24 hours prior to 00:00; b) minimum overnight temperature; c) relative humidity at 21:00; d) cloud cover at 21:00; e) wind speed at 21:00; f) moon illumination between 1700 and 2100; g) night of trapping within a single consecutive period

Appendices – Chapter 3

Appendix S3.1 – Variable contributions of each predictor variable

Table S3.1.1: Contribution (%) of each predictor variable to M1 for 17 mammal species. Variable contribution is the amount each variable contributed to the gain of M1 during training, converted to a percentage.

Taxon	Species	Time since fire	Area mid-old triodia	Area early age class	Diversity	Annual rainfall	Elevation	Vegetation type
Small marsupial	*Cercartetus concinnus*	57.71	7.95		4.95	21.64	3.66	4.09
Small marsupial	*Cercartetus lepidus*	0.13	1.82		0.29	62.48	32.04	3.25
Small marsupial	*Ningaui yvonneae*	16.02	61.46		11.81	1.28	8.09	1.35
Rodent	*Notomys mitchellii*	40.33	1.15		0.00	38.74	0.00	19.78
Rodent	*Pseudomys bolami*	5.60	26.88		6.22	58.54	2.77	0.00
Small marsupial	*Sminthopsis murina*	44.53	9.08		9.93	23.65	5.92	6.91
Microbat	*Chalinolobus gouldii*	8.08		5.21	15.24	35.15	16.86	19.46
Microbat	*Chalinolobus morio*	0.93		11.82	8.12	75.28	0.27	3.59
Microbat	*Chalinolobus picatus*	0.01		8.20	30.42	25.72	0.89	34.72
Microbat	*Ozimops petersi*	1.17		0.05	4.98	0.38	0.00	93.41
Microbat	*Nyctophilus corbeni*	6.20		18.02	8.89	53.95	1.48	11.44
Microbat	*Nyctophilus geoffroyi*	8.01		4.28	8.77	44.69	7.64	26.61
Microbat	*Scotorepens balstoni*	0.07		0.84	37.64	3.26	0.60	57.59
Microbat	*Vespadelus baverstocki*	0.60		18.17	12.07	13.35	5.85	49.96
Microbat	*Vespadelus regulus*	0.00		0.00	0.00	53.26	46.17	0.56
Macropod	*Macropus fuliginosus*	0.56		27.05	16.38	25.44	27.58	2.98
Monotreme	*Tachyglossus aculeatus*	9.14		0.96	0.28	74.86	3.06	11.70

Table S3.1.2: Contribution (%) of each predictor variable to M2 for 17 mammal species. Variable contribution is the amount each variable contributed to the gain of M2 during training, converted to a percentage.

Taxon	Species	Time since fire	ConFigure mid-old triodia	ConFigure early age class	Diversity	Annual rainfall	Elevation	Vegetation type
Small marsupial	*Cercartetus concinnus*	55.3	10.1		7.8	20.9	2.2	3.7
Small marsupial	*Cercartetus lepidus*	0.3	6.1		0.2	61	31.3	1.1
Small marsupial	*Ningaui yvonneae*	13.9	62.1		10	2.6	7.4	4.1
Rodent	*Notomys mitchellii*	41.2	0		0	39.4	0	19.4
Rodent	*Pseudomys bolami*	5.3	20.1		4.8	66.2	2.4	1.3
Small marsupial	*Sminthopsis murina*	42.9	13.2		11.44	21.3	6	5.2
Microbat	*Chalinolobus gouldii*	8.8		5.7	14.6	34.9	16.5	19.6
Microbat	*Chalinolobus morio*	0.8		9.2	6.2	79.2	0.2	4.4
Microbat	*Chalinolobus picatus*	0		5.3	31.3	26	1	36.3
Microbat	*Ozimops petersi*	1.2		0	5	0.4	0	93.5
Microbat	*Nyctophilus corbeni*	9.8		4	10.8	62.4	1.2	11.8
Microbat	*Nyctophilus geoffroyi*	5.9		11.4	6.5	42.7	7.1	26.3
Microbat	*Scotorepens balstoni*	0		1.1	35.8	4.4	1	57.6
Microbat	*Vespadelus baverstocki*	0.9		8.3	10.5	8.2	7.8	64.4
Microbat	*Vespadelus regulus*	0		0	0	53.3	46.2	0.6
Macropod	*Macropus fuliginosus*	4.1		27.7	3.9	31.9	28.9	3.4
Monotreme	*Tachyglossus aculeatus*	8.5		0.1	0.3	76	3.1	12

Table S3.1.3: Contribution (%) of each predictor variable to M2 for 17 mammal species. Variable contribution is the amount each variable contributed to the gain of M2 during training, converted to a percentage.

Taxon	Species	Fire freq.	Area mid-old triodia	Area early age class	Diversity	Annual rainfall	Elevation	Vegetation type
Small marsupial	*Cercartetus concinnus*	23.3	4.5		22.4	39.5	2.8	7.6
Small marsupial	*Cercartetus lepidus*	0	1.8		0.3	62.3	32.3	3.3
Small marsupial	*Ningaui yvonneae*	12.7	61.9		16.3	0.8	7	1.4
Rodent	*Notomys mitchellii*	0	0.9		0	73.3	0	25.8
Rodent	*Pseudomys bolami*	1.3	29.4		7.4	59.9	2.1	0
Small marsupial	*Sminthopsis murina*	32.6	8.1		31.5	20.4	1.3	6
Microbat	*Chalinolobus gouldii*	8.5		4.2	14.5	40.4	14.7	17.8
Microbat	*Chalinolobus morio*	12.7		7.4	10.7	60.7	4.6	3.8
Microbat	*Chalinolobus picatus*	0		8.2	30.4	25.7	0.9	34.7
Microbat	*Ozimops petersi*	0		0.1	4	0.3	0	95.7
Microbat	*Nyctophilus corbeni*	5.3		21	8.9	53.7	0.6	10.4
Microbat	*Nyctophilus geoffroyi*	19		2.4	10.6	38.5	5.6	23.9
Microbat	*Scotorepens balstoni*	0.1		0.9	37.6	3.6	0.6	57.3
Microbat	*Vespadelus baverstocki*	4.4		17.2	11.8	13	5.9	47.7
Microbat	*Vespadelus regulus*	1.4		0.8	0	53	43.9	0.8
Macropod	*Macropus fuliginosus*	1.5		27.2	15.5	25.1	27.8	3
Monotreme	*Tachyglossus aculeatus*	0		0	0	86.1	0	13.9

Appendix S3.2 – Response plots of mallee mammals to predictors

Preface: Figures in this appendix are formatted as embedded PDF files and open externally for enhanced viewing

a)

b)

c)

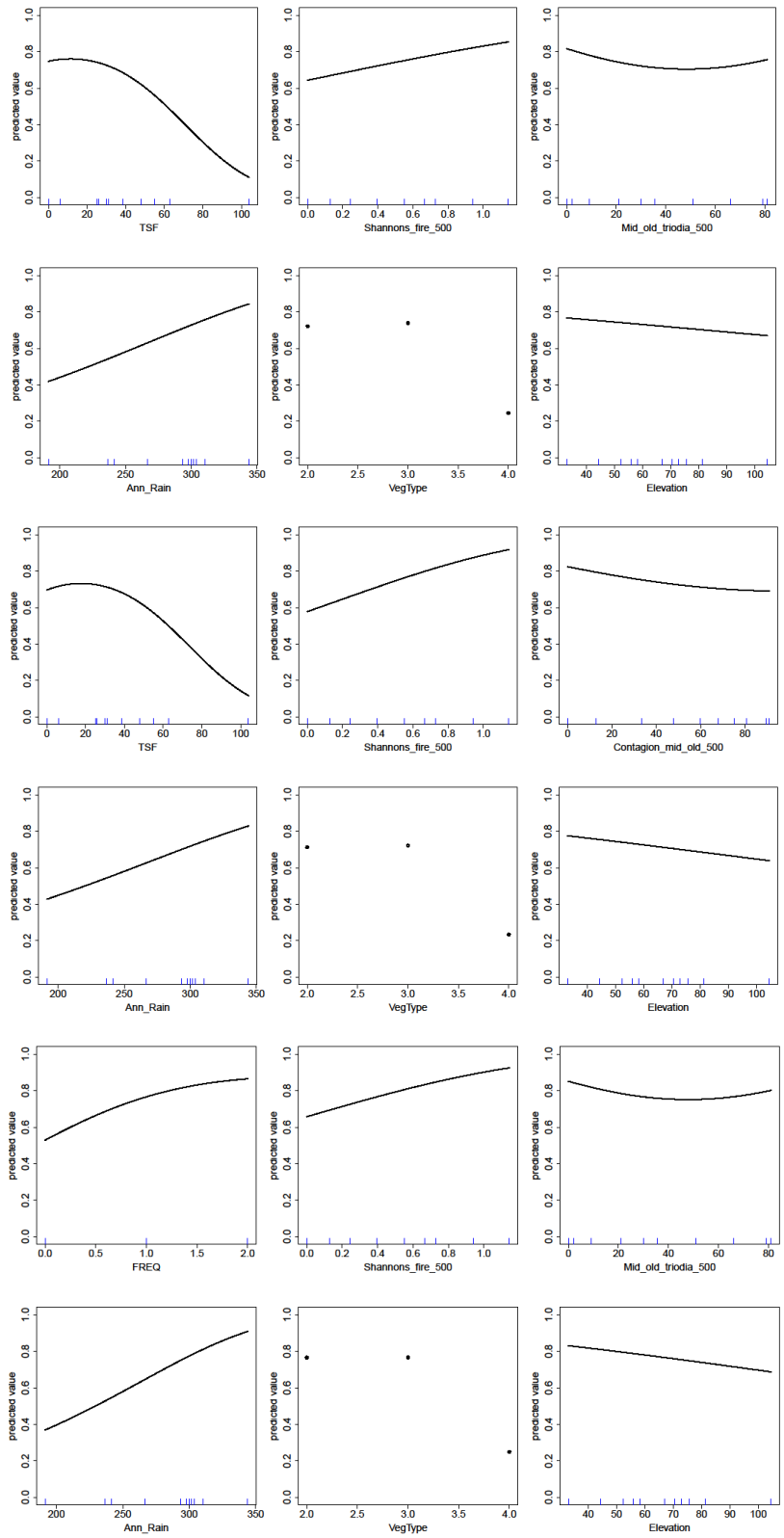
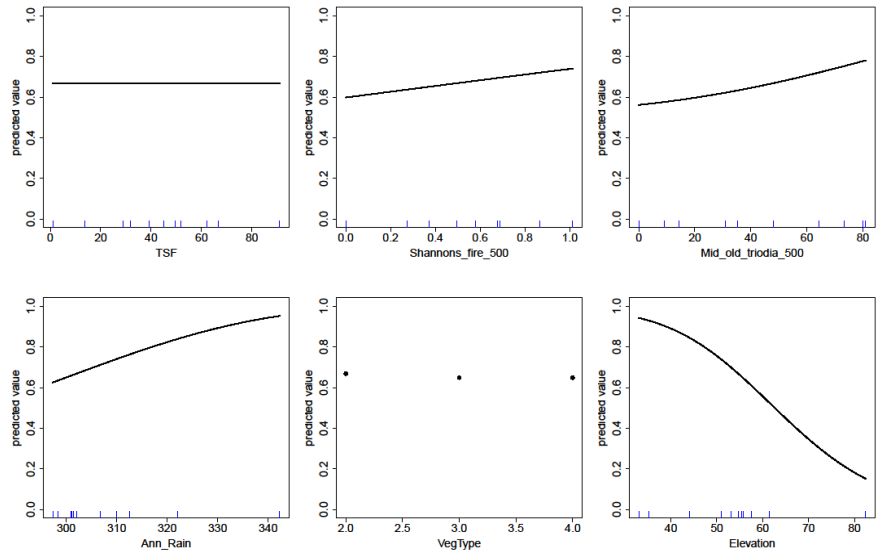


Figure S3.2.1: Response of *Cercartetus concinnus* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

a)



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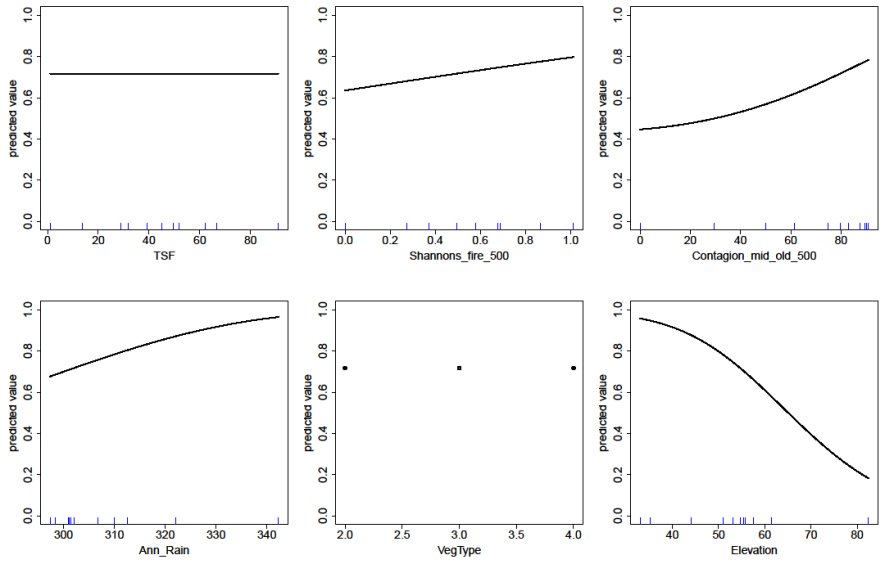

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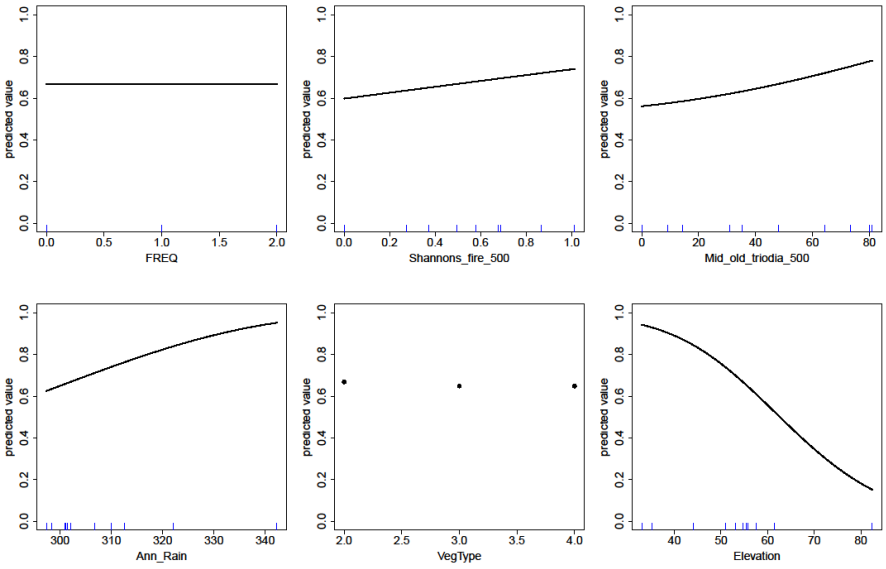
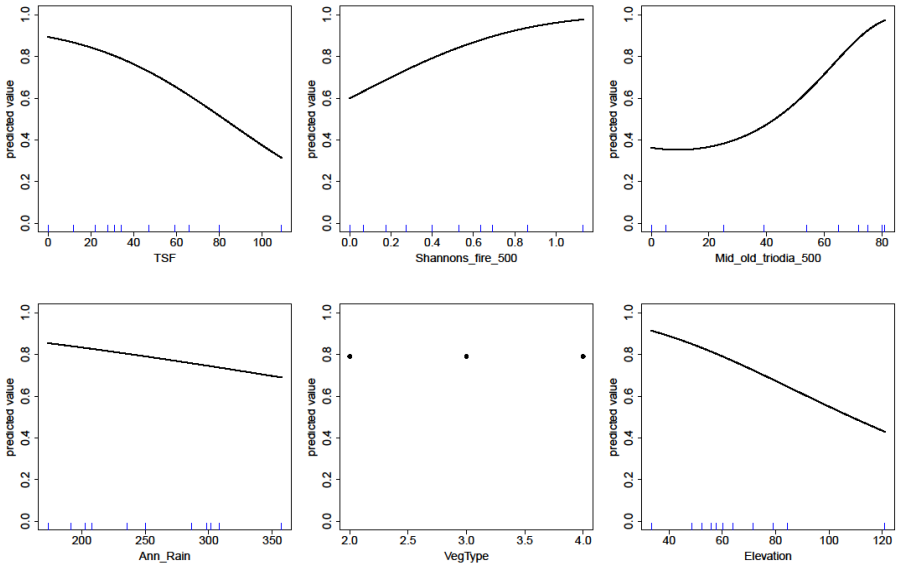


Figure S2.3.2: Response of *Cercartetus lepidus* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

a)



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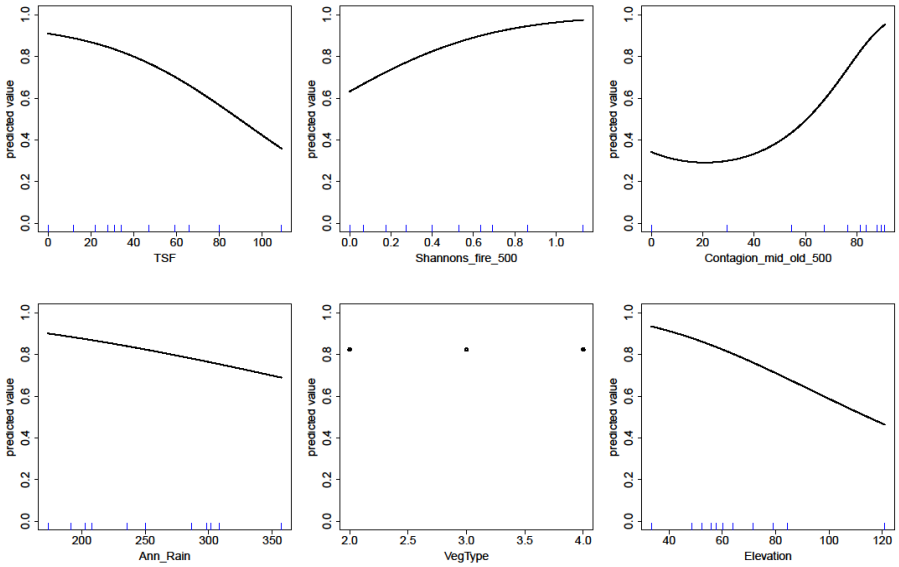

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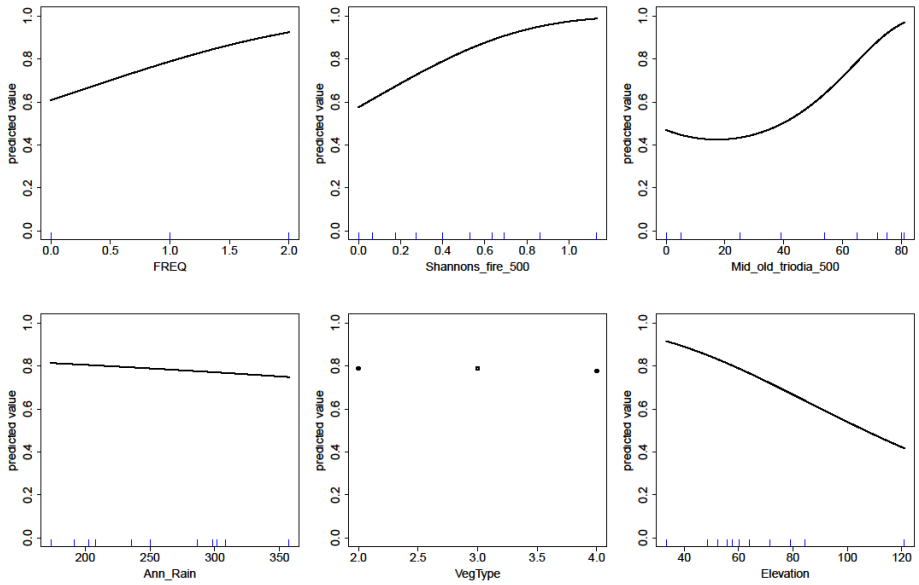


Figure S3.2.3: Response of *Ningaui yvonneae* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

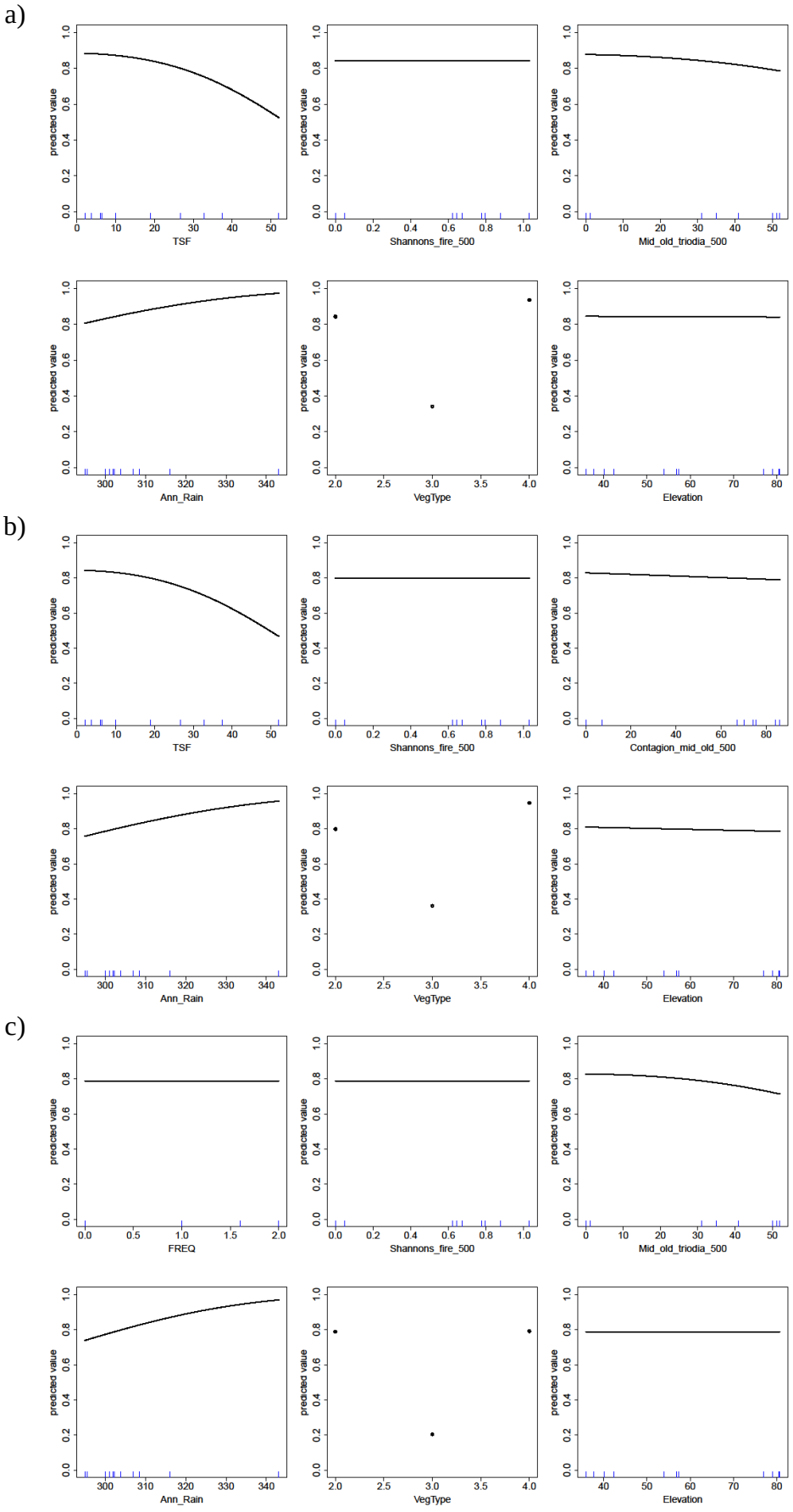
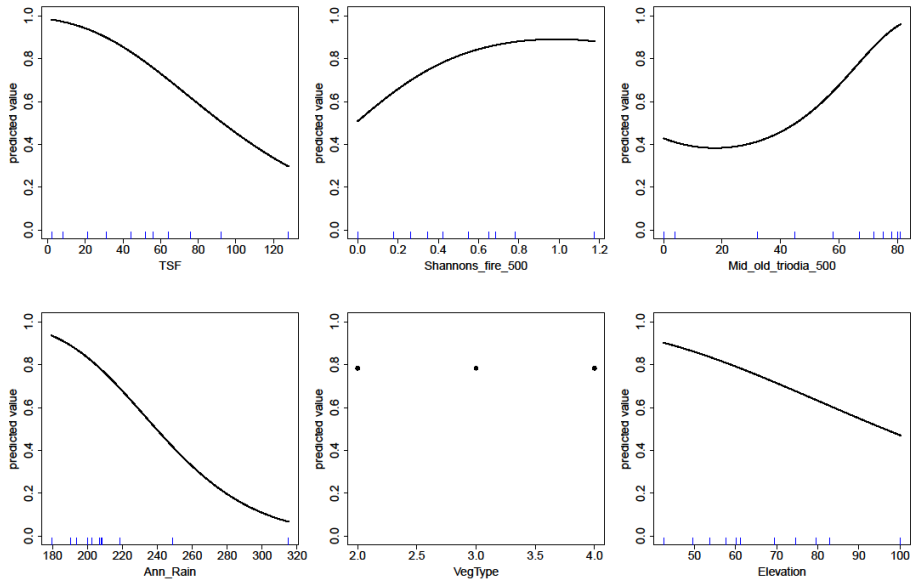


Figure S3.2.4: Response of *Notomys mitchellii* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

a)



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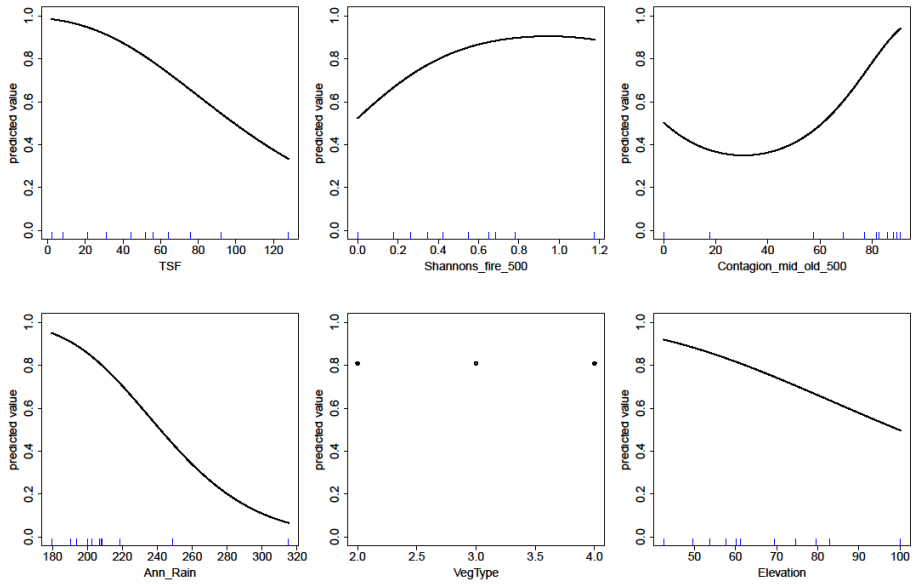

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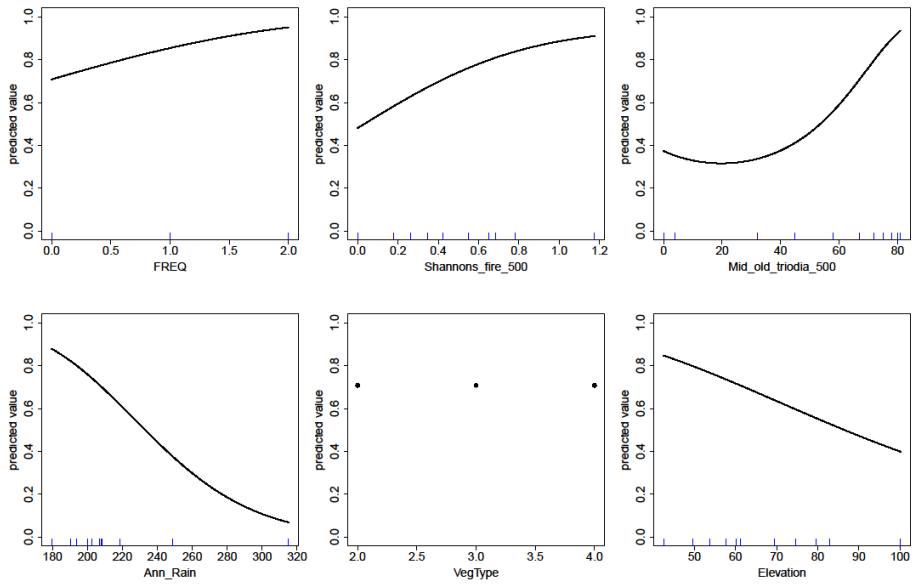


Figure S3.2.5: Response of *Pseudomys bolami* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

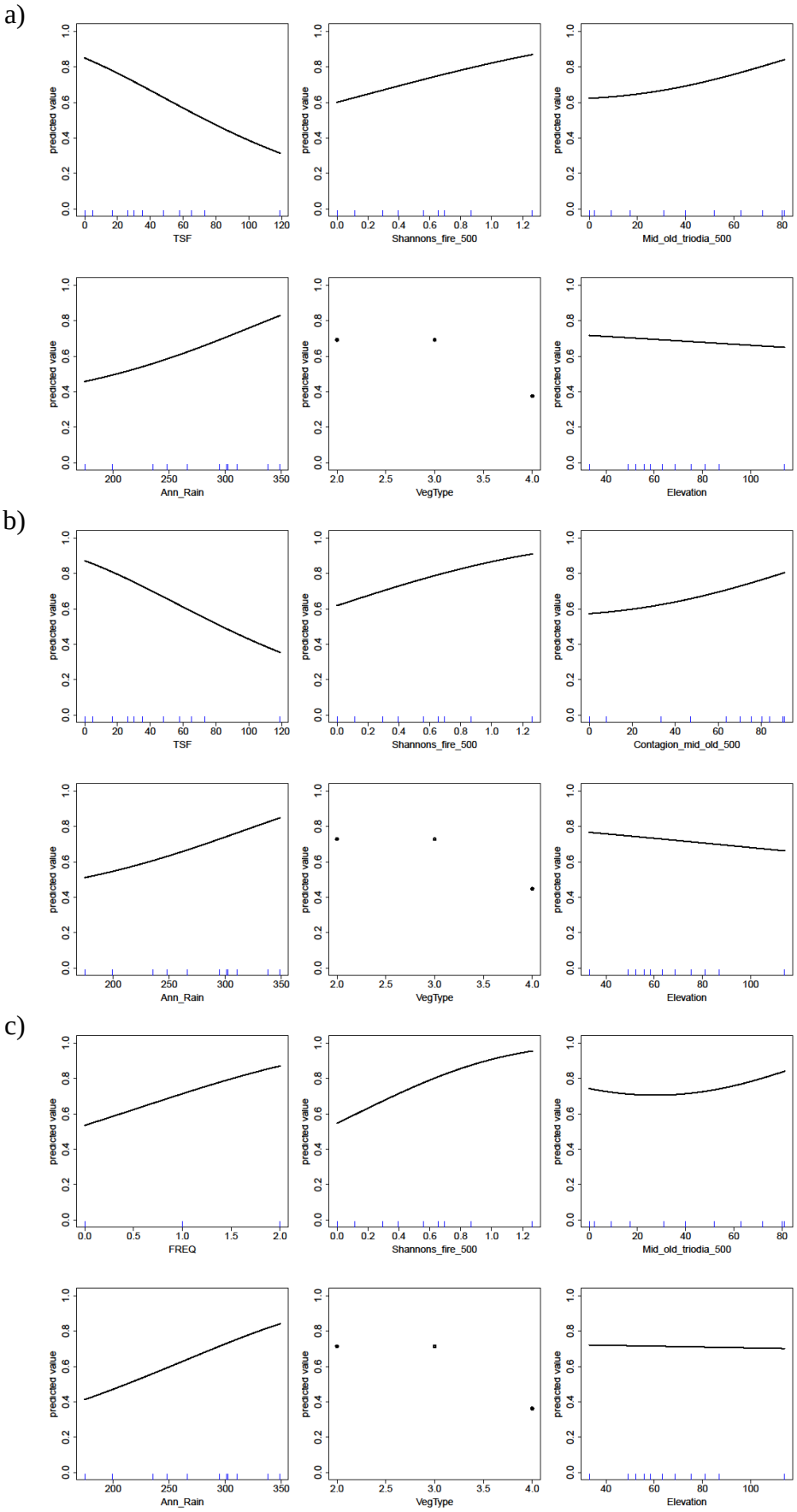


Figure S3.2.6: Response of *Sminthopsis murina* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

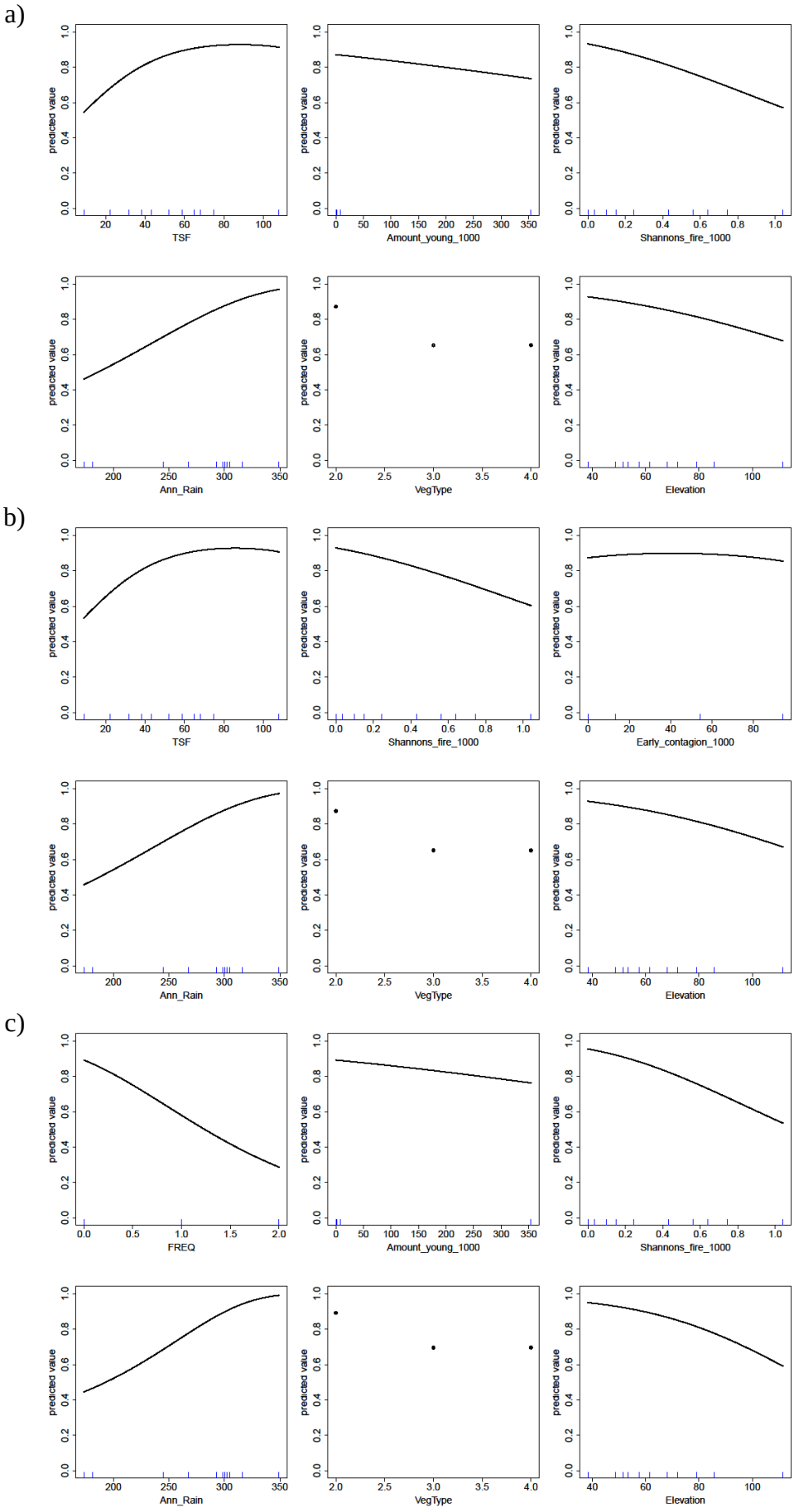


Figure S3.2.7: Response of *Chalinolobus gouldii* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

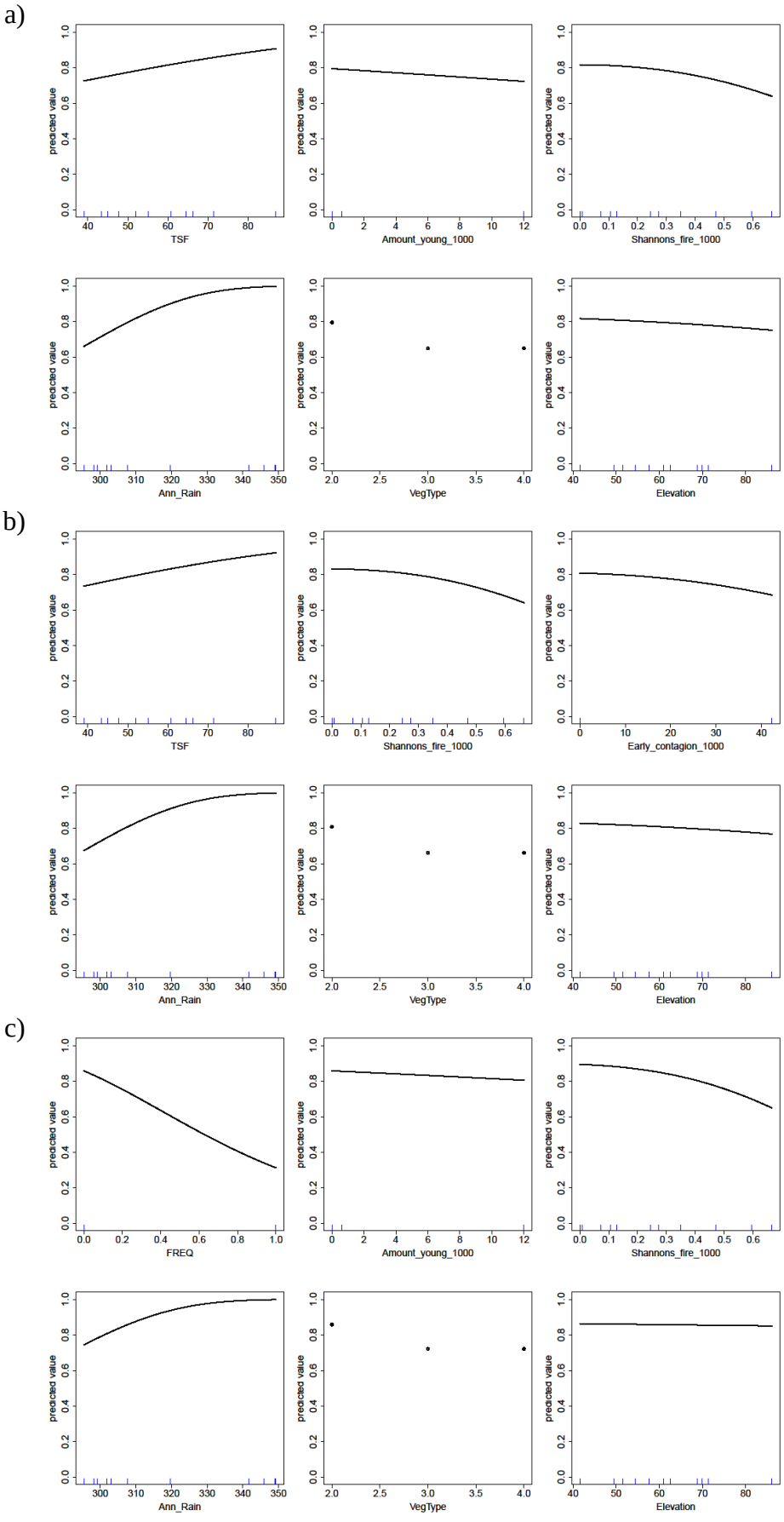


Figure S3.2.8: Response of *Chalinolobus morio* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

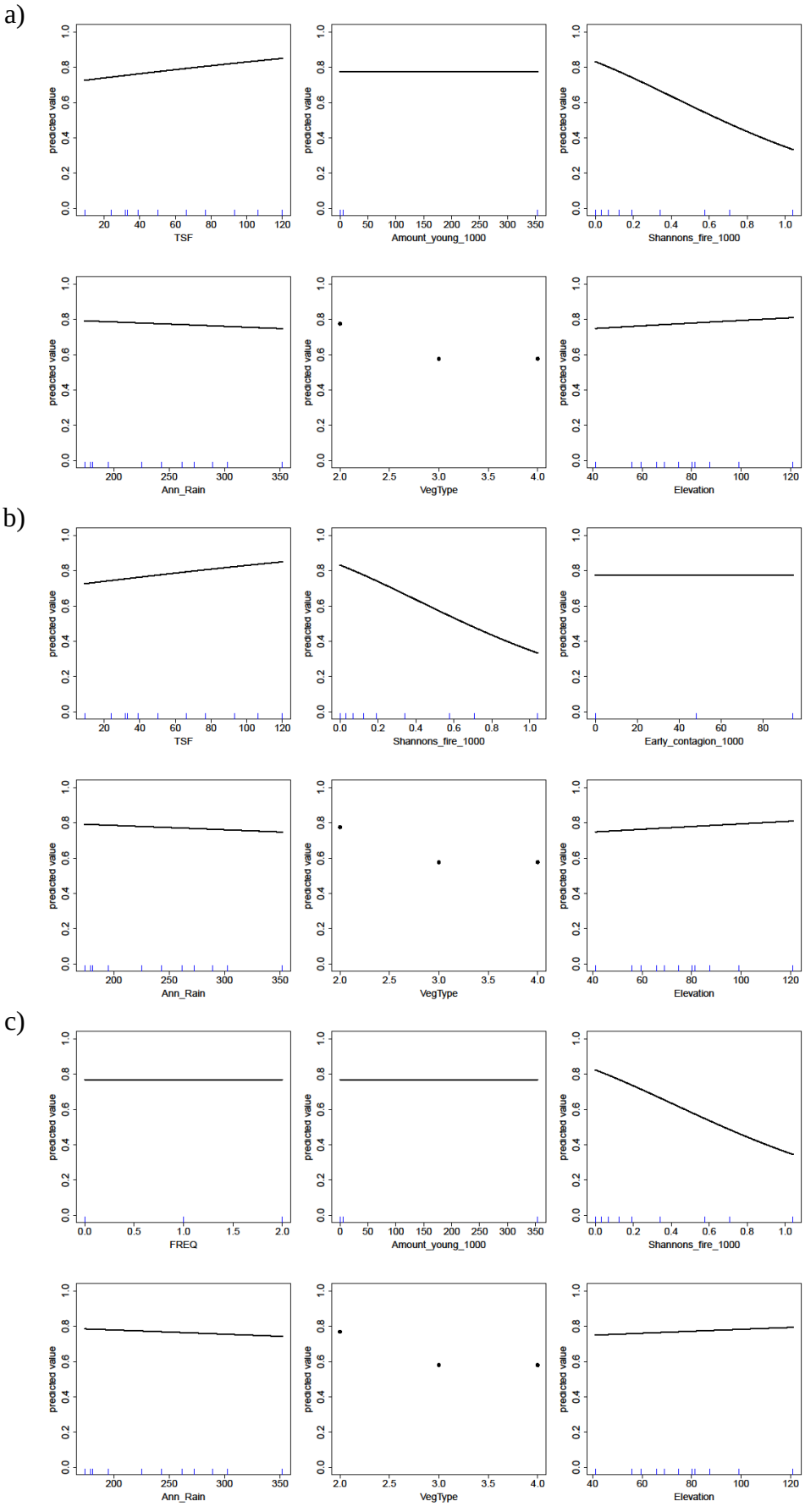
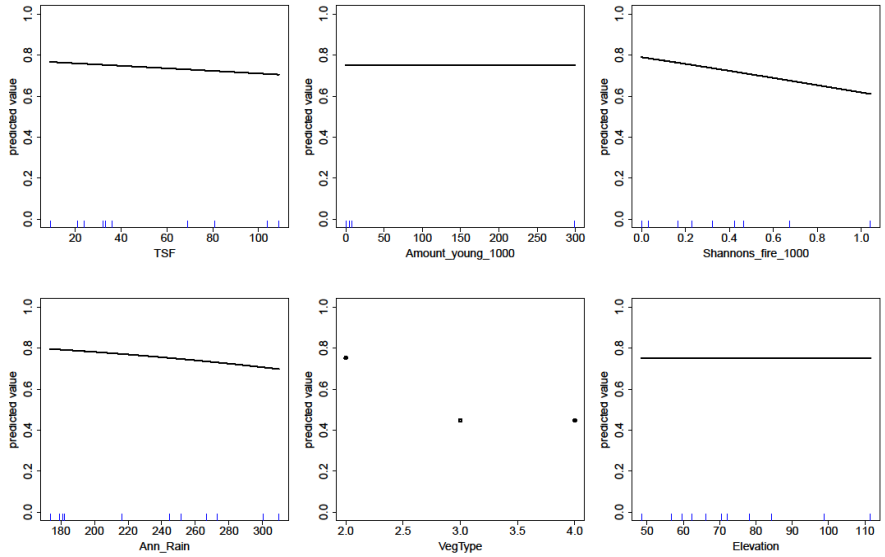


Figure S3.2.9: Response of *Chalinolobus picatus* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

a)



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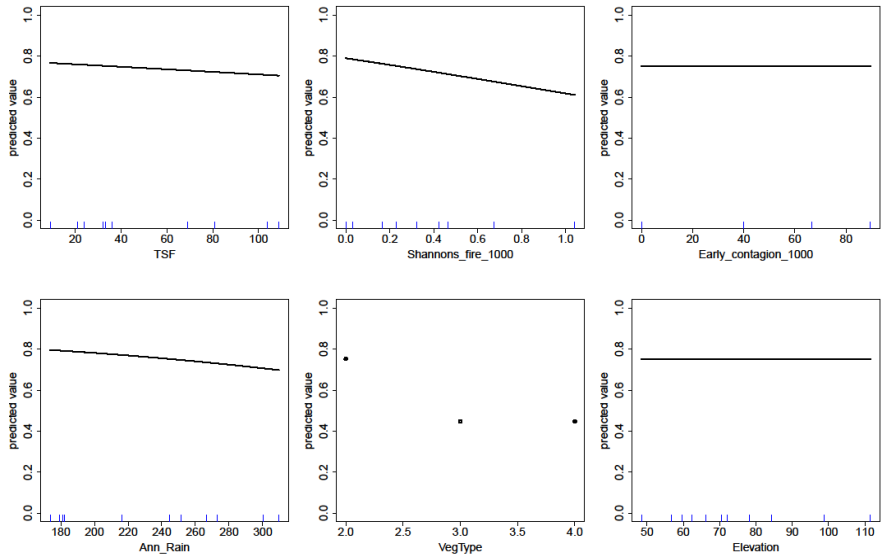


c)

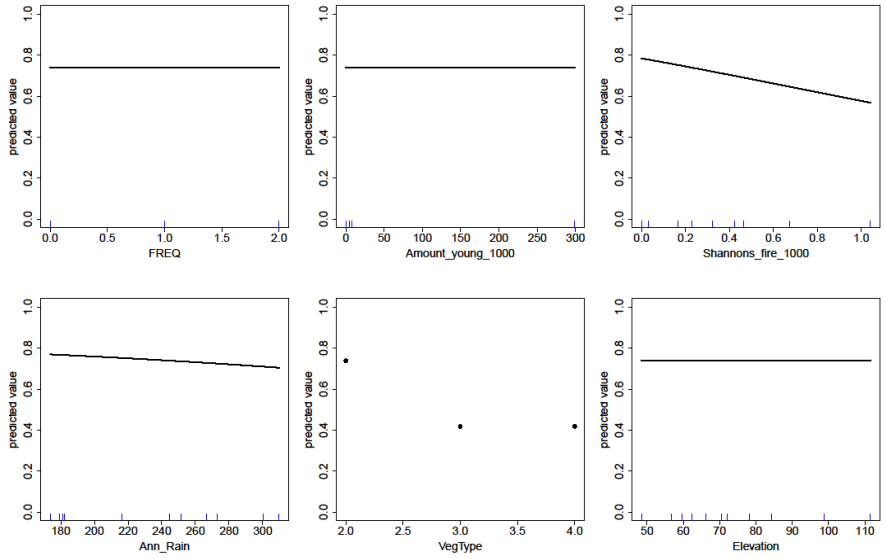


Figure S3.2.10: Response of *Ozimops petersi* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

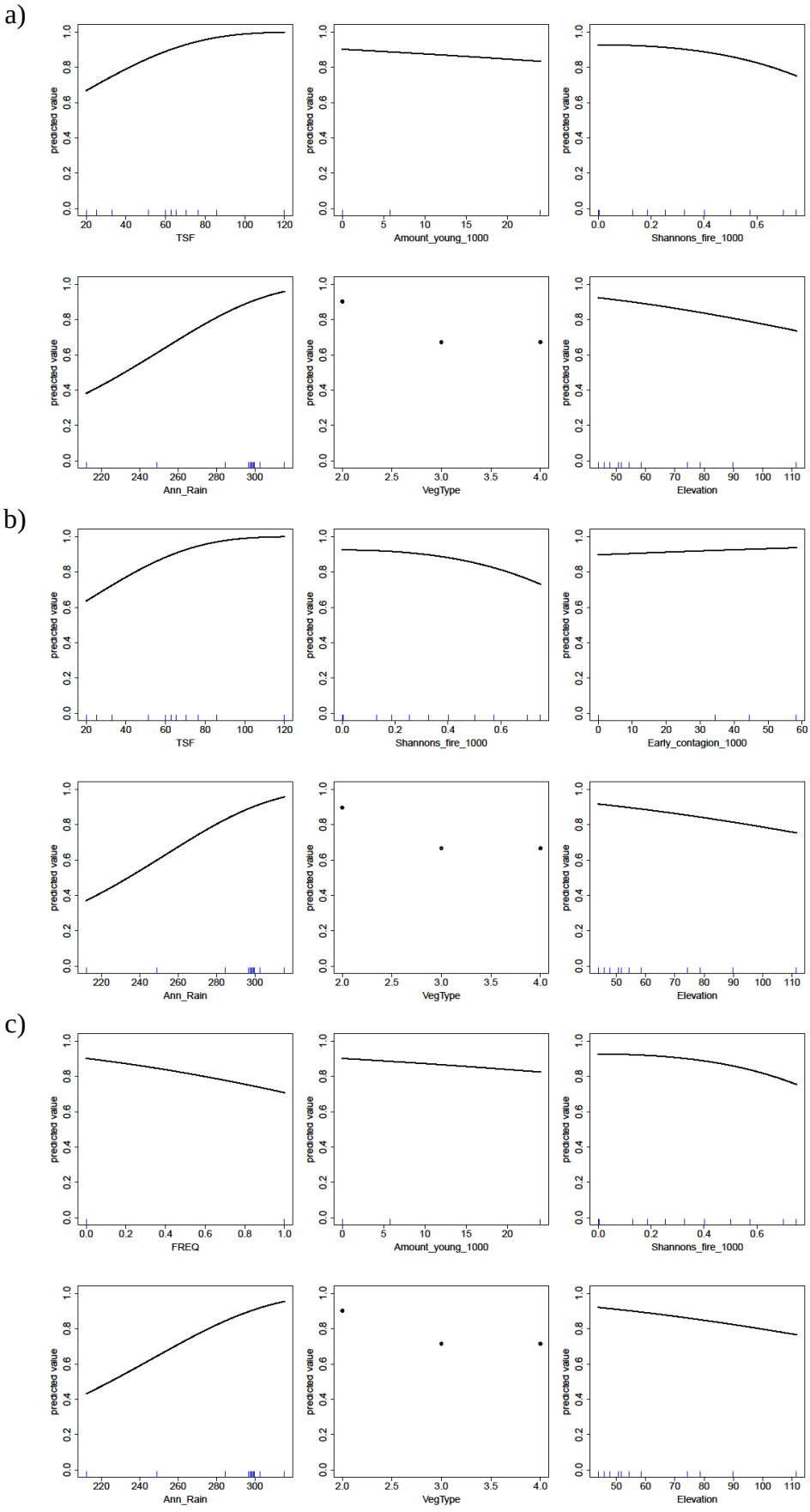


Figure S3.2.11: Response of *Nyctophilus corbeni* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

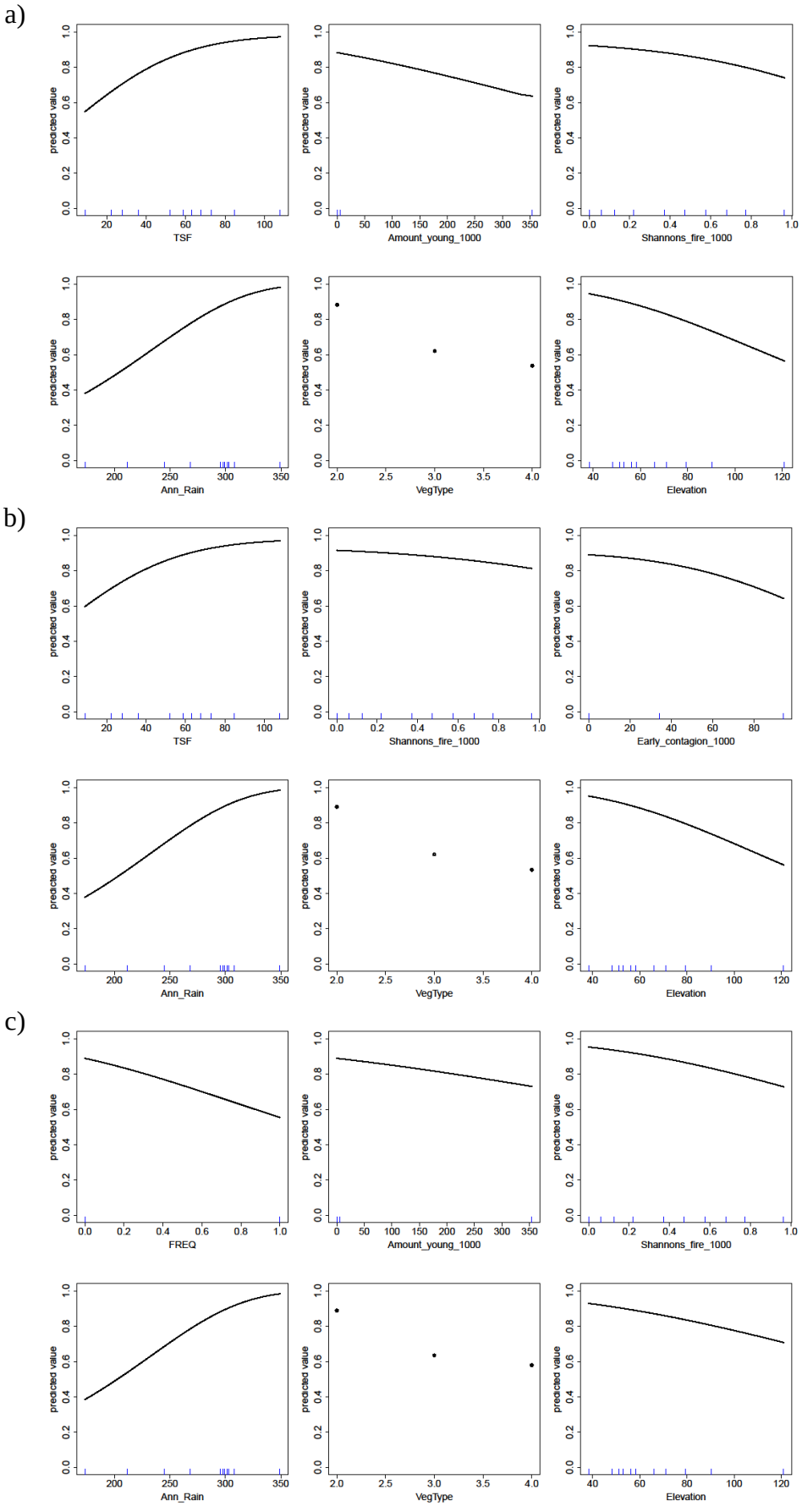
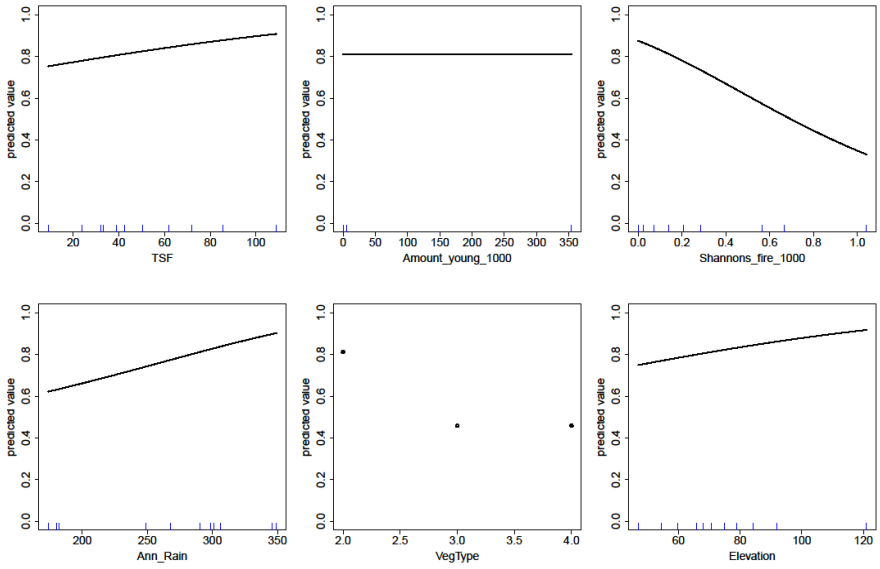
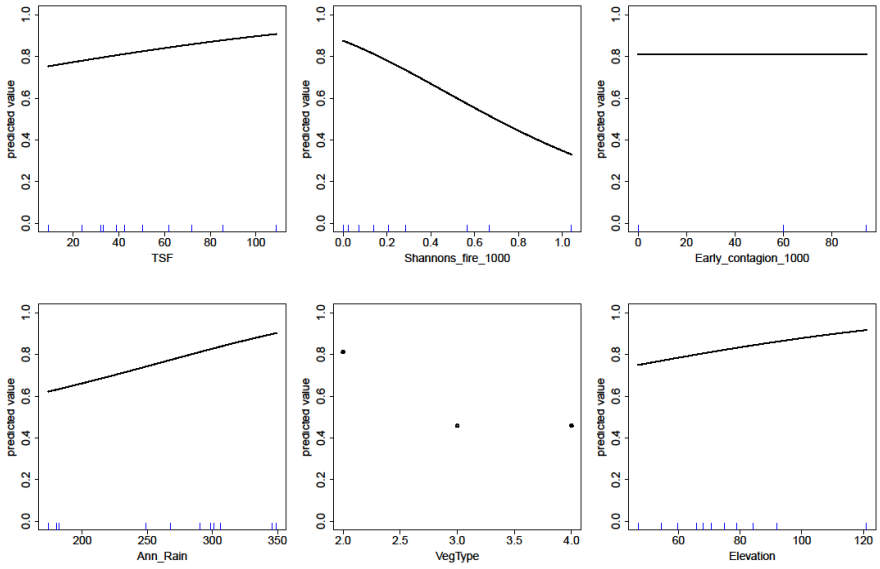


Figure S3.2.12: Response of *Nyctophilus geoffroyi* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

a)



b)



c)

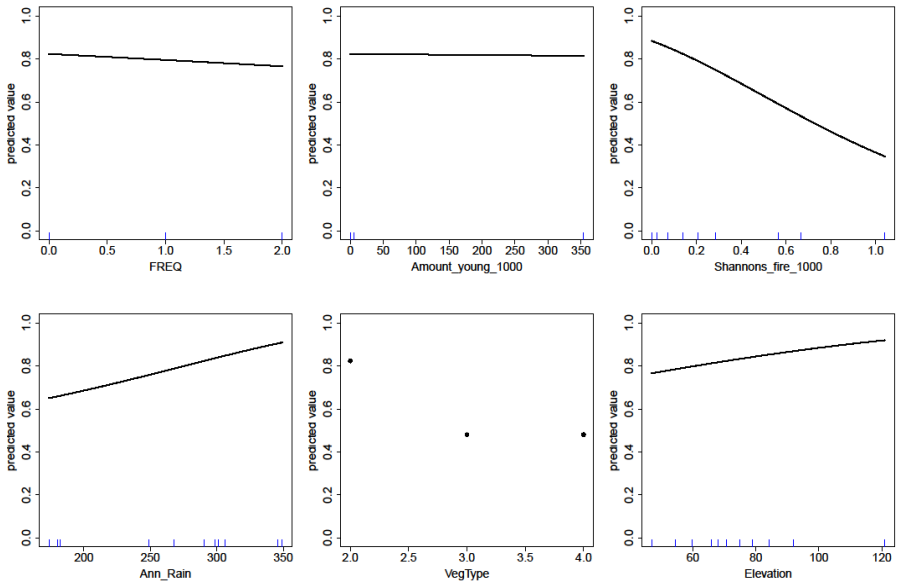


Figure S3.2.13: Response of *Scotorepens balstoni* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

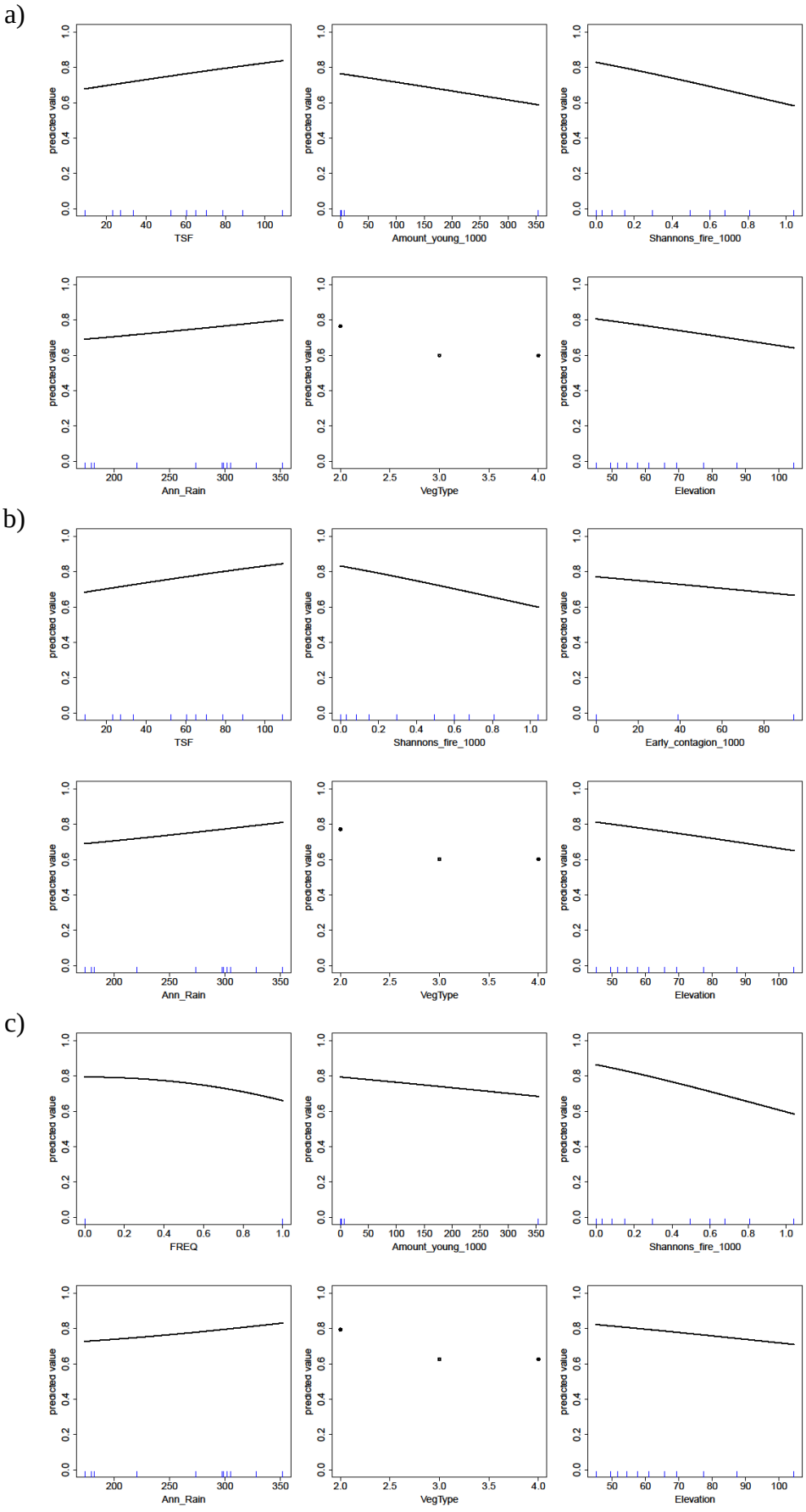


Figure S3.2.14: Response of *Vespadelus baverstocki* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

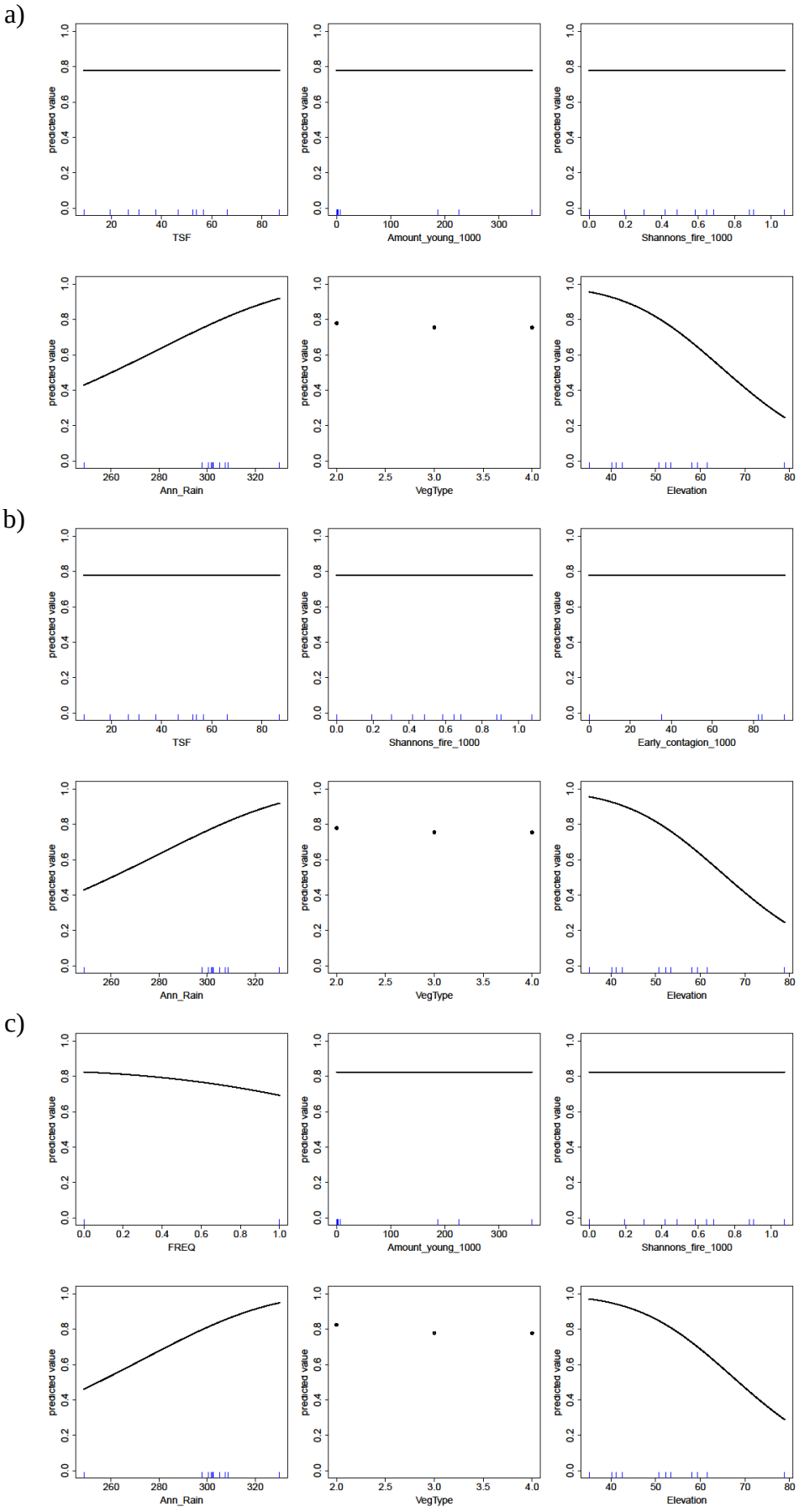


Figure S3.2.15: Response of *Vespadelus regulus* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

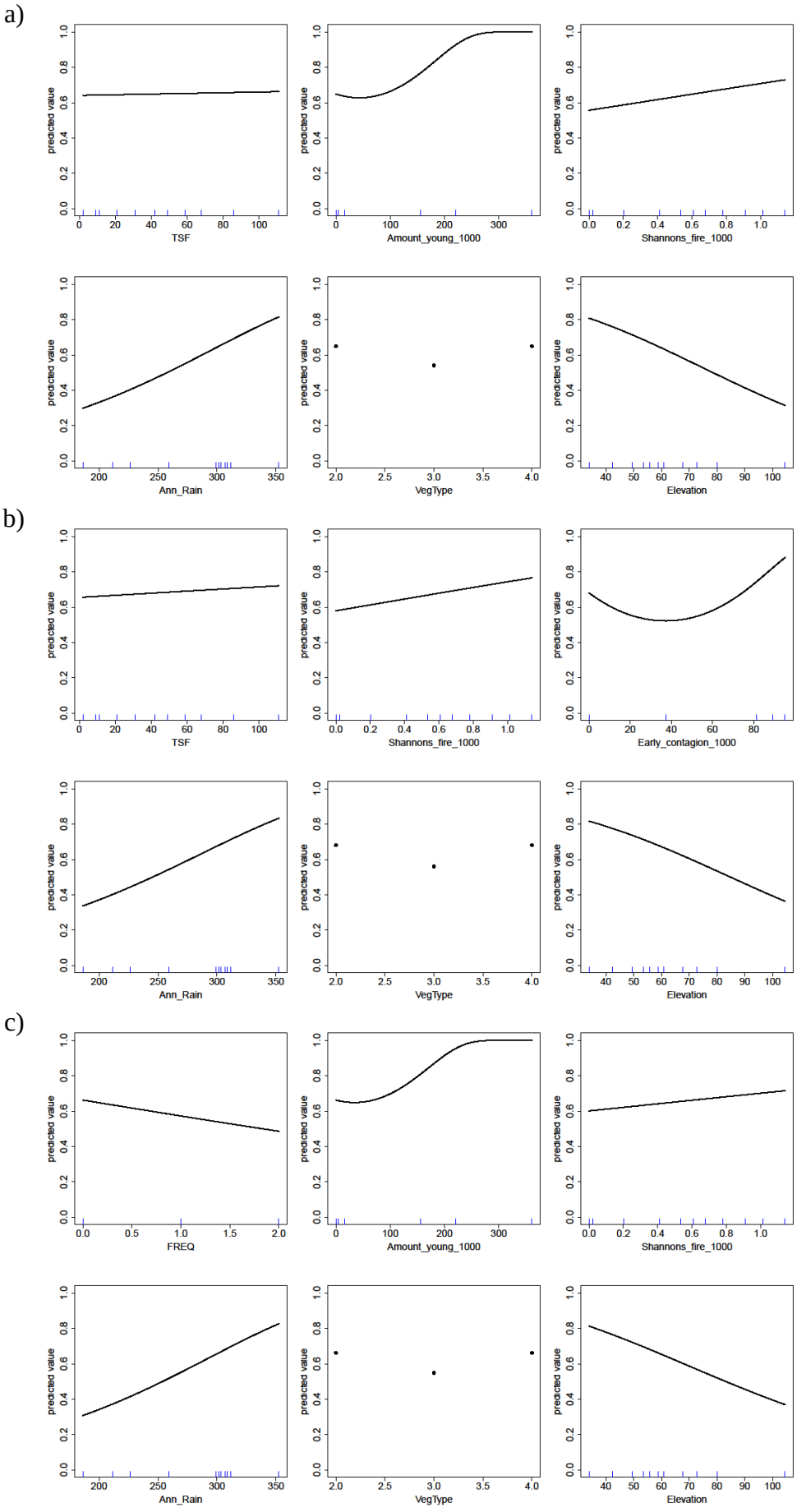
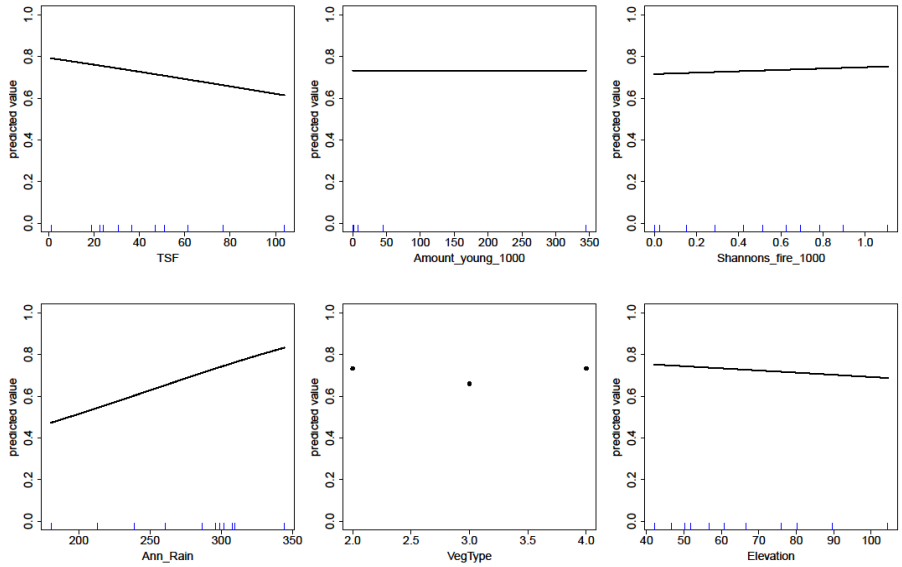
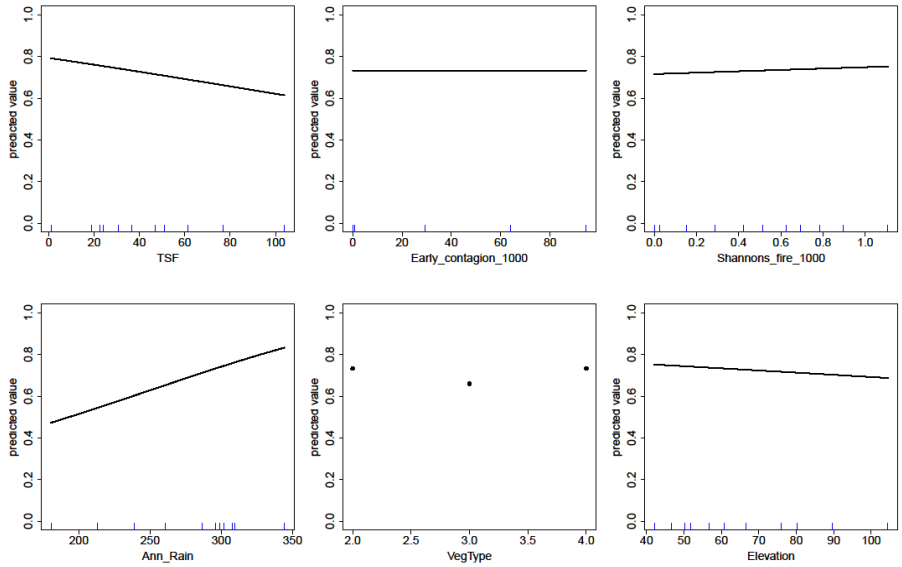


Figure S3.2.16: Response of *Macropus fuliginosus* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

a)



b)



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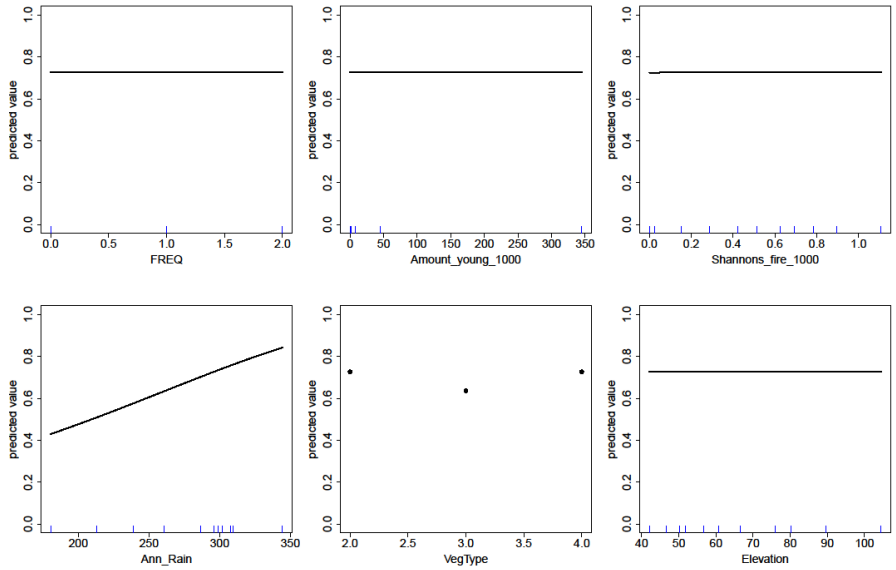
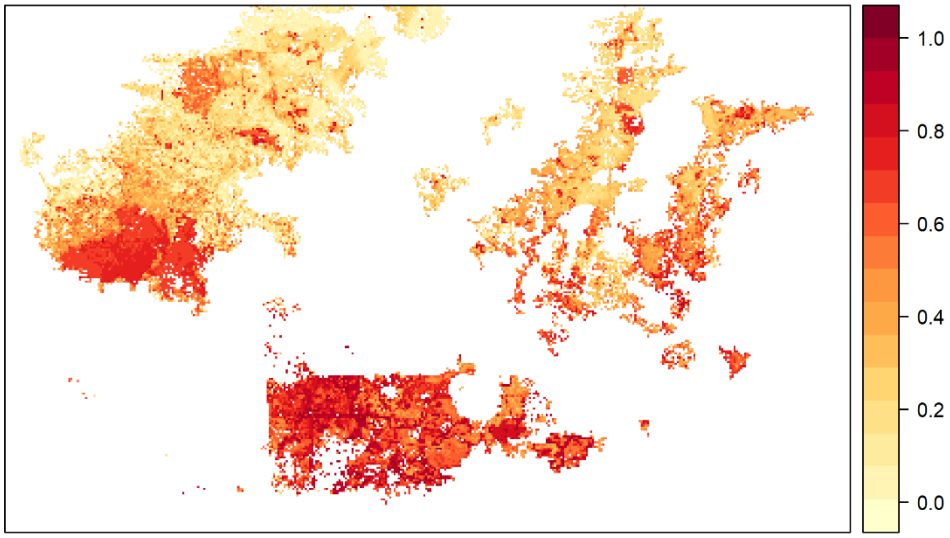
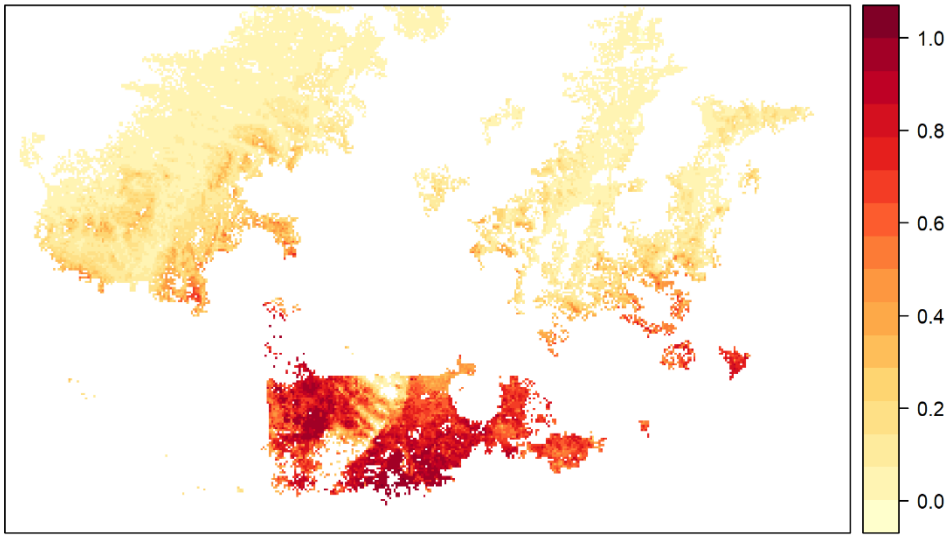


Figure S3.2.17: Response of *Tachyglossus aculeatus* to predictor variables in; a) model 1; b) model 2; c) model 3 when all other variables are held at their median value.

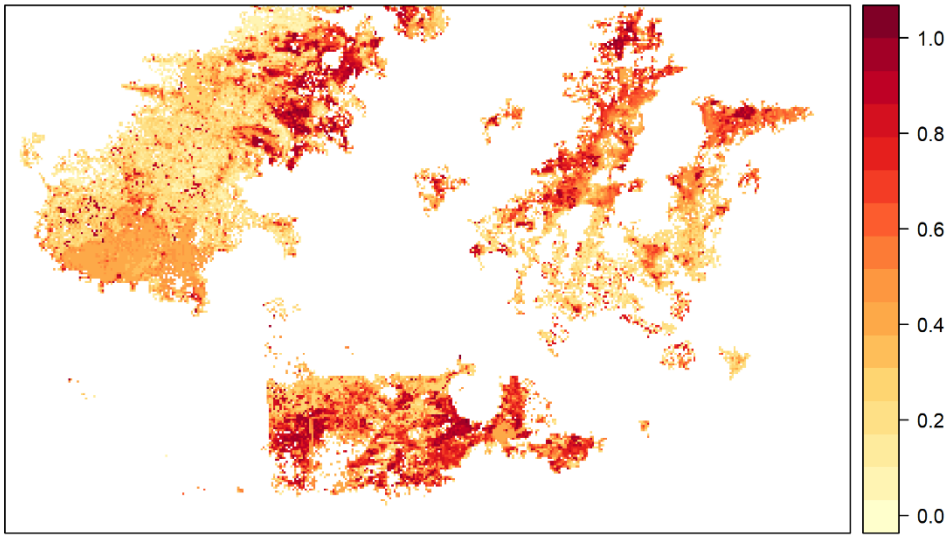
Appendix S3.3 – Maps of predicted distributions of mallee mammals



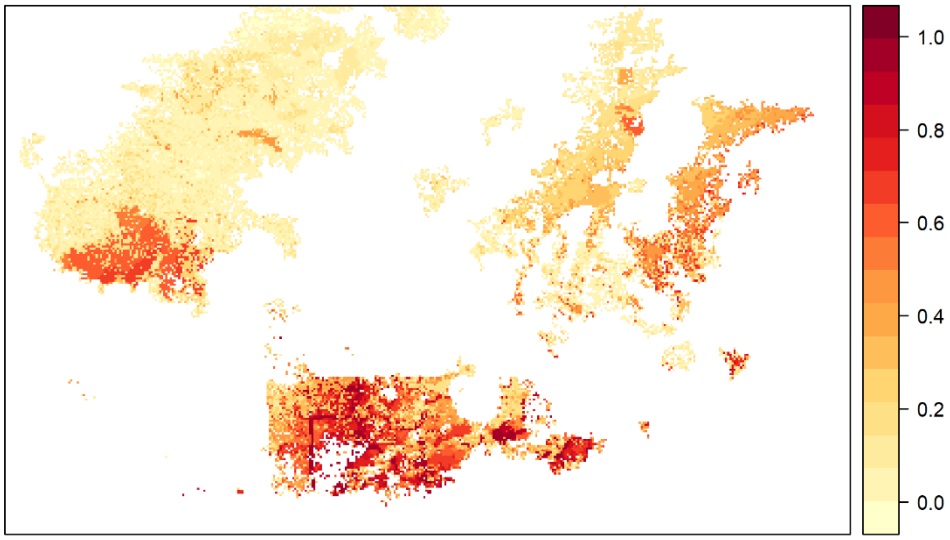
FigureS3.3.1: Relative likelihood of occurrence (cloglog function) of *Cercartetus concinnus* in relation to M1 predictor variables



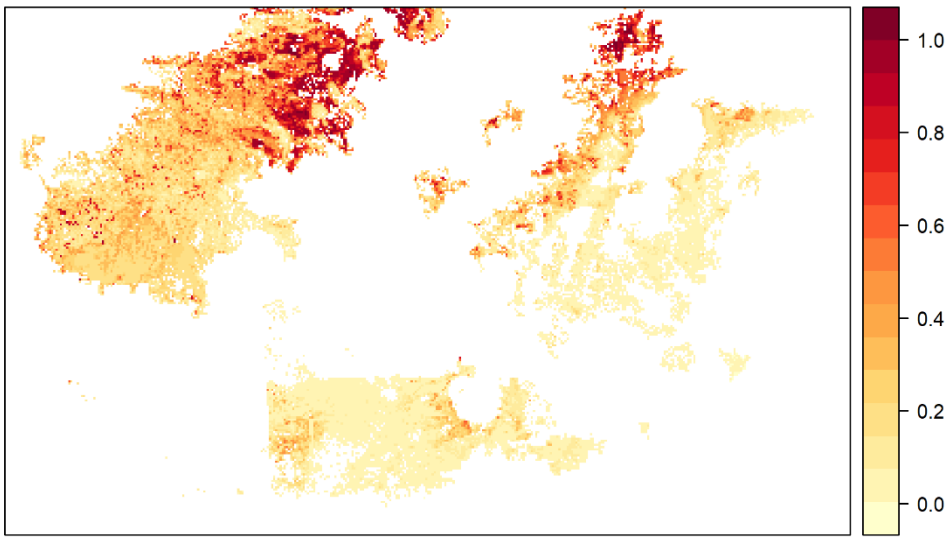
FigureS3.3.2: Relative likelihood of occurrence (cloglog function) of *Cercartetus lepidus* in relation to M1 predictor variables



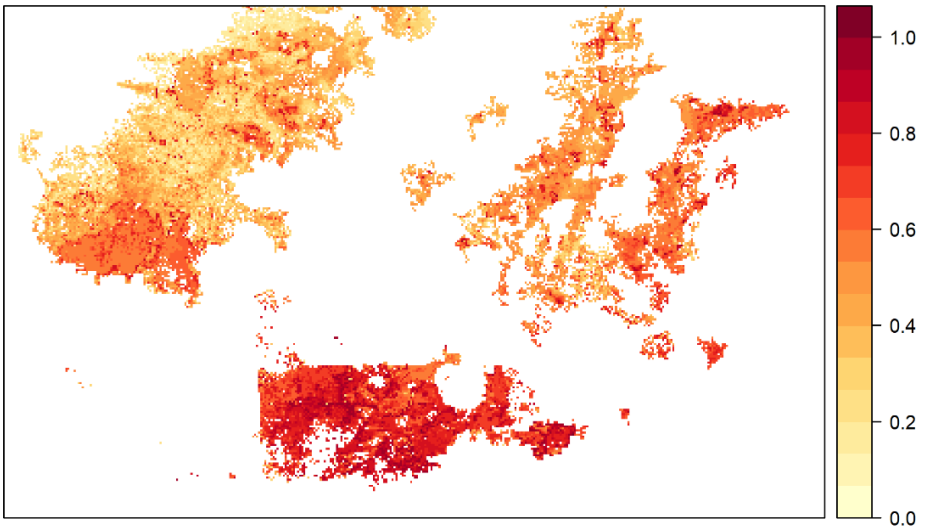
FigureS3.3.3: Relative likelihood of occurrence (cloglog function) of *Ningaui yvonneae* in relation to M1 predictor variables



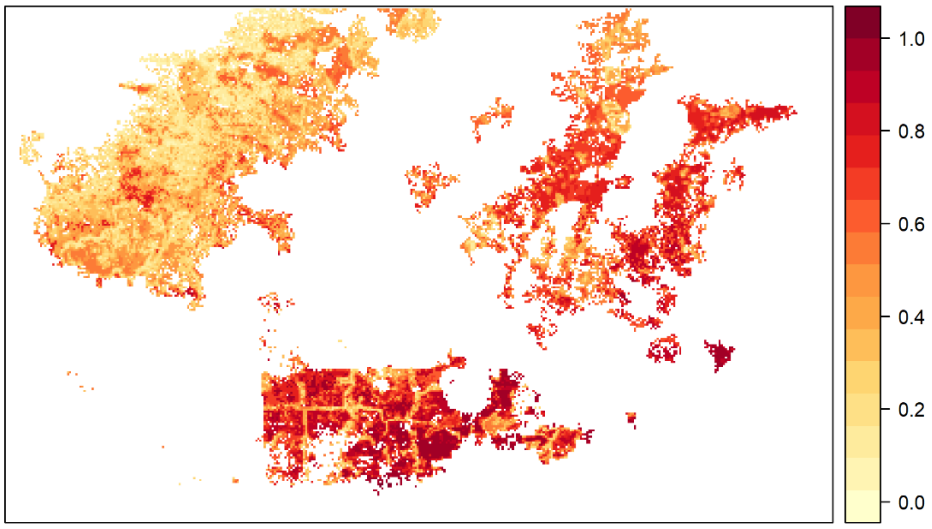
FigureS3.3.4: Relative likelihood of occurrence (cloglog function) of *Notomys mitchellii* in relation to M1 predictor variables



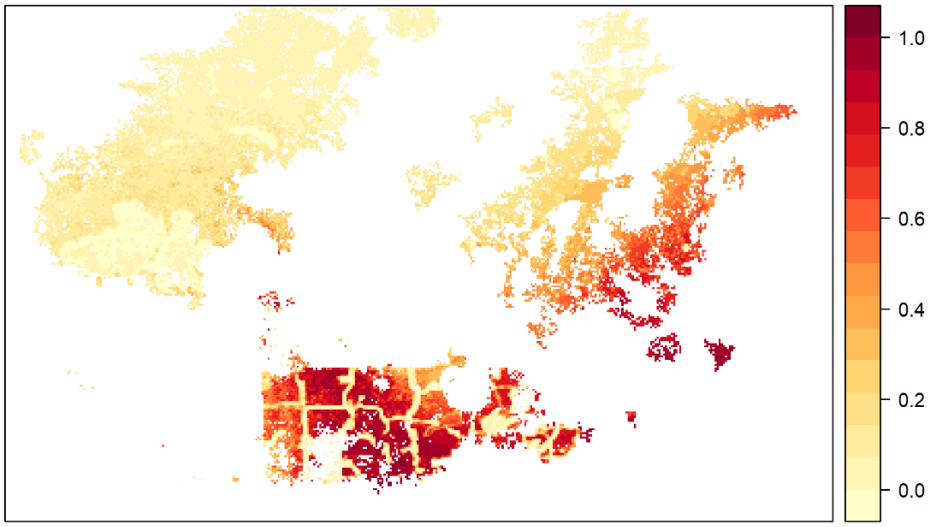
FigureS3.3.5: Relative likelihood of occurrence (cloglog function) of *Pseudomys bolami* in relation to M1 predictor variables



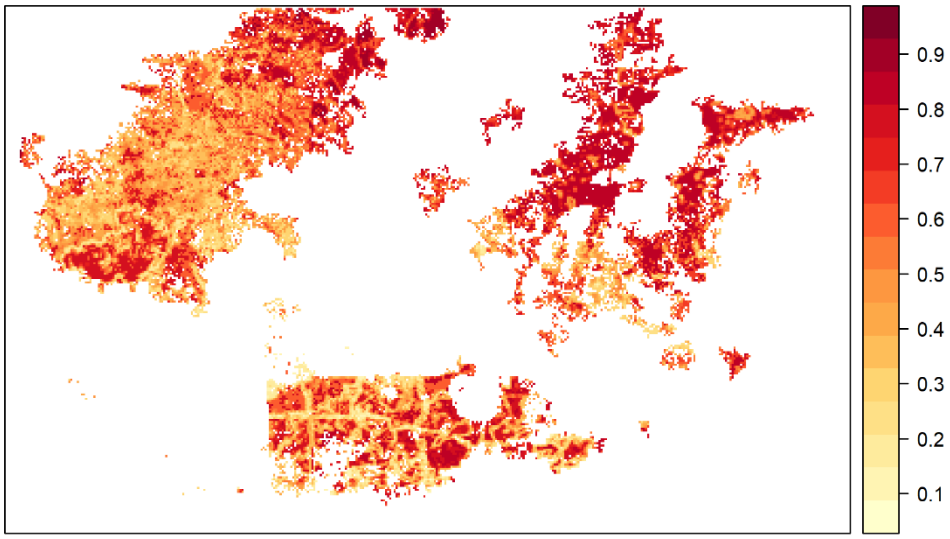
FigureS3.3.6: Relative likelihood of occurrence (cloglog function) of *Sminthopsis murina* in relation to M1 predictor variables



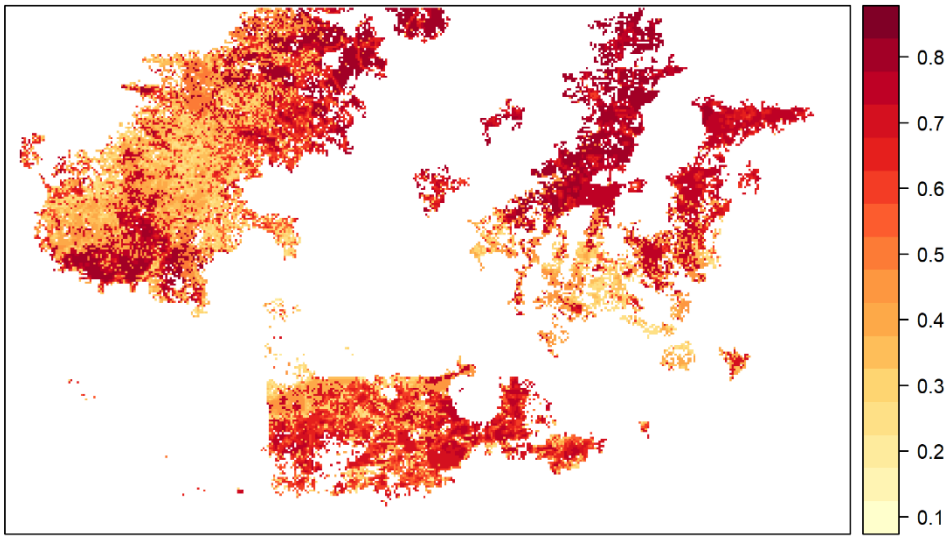
FigureS3.3.7: Relative likelihood of occurrence (cloglog function) of *Chalinolobus gouldii* in relation to M1 predictor variables



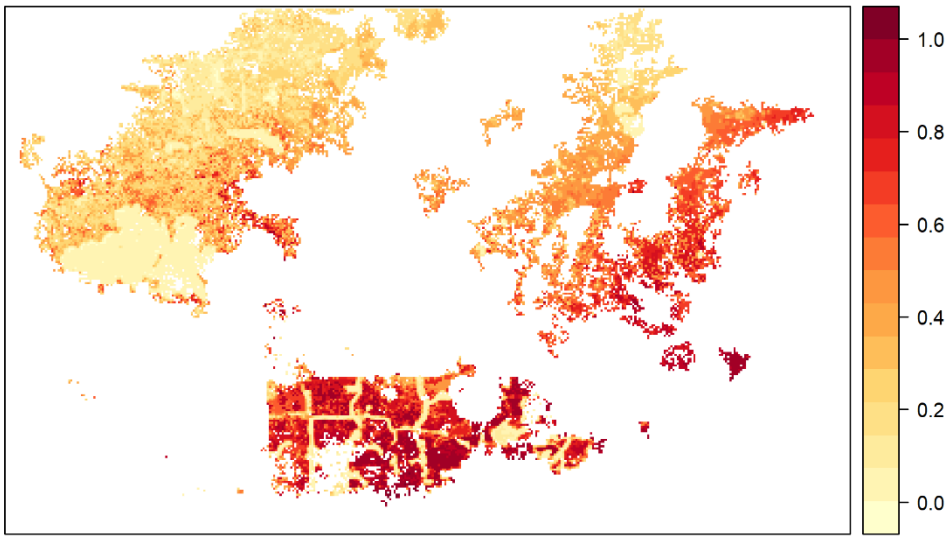
FigureS3.3.8: Relative likelihood of occurrence (cloglog function) of *Chalinolobus morio* in relation to M1 predictor variables



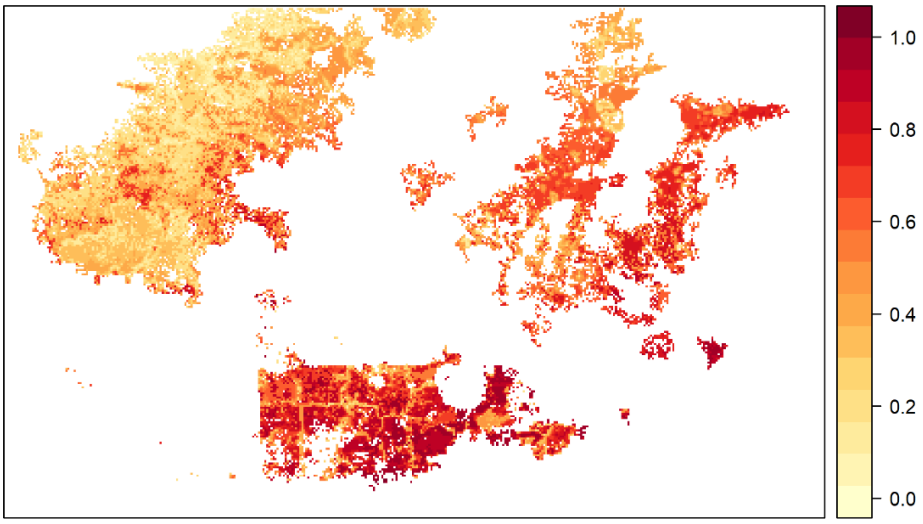
FigureS3.3.9: Relative likelihood of occurrence (cloglog function) of *Chalinolobus picatus* in relation to M1 predictor variables



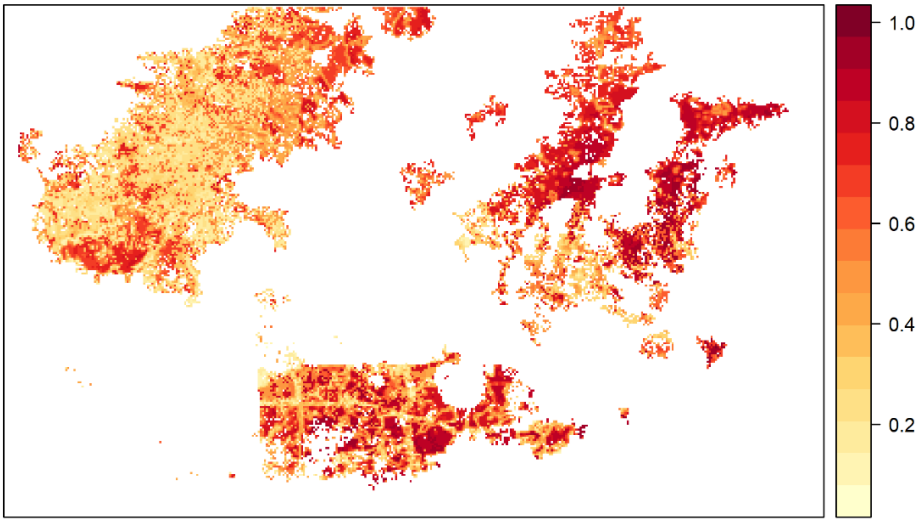
FigureS3.3.10: Relative likelihood of occurrence (cloglog function) of *Ozimops petersi* in relation to M1 predictor variables



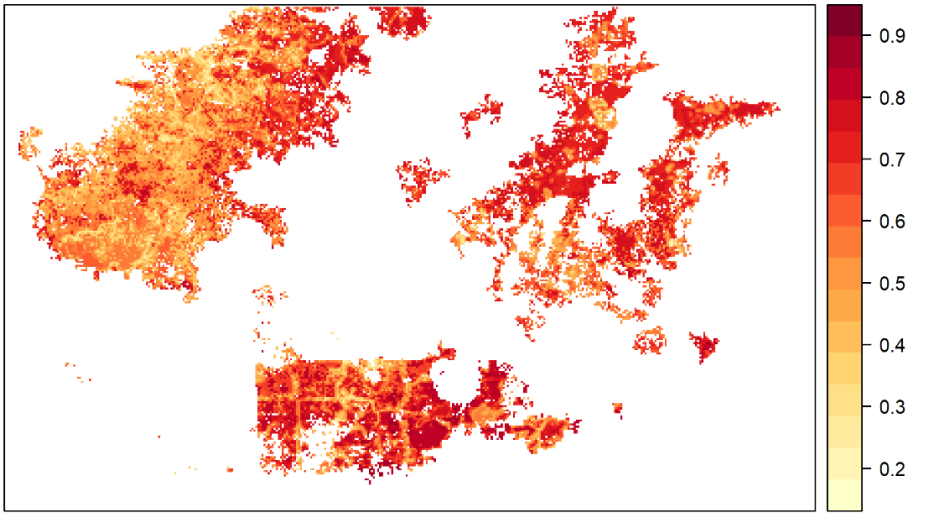
FigureS3.3.11: Relative likelihood of occurrence (cloglog function) of *Nyctophilus corbeni* in relation to M1 predictor variables



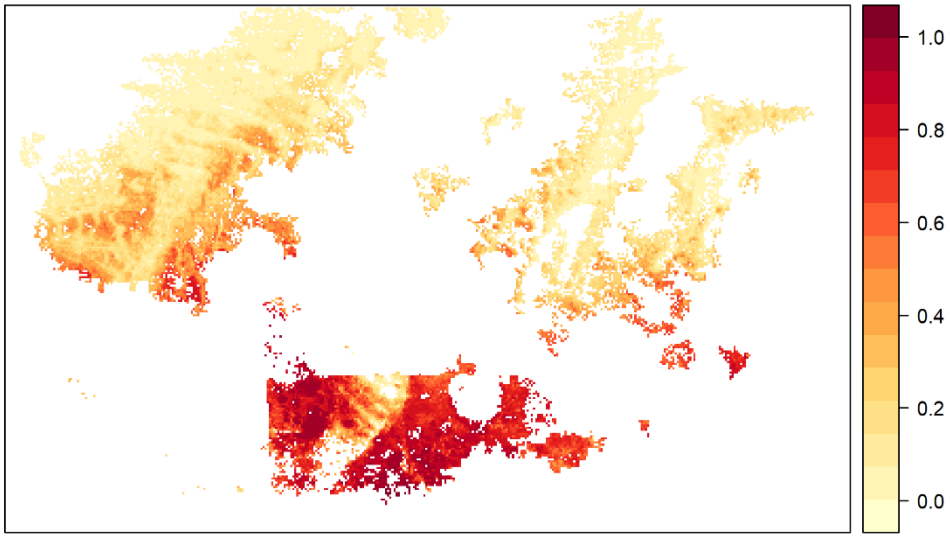
FigureS3.3.12: Relative likelihood of occurrence (cloglog function) of *Nyctophilus geoffroyi* in relation to M1 predictor variables



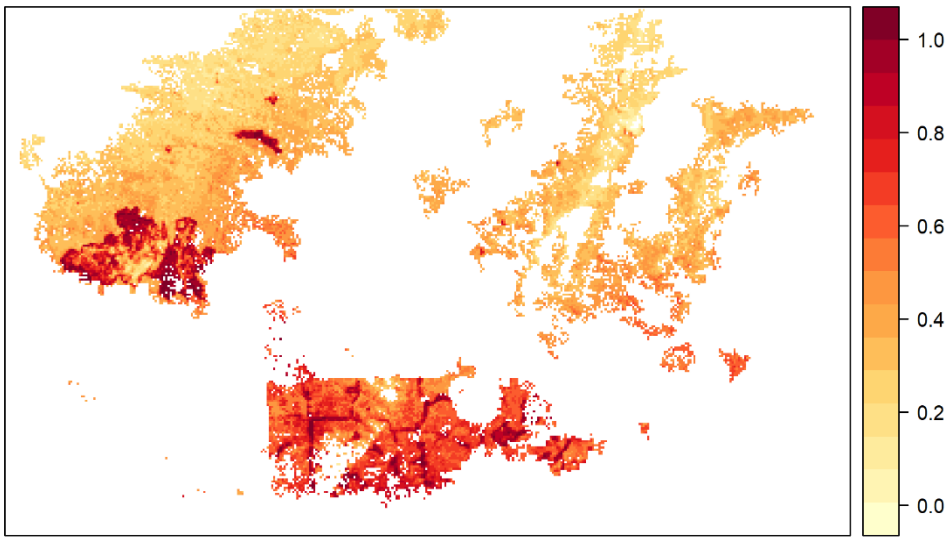
FigureS3.3.13: Relative likelihood of occurrence (cloglog function) of *Scotorepens balstoni* in relation to M1 predictor variables



FigureS3.3.14: Relative likelihood of occurrence (cloglog function) of *Vespadelus baverstocki* in relation to M1 predictor variables



FigureS3.3.15: Relative likelihood of occurrence (cloglog function) of *Vespadelus regulus* in relation to M1 predictor variables



FigureS3.3.16: Relative likelihood of occurrence (cloglog function) of *Macropus fuliginosus* in relation to M1 predictor variables

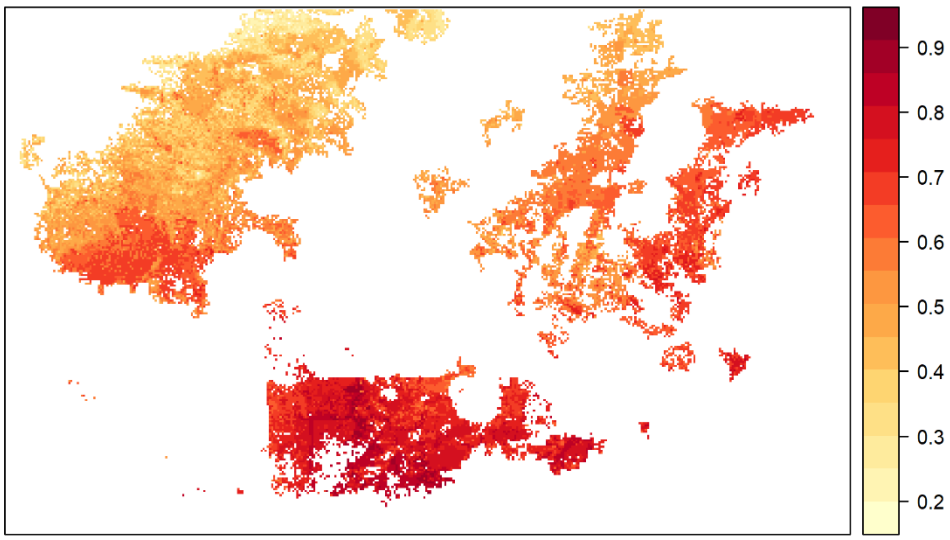


Figure3.3.17: Relative likelihood of occurrence (cloglog function) of *Tachyglossus aculeatus* in relation to M1 predictor variables

Appendices – Chapter 4

Appendix S4.1 – Conservation status of reptile species included in species distribution modelling

Table S4.1.1 – Species with greater than or equal to 15 presence records for tree mallee vegetation once filtered by year and grid cell. Classification and names from Cogger (2014) and ALA (2020). Conservation status: R = Rare, NT = Near Threatened, V = Vulnerable, E = Endangered, CE = Critically Endangered. Relevant conservation legislation: South Australia = *National Parks and Wildlife Act 1972*; New South Wales (NSW) = *Biodiversity Conservation Act 2016*; Victoria = *Advisory List of Threatened Vertebrate Fauna 2013*. No species are federally listed under the *Environmental Protection and Biodiversity Conservation Act 1999*.

Family	**Scientific**	**Common name/s**	**Conservation status**		
			SA	**NSW**	**Victoria**
Agamidae	*Ctenophorus fordi/ spinodomus*	Mallee dragon			
Agamidae	*Ctenophorus pictus*	Painted dragon			
Agamidae	*Diporiphora nobbi*	Nobbi dragon			
Agamidae	*Pogona vitticeps*	Central bearded dragon			
Carphodactylidae	*Nephrurus levis*	Smooth knob-tailed gecko			
Carphodactylidae	*Underwoodisaurus milii*	Thick-tailed gecko			
Diplodactylidae	*Diplodactylus vittatus*	Wood gecko			
Diplodactylidae	*Lucasium damaeum*	Beaded gecko			
Diplodactylidae	*Rhynchoedura ormsbyi*	Eastern beaked gecko			
Diplodactylidae	*Rhynchoedura ornata*	Western beaked gecko			CE
Diplodactylidae	*Strophurus elderi*	Jewelled gecko		V	
Diplodactylidae	*Strophurus intermedius*	Southern spiny-tailed gecko			
Diplodactylidae	*Strophurus williamsi*	Eastern spiny-tailed gecko			
Elapidae	*Brachyurophis australis*	Coral snake			
Elapidae	*Demansia psammophis*	Yellow-faced whip snake			NT
Elapidae	*Parasuta nigriceps*	Short-tailed snake			

Family	Scientific	Common name/s	Conservation status		
			SA	NSW	Victoria
Elapidae	*Pseudonaja modesta*	Ringed brown snake		E	
Elapidae	*Pseudonaja textilis*	Eastern brown snake, Warrala			
Elapidae	*Vermicella annulata*	Bandy-bandy	R		V
Gekkonidae	*Gehyra variegata*	Tree Dtella, Wiibidi			
Gekkonidae	*Heteronotia binoei*	Bynoe's gecko, Garragarraandi			
Pygopodidae	*Aprasia inaurita*	Mallee worm lizard		E	
Pygopodidae	*Delma australis*	Marble-faced delma		E	
Pygopodidae	*Delma butleri*	Unbanded delma			
Pygopodidae	*Lialis burtonis*	Burton's snake-lizard			
Pygopodidae	*Pygopus lepidopodus*	Common scaly-foot			
Pygopodidae	*Pygopus schraderi*	Eastern hooded scaly-foot			CE
Scincidae	*Cryptoblepharus australis*	Inland snake-eyed skink			
Scincidae	*Ctenotus atlas*	Southern mallee ctenotus			
Scincidae	*Ctenotus brachyonyx*	Short-clawed ctenotus			
Scincidae	*Ctenotus brooksi iridis*	Wedgesnout ctenotus		V	
Scincidae	*Ctenotus inornatus*	Bar-shouldered ctenotus			
Scincidae	*Ctenotus orientalis*	Eastern ctenotus			
Scincidae	*Ctenotus regius*	Pale-rumped ctenotus			
Scincidae	*Ctenotus robustus*	Robust ctenotus			
Scincidae	*Ctenotus schomburgkii*	Barred wedgesnout ctenotus			
Scincidae	*Cyclodomorphus melanops*	Spinifex slender bluetongue		E	
Scincidae	*Egernia striolata*	Tree skink, Waluubaal			
Scincidae	*Eremiascincus richardsonii*	Broad-banded sand-swimmer			
Scincidae	*Lerista aericeps*	Desert plain slider			
Scincidae	*Lerista bougainvillii*	South-eastern slider			
Scincidae	*Lerista labialis*	Southern sandslider			
Scincidae	*Lerista muelleri*	Wood mulch-slider			

Family	Scientific	Common name/s	Conservation status		
			SA	NSW	Victoria
Scincidae	*Lerista punctatovittata*	Eastern robust slider			
Scincidae	*Lerista xanthura*	Yellow-tailed plain slider		V	
Scincidae	*Liopholis inornata*	Desert skink			
Scincidae	*Menetia greyii*	Common dwarf skink			
Scincidae	*Morethia boulengeri*	South-eastern morethia skink			
Scincidae	*Morethia obscura*	Shrubland morethia skink			
Scincidae	*Tiliqua occipitalis*	Western blue-tongued lizard		V	NT
Scincidae	*Tiliqua rugosa*	Shingle-back, Garrbaali, Manggaay			
Typhlopidae	*Anilios bicolor*	Dark-spined blind snake			
Typhlopidae	*Anilios bituberculatus*	Prong-snouted blind snake			
Varanidae	*Varanus gouldii*	Sand goanna, Biiwii, Dhulii			

Appendix S4.2 – Variable contributions of each predictor variable for main and alternative models

Table S4.2.1: Cross validated variable contributions for model M1 with spatial fire variables based on mid-old triodia mallee age classes

Family	Scientific name	Common name	Time since fire	Area of mid-old triodia mallee	Diversity	Annual Rainfall	Elevation	Veg. Type
Blind snakes	*Anilios bicolor*	Dark-spined blind snake	30.72	21.33	20.33	0.58	7.14	19.90
	Anilios bituberculatus	Prong-snouted blind snake	15.08	38.27	8.85	5.77	23.24	8.80
Dragons	*Ctenophorus fordi*	Mallee dragon	23.03	31.09	12.97	2.43	9.44	21.04
	Ctenophorus pictus	Painted dragon	54.12	8.12	6.29	8.27	16.47	6.73
	Diporiphora nobbi	Nobbi dragon	12.40	26.62	8.62	31.97	14.27	6.13
	Pogona vitticeps	Central bearded dragon	26.02	27.13	9.47	6.70	26.98	3.70
Elapid snakes	*Brachyurophis australis*	Coral snake	14.46	42.02	7.41	19.56	14.56	1.99
	Parasuta nigriceps	Short-tailed snake	13.55	20.54	14.24	48.53	3.14	0.00
	Pseudonaja modesta	Ringed brown snake	6.73	0.10	2.44	87.39	0.66	2.67
	Pseudonaja textilis	Eastern brown snake	0.00	0.00	0.00	0.00	0.00	60.00
	Vermicella annulata	Bandy-bandy	40.67	2.60	8.65	48.07	0.00	0.01
Geckos	*Diplodactylus vittatus*	Wood gecko	9.49	19.79	10.19	56.01	4.46	0.05
	Gehyra variegata	Tree Dtella	4.90	65.32	12.49	12.46	1.32	3.51
	Lucasium damaeum	Smooth knob-tailed gecko	23.66	17.61	10.02	9.81	21.92	16.98
	Nephrurus levis	Beaded gecko	9.24	46.18	2.33	38.21	4.00	0.04
	Rhynchoedura ormsbyi	Eastern beaked gecko	1.78	60.03	6.88	13.52	12.05	5.74
	Rhynchoedura ornata	Western beaked gecko	17.44	6.83	49.05	16.34	7.04	3.29
	Strophurus elderi	Jewelled gecko	17.38	23.77	4.08	22.41	0.08	32.28
	Strophurus intermedius	Southern spiny-tailed gecko	10.78	23.86	8.35	45.50	9.78	1.74
	Strophurus williamsi	Eastern spiny-tailed gecko	0.00	47.12	2.63	45.63	1.54	3.08
	Underwoodisaurus milii	Thick-tailed gecko	0.02	0.61	0.02	98.73	0.61	0.00
Goanna	*Varanus gouldii*	Sand goanna	21.70	54.65	13.18	4.16	1.81	4.51

Family	Scientific name	Common name	Time since fire	Area of mid-old triodia mallee	Diversity	Annual Rainfall	Elevation	Veg. Type
Legless lizards	*Aprasia inaurita*	Mallee worm lizard	0.21	7.29	0.00	89.01	0.68	2.80
	Delma australis	Marble-faced delma	34.51	25.31	16.67	13.08	4.44	5.99
	Delma butleri	Unbanded delma	15.24	36.96	11.19	12.84	23.78	0.00
	Lialis burtonis	Burton's snake-lizard	29.63	46.30	13.56	4.89	4.31	1.30
	Pygopus lepidopodus	Common scaly-foot	25.46	46.24	8.78	3.79	11.84	3.89
Skinks	*Ctenotus atlas*	Southern mallee ctenotus	15.61	35.57	8.37	19.28	1.14	20.04
	Ctenotus brachyonyx	Short-clawed ctenotus	15.19	43.18	14.67	1.53	6.55	18.88
	Ctenotus brooksi iridis	Wedgesnout ctenotus	41.73	0.00	0.68	0.00	0.00	57.59
	Ctenotus orientalis	Eastern ctenotus	14.13	4.33	1.47	79.20	0.50	0.36
	Ctenotus regius	Pale-rumped ctenotus	30.17	24.75	5.85	9.13	8.68	21.43
	Ctenotus schomburgkii	Barred wedgesnout ctenotus	5.03	30.88	12.89	49.18	1.91	0.11
	Cyclodomorphus melanops	Spinifex slender bluetongue	34.21	0.44	0.00	46.51	0.29	18.55
	Eremiascincus richardsonii	Broad-banded sand-swimmer	1.05	4.00	2.52	83.53	0.00	8.91
	Lerista aericeps	Desert plain slider	19.13	43.30	0.00	16.65	15.50	5.42
	Lerista bougainvillii	South-eastern slider	6.83	6.13	2.75	80.46	3.82	0.00
	Lerista labialis	Southern sandslider	0.41	21.99	0.81	75.22	0.55	1.02
	Lerista muelleri	Wood mulch-slider	15.63	5.76	5.94	45.80	26.78	0.10
	Lerista punctatovittata	Eastern robust slider	27.26	42.52	17.62	1.32	1.98	9.31
	Lerista xanthura	Yellow-tailed plain slider	0.00	0.00	0.00	0.48	0.48	99.03
	Liopholis inornata	Desert skink	44.34	19.24	10.47	3.09	2.57	20.30
	Menetia greyii	Common dwarf skink	51.48	24.15	16.69	6.36	0.74	0.58
	Morethia boulengeri	South-eastern morethia skink	9.86	47.63	20.48	3.15	4.45	14.44
	Morethia obscura	Shrubland morethia skink	35.77	14.26	14.33	2.85	1.96	30.84
	Tiliqua occipitalis	Western blue-tongued lizard	21.05	7.16	14.95	0.23	19.85	36.77
	Tiliqua rugosa	Shingle-back	24.19	5.69	4.67	25.92	14.15	25.39

Table S4.2.2: Cross validated variable contributions for model M2 with spatial fire variables based on mid-old triodia mallee age classes

Family	Scientific name	Common name	Time since fire	ConFigure of mid-old triodia mallee	Diversity	Annual Rainfall	Elevation	Veg. Type
Blind snakes	*Anilios bicolor*	Dark-spined blind snake	19.61	24.25	6.65	2.20	35.74	11.55
	Anilios bituberculatus	Prong-snouted blind snake	11.29	47.27	7.12	18.70	14.28	1.34
Dragons	*Ctenophorus fordi*	Mallee dragon	21.02	39.45	11.81	0.77	9.33	17.62
	Ctenophorus pictus	Painted dragon	57.15	12.12	7.94	7.19	10.95	4.64
	Diporiphora nobbi	Nobbi dragon	12.24	32.90	7.64	27.28	14.85	5.08
	Pogona vitticeps	Central bearded dragon	23.69	31.12	9.86	5.27	26.18	3.88
Elapid snakes	*Brachyurophis australis*	Coral snake	26.73	40.16	17.32	8.92	4.42	2.46
	Parasuta nigriceps	Short-tailed snake	11.99	25.40	13.25	47.32	2.03	0.01
	Pseudonaja modesta	Ringed brown snake	8.13	0.89	3.11	82.36	1.07	4.45
	Pseudonaja textilis	Eastern brown snake	0.00	0.00	0.00	0.00	0.00	100.00
	Vermicella annulata	Bandy-bandy	25.15	30.16	30.28	12.60	0.00	1.81
Geckos	*Diplodactylus vittatus*	Wood gecko	7.24	24.94	9.60	52.83	5.25	0.13
	Gehyra variegata	Tree Dtella	4.36	62.67	10.17	19.66	0.61	2.54
	Lucasium damaeum	Smooth knob-tailed gecko	20.65	24.62	11.42	8.24	21.01	14.06
	Nephrurus levis	Beaded gecko	10.01	38.14	1.74	46.24	3.85	0.01
	Rhynchoedura ormsbyi	Eastern beaked gecko	1.02	59.53	5.56	15.33	11.22	7.35
	Rhynchoedura ornata	Western beaked gecko	25.26	18.51	32.13	14.62	5.16	4.32
	Strophurus elderi	Jewelled gecko	14.61	32.55	2.19	21.30	0.06	29.30
	Strophurus intermedius	Southern spiny-tailed gecko	6.50	31.77	5.03	44.47	11.79	0.43
	Strophurus williamsi	Eastern spiny-tailed gecko	0.00	26.14	1.39	47.67	0.57	24.22
	Underwoodisaurus milii	Thick-tailed gecko	0.00	0.00	0.00	99.41	0.00	0.59
Goanna	*Varanus gouldii*	Sand goanna	20.15	54.03	11.98	6.79	1.33	5.73
Legless lizards	*Aprasia inaurita*	Mallee worm lizard	0.10	17.07	0.00	81.14	1.03	0.66
	Delma australis	Marble-faced delma	13.50	44.11	8.43	10.86	23.05	0.04
	Delma butleri	Unbanded delma	30.35	41.02	12.31	3.88	5.02	7.41

Family	Scientific name	Common name	Time since fire	ConFigure of mid-old triodia mallee	Diversity	Annual Rainfall	Elevation	Veg. Type
Skinks	*Lialis burtonis*	Burton's snake-lizard	24.73	52.86	8.27	1.47	12.23	0.44
	Pygopus lepidopodus	Common scaly-foot	29.33	19.78	18.49	0.11	6.80	25.50
	Ctenotus atlas	Southern mallee ctenotus	15.95	31.90	6.62	20.27	0.55	24.71
	Ctenotus brachyonyx	Short-clawed ctenotus	11.51	50.51	13.01	0.61	5.90	18.45
	Ctenotus brooksi iridis	Wedgesnout ctenotus	43.01	0.00	0.16	0.00	0.00	56.83
	Ctenotus orientalis	Eastern ctenotus	14.11	0.09	2.69	80.91	0.32	1.86
	Ctenotus regius	Pale-rumped ctenotus	27.40	33.41	6.44	6.53	8.17	18.04
	Ctenotus schomburgkii	Barred wedgesnout ctenotus	5.07	18.74	9.55	58.55	2.92	5.19
	Cyclodomorphus melanops	Spinifex slender bluetongue	31.85	2.07	0.00	36.13	0.54	29.42
	Eremiascincus richardsonii	Broad-banded sand-swimmer	1.84	2.01	2.58	87.21	0.05	6.32
	Lerista aericeps	Desert plain slider	20.15	43.88	0.00	11.86	20.15	3.96
	Lerista bougainvillii	South-eastern slider	5.96	6.15	4.26	80.44	3.19	0.00
	Lerista labialis	Southern sandslider	0.48	8.62	0.39	83.31	0.45	6.75
	Lerista muelleri	Wood mulch-slider	16.74	6.70	4.67	44.77	27.12	0.00
	Lerista punctatovittata	Eastern robust slider	25.95	46.34	15.96	1.05	0.68	10.01
	Lerista xanthura	Yellow-tailed plain slider	0.00	0.00	0.00	0.35	0.35	99.31
	Liopholis inornata	Desert skink	42.93	19.89	9.10	2.89	3.43	21.76
	Menetia greyii	Common dwarf skink	48.30	27.23	18.02	3.96	1.08	1.41
	Morethia boulengeri	South-eastern morethia skink	9.81	44.93	18.07	6.57	4.33	16.28
	Morethia obscura	Shrubland morethia skink	33.81	14.65	12.85	3.48	0.87	34.34
	Tiliqua occipitalis	Western blue-tongued lizard	21.39	14.35	15.65	0.01	24.08	24.53
	Tiliqua rugosa	Shingle-back	25.61	5.81	5.47	21.97	18.08	23.05

Table S4.2.3: Cross validated variable contributions for model M3 with spatial fire variables based on mid-old triodia mallee age classes

Family	Scientific name	Common name	Fire frequency	Area of mid-old triodia mallee	Diversity	Annual Rainfall	Elevation	Veg. Type
Blind snakes	*Anilios bicolor*	Dark-spined blind snake	6.63	40.48	12.96	7.94	22.85	9.14
	Anilios bituberculatus	Prong-snouted blind snake	7.21	46.85	7.53	21.90	15.06	1.44
Dragons	*Ctenophorus fordi*	Mallee dragon	17.56	31.35	21.01	1.93	7.70	20.45
	Ctenophorus pictus	Painted dragon	34.66	12.70	12.13	21.29	10.80	8.42
	Diporiphora nobbi	Nobbi dragon	10.15	27.60	14.86	32.35	9.82	5.22
	Pogona vitticeps	Central bearded dragon	17.10	31.02	19.82	10.31	18.29	3.45
Elapid snakes	*Brachyurophis australis*	Coral snake	25.58	24.94	30.76	11.16	3.20	4.36
	Parasuta nigriceps	Short-tailed snake	4.39	21.71	17.06	53.69	3.14	0.00
	Pseudonaja modesta	Ringed brown snake	1.71	0.02	2.55	91.52	0.39	3.81
	Pseudonaja textilis	Eastern brown snake	95.99	0.00	0.00	0.00	0.00	4.01
	Vermicella annulata	Bandy-bandy	0.00	6.65	17.03	71.89	0.00	4.42
Geckos	*Diplodactylus vittatus*	Wood gecko	6.31	19.70	13.32	56.93	3.71	0.04
	Gehyra variegata	Tree Dtella	1.69	70.33	12.72	11.13	0.96	3.17
	Lucasium damaeum	Smooth knob-tailed gecko	22.78	19.51	21.26	10.63	10.49	15.33
	Nephrurus levis	Beaded gecko	3.45	50.67	3.24	38.67	3.97	0.01
	Rhynchoedura ormsbyi	Eastern beaked gecko	3.23	57.69	8.22	14.22	11.17	5.47
	Rhynchoedura ornata	Western beaked gecko	32.41	1.82	56.35	4.76	4.40	0.26
	Strophurus elderi	Jewelled gecko	19.34	24.41	14.34	11.41	0.13	30.37
	Strophurus intermedius	Southern spiny-tailed gecko	2.10	25.07	10.03	50.49	10.34	1.98
	Strophurus williamsi	Eastern spiny-tailed gecko	0.00	47.12	2.63	45.63	1.54	3.08
	Underwoodisaurus milii	Thick-tailed gecko	3.19	3.19	0.00	93.62	0.00	0.00
Goanna	*Varanus gouldii*	Sand goanna	6.99	64.61	18.84	2.25	1.97	5.34
Legless lizards	*Aprasia inaurita*	Mallee worm lizard	0.41	7.25	0.41	88.65	0.48	2.81
	Delma australis	Marble-faced delma	8.85	39.81	15.23	14.58	21.52	0.00
	Delma butleri	Unbanded delma	21.40	47.55	23.73	2.15	4.43	0.73

Family	Scientific name	Common name	Fire frequency	Area of mid-old triodia mallee	Diversity	Annual Rainfall	Elevation	Veg. Type
	Lialis burtonis	Burton's snake-lizard	13.56	50.22	15.06	5.38	11.84	3.94
	Pygopus lepidopodus	Common scaly-foot	6.84	26.13	29.88	0.02	8.03	29.10
Skinks	*Ctenotus atlas*	Southern mallee ctenotus	13.17	36.82	15.68	13.18	0.65	20.49
	Ctenotus brachyonyx	Short-clawed ctenotus	10.77	43.38	19.98	1.49	5.60	18.78
	Ctenotus brooksi iridis	Wedgesnout ctenotus	0.01	0.00	2.99	0.00	0.00	96.99
	Ctenotus orientalis	Eastern ctenotus	7.11	4.16	6.93	81.55	0.16	0.09
	Ctenotus regius	Pale-rumped ctenotus	21.43	26.64	15.85	7.17	7.71	21.20
	Ctenotus schomburgkii	Barred wedgesnout ctenotus	4.95	30.02	15.45	47.30	2.17	0.11
	Cyclodomorphus melanops	Spinifex slender bluetongue	0.00	1.28	0.00	48.23	0.31	50.19
	Eremiascincus richardsonii	Broad-banded sand-swimmer	1.12	4.50	3.20	82.88	0.02	8.27
	Lerista aericeps	Desert plain slider	3.01	70.51	0.00	12.72	0.19	13.56
	Lerista bougainvillii	South-eastern slider	6.14	4.68	3.49	82.20	3.50	0.01
	Lerista labialis	Southern sandslider	0.01	22.11	0.77	75.49	0.52	1.10
	Lerista muelleri	Wood mulch-slider	16.19	4.55	14.04	39.91	25.22	0.09
	Lerista punctatovittata	Eastern robust slider	19.83	42.10	28.01	1.53	0.90	7.63
	Lerista xanthura	Yellow-tailed plain slider	0.00	0.00	0.00	0.48	0.48	99.03
	Liopholis inornata	Desert skink	30.17	20.03	26.66	1.51	2.70	18.93
	Menetia greyii	Common dwarf skink	12.52	36.61	30.52	16.51	1.21	2.63
	Morethia boulengeri	South-eastern morethia skink	1.97	52.54	22.77	2.59	4.66	15.46
	Morethia obscura	Shrubland morethia skink	16.18	13.69	24.37	0.77	2.10	42.90
	Tiliqua occipitalis	Western blue-tongued lizard	1.95	8.43	16.38	0.65	24.19	48.40
	Tiliqua rugosa	Shingle-back	5.27	8.20	5.23	36.98	13.52	30.80

Table S4.2.4: Cross-validated AUC values for alternative models built with spatial fire variable based on the early post-fire age class, for species presented in the main text

Taxa.	Scientific name	Common name	M1	M2	M3
Blind snakes	*Anilios bicolor*	Dark-spined blind snake	0.67	0.67	0.66
	Anilios bituberculatus	Prong-snouted blind snake	0.72	0.72	0.68
Dragons	*Ctenophorus fordi*	Mallee dragon	0.72	0.72	0.71
	Ctenophorus pictus	Painted dragon	0.76	0.76	0.74
	Diporiphora nobbi	Nobbi dragon	0.74	0.74	0.72
	Pogona vitticeps	Central bearded dragon	0.72	0.72	0.71
Elapid snakes	*Brachyurophis australis*	Coral snake	0.72	0.73	0.69
	Parasuta nigriceps	Short-tailed snake	0.75	0.75	0.73
	Pseudonaja modesta	Ringed brown snake	0.87	0.87	0.85
	Pseudonaja textilis	Eastern brown snake	0.53	0.47	0.77
	Vermicella annulata	Bandy-bandy	0.58	0.62	0.60
Geckos	*Diplodactylus vittatus*	Wood gecko	0.75	0.75	0.73
	Gehyra variegata	Tree Dtella	0.60	0.59	0.59
	Nephrurus levis	Smooth knob-tailed gecko	0.69	0.69	0.68
	Lucasium damaeum	Beaded gecko	0.89	0.89	0.87
	Rhynchoedura ormsbyi	Eastern beaked gecko	0.75	0.75	0.76
	Rhynchoedura ornata	Western beaked gecko	0.71	0.73	0.71
	Strophurus elderi	Jewelled gecko	0.81	0.81	0.82
	Strophurus intermedius	Southern spiny-tailed gecko	0.72	0.72	0.72
	Strophurus williamsi	Eastern spiny-tailed gecko	0.78	0.78	0.79
	Underwoodisaurus milii	Thick-tailed gecko	0.77	0.77	0.77
Goanna	*Varanus gouldii*	Sand goanna	0.65	0.66	0.62
Legless lizards	*Aprasia inaurita*	Mallee worm lizard	0.72	0.72	0.72
	Delma australis	Marble-faced delma	0.75	0.75	0.75
	Delma butleri	Unbanded delma	0.77	0.78	0.77
	Lialis burtonis	Burton's snake-lizard	0.72	0.73	0.70
	Pygopus lepidopodus	Common scaly-foot	0.76	0.77	0.74
Skinks	*Ctenotus atlas*	Southern mallee ctenotus	0.74	0.74	0.72
	Ctenotus brachyonyx	Short-clawed ctenotus	0.76	0.76	0.75
	Ctenotus brooksi iridis	Wedgesnout ctenotus	0.86	0.87	0.79
	Ctenotus orientalis	Eastern ctenotus	0.92	0.91	0.91
	Ctenotus regius	Pale-rumped ctenotus	0.67	0.67	0.66
	Ctenotus schomburgkii	Barred wedgesnout ctenotus	0.70	0.70	0.70
	Cyclodomorphus melanops	Spinifex slender bluetongue	0.73	0.77	0.68
	Eremiascincus richardsonii	Broad-banded sand-swimmer	0.69	0.82	0.80
	Lerista aericeps	Desert plain slider	0.86	0.88	0.86
	Lerista bougainvillii	South-eastern slider	0.87	0.87	0.87
	Lerista labialis	Southern sandslider	0.92	0.92	0.92
	Lerista muelleri	Wood mulch-slider	0.81	0.81	0.81
	Lerista punctatovittata	Eastern robust slider	0.65	0.66	0.64
	Lerista xanthura	Yellow-tailed plain slider	0.66	0.63	0.66
	Liopholis inornata	Desert skink	0.76	0.77	0.74
	Menetia greyii	Common dwarf skink	0.65	0.65	0.61
	Morethia boulengeri	South-eastern morethia skink	0.59	0.58	0.56
	Morethia obscura	Shrubland morethia skink	0.75	0.75	0.70
	Tiliqua occipitalis	Western blue-tongued lizard	0.78	0.78	0.73
	Tiliqua rugosa	Shingle-back	0.72	0.72	0.70

Table S4.2.5: Cross validated variable contributions for alternative model M1 with spatial fire variables based on early post-fire age class

Family	Scientific name	Common name	Time since fire	Area early post-fire age class	Diversity	Annual Rainfall	Elevation	Veg. Type
Blind snakes	*Anilios bicolor*	Dark-spined blind snake	22.64	9.20	1.37	4.09	40.02	22.68
	Anilios bituberculatus	Prong-snouted blind snake	23.14	0.48	0.51	33.38	34.88	7.60
Dragons	*Ctenophorus fordi*	Mallee dragon	23.20	10.04	11.35	2.47	10.34	42.60
	Ctenophorus pictus	Painted dragon	5.22	64.52	7.08	17.35	2.38	3.46
	Diporiphora nobbi	Nobbi dragon	12.95	7.28	7.12	36.77	15.59	20.29
	Pogona vitticeps	Central bearded dragon	27.40	10.00	8.51	6.64	31.68	15.77
Elapid snakes	*Brachyurophis australis*	Coral snake	25.91	20.94	19.49	11.25	4.91	17.49
	Parasuta nigriceps	Short-tailed snake	12.51	7.70	6.11	67.47	4.45	1.75
	Pseudonaja modesta	Ringed brown snake	6.35	1.58	3.20	84.15	0.85	3.88
	Pseudonaja textilis	Eastern brown snake	21.09	0.00	0.00	0.00	0.00	71.16
	Vermicella annulata	Bandy-bandy	18.54	27.96	34.54	17.18	0.00	1.78
Geckos	*Diplodactylus vittatus*	Wood gecko	9.45	4.04	4.11	70.69	8.99	2.72
	Gehyra variegata	Tree Dtella	18.03	27.87	5.68	40.78	4.79	2.85
	Lucasium damaeum	Smooth knob-tailed gecko	18.08	22.82	11.50	12.69	10.54	24.36
	Nephrurus levis	Beaded gecko	17.18	1.30	0.94	63.38	5.25	11.96
	Rhynchoedura ormsbyi	Eastern beaked gecko	5.53	13.87	3.96	25.67	12.27	38.69
	Rhynchoedura ornata	Western beaked gecko	9.91	26.17	43.98	8.47	7.76	3.72
	Strophurus elderi	Jewelled gecko	17.97	0.94	0.43	28.31	0.19	52.16
	Strophurus intermedius	Southern spiny-tailed gecko	9.89	1.82	0.38	58.80	17.11	12.00
	Strophurus williamsi	Eastern spiny-tailed gecko	0.00	0.42	0.20	59.38	0.51	39.49
	Underwoodisaurus milii	Thick-tailed gecko	0.00	0.00	0.00	99.41	0.00	0.59
Goanna	*Varanus gouldii*	Sand goanna	40.48	4.41	5.90	12.80	6.17	30.24
Legless lizards	*Aprasia inaurita*	Mallee worm lizard	1.49	0.78	0.06	90.89	1.36	5.42
	Delma australis	Marble-faced delma	23.64	6.30	3.50	15.30	41.08	10.18
	Delma butleri	Unbanded delma	40.24	7.78	11.45	8.49	6.21	25.83
	Lialis burtonis	Burton’s snake-lizard	34.17	16.42	11.78	1.04	11.95	24.64

Family	Scientific name	Common name	Time since fire	Area early post-fire age class	Diversity	Annual Rainfall	Elevation	Veg. Type
	Pygopus lepidopodus	Common scaly-foot	37.61	0.06	9.32	2.75	11.68	38.58
Skinks	*Ctenotus atlas*	Southern mallee ctenotus	20.46	2.26	4.62	27.56	0.66	44.43
	Ctenotus brachyonyx	Short-clawed ctenotus	17.22	7.00	8.03	5.47	8.75	53.53
	Ctenotus brooksi iridis	Wedgesnout ctenotus	40.66	9.12	2.85	2.03	0.00	45.35
	Ctenotus orientalis	Eastern ctenotus	7.37	9.23	8.39	72.66	0.07	2.29
	Ctenotus regius	Pale-rumped ctenotus	23.40	16.27	10.65	8.56	5.82	35.29
	Ctenotus schomburgkii	Barred wedgesnout ctenotus	5.32	4.36	8.11	69.52	3.42	9.27
	Cyclodomorphus melanops	Spinifex slender bluetongue	32.37	0.77	0.00	36.68	0.58	29.60
	Eremiascincus richardsonii	Broad-banded sand-swimmer	1.43	6.36	1.43	83.82	0.01	6.95
	Lerista aericeps	Desert plain slider	1.28	89.16	0.00	6.93	2.62	0.01
	Lerista bougainvillii	South-eastern slider	1.93	2.12	3.47	87.72	4.61	0.16
	Lerista labialis	Southern sandslider	0.56	0.02	0.20	89.07	0.20	9.94
	Lerista muelleri	Wood mulch-slider	15.82	6.58	4.76	45.86	26.55	0.43
	Lerista punctatovittata	Eastern robust slider	32.20	12.30	15.45	0.24	1.94	37.88
	Lerista xanthura	Yellow-tailed plain slider	0.00	65.23	3.50	0.13	0.13	31.01
	Liopholis inornata	Desert skink	39.24	12.37	9.07	5.46	2.97	30.88
	Menetia greyii	Common dwarf skink	67.54	4.41	9.32	4.84	4.12	9.78
	Morethia boulengeri	South-eastern morethia skink	26.64	1.94	13.56	10.34	10.99	36.54
	Morethia obscura	Shrubland morethia skink	35.49	4.61	13.07	4.03	2.78	40.02
	Tiliqua occipitalis	Western blue-tongued lizard	22.80	0.00	15.79	0.01	31.79	29.61
	Tiliqua rugosa	Shingle-back	25.32	1.54	4.84	22.68	22.07	23.55

Table S4.2.6: Cross validated variable contributions for alternative model M2 with spatial fire variables based on early post-fire age class

Family	Scientific name	Common name	Time since fire	ConFigure of early post-fire age class	Diversity	Annual Rainfall	Elevation	Veg. Type
Blind snakes	*Anilios bicolor*	Dark-spined blind snake	16.07	21.05	0.86	5.51	35.44	21.07
	Anilios bituberculatus	Prong-snouted blind snake	22.08	2.37	0.56	34.27	33.19	7.53
Dragons	*Ctenophorus fordi*	Mallee dragon	18.56	17.03	8.45	2.57	11.63	41.76
	Ctenophorus pictus	Painted dragon	7.18	68.41	1.68	15.96	3.38	3.39
	Diporiphora nobbi	Nobbi dragon	13.57	10.83	4.39	38.79	13.15	19.27
	Pogona vitticeps	Central bearded dragon	25.71	18.01	10.17	4.97	26.41	14.73
Elapid snakes	*Brachyurophis australis*	Coral snake	20.61	27.24	21.78	10.22	4.91	15.23
	Parasuta nigriceps	Short-tailed snake	11.24	11.32	10.16	61.59	4.19	1.50
	Pseudonaja modesta	Ringed brown snake	4.92	4.98	4.04	80.63	1.79	3.65
	Pseudonaja textilis	Eastern brown snake	0.00	29.09	0.00	0.00	0.00	70.91
	Vermicella annulata	Bandy-bandy	2.81	46.15	46.15	3.84	0.00	1.04
Geckos	*Diplodactylus vittatus*	Wood gecko	8.30	7.93	7.40	65.20	8.85	2.33
	Gehyra variegata	Tree Dtella	19.34	21.37	8.66	43.95	3.93	2.76
	Lucasium damaeum	Smooth knob-tailed gecko	13.37	34.72	3.80	12.28	12.31	23.53
	Nephrurus levis	Beaded gecko	18.37	3.18	1.00	61.27	4.60	11.58
	Rhynchoedura ormsbyi	Eastern beaked gecko	5.19	10.49	6.27	25.82	12.87	39.36
	Rhynchoedura ornata	Western beaked gecko	2.45	61.08	29.48	4.46	1.92	0.61
	Strophurus elderi	Jewelled gecko	17.50	0.10	0.37	28.14	0.34	53.54
	Strophurus intermedius	Southern spiny-tailed gecko	8.54	4.67	0.47	58.05	16.72	11.56
	Strophurus williamsi	Eastern spiny-tailed gecko	0.00	0.88	0.20	58.58	1.24	39.10
	Underwoodisaurus milii	Thick-tailed gecko	0.00	0.00	0.00	99.41	0.00	0.59
Goanna	*Varanus gouldii*	Sand goanna	36.27	10.06	8.46	12.27	3.97	28.97
Legless lizards	*Aprasia inaurita*	Mallee worm lizard	0.99	0.14	0.00	91.93	1.48	5.46
	Delma australis	Marble-faced delma	22.61	7.35	4.64	14.30	41.01	10.09
	Delma butleri	Unbanded delma	34.89	16.87	12.90	5.80	6.21	23.33

Family	Scientific name	Common name	Time since fire	ConFigure of early post-fire age class	Diversity	Annual Rainfall	Elevation	Veg. Type
	Lialis burtonis	Burton's snake-lizard	30.00	16.90	16.12	1.47	11.20	24.32
	Pygopus lepidopodus	Common scaly-foot	32.67	5.60	10.39	1.34	10.50	39.50
Skinks	*Ctenotus atlas*	Southern mallee ctenotus	19.60	13.24	3.92	21.75	0.39	41.09
	Ctenotus brachyonyx	Short-clawed ctenotus	14.37	11.80	10.36	2.66	9.59	51.22
	Ctenotus brooksi iridis	Wedgesnout ctenotus	40.21	11.31	3.30	2.04	0.00	43.15
	Ctenotus orientalis	Eastern ctenotus	10.41	5.59	1.71	79.73	0.42	2.14
	Ctenotus regius	Pale-rumped ctenotus	22.27	24.30	6.52	5.77	5.61	35.52
	Ctenotus schomburgkii	Barred wedgesnout ctenotus	4.44	10.51	6.39	66.94	3.35	8.37
	Cyclodomorphus melanops	Spinifex slender bluetongue	32.59	0.00	0.00	36.90	0.50	30.00
	Eremiascincus richardsonii	Broad-banded sand-swimmer	2.21	0.00	2.21	88.66	0.00	6.93
	Lerista aericeps	Desert plain slider	0.16	93.04	0.00	6.25	0.52	0.03
	Lerista bougainvillii	South-eastern slider	2.17	1.23	2.41	88.83	5.15	0.21
	Lerista labialis	Southern sandslider	0.58	0.01	0.19	89.07	0.20	9.95
	Lerista muelleri	Wood mulch-slider	15.57	5.56	6.03	44.82	27.65	0.37
	Lerista punctatovittata	Eastern robust slider	25.75	17.96	17.86	0.38	1.89	36.16
	Lerista xanthura	Yellow-tailed plain slider	0.05	12.91	0.00	0.30	0.30	86.45
	Liopholis inornata	Desert skink	23.36	36.36	4.57	7.39	0.91	27.41
	Menetia greyii	Common dwarf skink	59.94	11.00	12.05	3.73	4.15	9.15
	Morethia boulengeri	South-eastern morethia skink	25.51	8.77	10.43	10.17	10.91	34.21
	Morethia obscura	Shrubland morethia skink	32.67	9.85	12.11	3.62	2.25	39.50
	Tiliqua occipitalis	Western blue-tongued lizard	22.48	2.70	12.84	0.00	30.05	31.92
	Tiliqua rugosa	Shingle-back	25.94	2.58	5.82	23.17	19.70	22.80

Table S4.2.7: Cross validated variable contributions for alternative model M3 with spatial fire variables based on early post-fire age class

Family	Scientific name	Common name	Fire frequency	Area of mid-old triodia mallee	Diversity	Annual Rainfall	Elevation	Veg. Type
Blind snakes	*Anilios bicolor*	Dark-spined blind snake	9.79	10.36	7.95	9.52	38.21	24.17
	Anilios bituberculatus	Prong-snouted blind snake	1.82	0.47	0.34	44.79	41.42	11.16
Dragons	*Ctenophorus fordi*	Mallee dragon	19.54	4.67	21.16	1.92	10.06	42.66
	Ctenophorus pictus	Painted dragon	10.24	62.90	8.47	13.56	2.24	2.59
	Diporiphora nobbi	Nobbi dragon	8.27	4.93	11.69	38.96	14.99	21.16
	Pogona vitticeps	Central bearded dragon	15.90	8.47	20.74	10.88	24.90	19.11
Elapid snakes	*Brachyurophis australis*	Coral snake	28.68	5.88	32.22	11.75	4.51	16.97
	Parasuta nigriceps	Short-tailed snake	2.75	6.84	8.05	75.18	4.82	2.36
	Pseudonaja modesta	Ringed brown snake	0.66	2.72	3.89	87.68	0.28	4.76
	Pseudonaja textilis	Eastern brown snake	92.62	0.33	0.00	0.00	0.00	7.05
	Vermicella annulata	Bandy-bandy	0.00	34.29	41.76	21.94	0.00	2.00
Geckos	*Diplodactylus vittatus*	Wood gecko	6.95	1.38	8.28	71.51	8.97	2.92
	Gehyra variegata	Tree Dtella	0.15	33.14	9.05	47.52	5.51	4.63
	Lucasium damaeum	Smooth knob-tailed gecko	17.77	17.44	20.06	12.15	8.76	23.82
	Nephrurus levis	Beaded gecko	6.84	1.05	1.92	69.11	5.90	15.18
	Rhynchoedura ormsbyi	Eastern beaked gecko	6.80	15.11	5.94	24.37	11.33	36.45
	Rhynchoedura ornata	Western beaked gecko	23.17	16.89	50.60	2.54	5.02	1.78
	Strophurus elderi	Jewelled gecko	23.85	0.09	11.47	15.19	1.54	47.86
	Strophurus intermedius	Southern spiny-tailed gecko	2.81	2.65	3.03	62.26	16.30	12.95
	Strophurus williamsi	Eastern spiny-tailed gecko	4.33	4.58	0.23	53.76	0.86	36.24
	Underwoodisaurus milii	Thick-tailed gecko	0.00	0.00	0.00	99.41	0.00	0.59
Goanna	*Varanus gouldii*	Sand goanna	16.28	2.37	15.74	11.61	5.84	48.16
Legless lizards	*Aprasia inaurita*	Mallee worm lizard	0.22	0.15	0.37	93.00	0.64	5.61
	Delma australis	Marble-faced delma	13.74	4.61	12.32	18.08	38.88	12.36
	Delma butleri	Unbanded delma	32.16	4.17	26.35	2.59	7.93	26.80

Family	Scientific name	Common name	Fire frequency	Area of mid-old triodia mallee	Diversity	Annual Rainfall	Elevation	Veg. Type
	Lialis burtonis	Burton's snake-lizard	11.65	15.28	21.38	4.25	15.76	31.68
	Pygopus lepidopodus	Common scaly-foot	14.05	0.01	18.12	0.05	15.11	52.66
Skinks	*Ctenotus atlas*	Southern mallee ctenotus	14.24	0.45	13.64	21.71	2.21	47.76
	Ctenotus brachyonyx	Short-clawed ctenotus	15.95	1.24	15.86	1.63	11.93	53.38
	Ctenotus brooksi iridis	Wedgesnout ctenotus	1.01	12.12	6.39	0.00	0.00	80.48
	Ctenotus orientalis	Eastern ctenotus	0.84	9.84	9.35	78.07	0.24	1.66
	Ctenotus regius	Pale-rumped ctenotus	13.20	15.68	13.76	10.78	10.32	36.26
	Ctenotus schomburgkii	Barred wedgesnout ctenotus	6.69	2.38	10.32	67.61	4.56	8.45
	Cyclodomorphus melanops	Spinifex slender bluetongue	0.00	0.88	0.00	23.88	0.15	75.09
	Eremiascincus richardsonii	Broad-banded sand-swimmer	0.13	6.77	0.76	85.59	0.49	6.25
	Lerista aericeps	Desert plain slider	0.00	92.91	0.00	5.41	1.60	0.09
	Lerista bougainvillii	South-eastern slider	2.74	1.44	4.08	87.16	4.38	0.18
	Lerista labialis	Southern sandslider	0.00	0.02	0.18	89.50	0.18	10.12
	Lerista muelleri	Wood mulch-slider	14.85	4.59	15.86	38.78	25.46	0.47
	Lerista punctatovittata	Eastern robust slider	28.17	2.45	30.61	0.38	1.20	37.18
	Lerista xanthura	Yellow-tailed plain slider	0.00	65.25	3.55	0.13	0.13	30.94
	Liopholis inornata	Desert skink	32.01	6.44	25.75	1.92	3.16	30.73
	Menetia greyii	Common dwarf skink	23.93	4.76	28.52	15.16	5.53	22.10
	Morethia boulengeri	South-eastern morethia skink	4.42	0.28	15.23	9.24	15.34	55.48
	Morethia obscura	Shrubland morethia skink	13.79	4.34	21.75	0.77	2.86	56.48
	Tiliqua occipitalis	Western blue-tongued lizard	3.38	0.00	17.86	0.70	36.25	41.81
	Tiliqua rugosa	Shingle-back	0.44	2.98	6.10	35.93	24.30	30.25

Appendix S4.3 – Response plots of mallee reptiles by taxonomic group

Preface: Figures in this appendix are formatted as embedded PDF files and open externally for enhanced viewing

Blind snakes (Typhlopidae)

a)

b)

c)

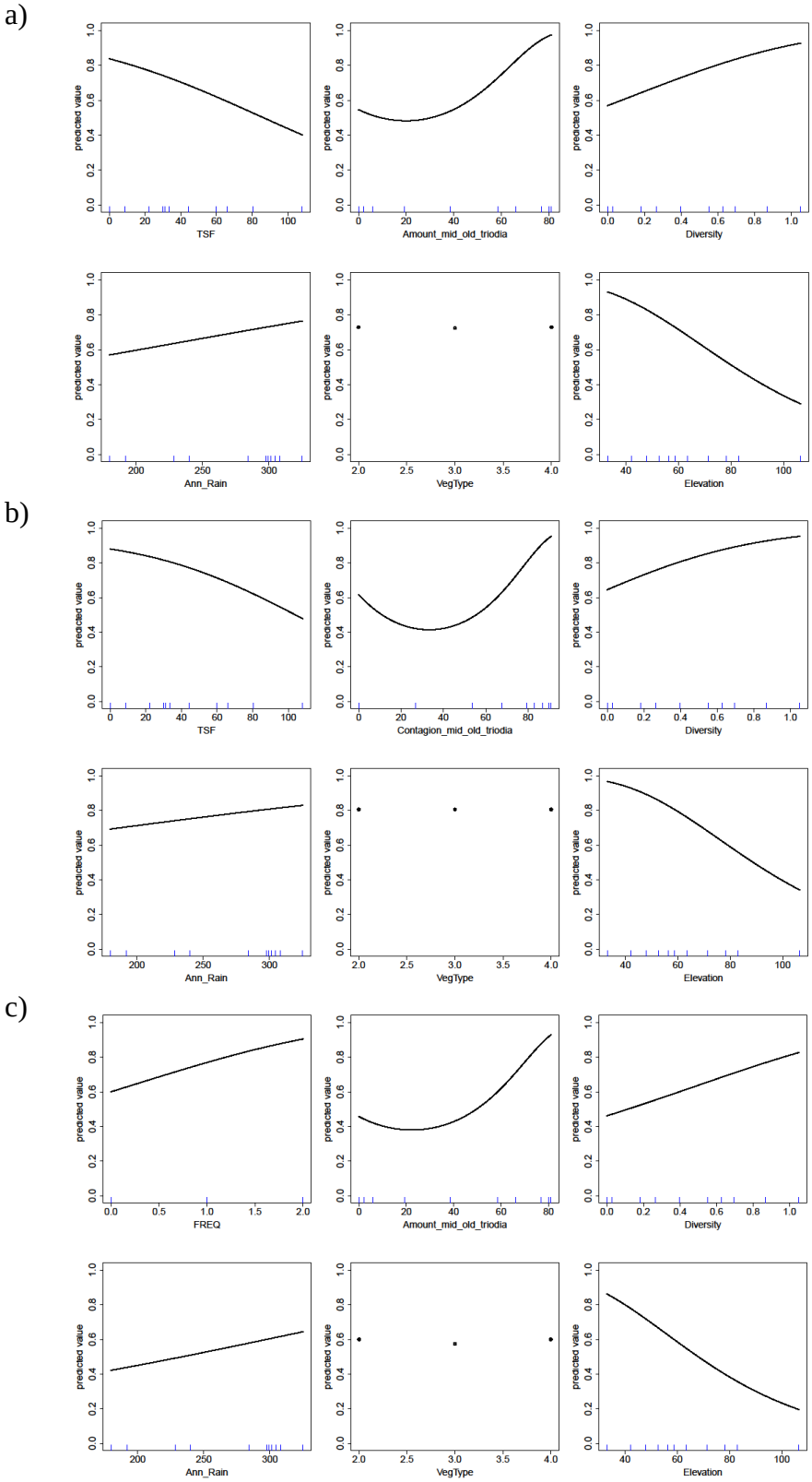
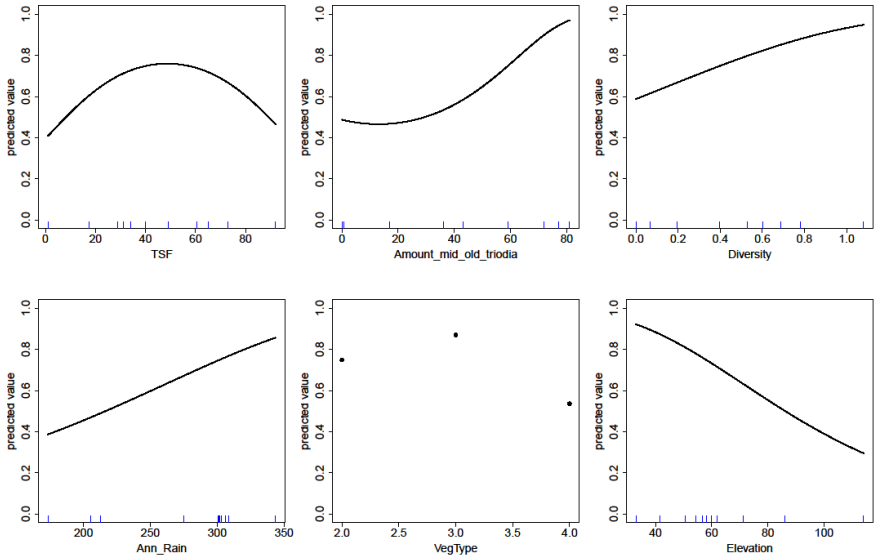


Figure S4.3.1: Response of *Anilios bicolor* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



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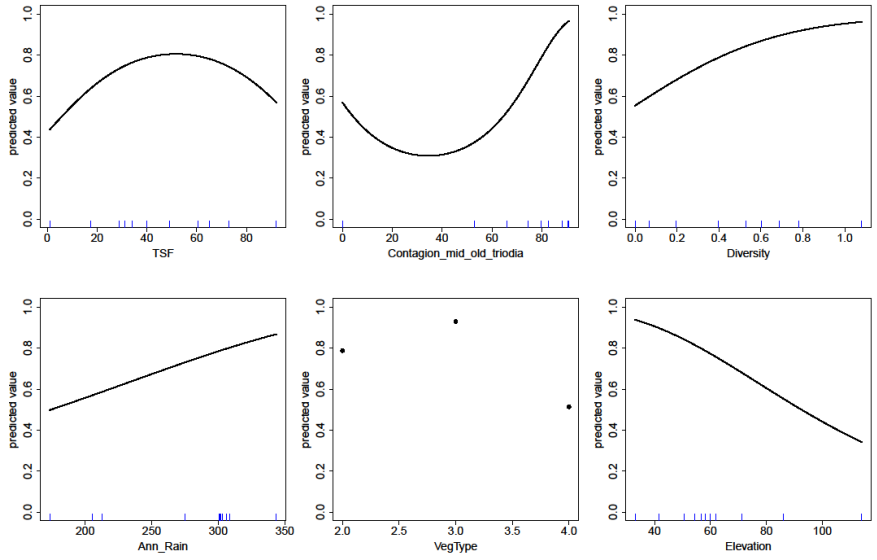

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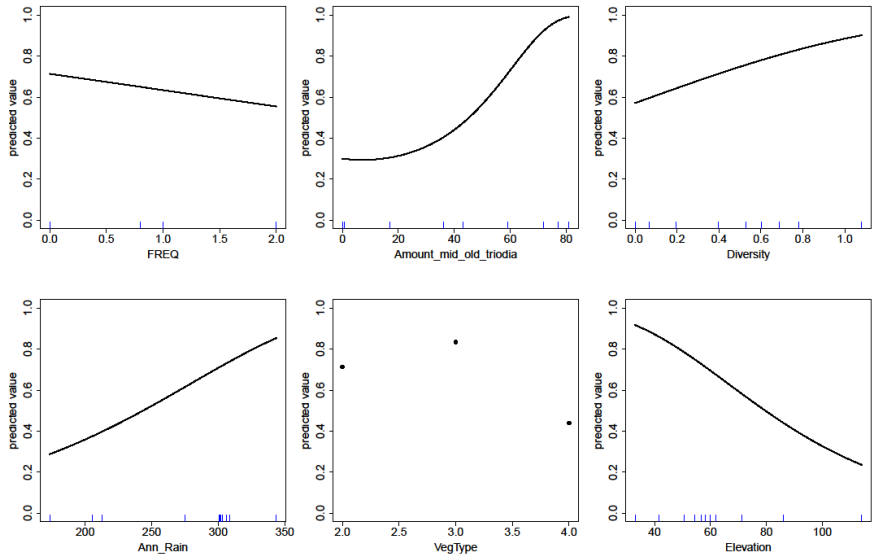
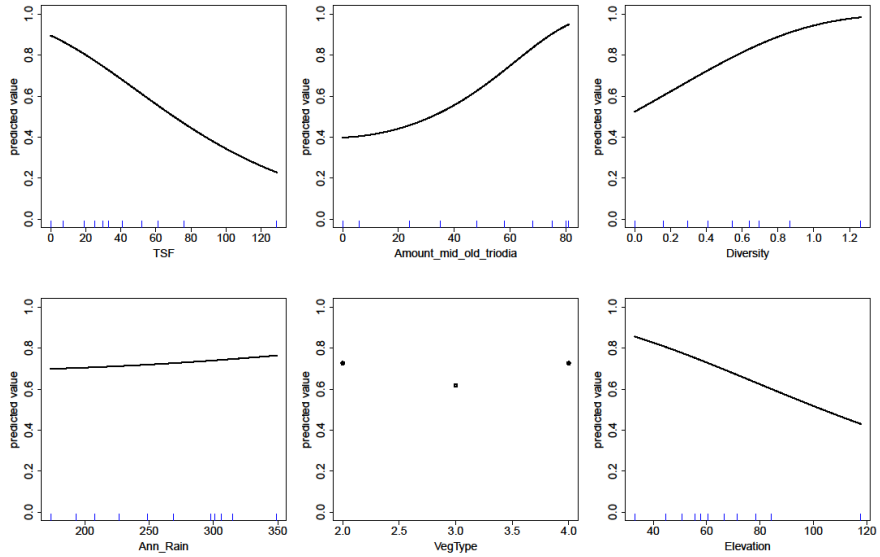


Figure S4.3.2: Response of *Anilios bituberculatus* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

Dragons (Agamidae)

a)



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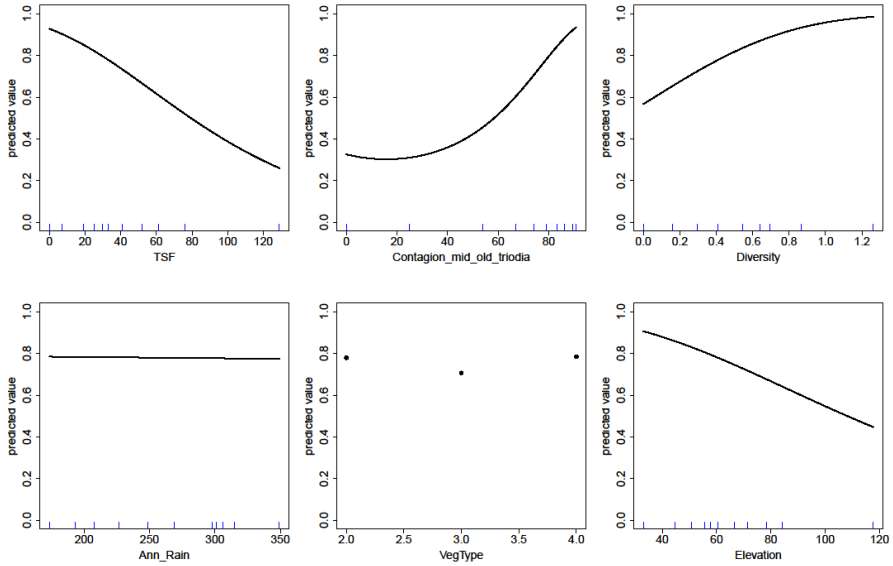

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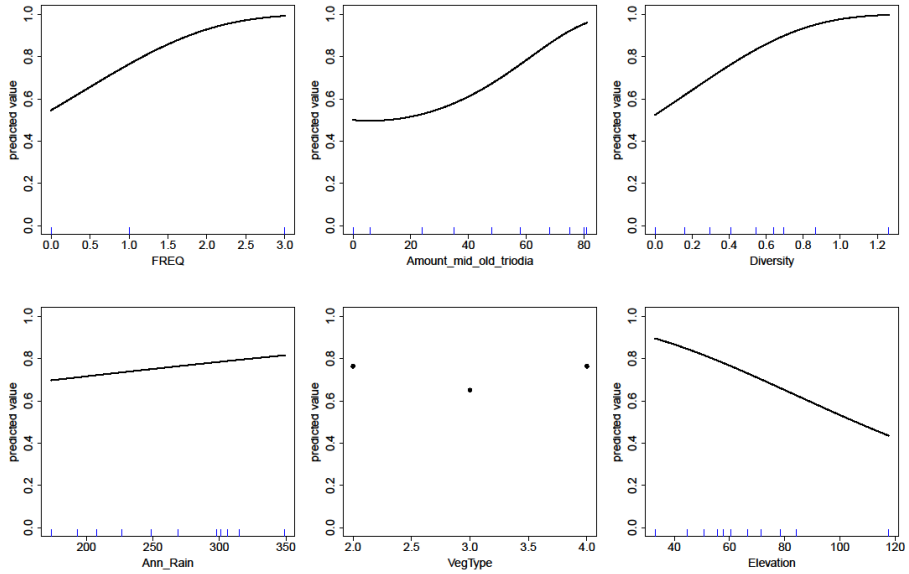
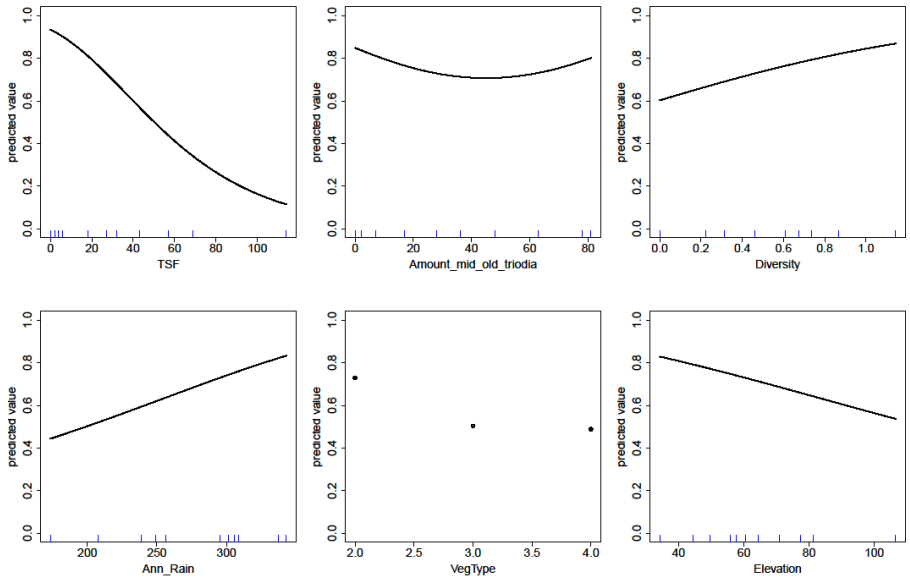


Figure S4.3.3: Response of *Ctenophorus fordi* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



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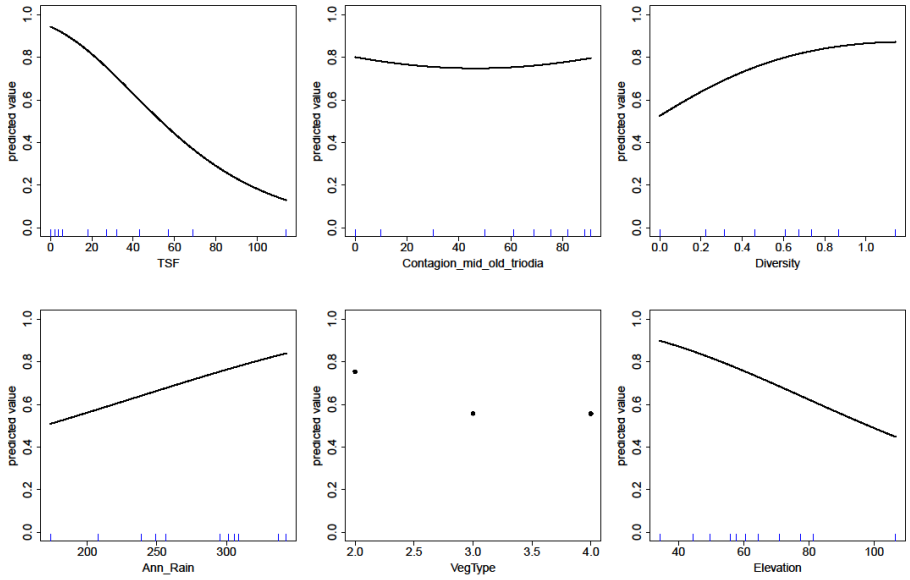

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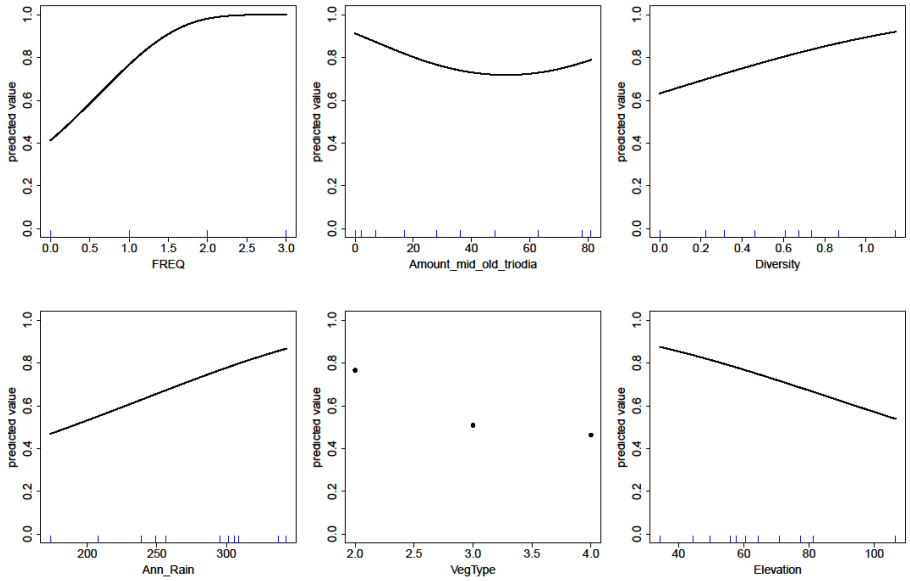
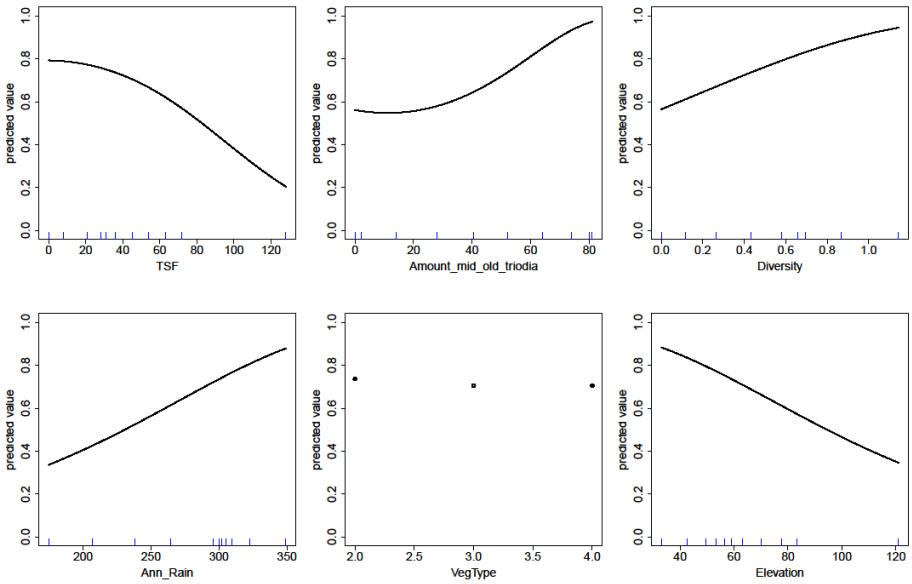


Figure S4.3.4: Response of *Ctenophorus pictus* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



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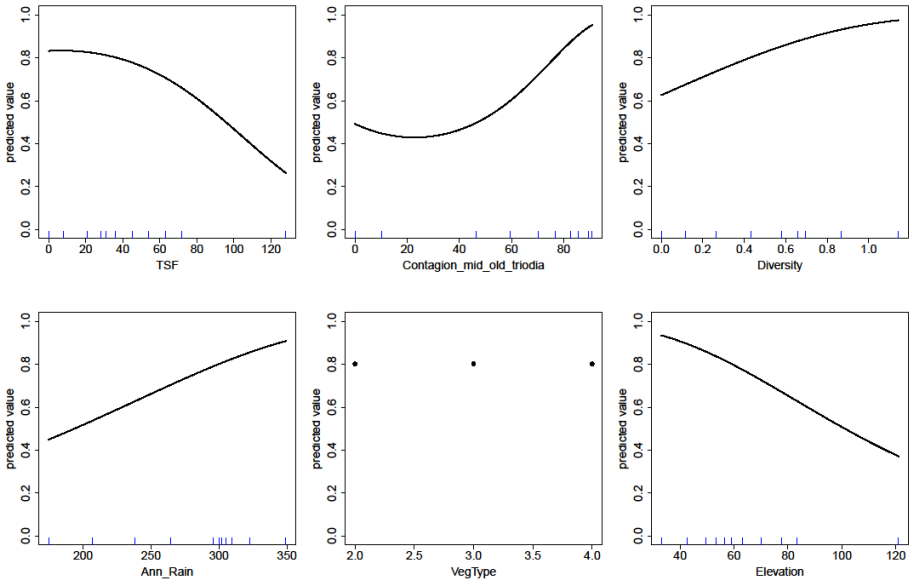

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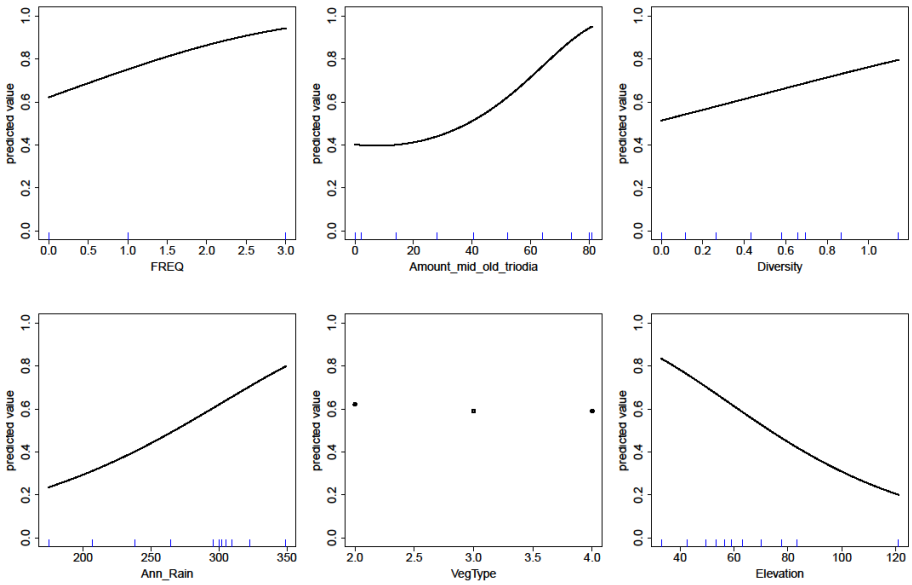
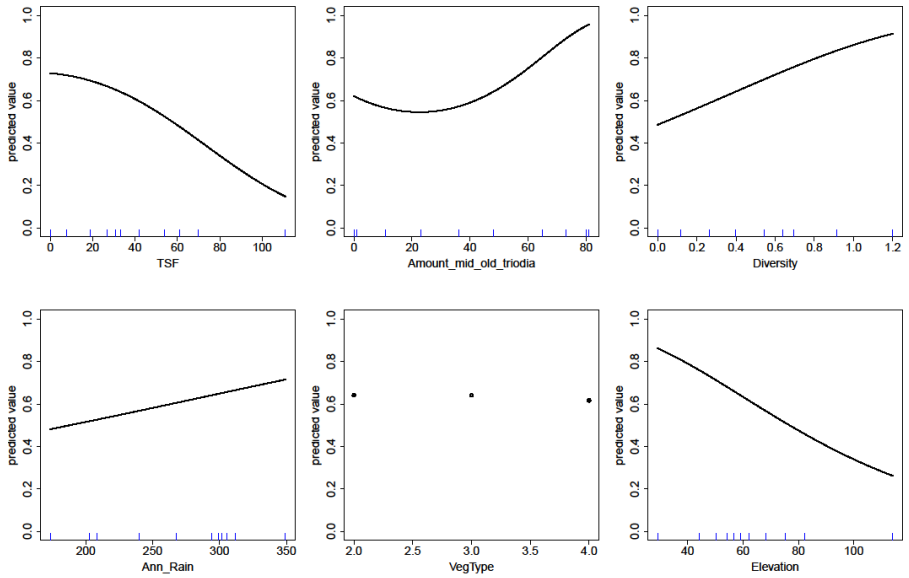


Figure S4.3.5: Response of *Diporiphora nobbi* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



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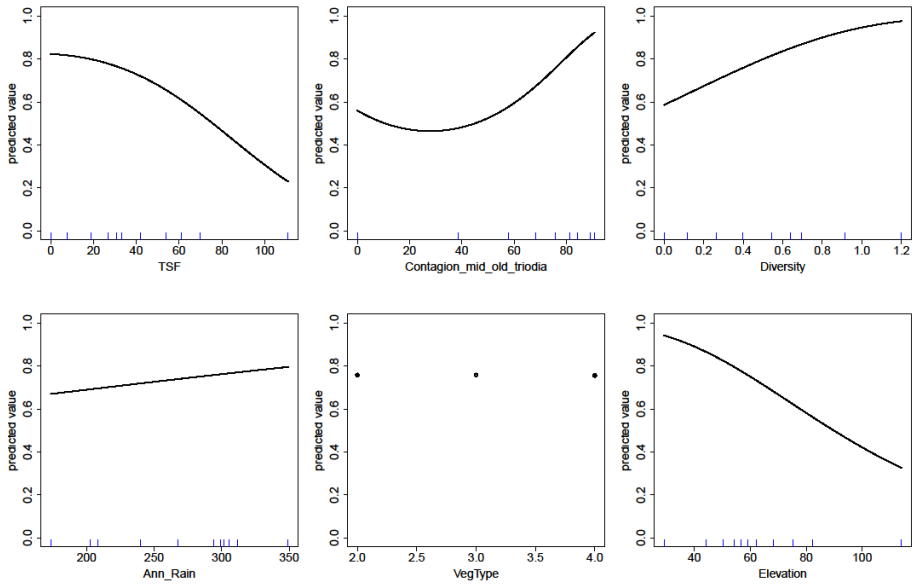

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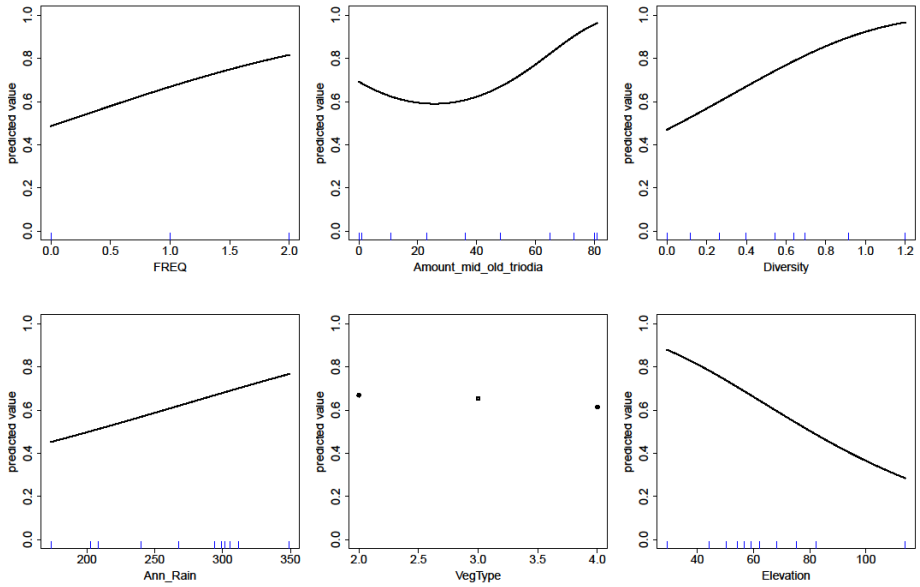


Figure S4.3.6: Response of *Pogona vitticeps* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

Elapid snakes (Elapidae)

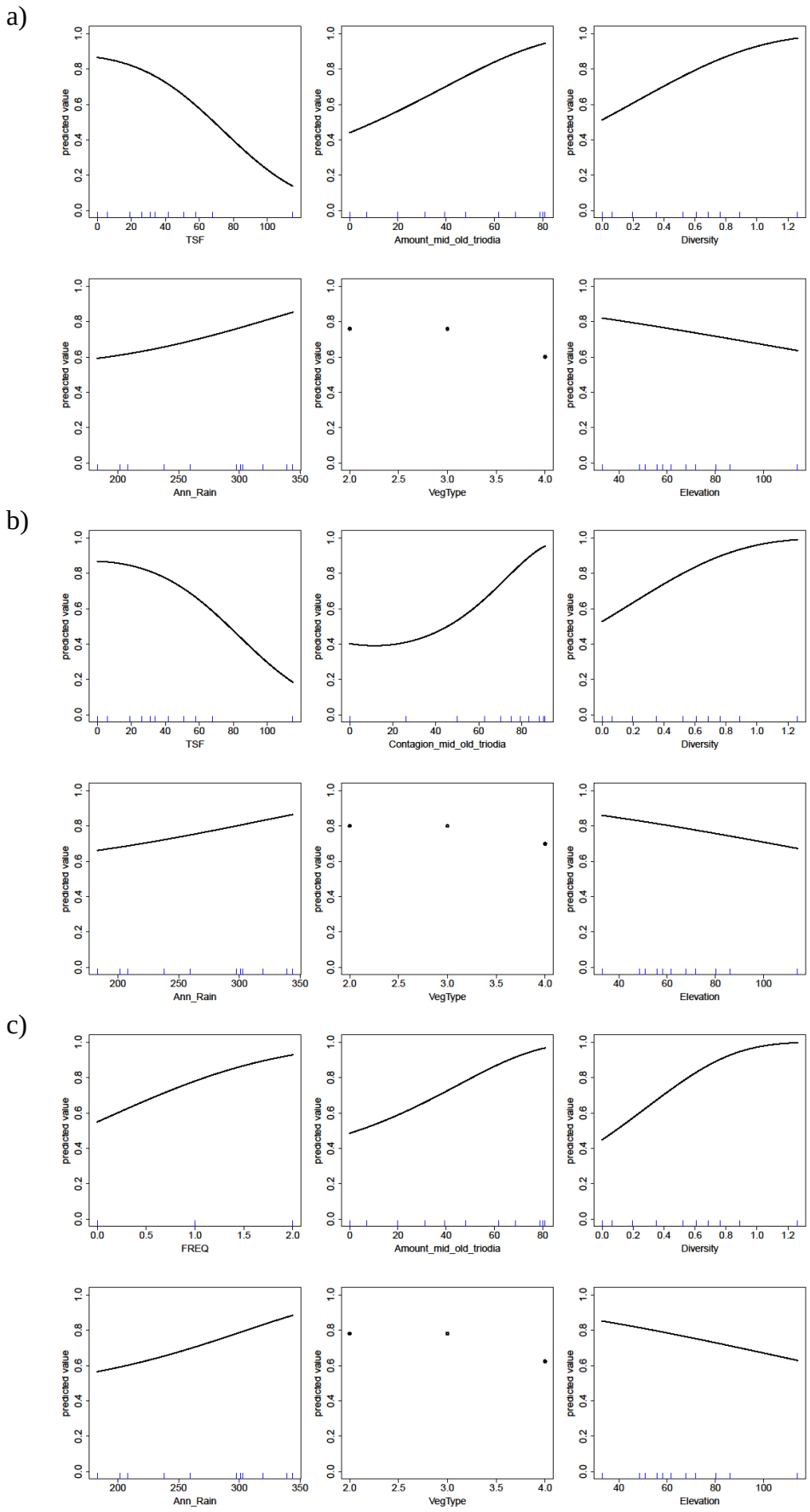


Figure S4.3.7: Response of *Brachyurophis australis* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

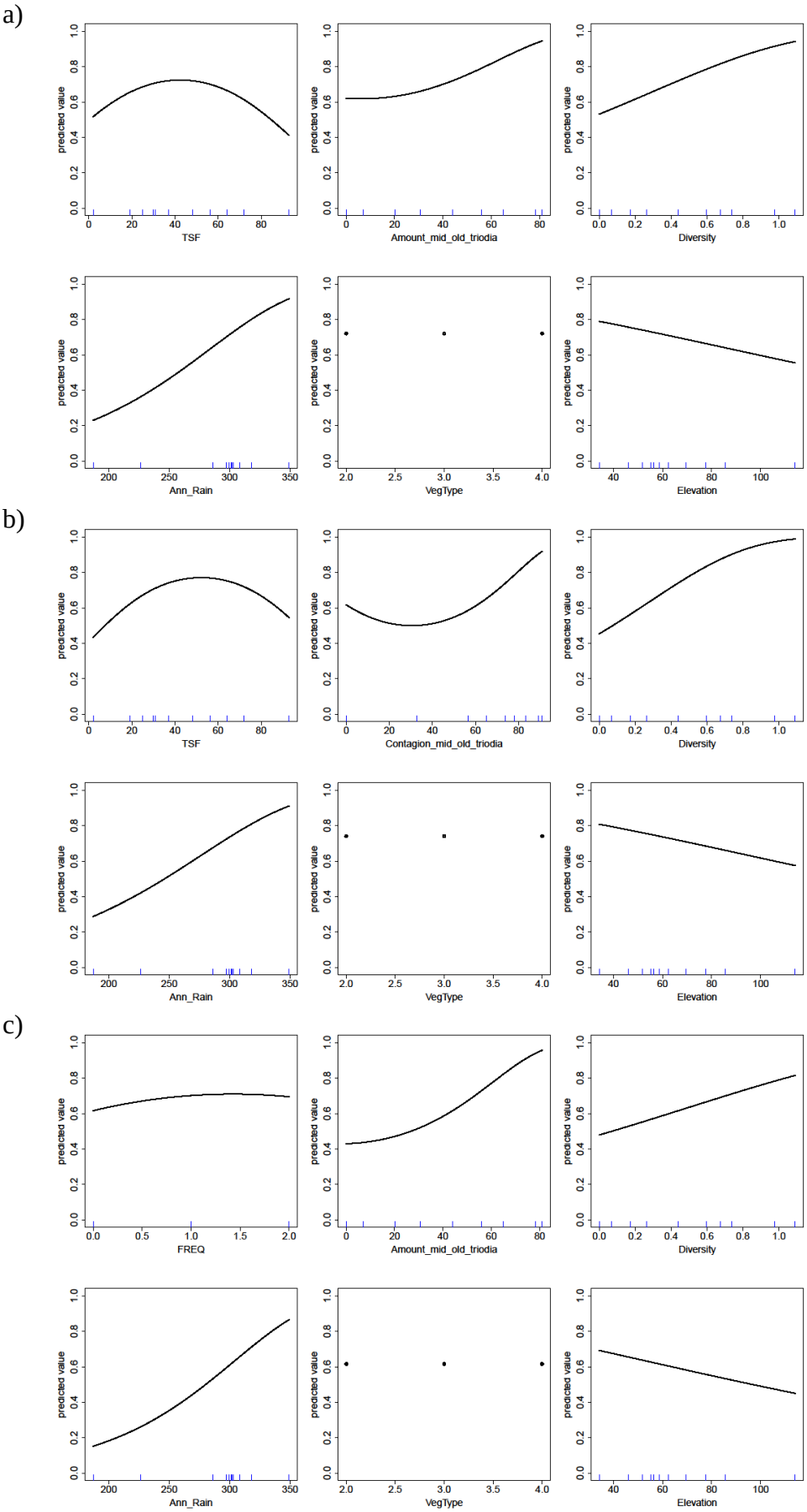


Figure S4.3.8: Response of *Parasuta nigriceps* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

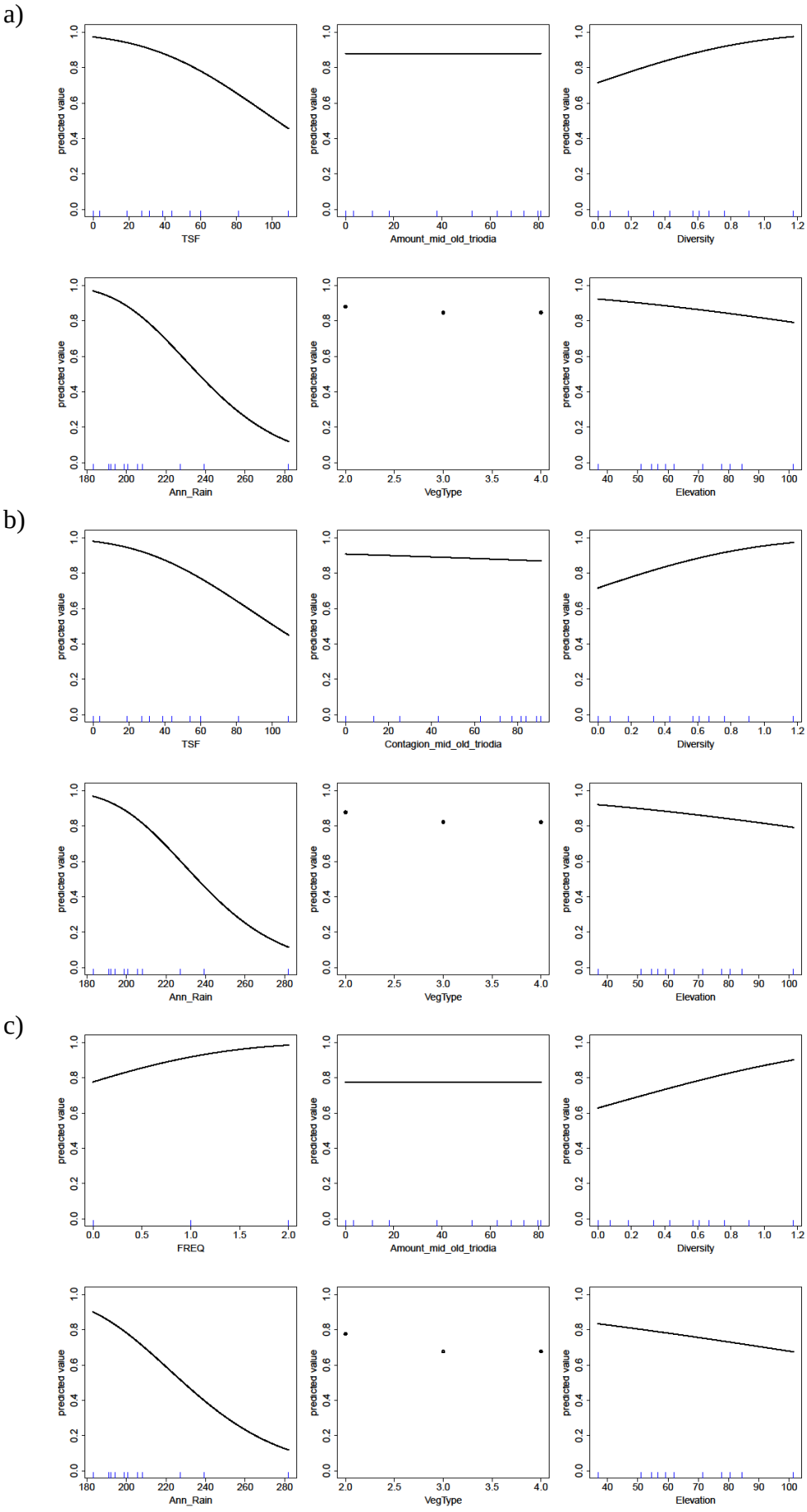
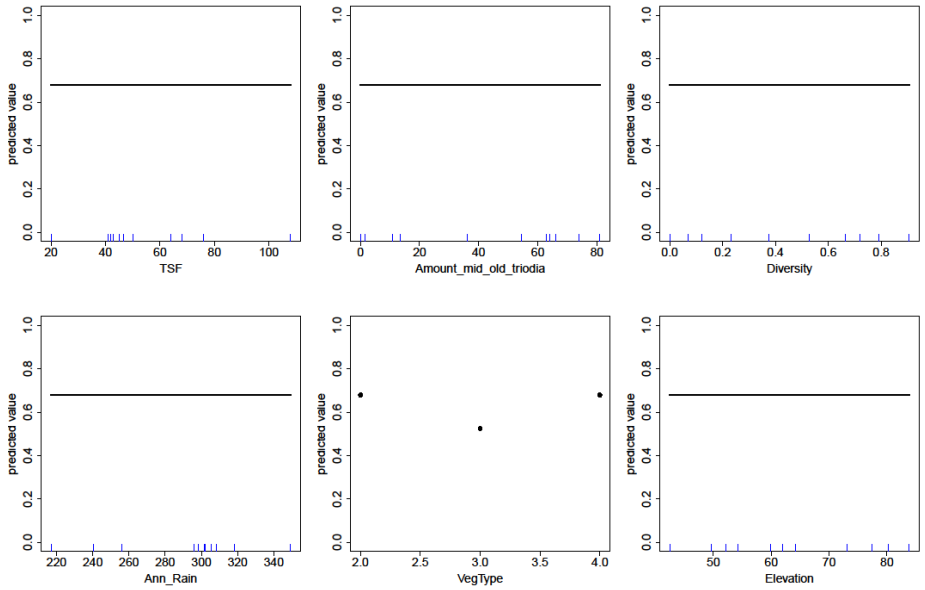


Figure S4.3.9: Response of *Pseudonaja modesta* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



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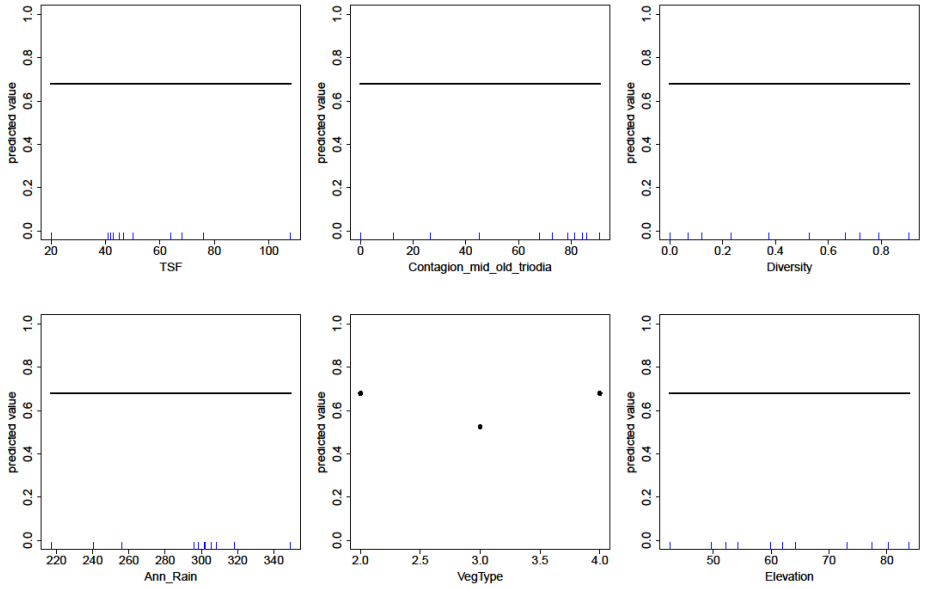

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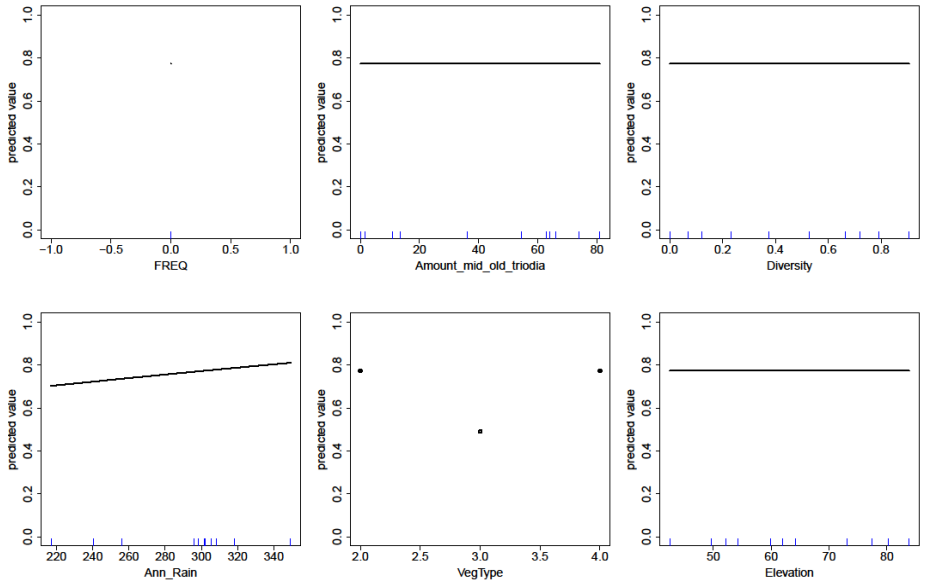

Figure S4.3.10: Response of *Pseudonaja textilis* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

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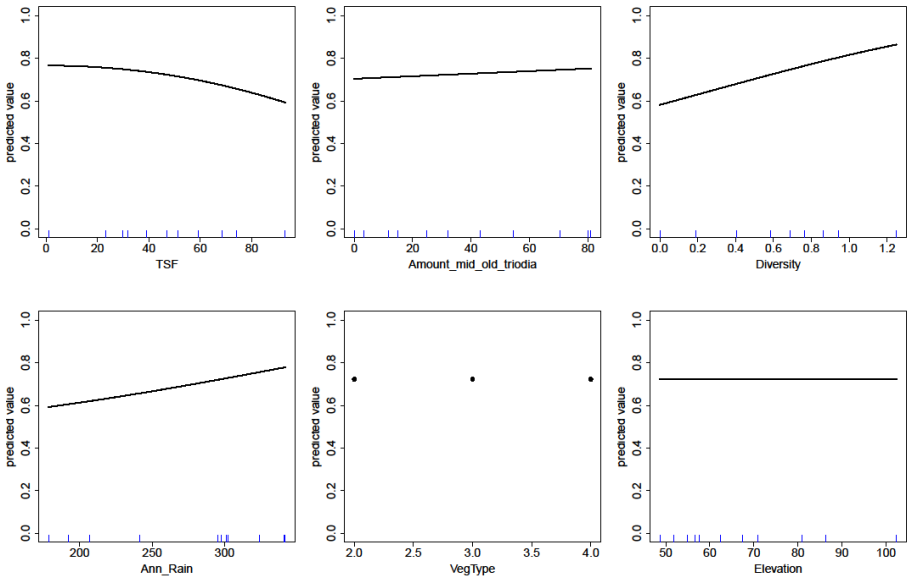

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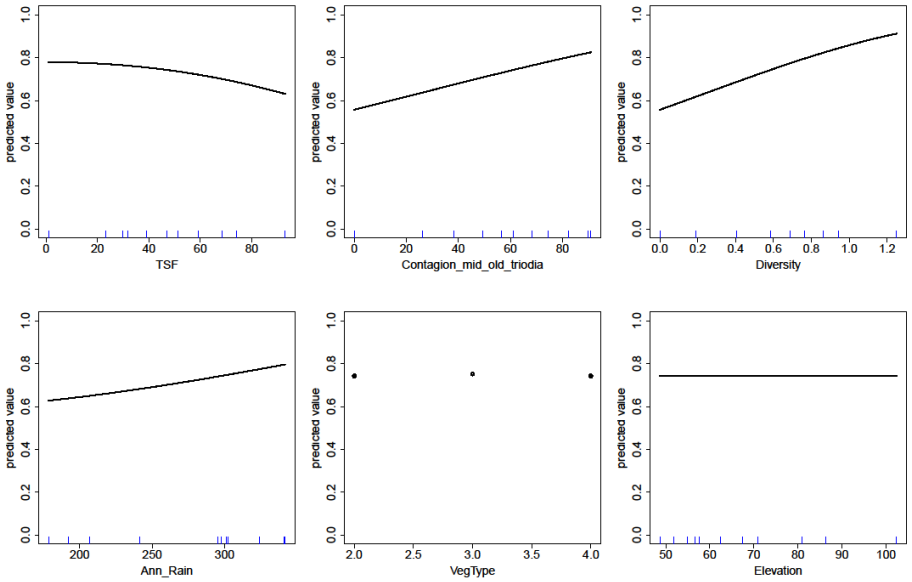


c)

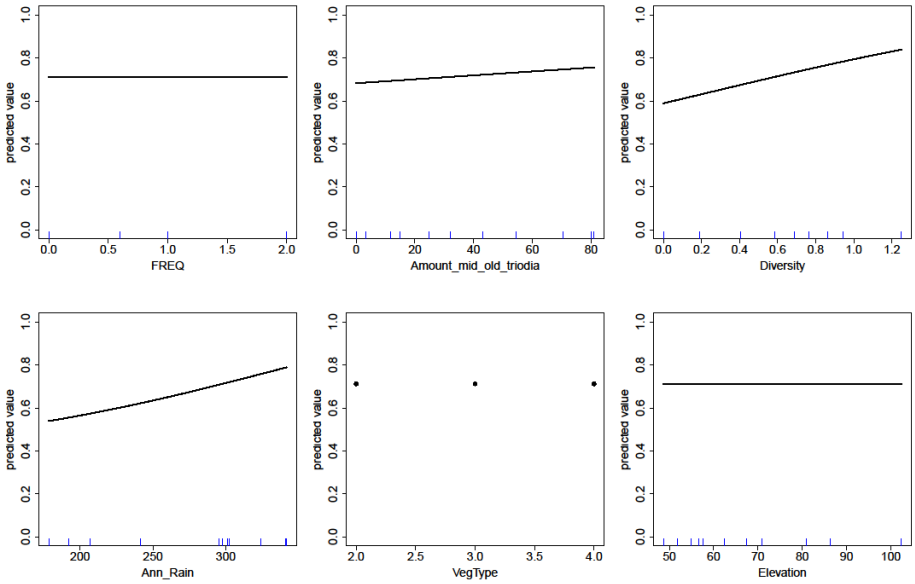


Figure S4.3.11: Response of *Vermicella annulata* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

Geckos (Carphodactylidae and Diplodactylidae)

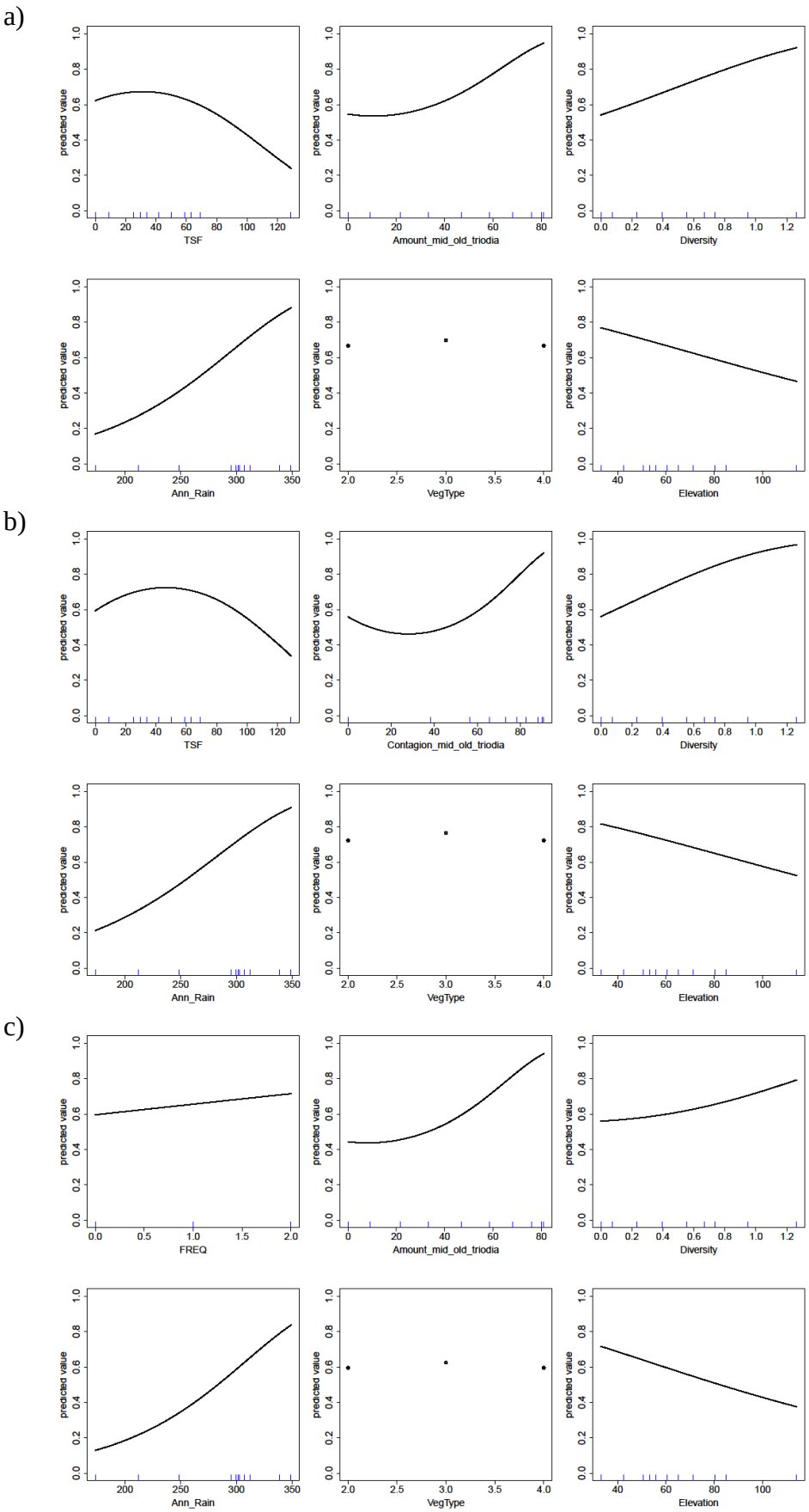
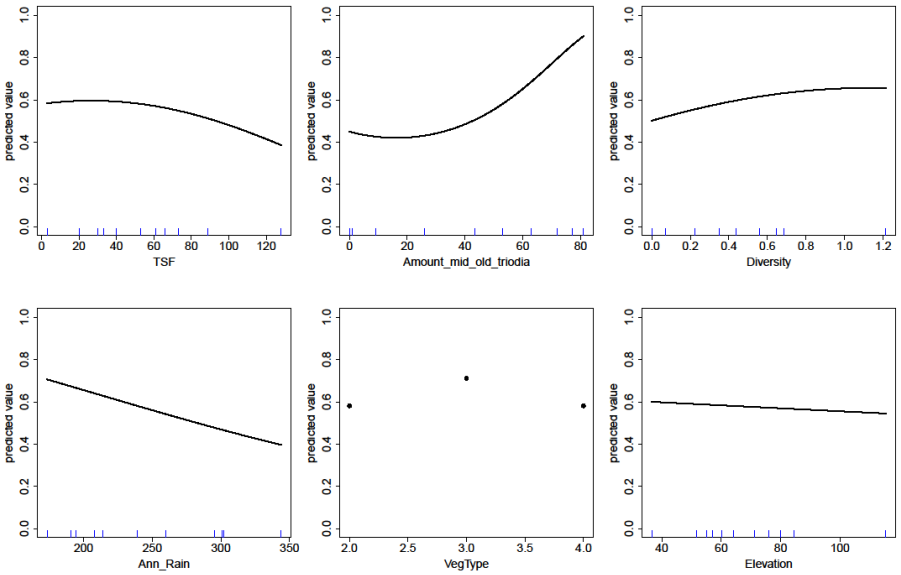
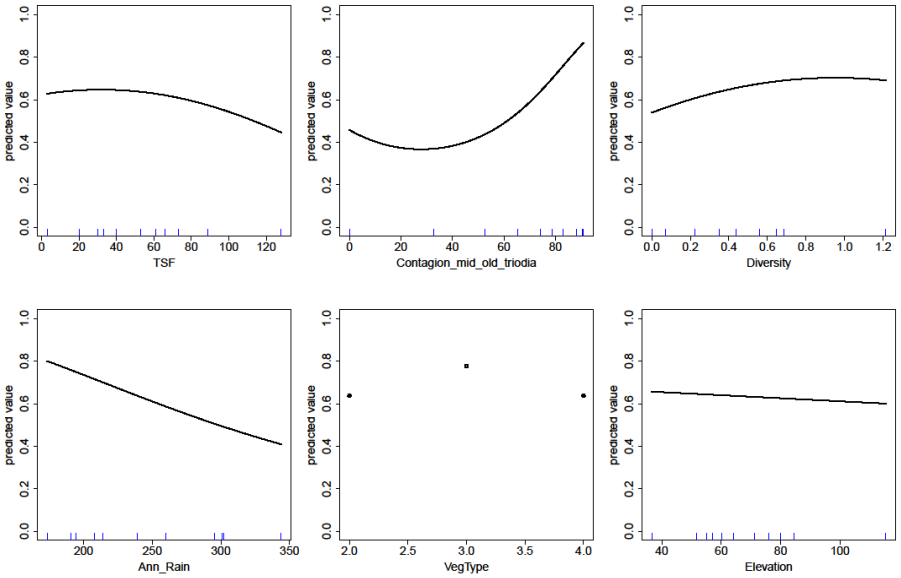


Figure S4.3.12: Response of *Diplodactylus vittatus* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

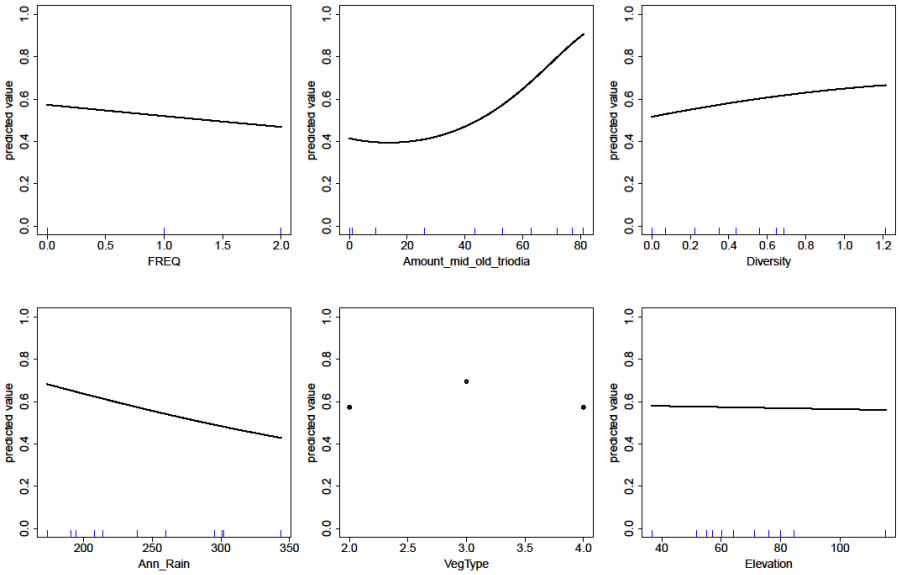
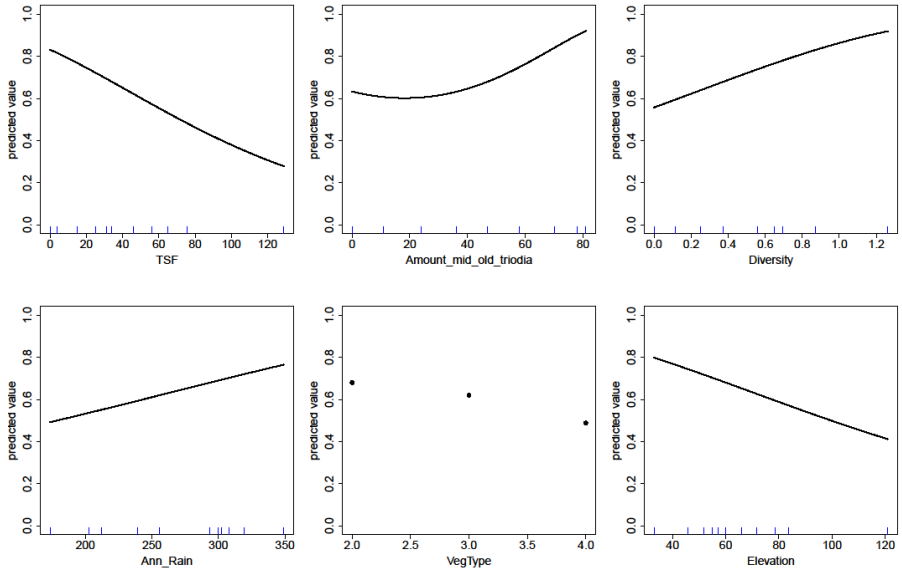
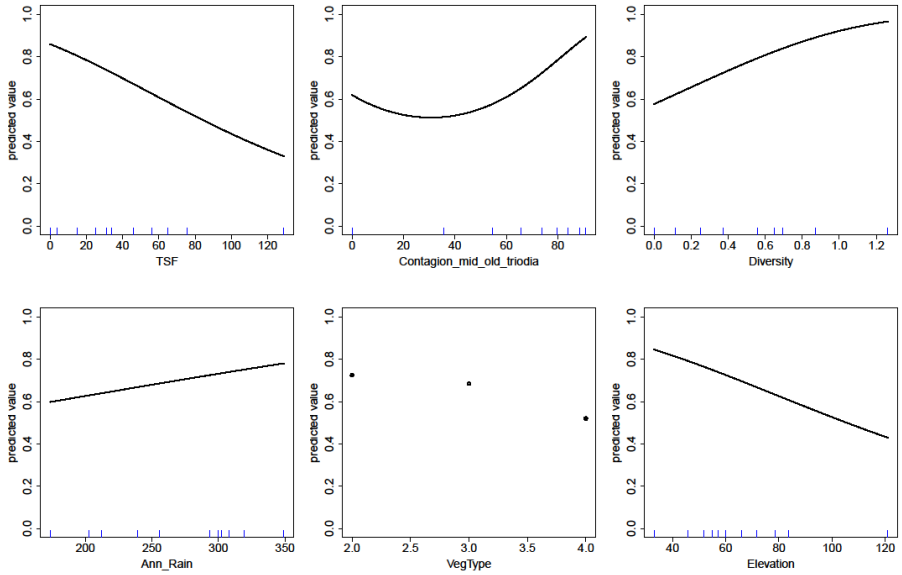


Figure S4.3.13: Response of *Gehyra variegata* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

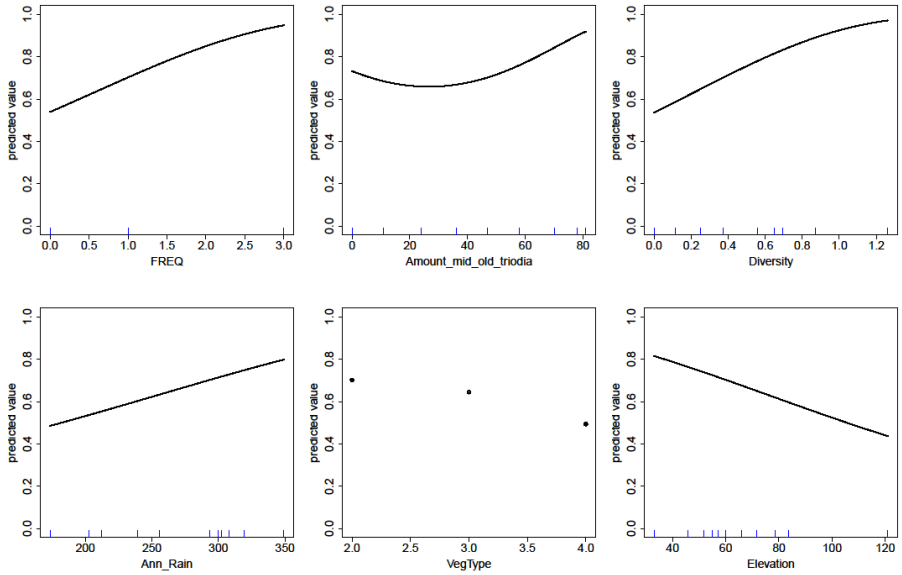
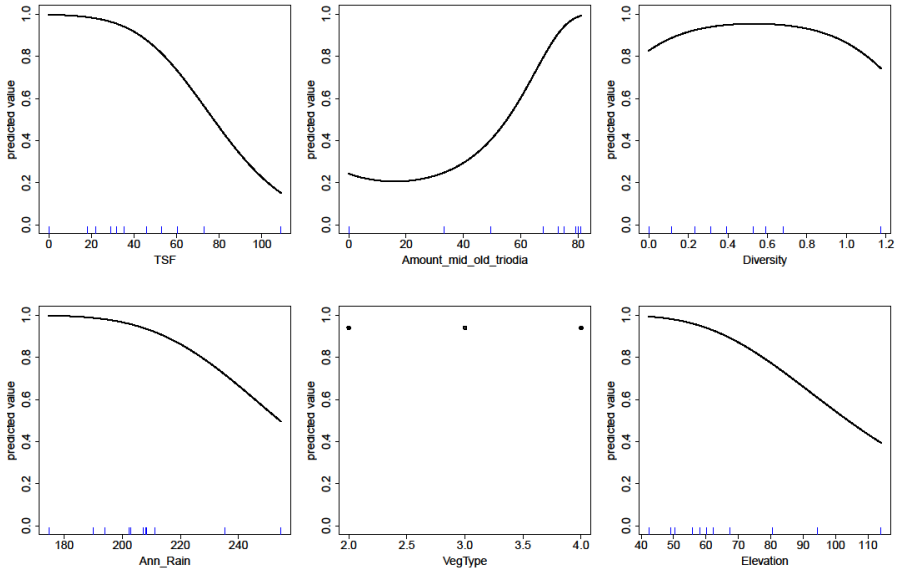
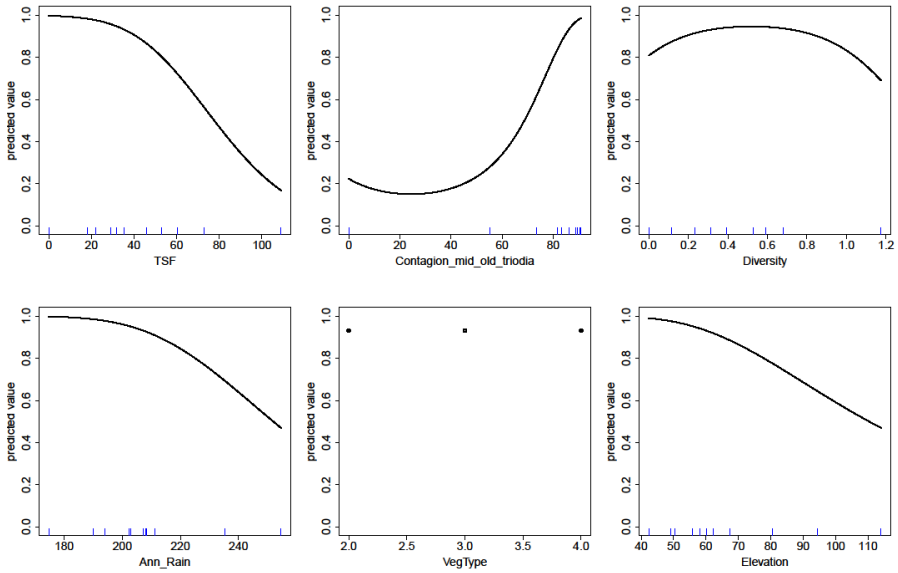


Figure S4.3.14: Response of *Lucasium damaeum* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



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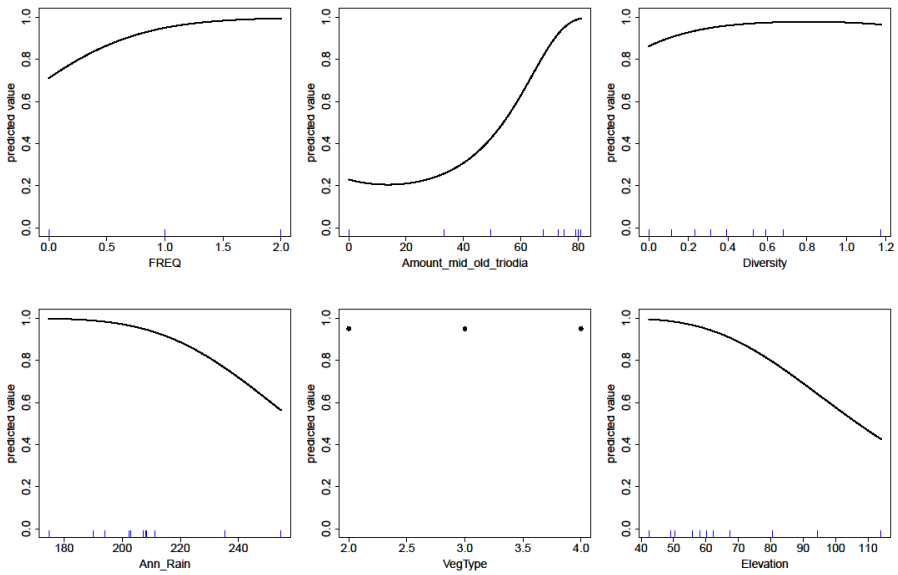
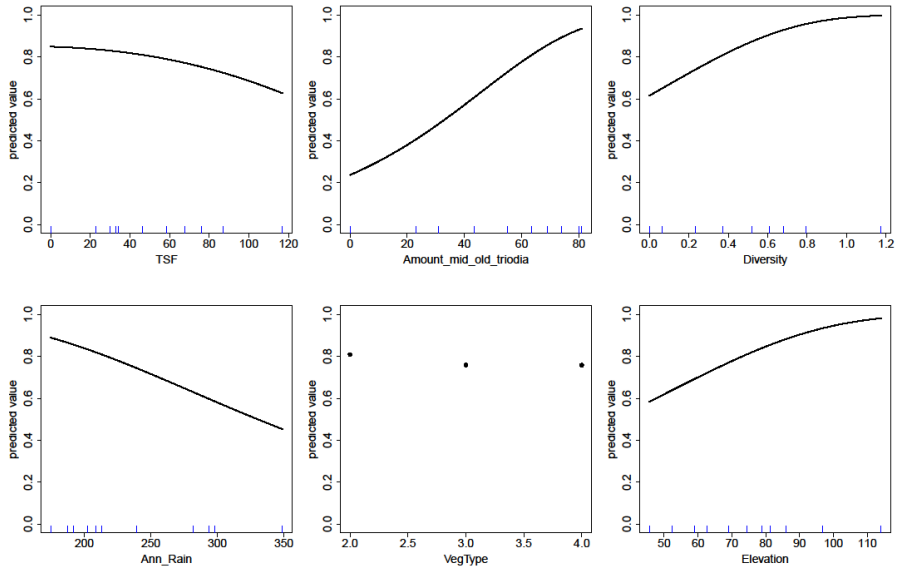


Figure S4.3.15: Response of *Nephrurus levis* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



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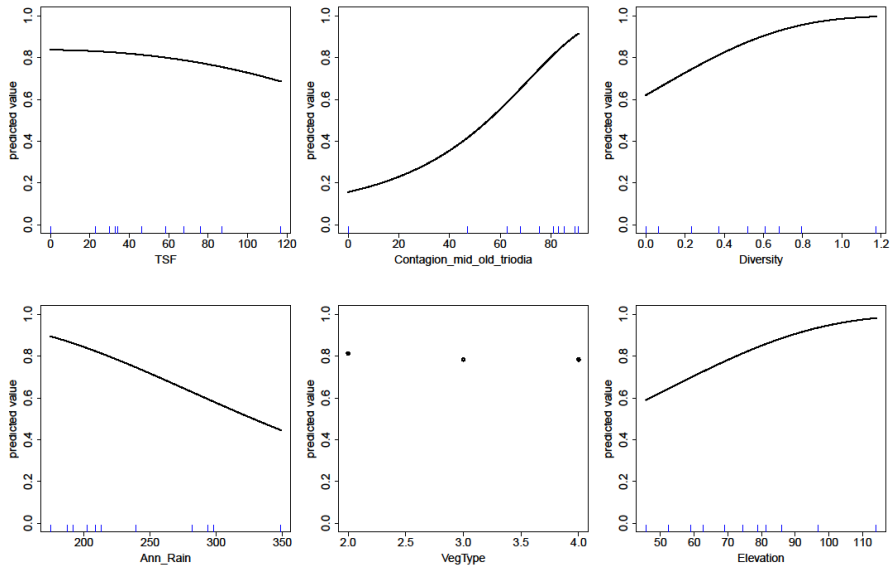

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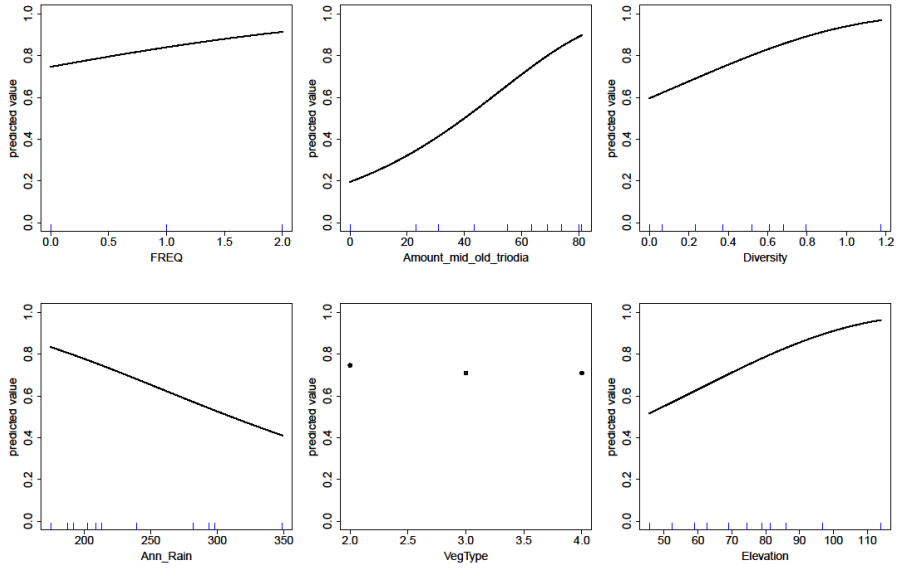
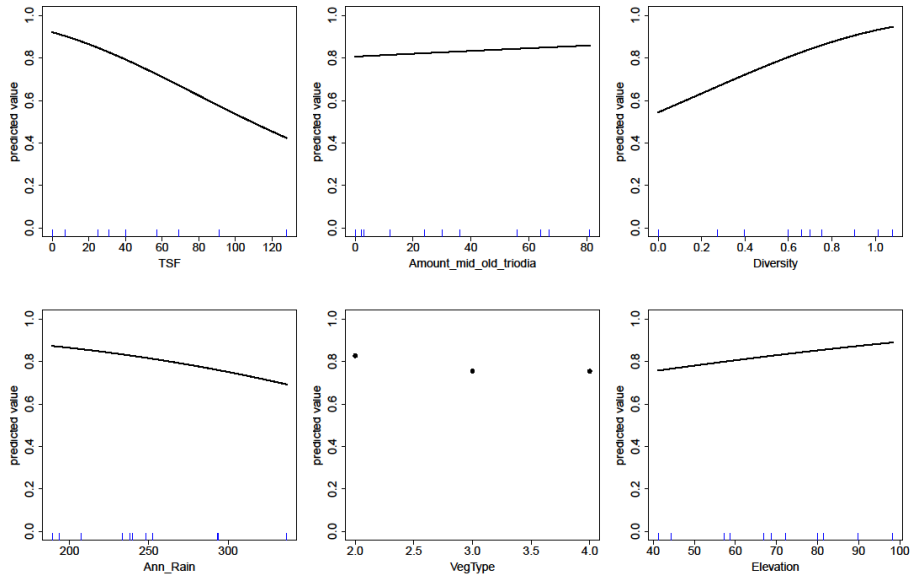
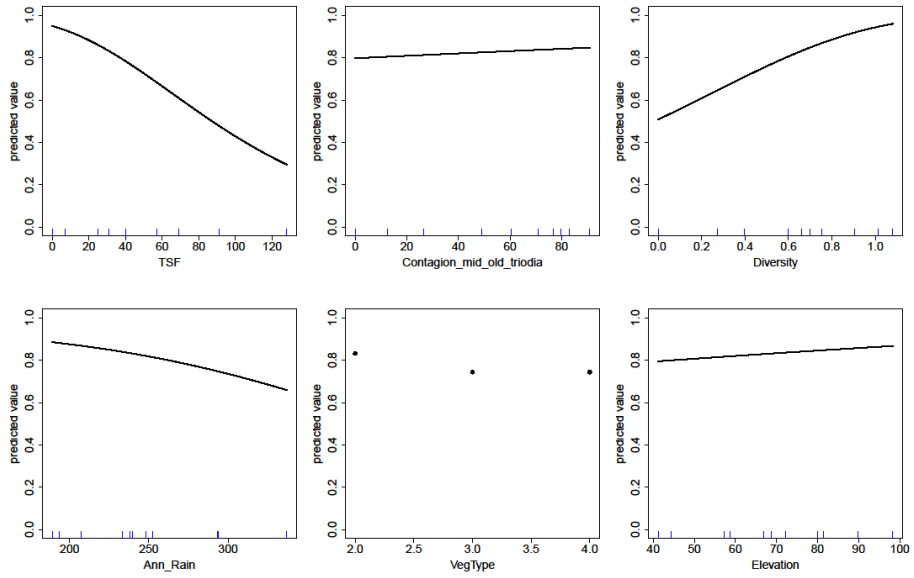


Figure S4.3.16: Response of *Rhynchoedura ormsbyi* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

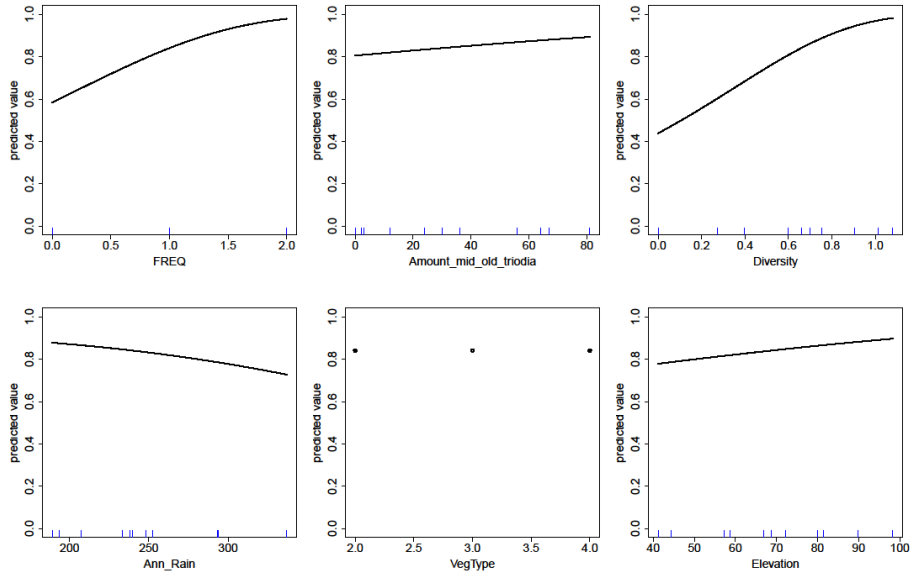
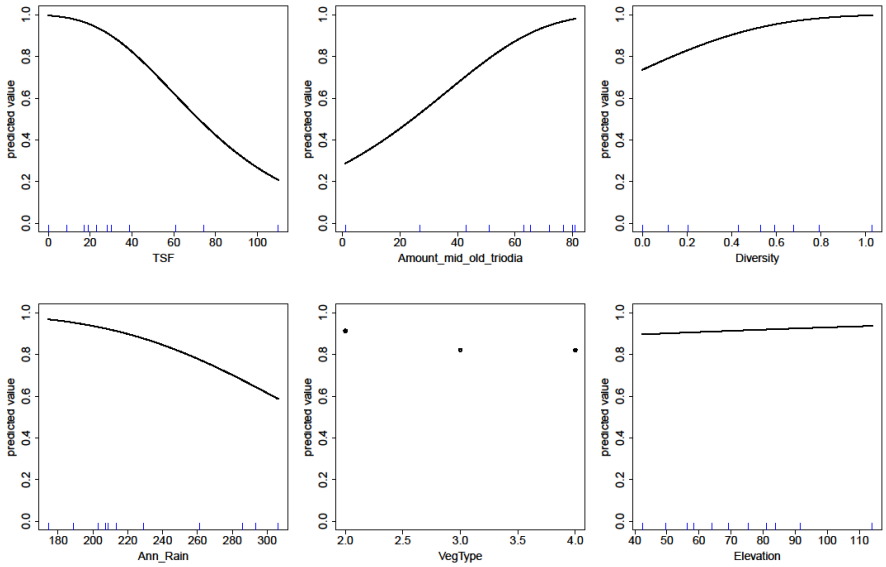
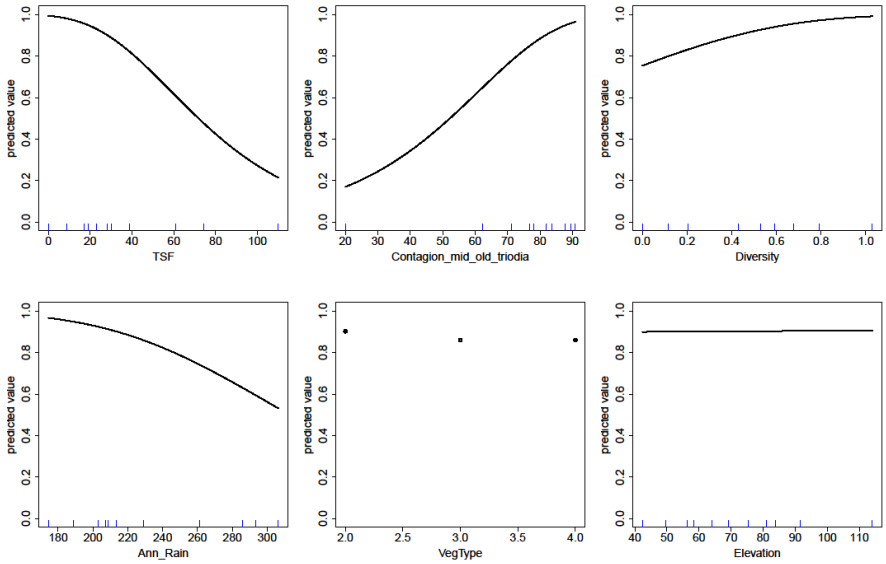


Figure S4.3.17: Response of *Rhynchoedura ornata* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



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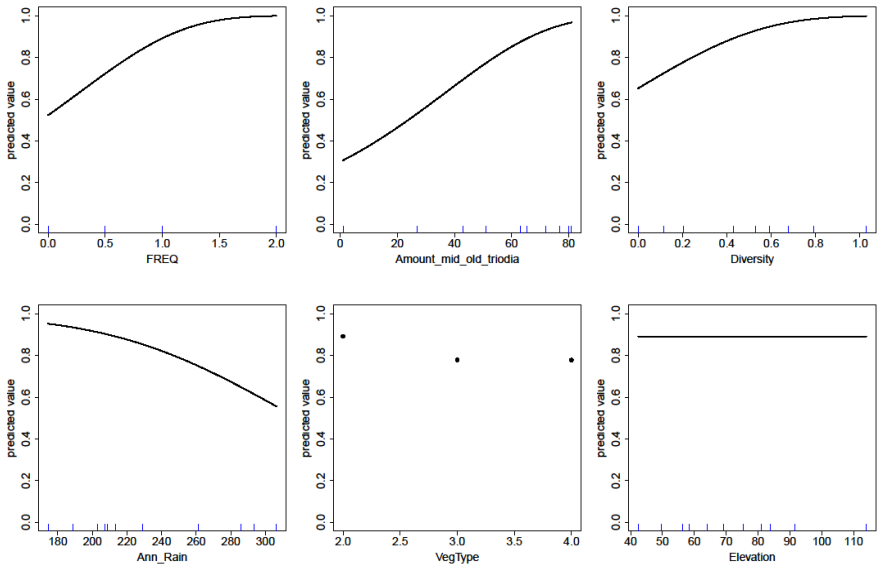
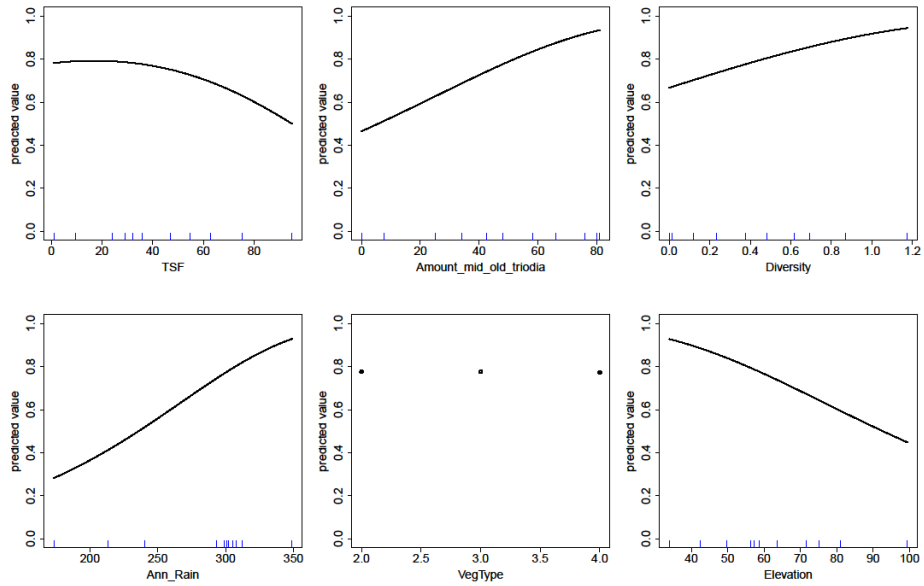
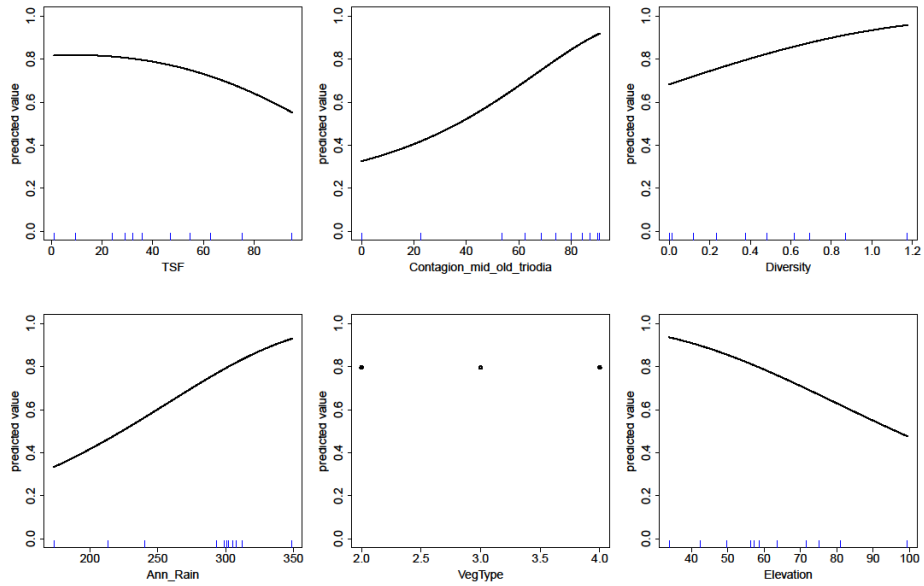


Figure S4.3.18: Response of *Strophurus elderi* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

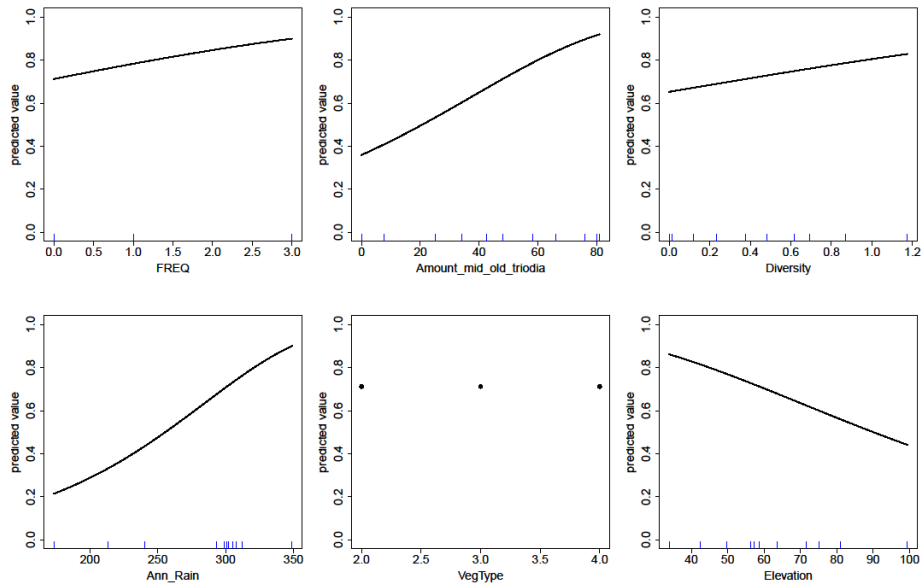
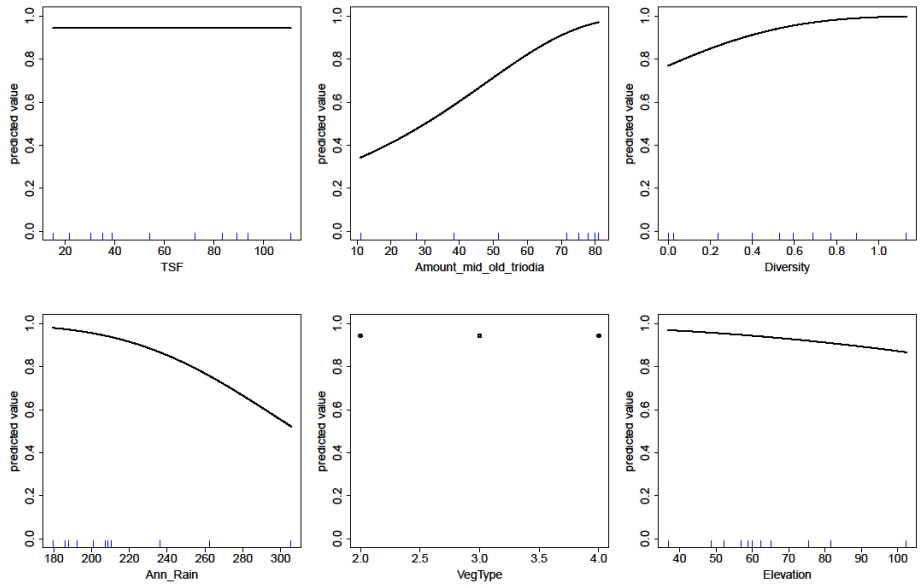
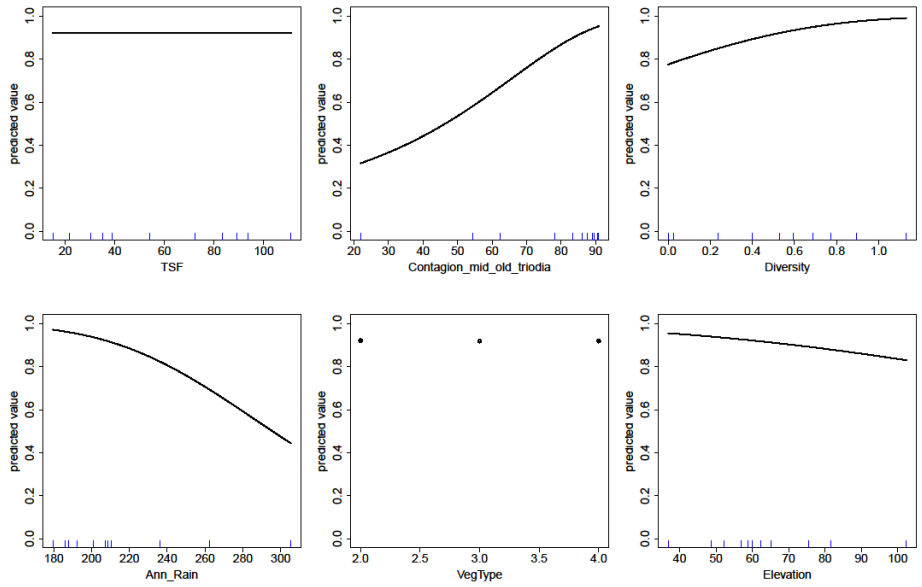


Figure S4.3.19: Response of *Strophurus intermedius* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



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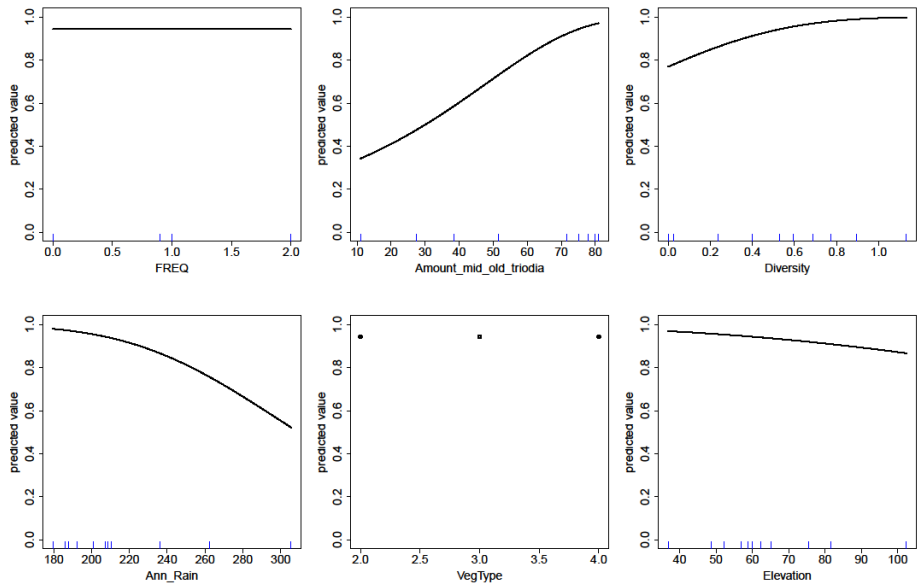
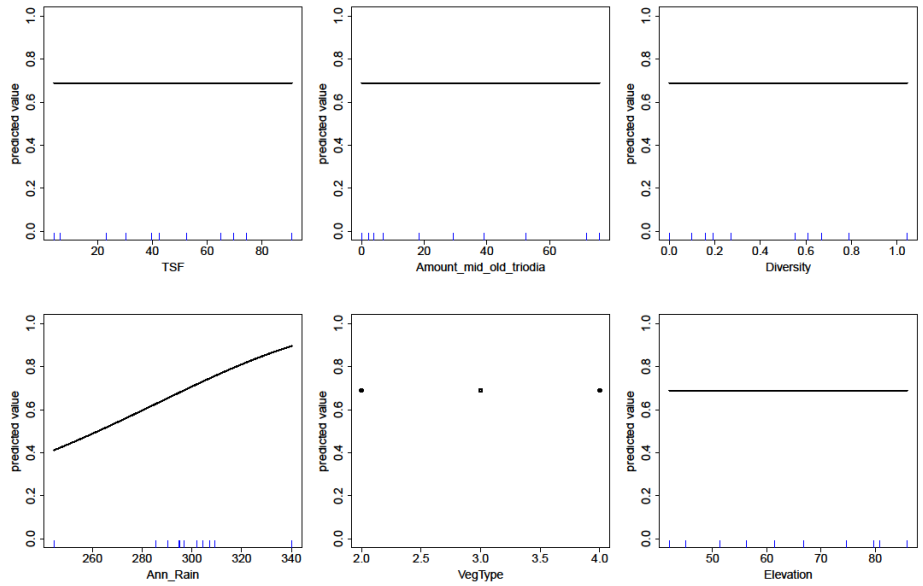
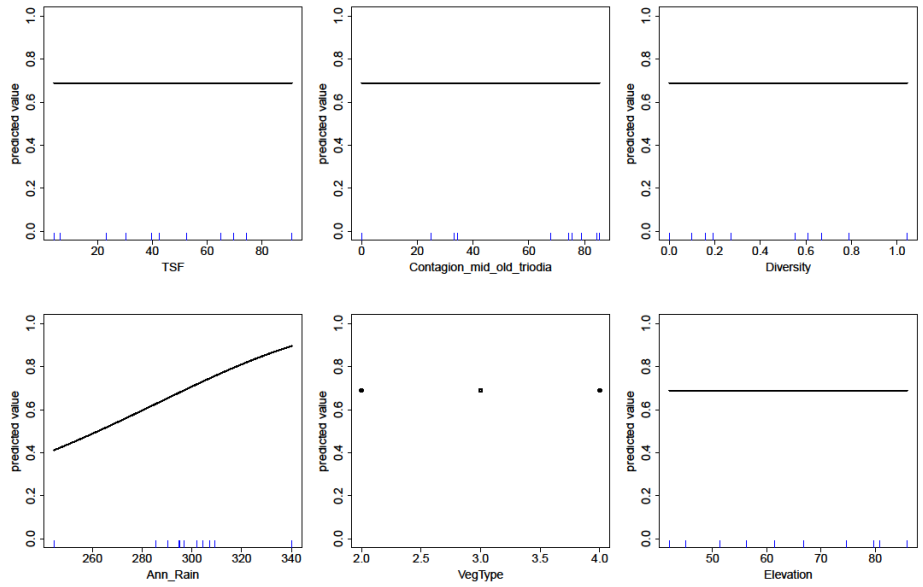


Figure S4.3.20: Response of *Strophurus williamsi* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



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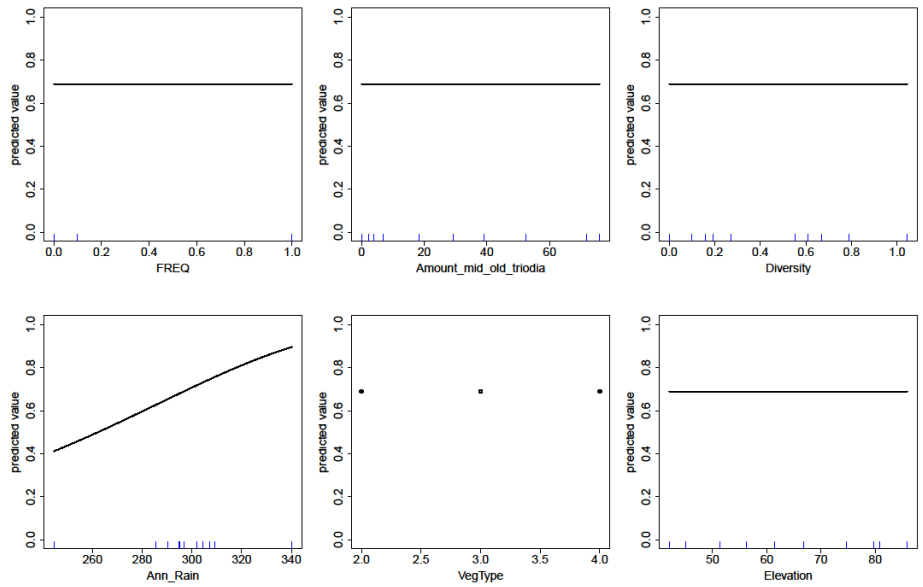


Figure S4.3.21: Response of *Underwoodisaurus milii* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

Goannas (Varanidae)

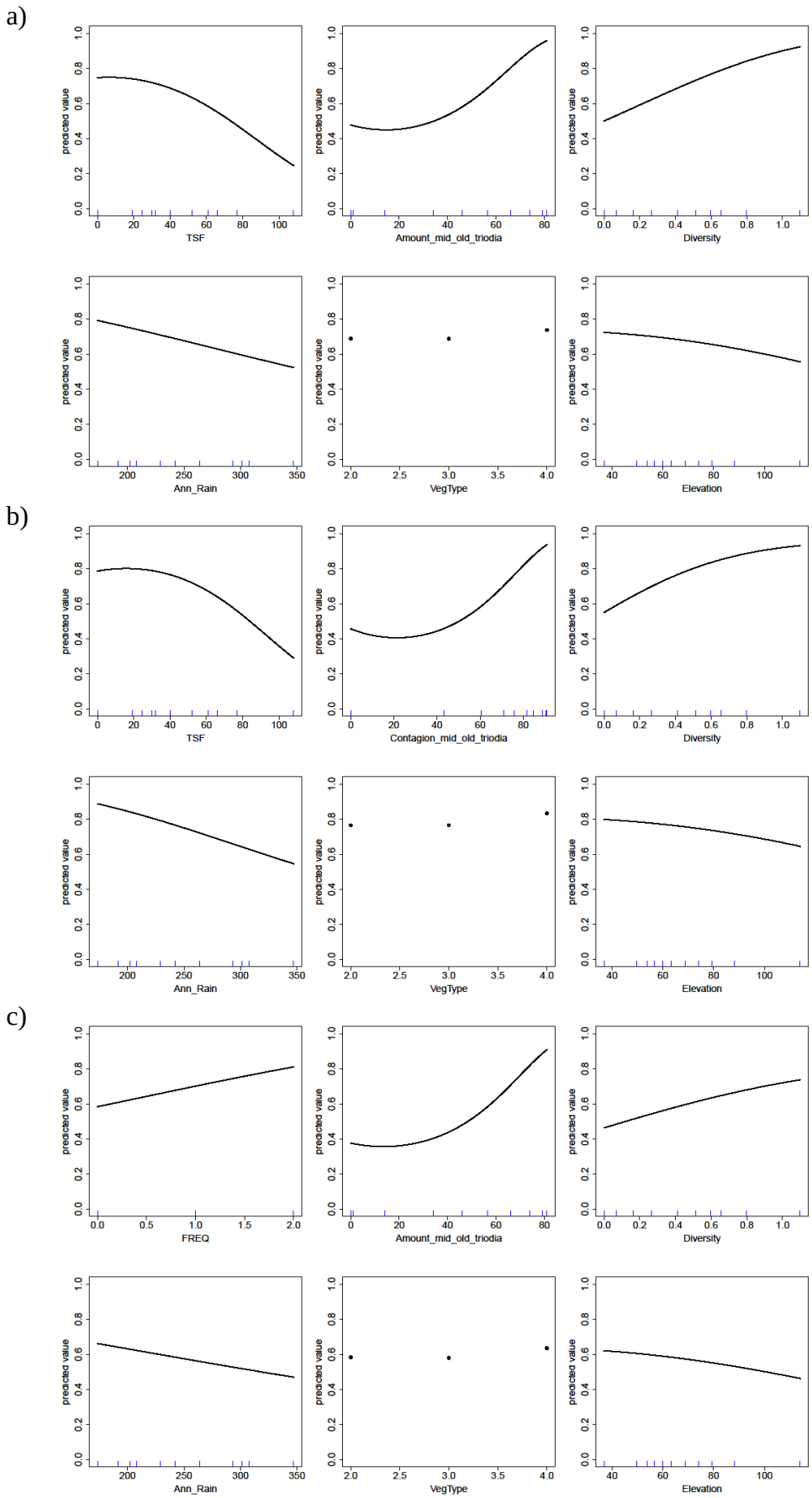


Figure S4.3.22: Response of *Varanus gouldii* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

Legless lizards (Pygopodidae)

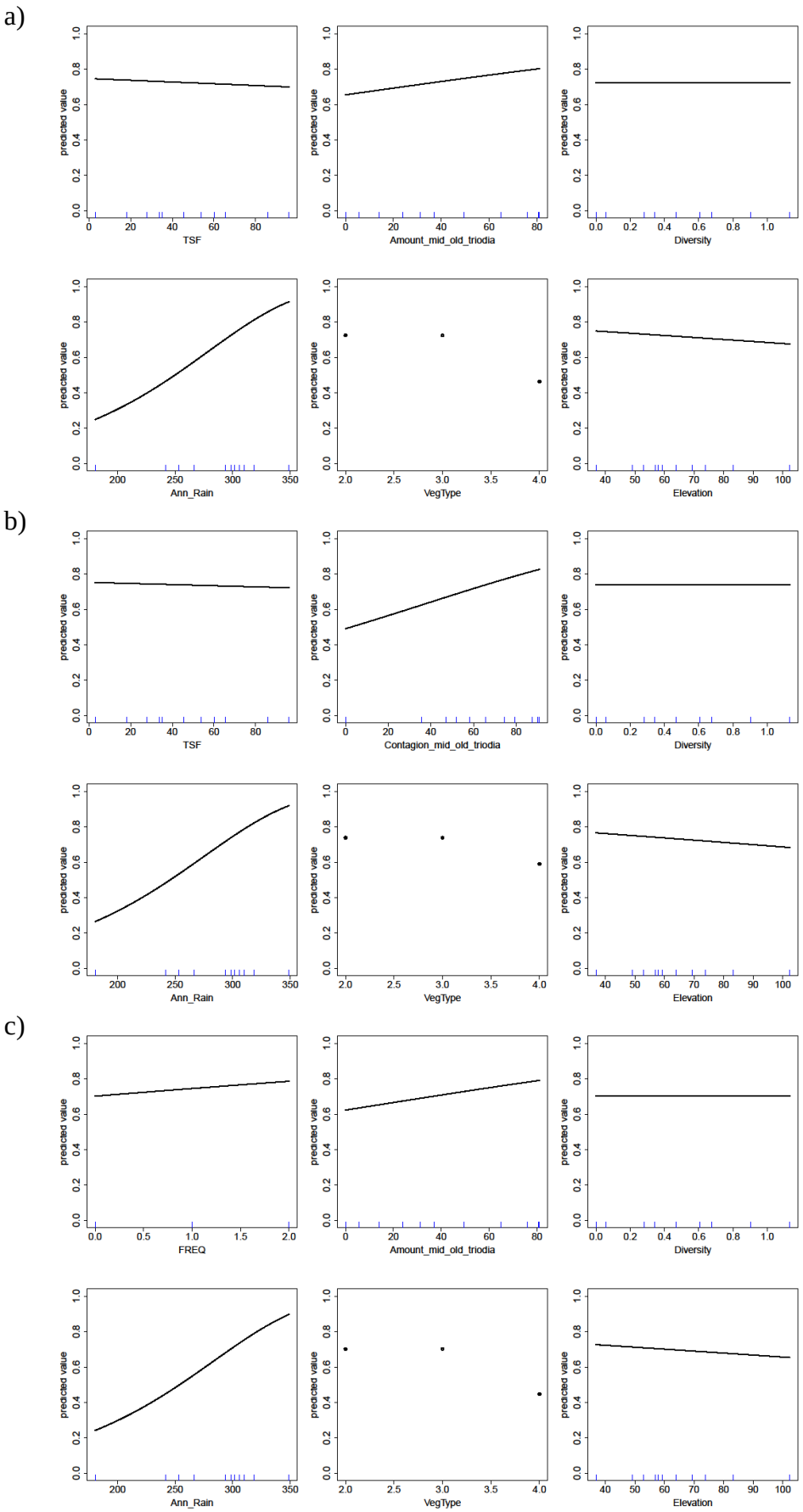
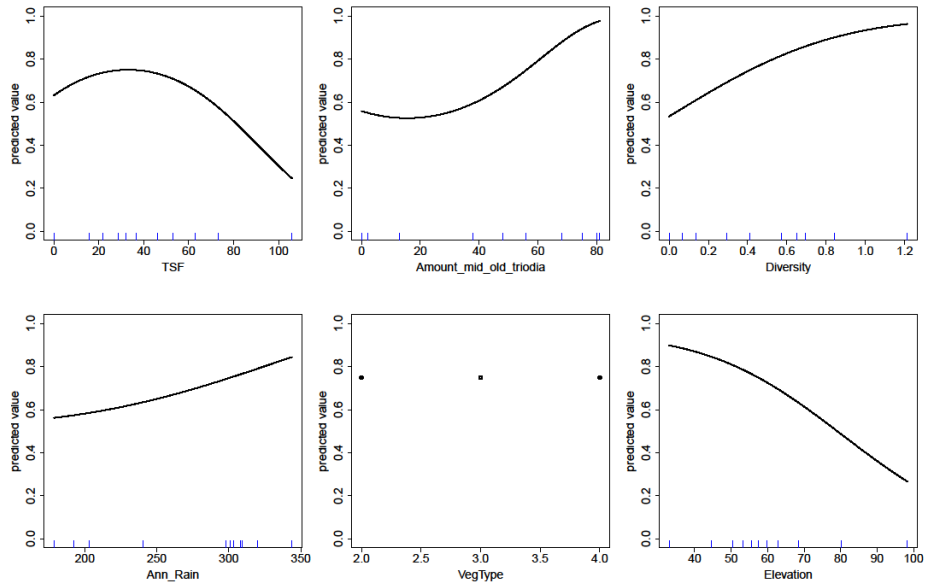
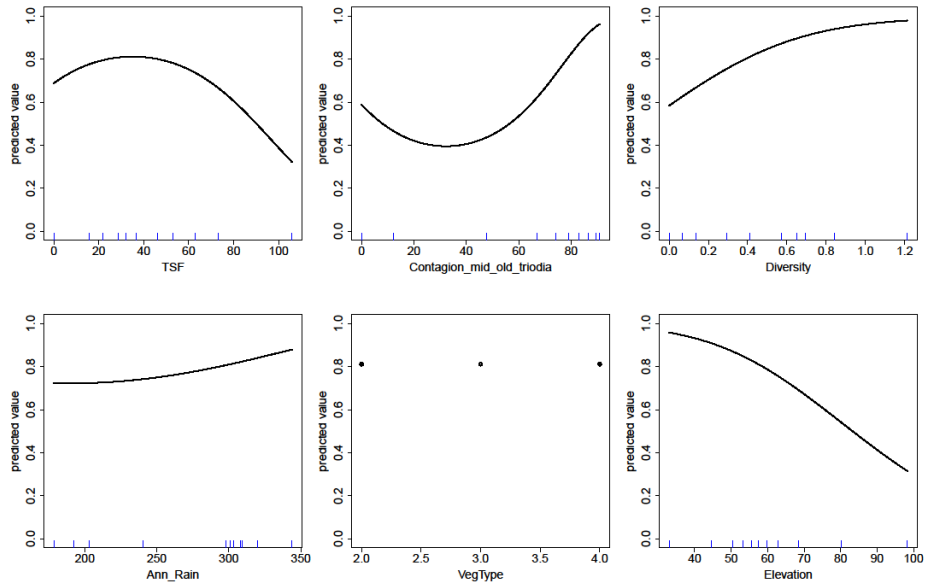


Figure S4.3.23: Response of *Aprasia inaurita* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

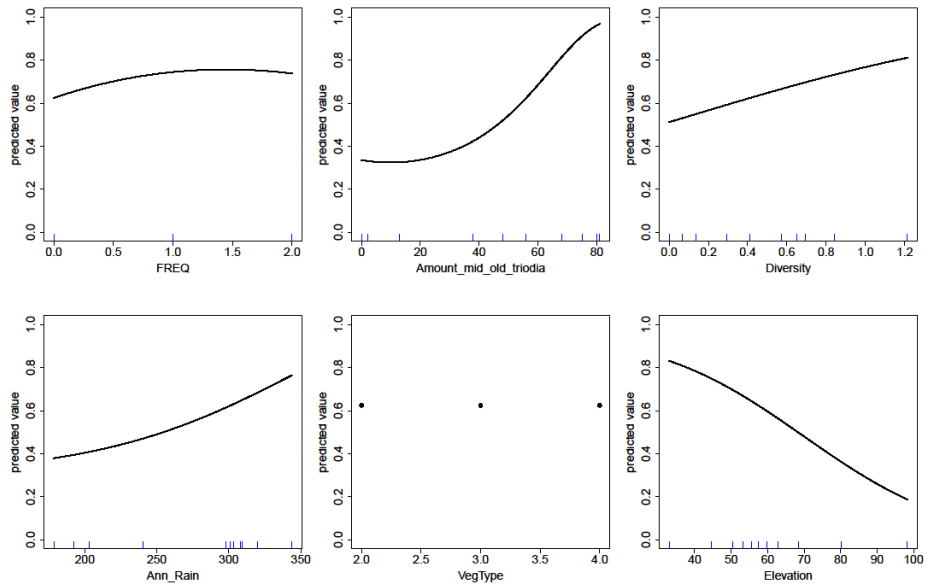
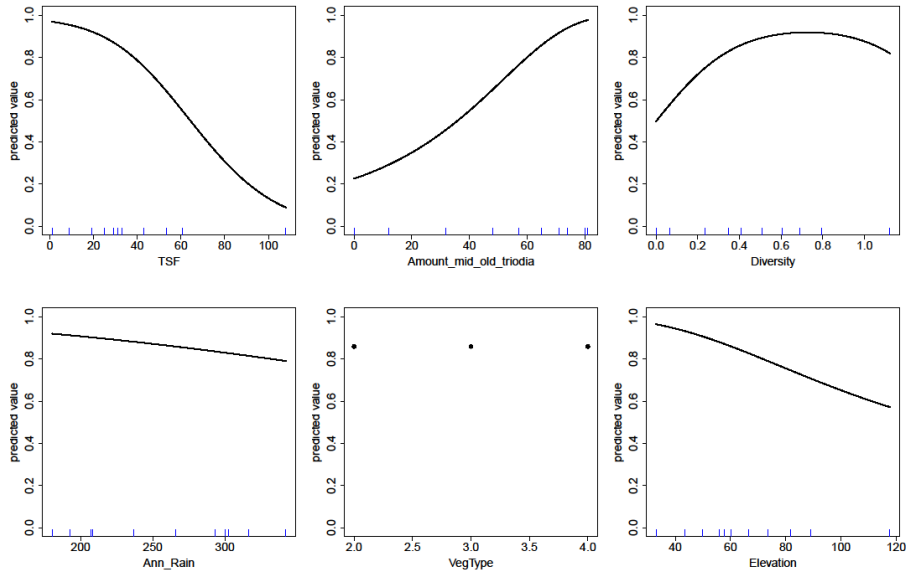


Figure S4.3.24: Response of *Delma australis* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



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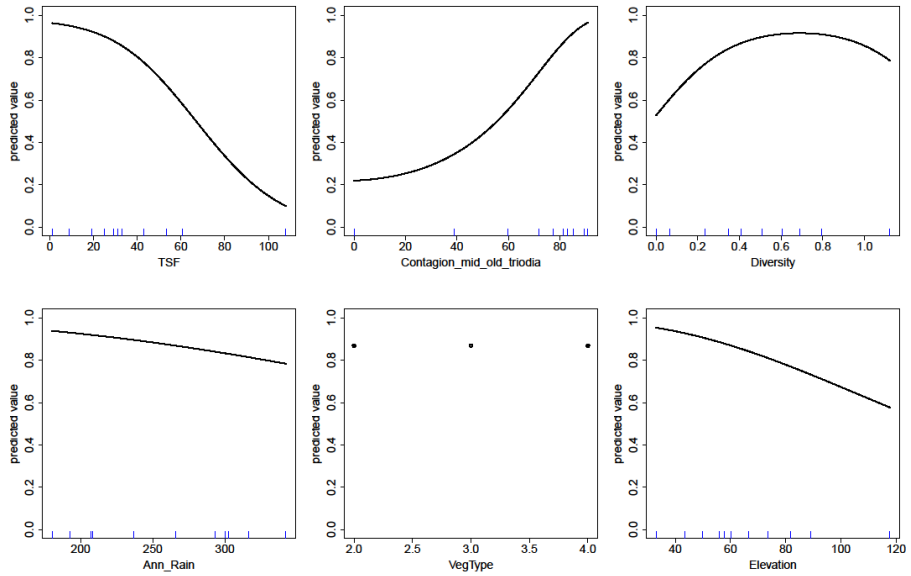


c)

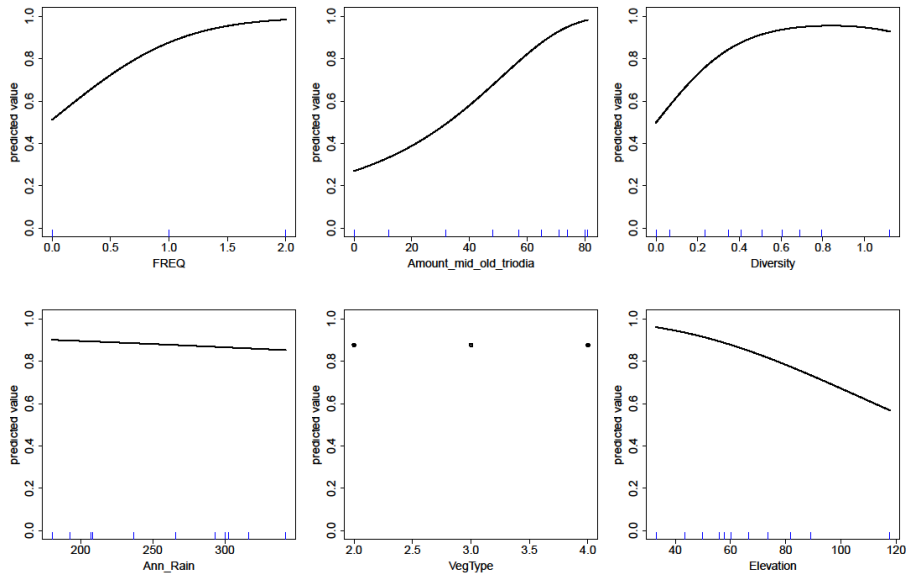
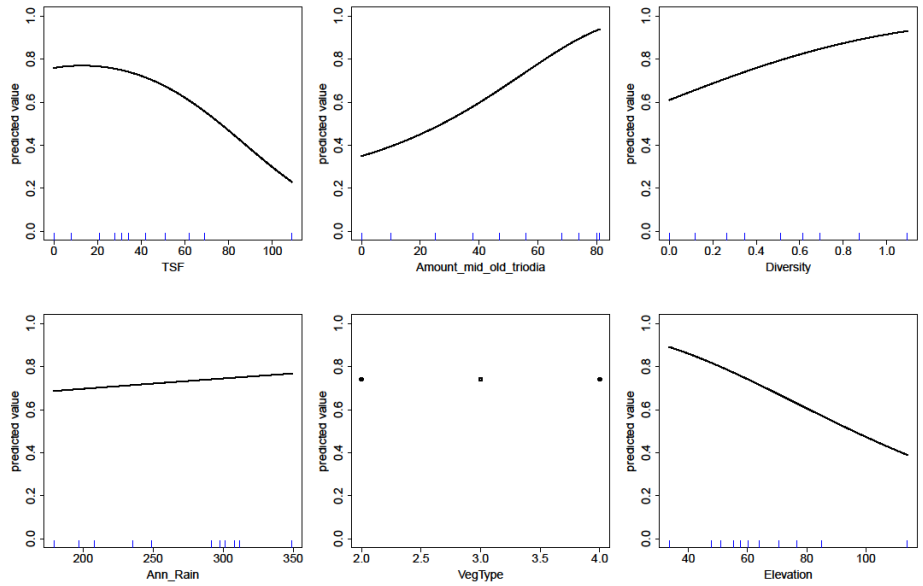
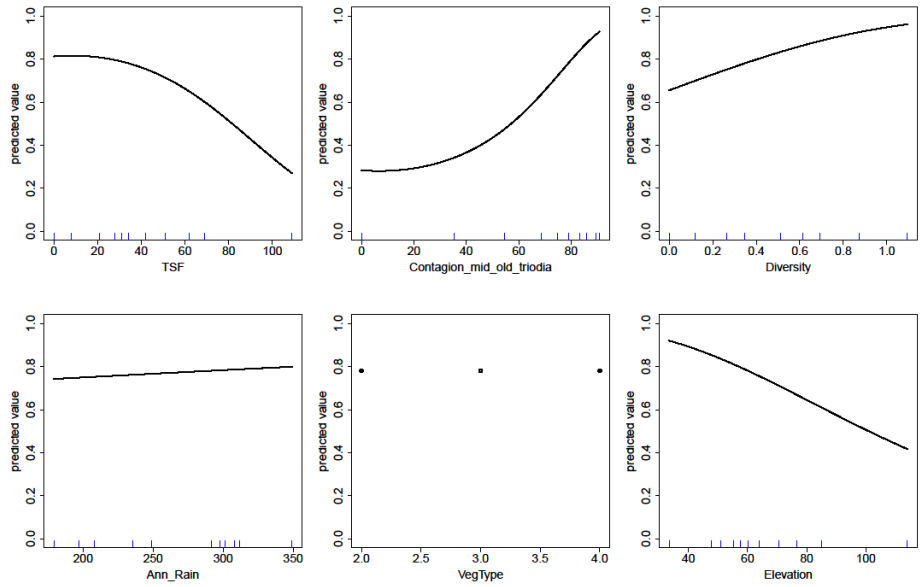


Figure S4.3.25: Response of *Delma butleri* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

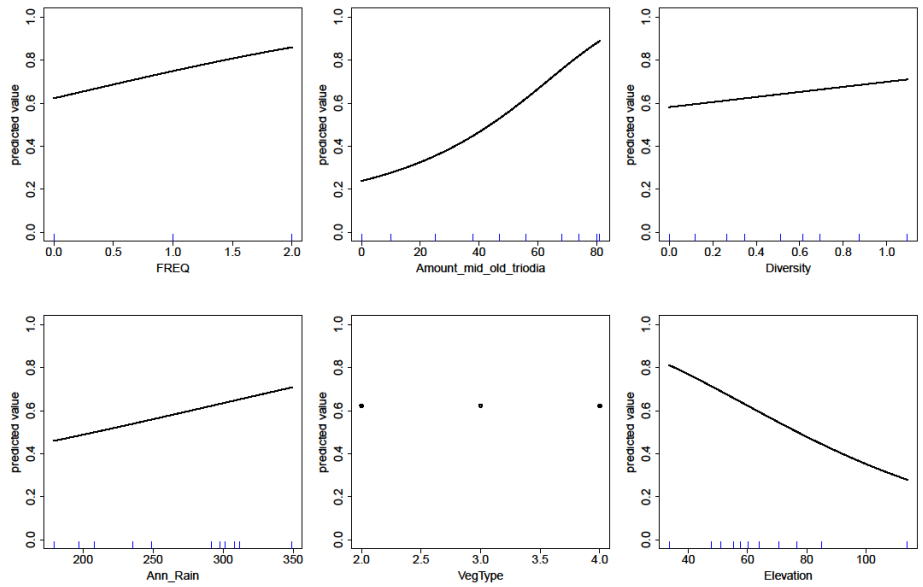
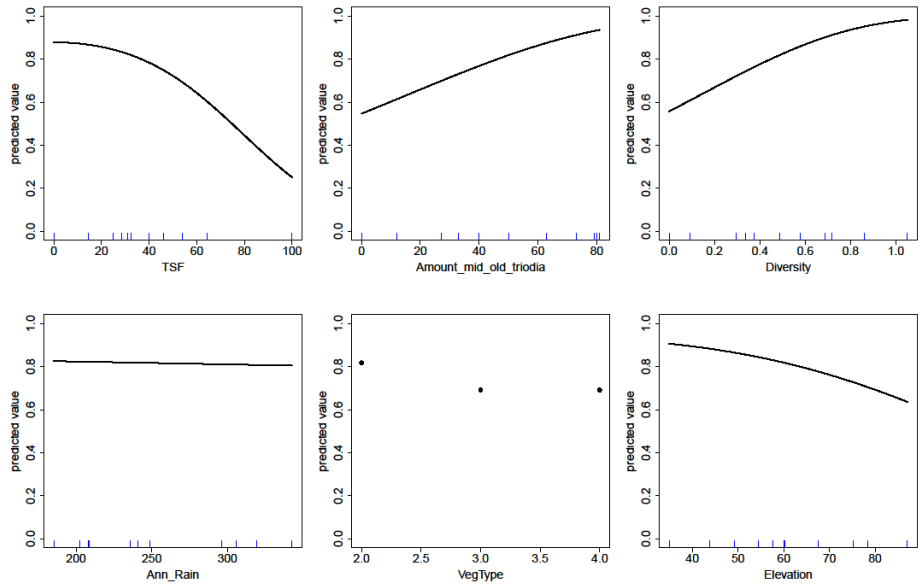


Figure S4.3.26: Response of *Lialis burtonis* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



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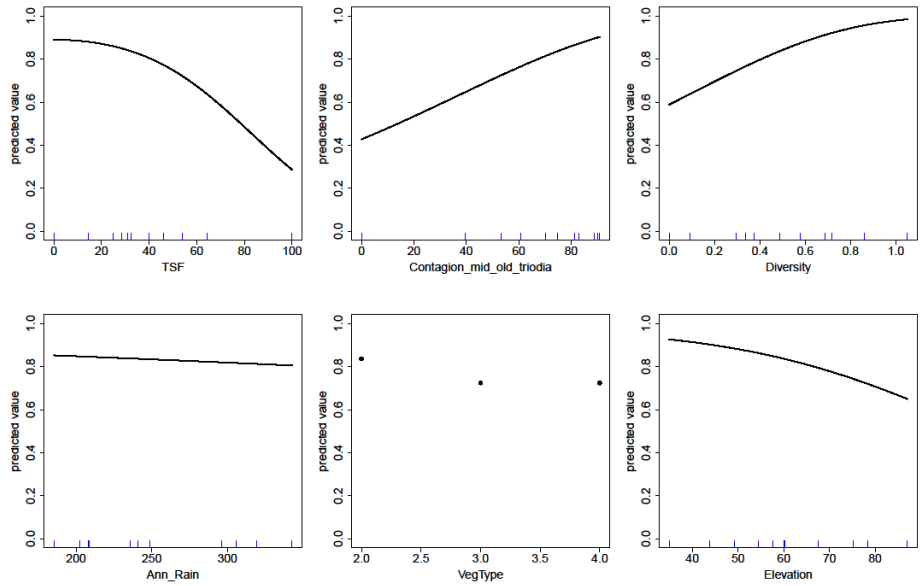


c)

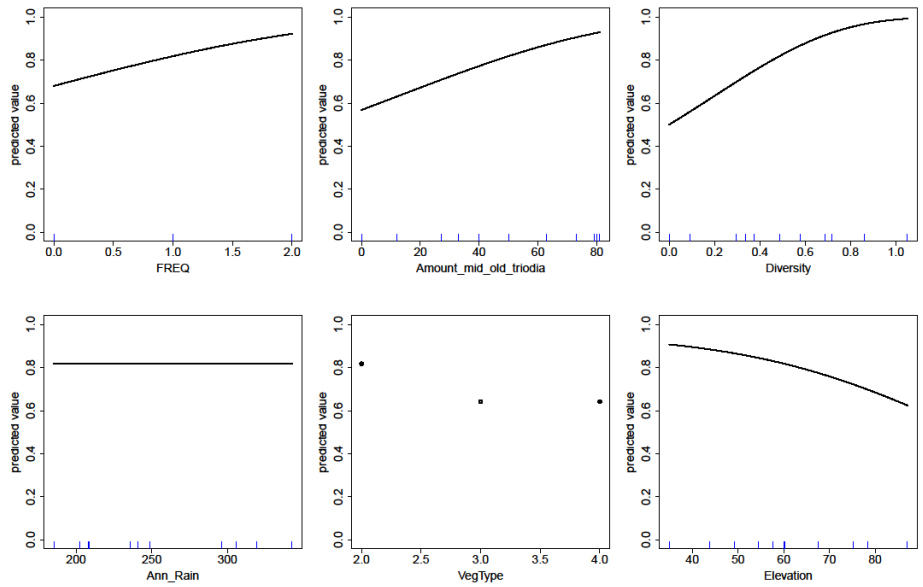
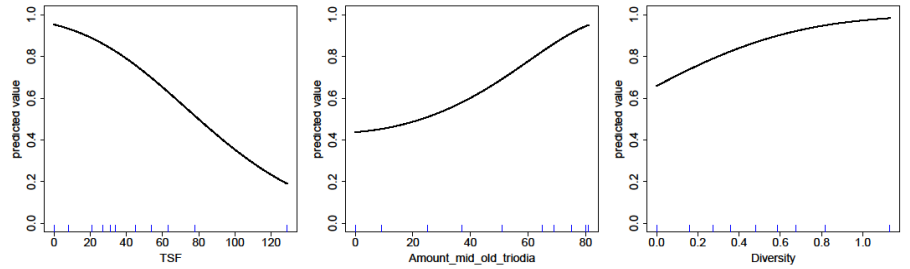
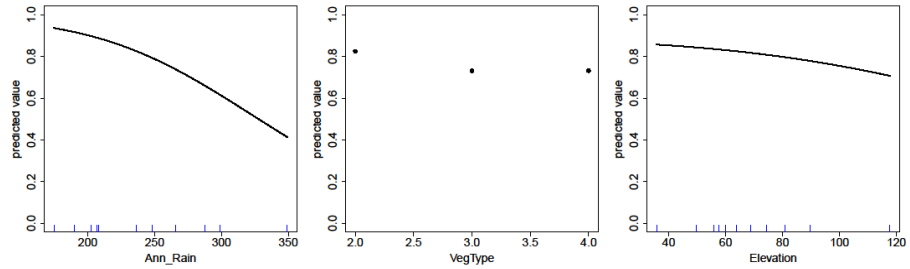


Figure S4.3.27: Response of *Pygopus lepidopodus* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

Skinks (Scincidae)

a)





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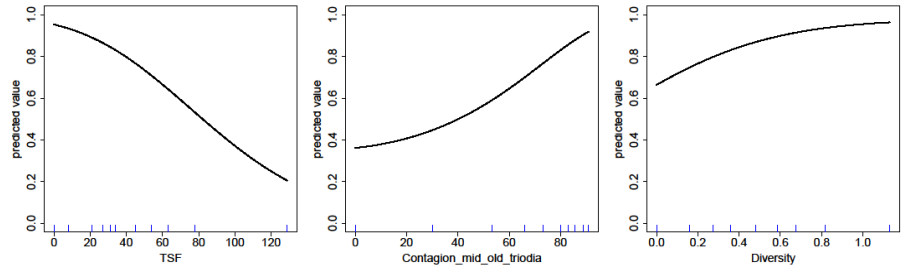

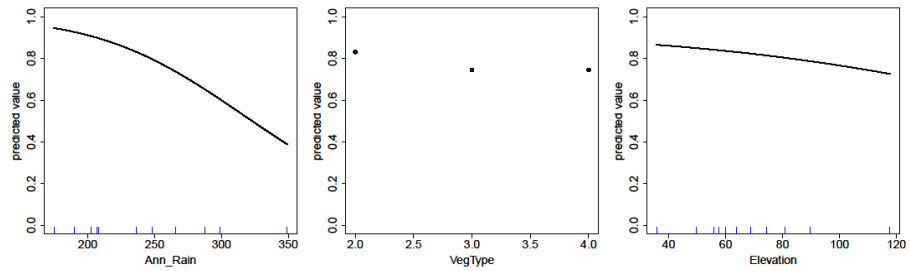

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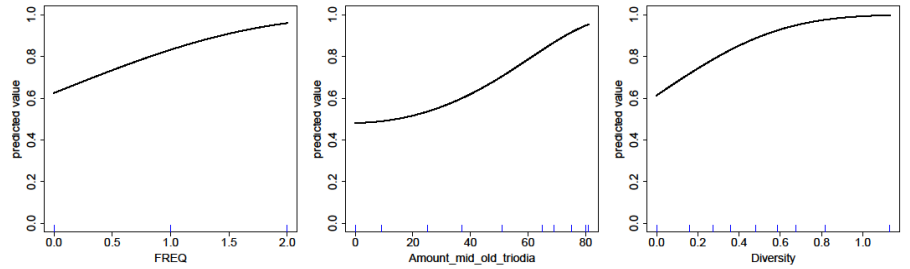


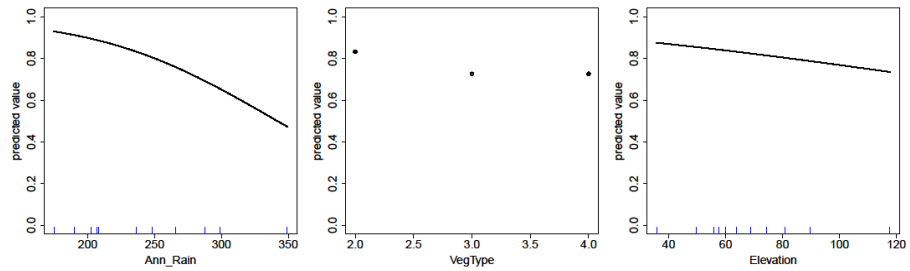
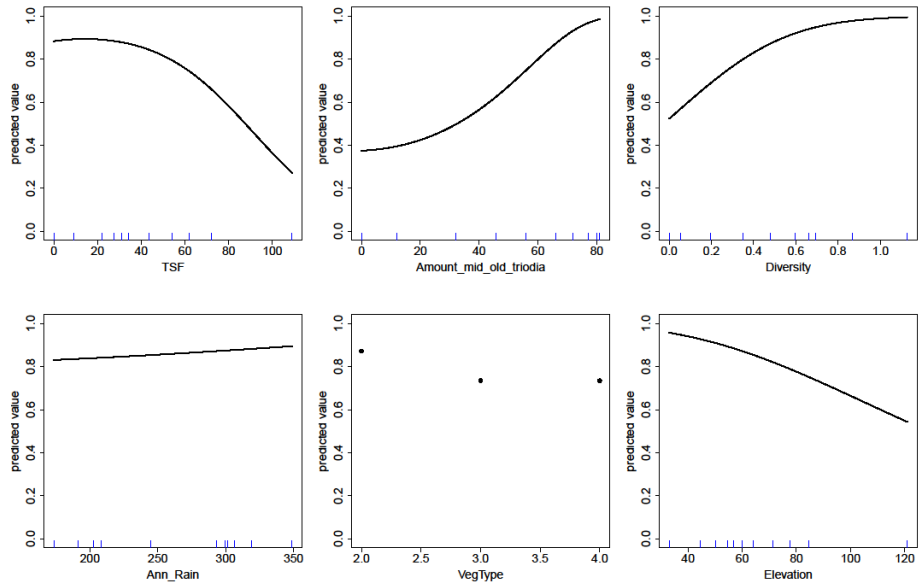
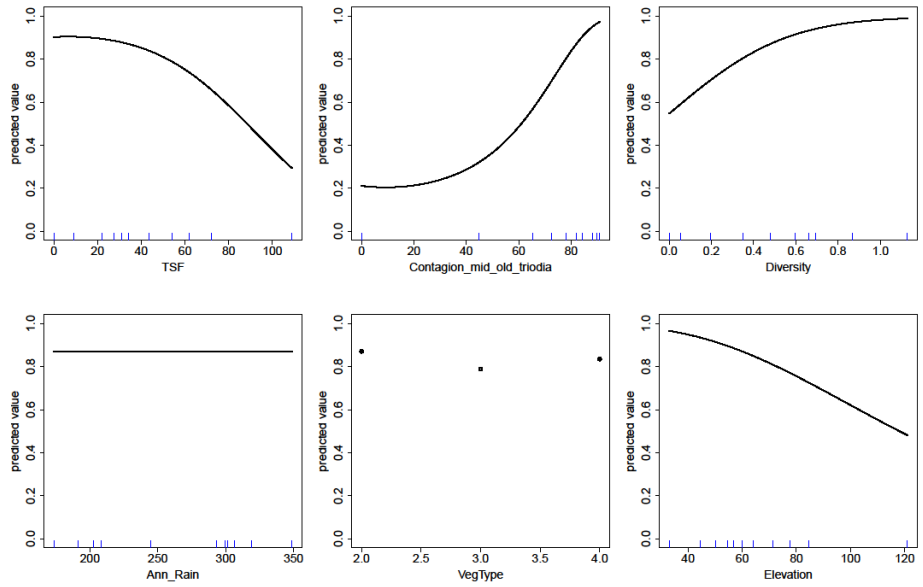


Figure S4.3.28: Response of *Ctenotus atlas* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

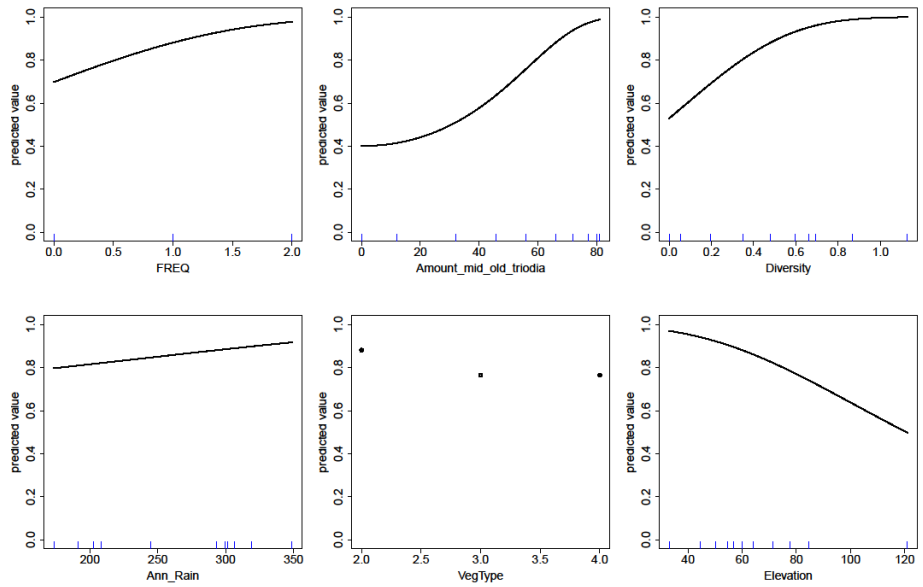
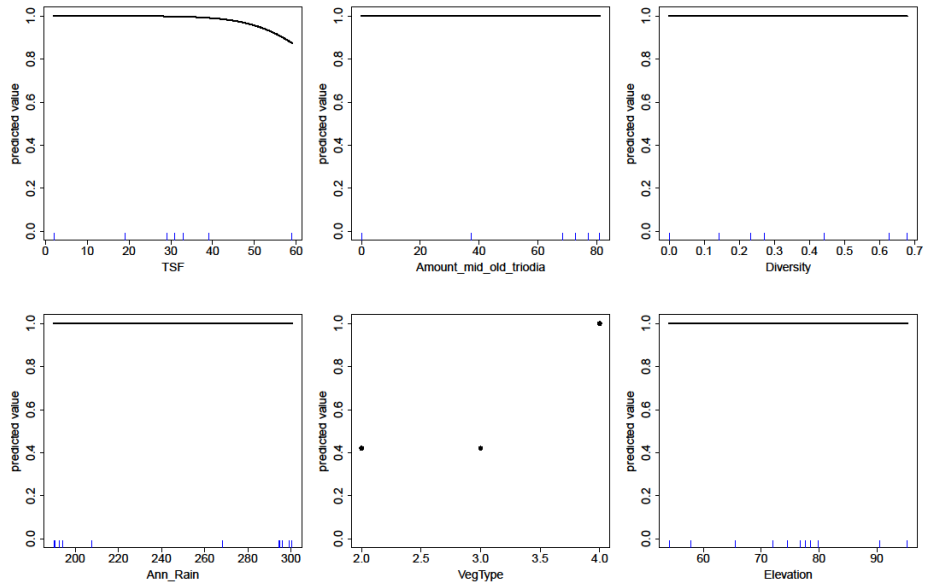
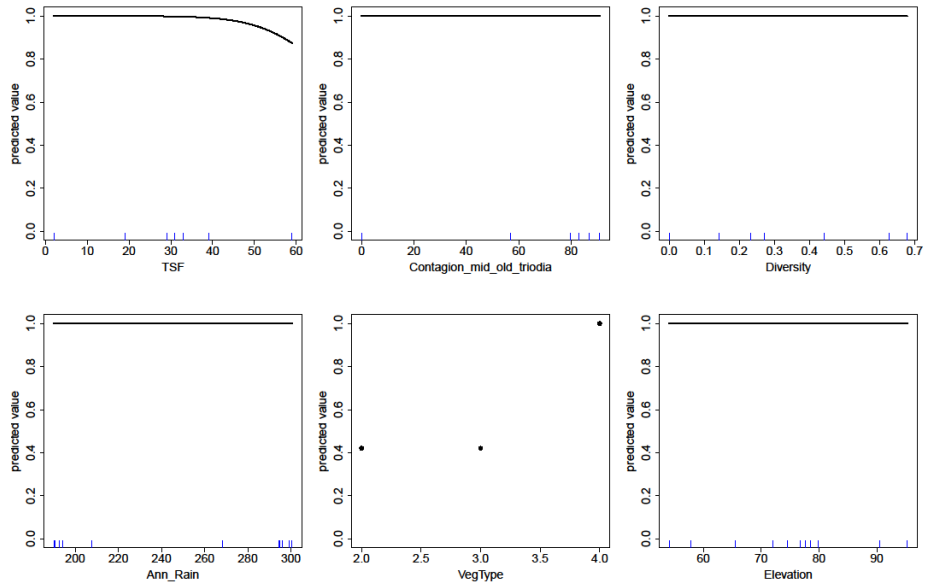


Figure S4.3.29: Response of *Ctenotus brachyonyx* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

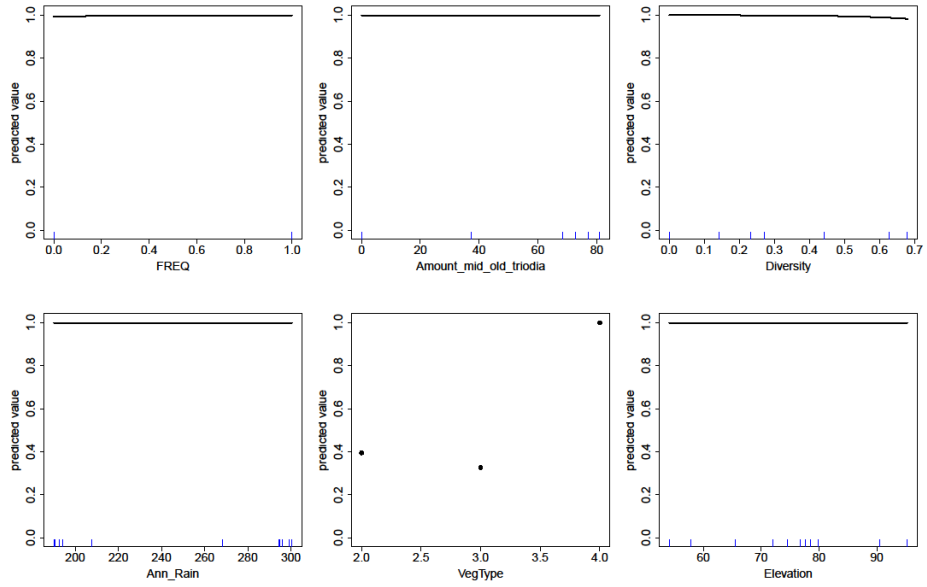
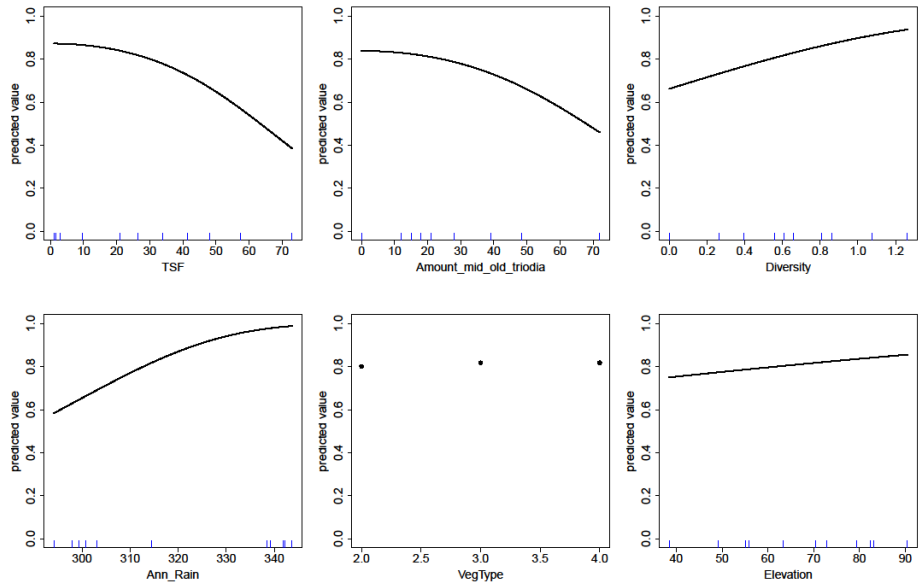
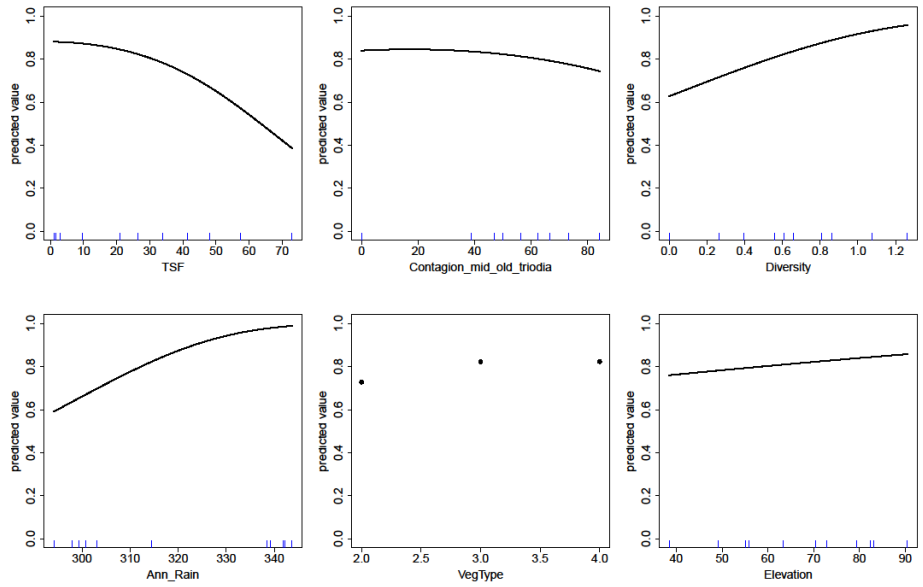


Figure S4.3.30: Response of *Ctenotus brooksi iridis* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

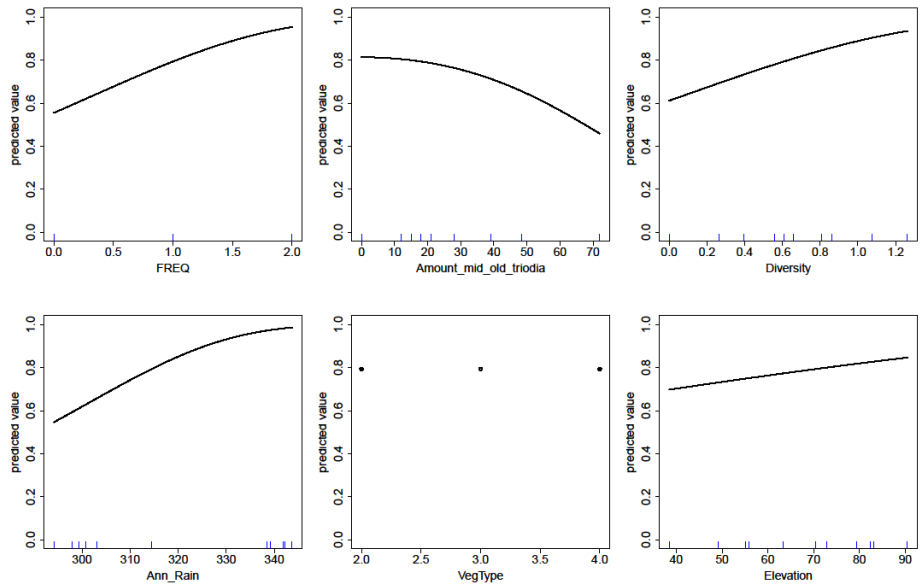
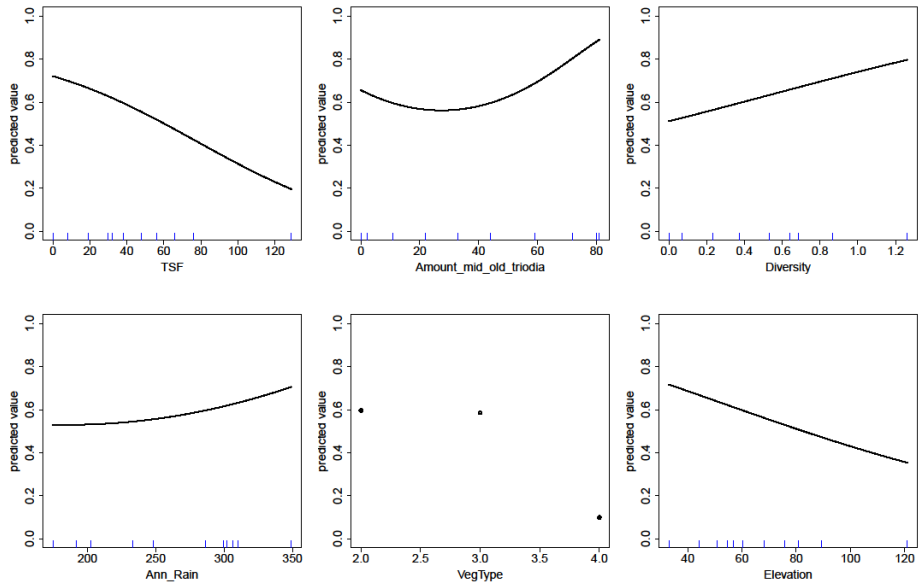
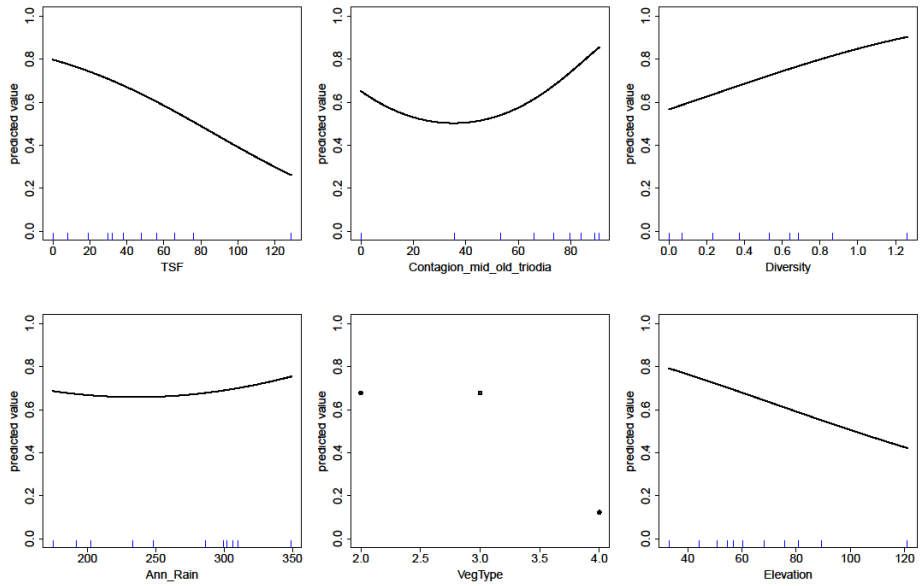


Figure S4.3.31: Response of *Ctenotus orientalis* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

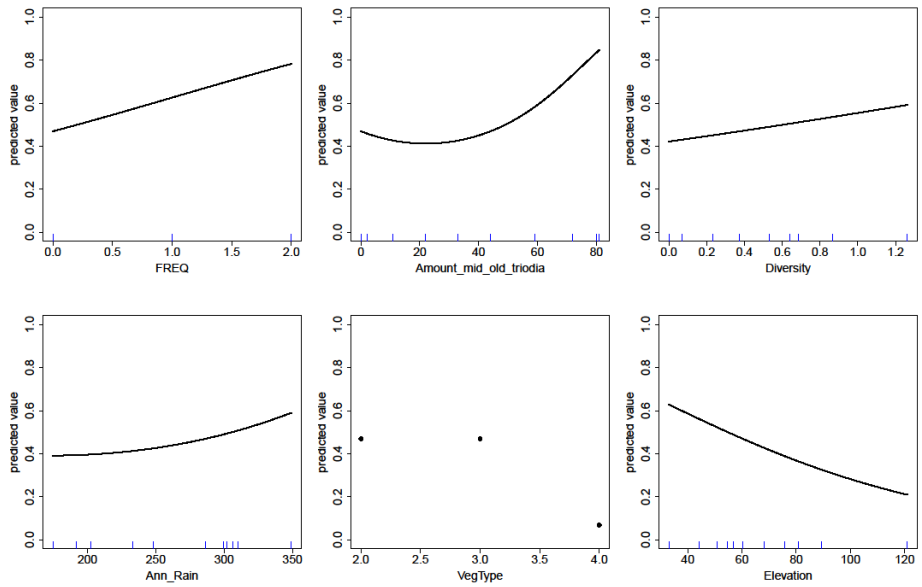
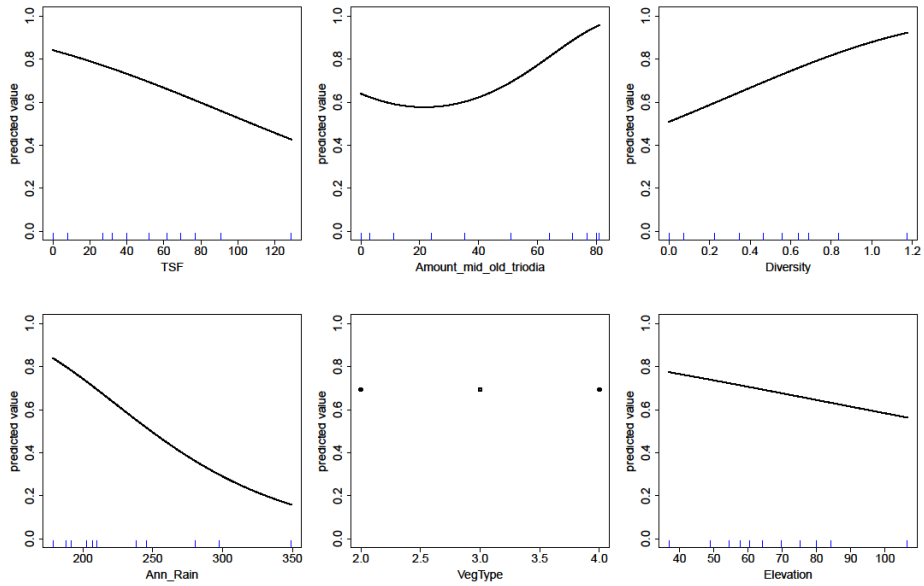
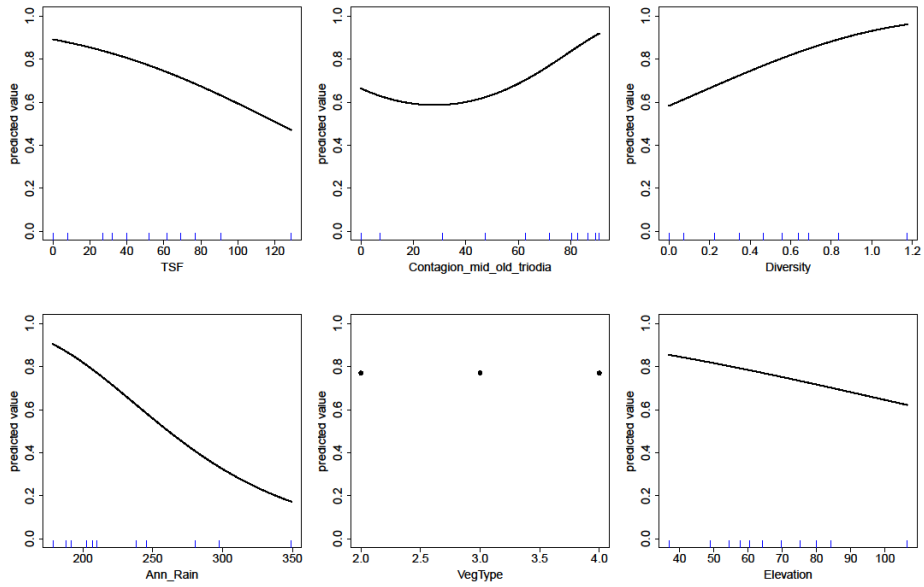


Figure S4.3.32: Response of *Ctenotus regius* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

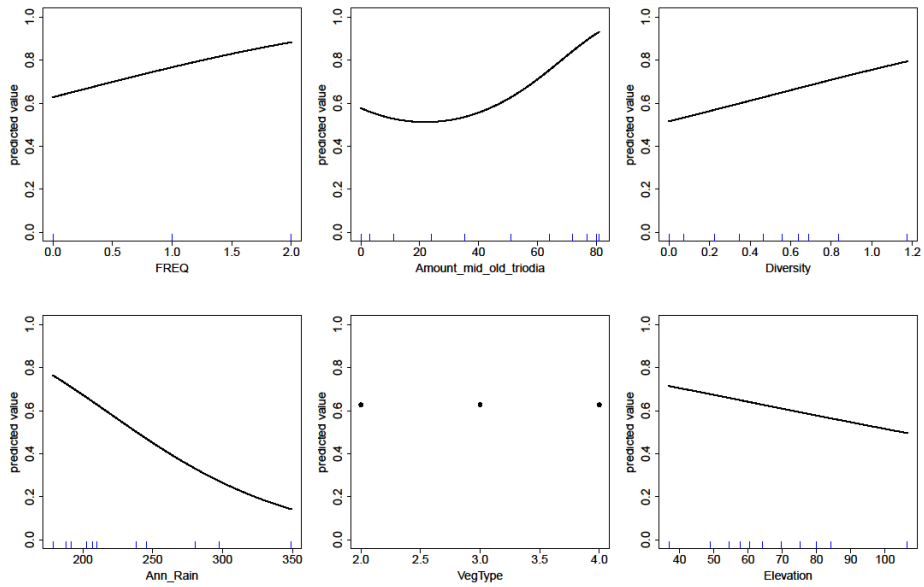
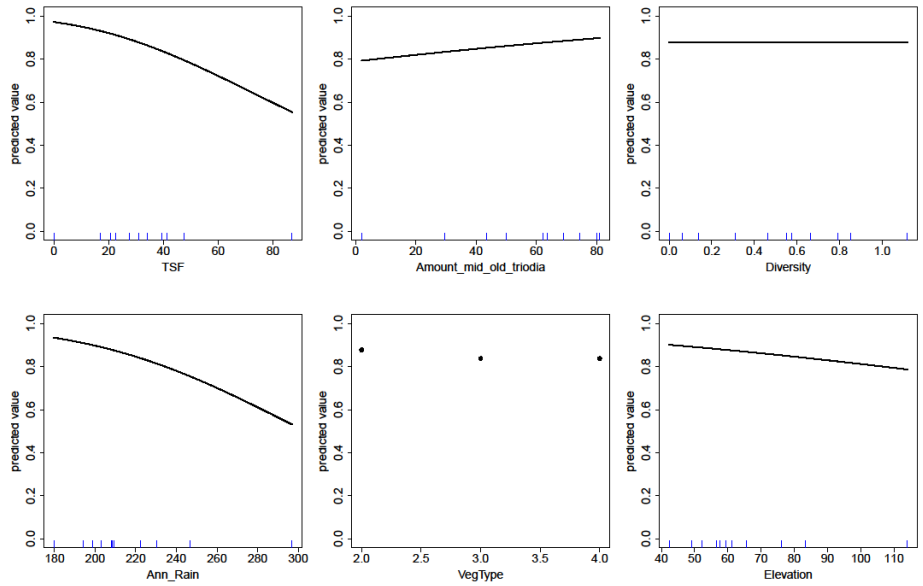
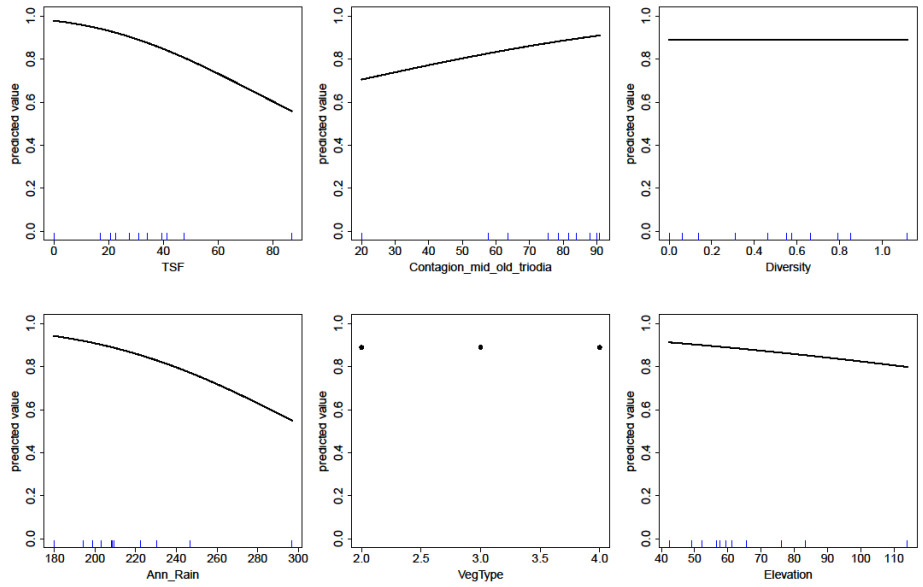


Figure S4.3.33: Response of *Ctenotus schomburgkii* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

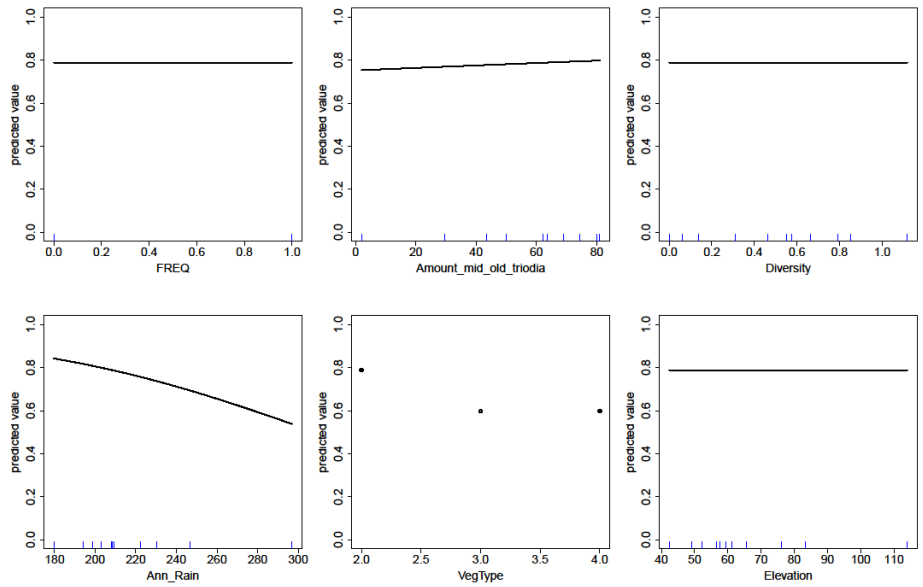
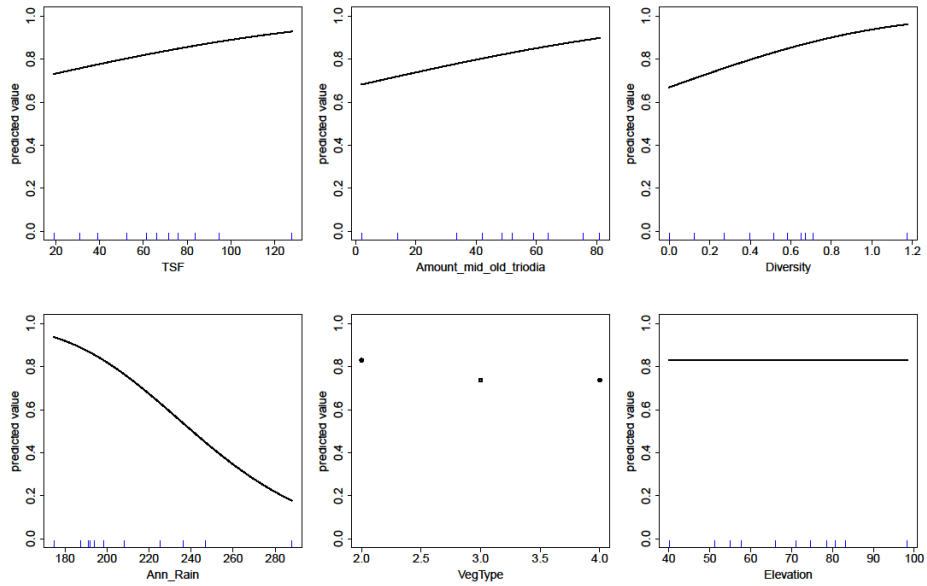
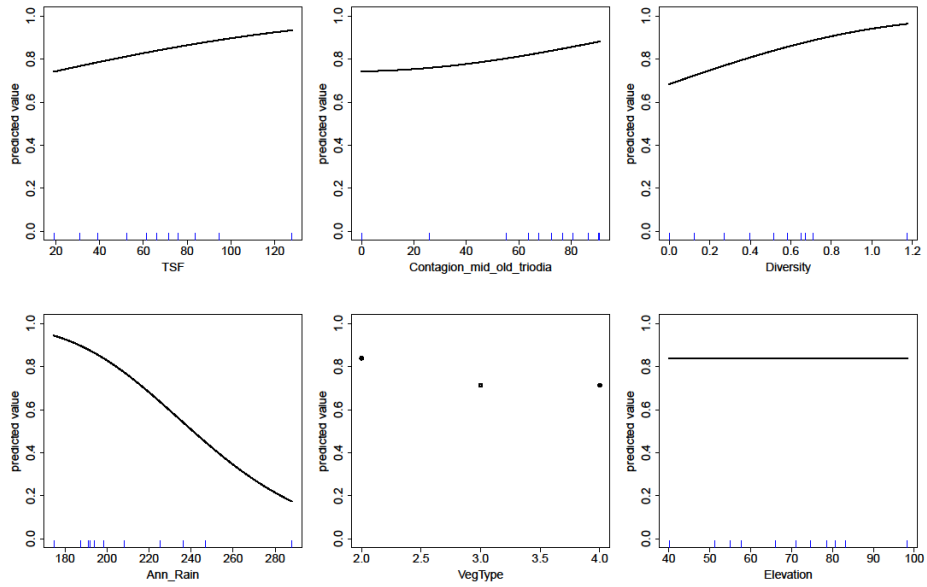


Figure S4.3.34: Response of *Cyclodomorphus melanops* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

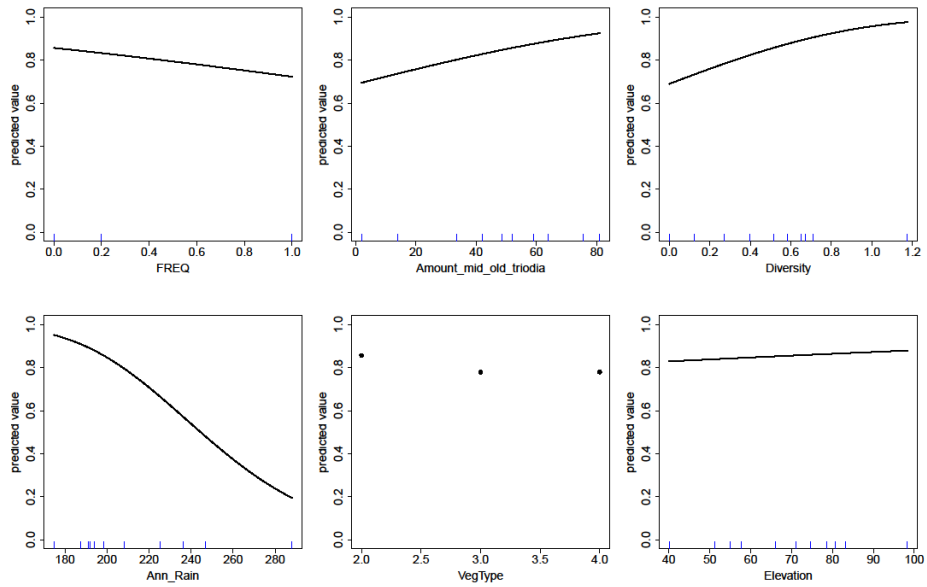
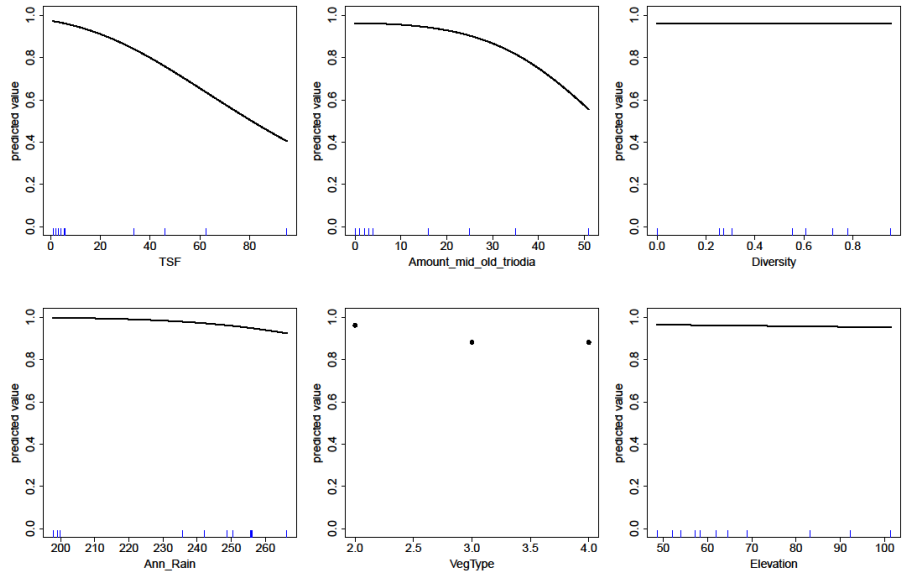
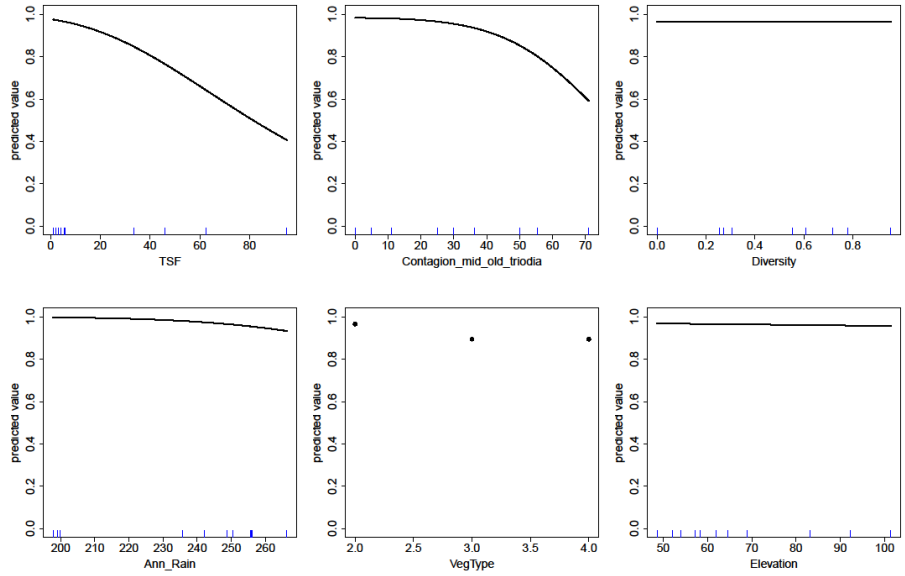


Figure S4.3.35: Response of *Eremiascincus richardsonii* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

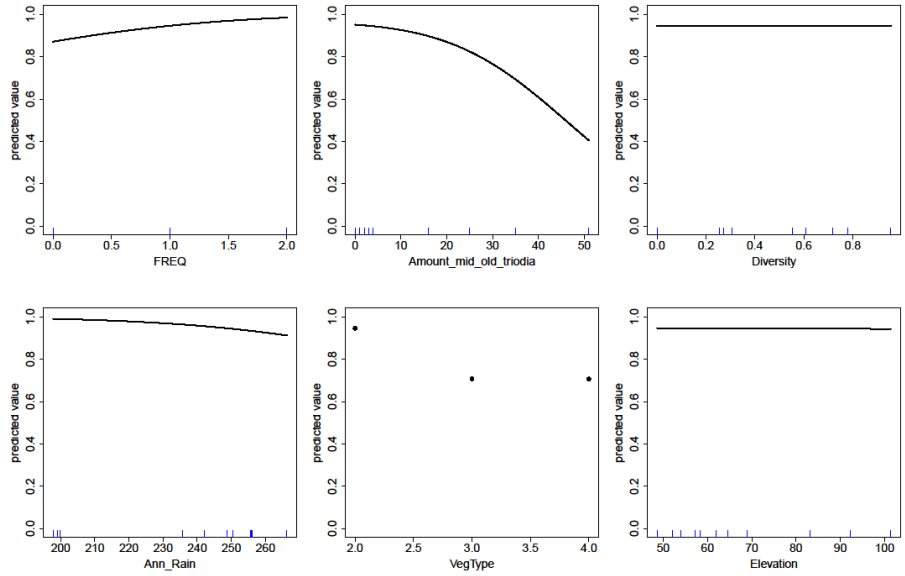
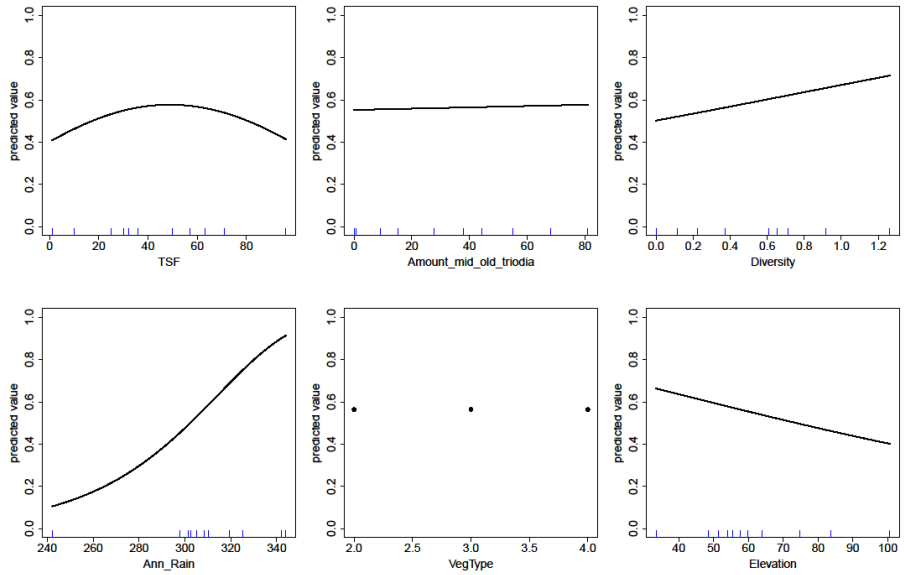
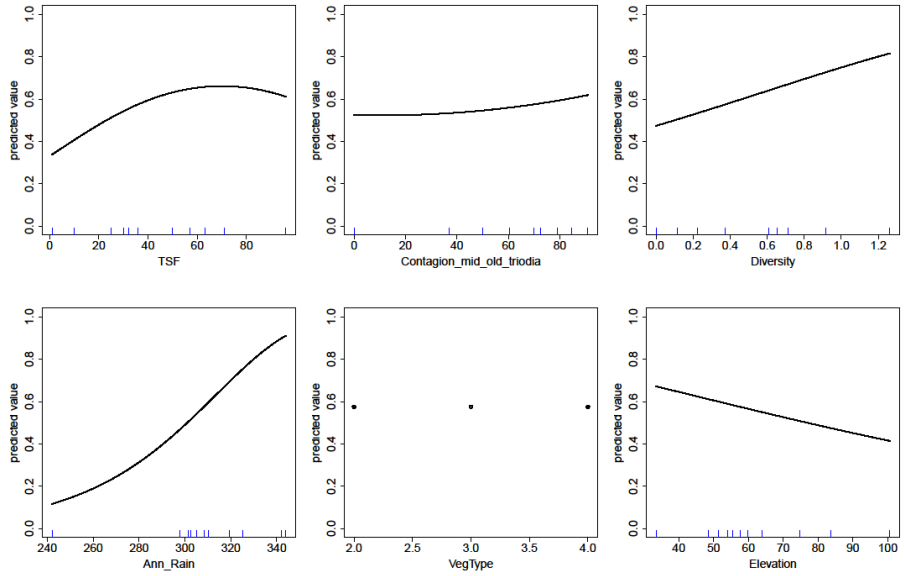


Figure S4.3.36: Response of *Lerista aericeps* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

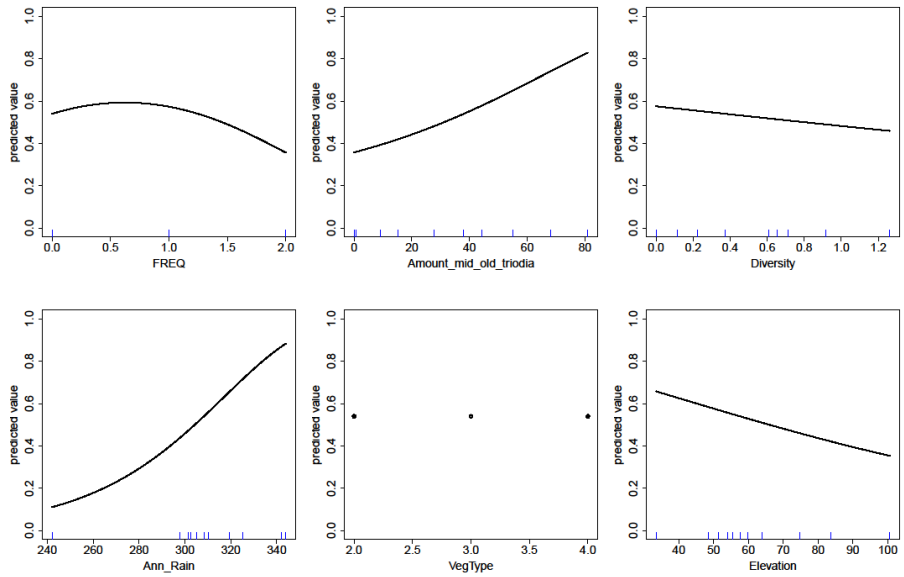
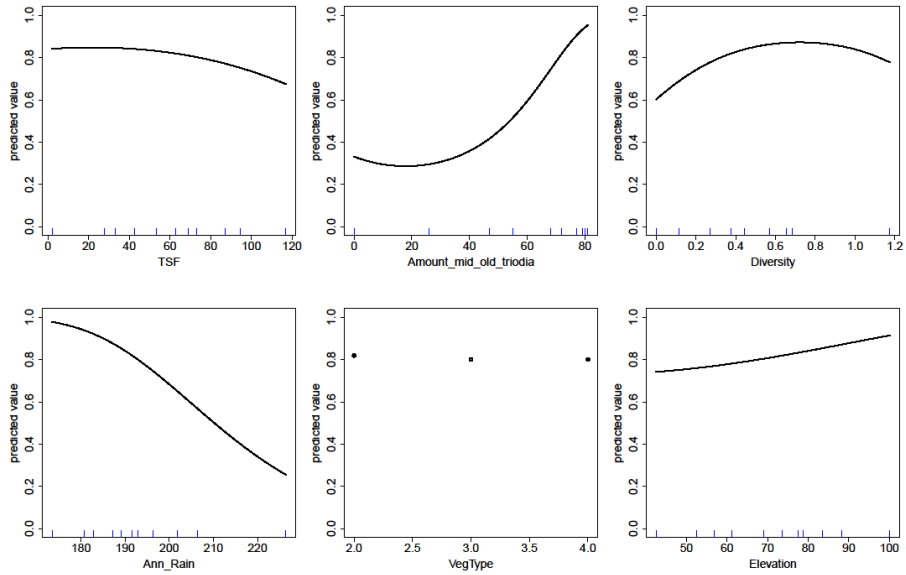
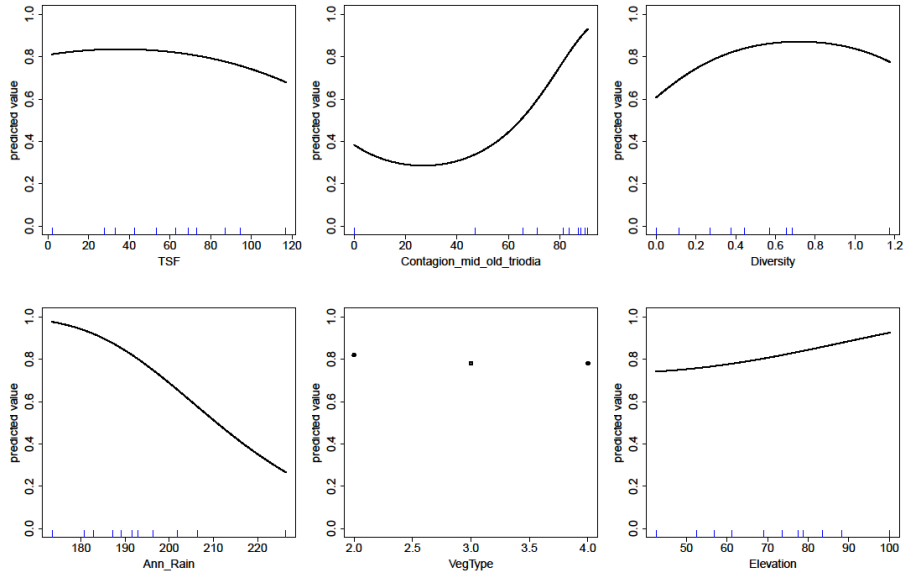


Figure S4.3.37: Response of *Lerista bougainvillii* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

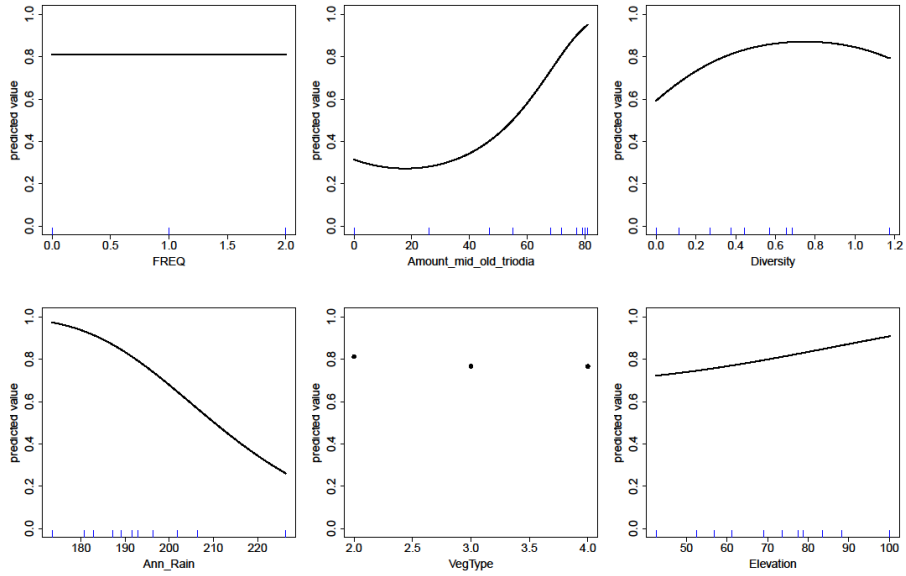
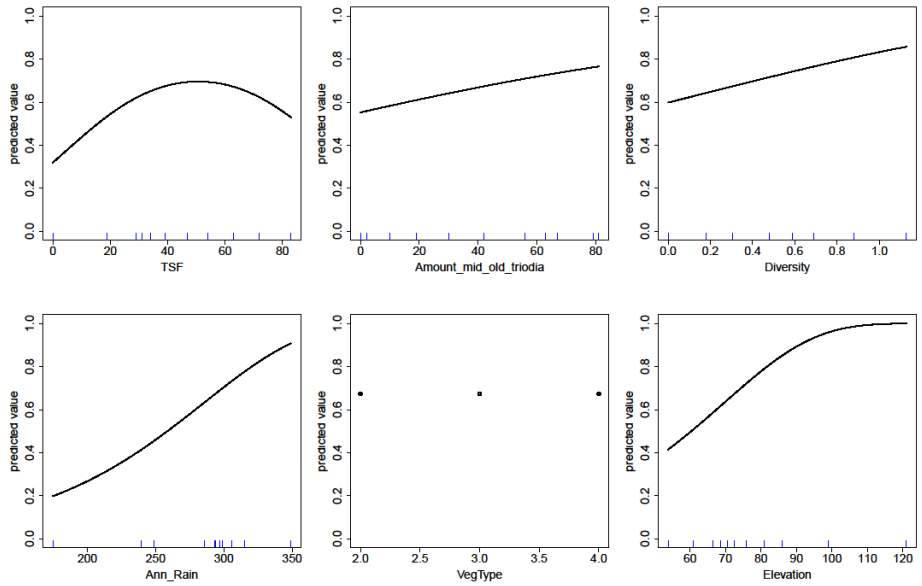
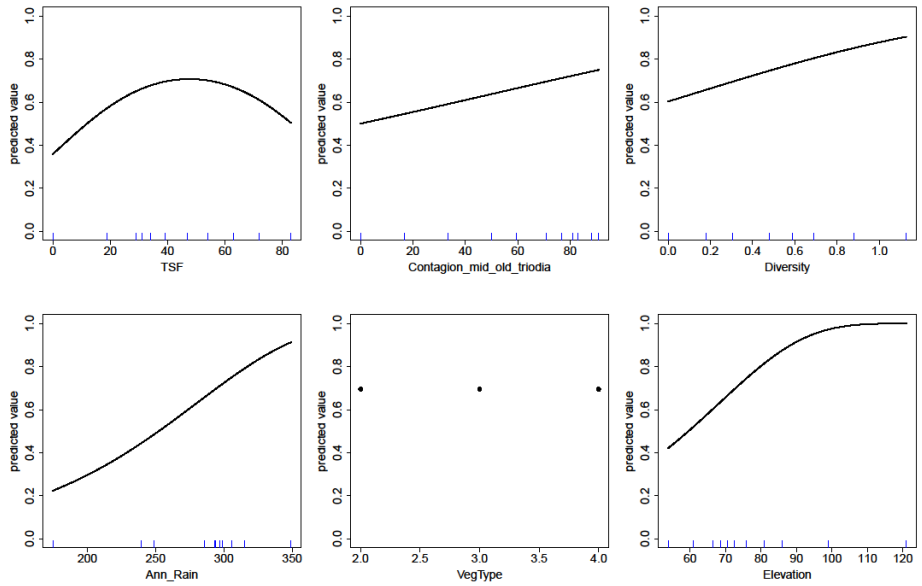


Figure S4.3.38: Response of *Lerista labialis* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

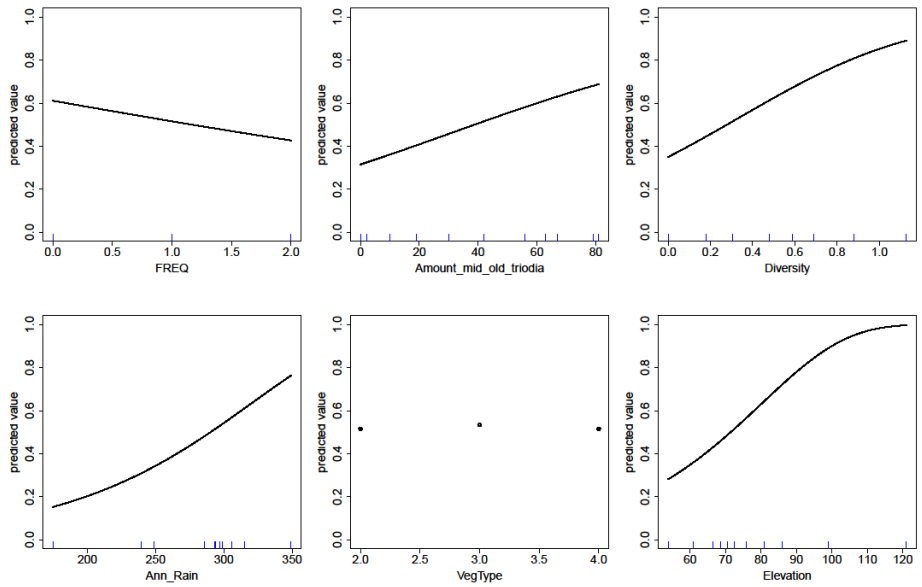
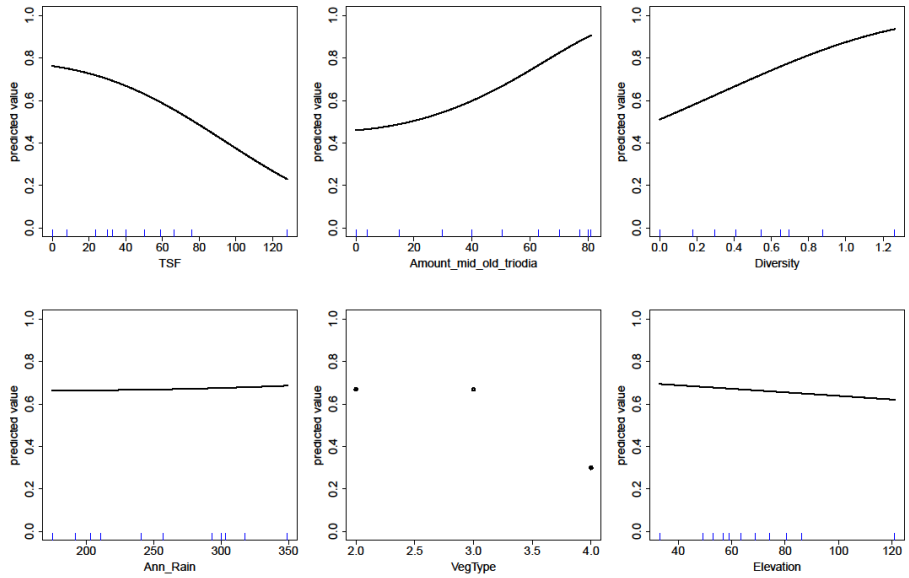
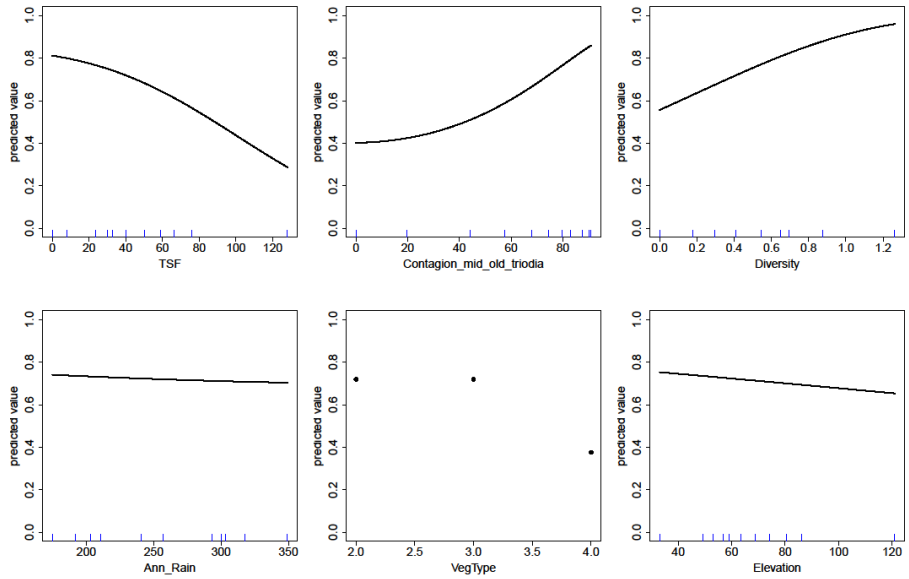


Figure S4.3.39: Response of *Lerista muelleri* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

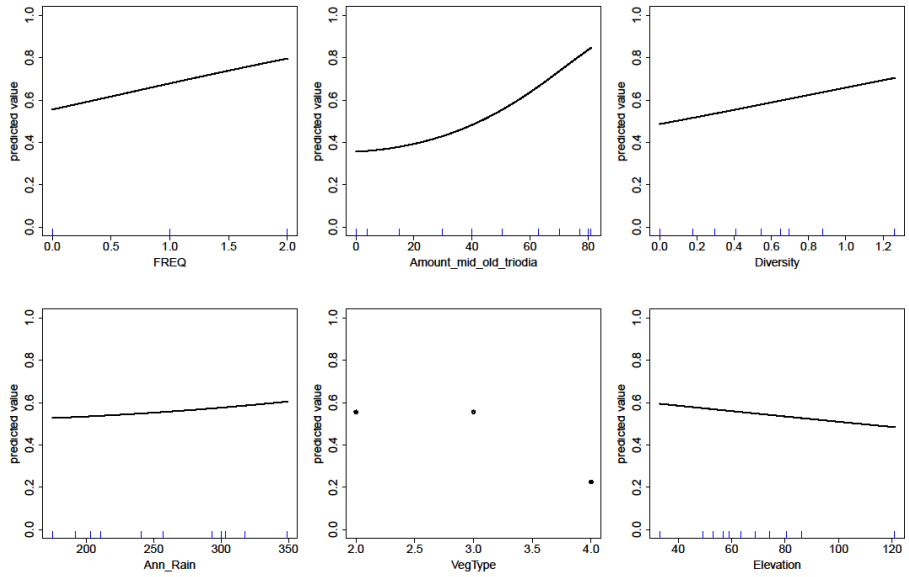
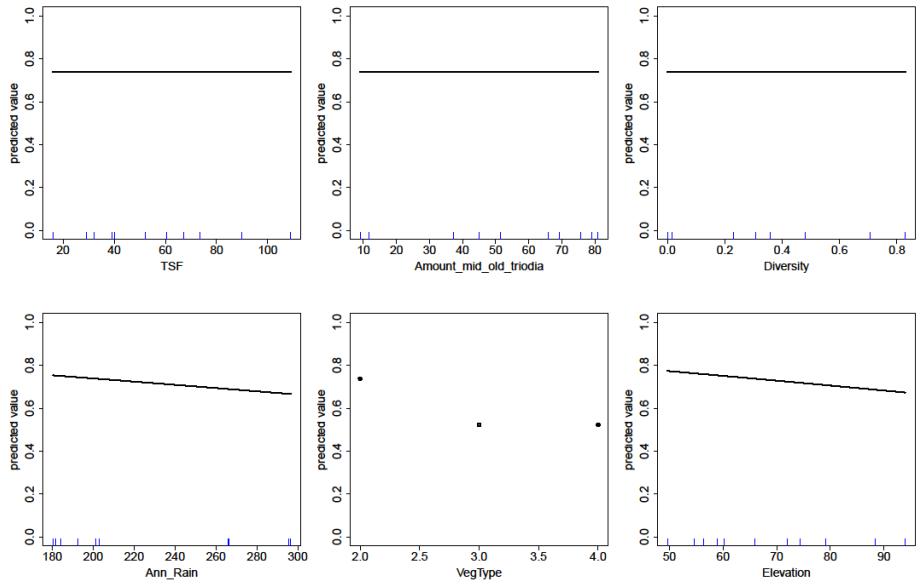
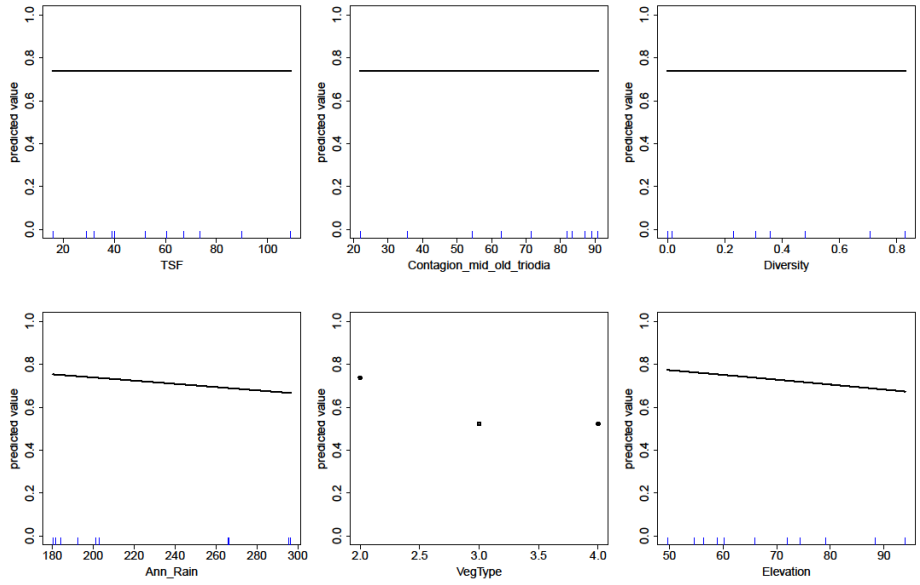


Figure S4.3.40: Response of *Lerista punctatovittata* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

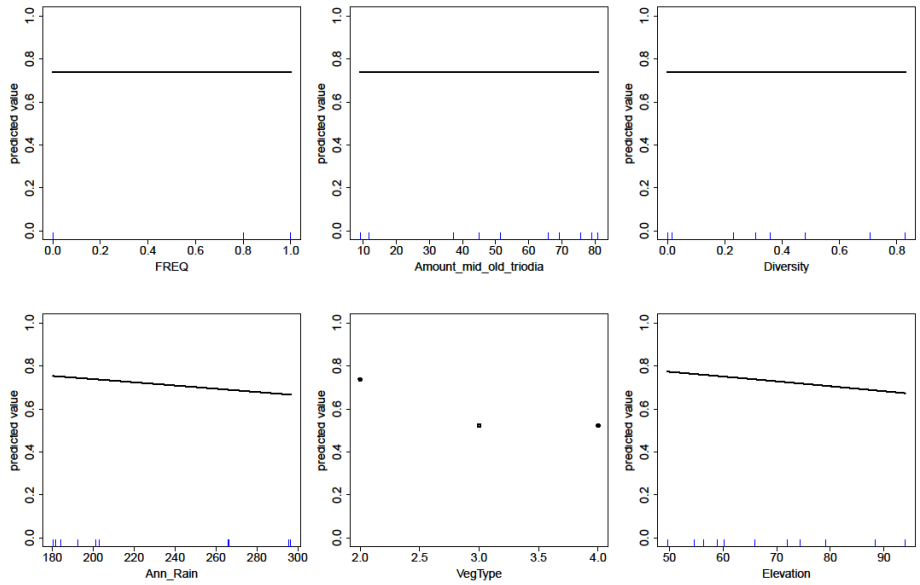
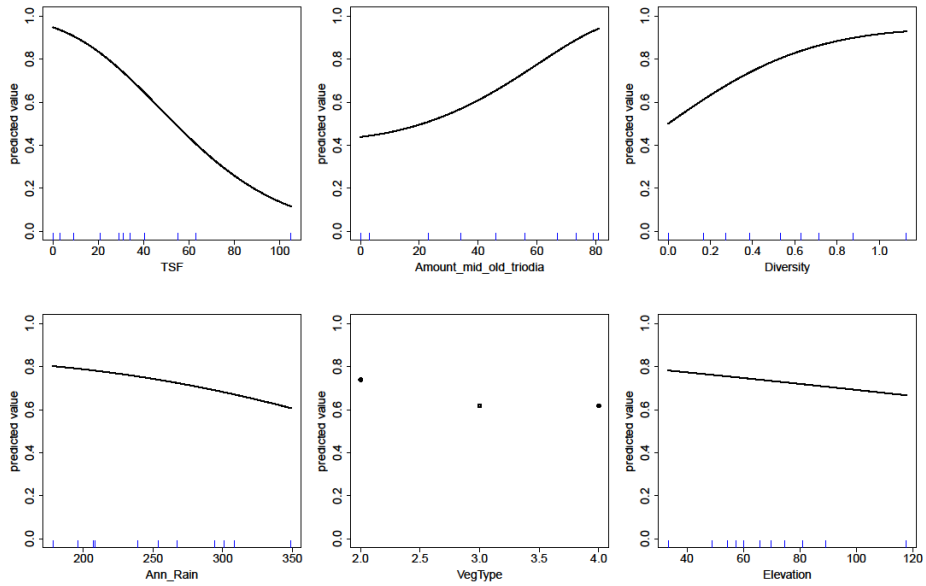
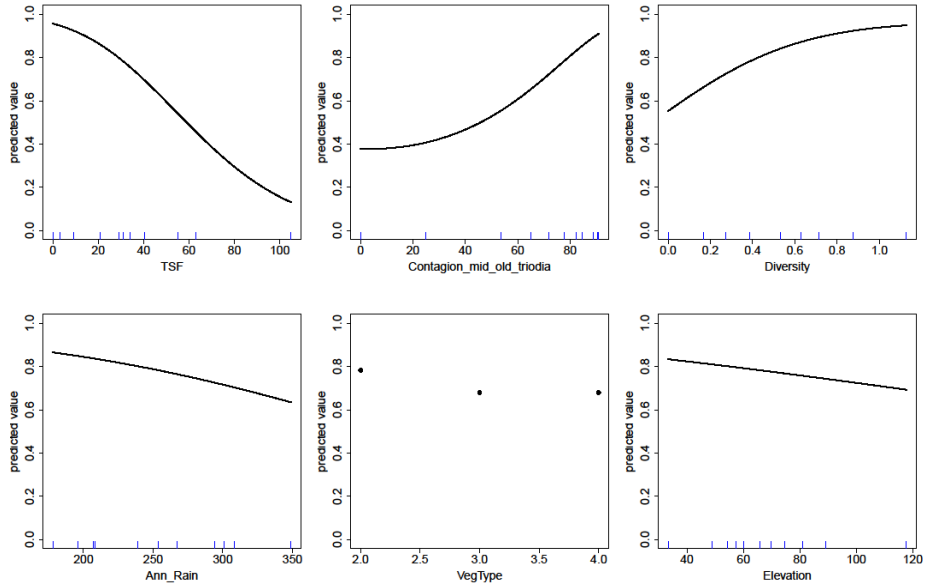


Figure S4.3.41: Response of *Lerista xanthura* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

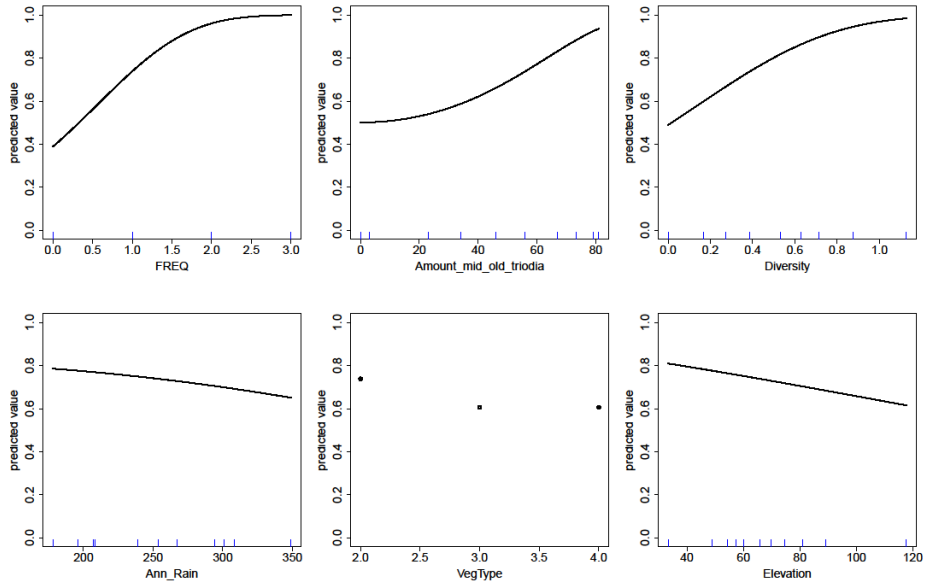
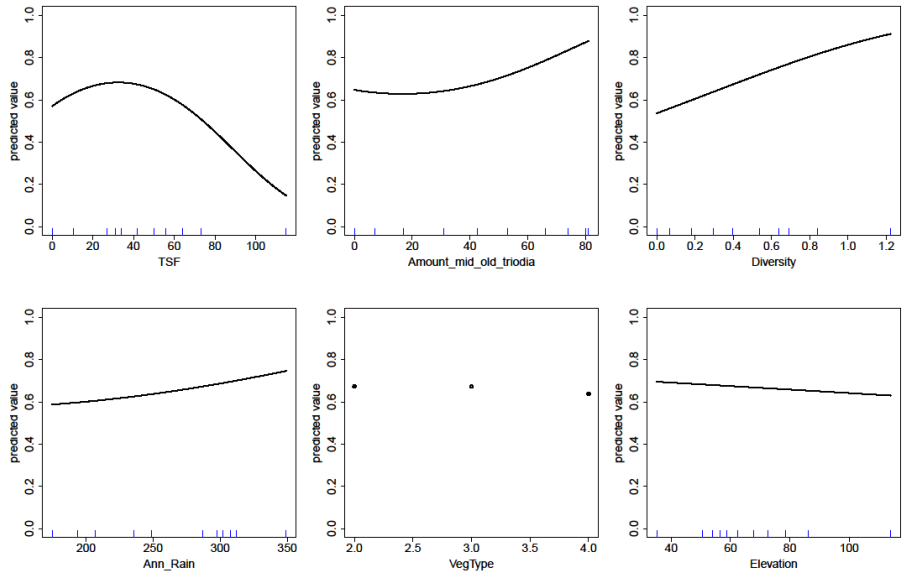
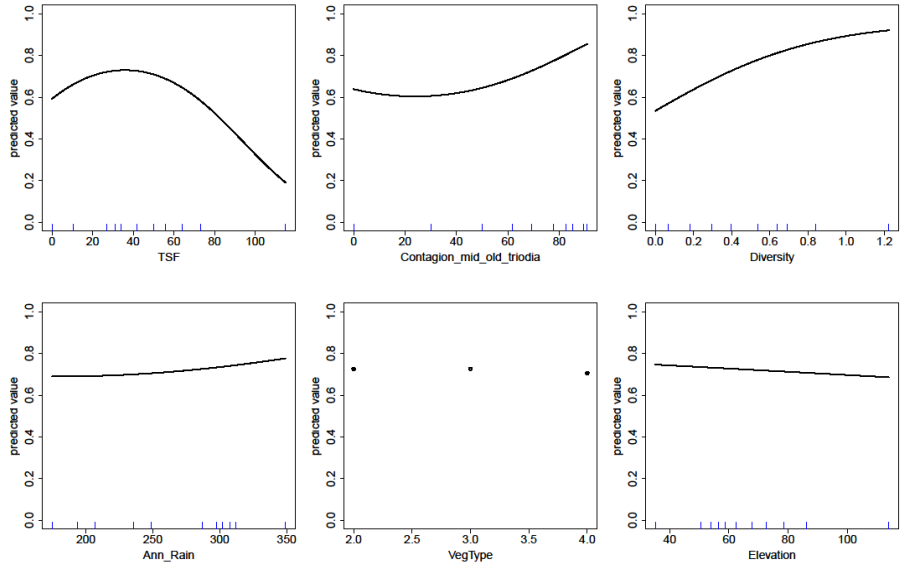


Figure S4.3.42: Response of *Liopholis inornata* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

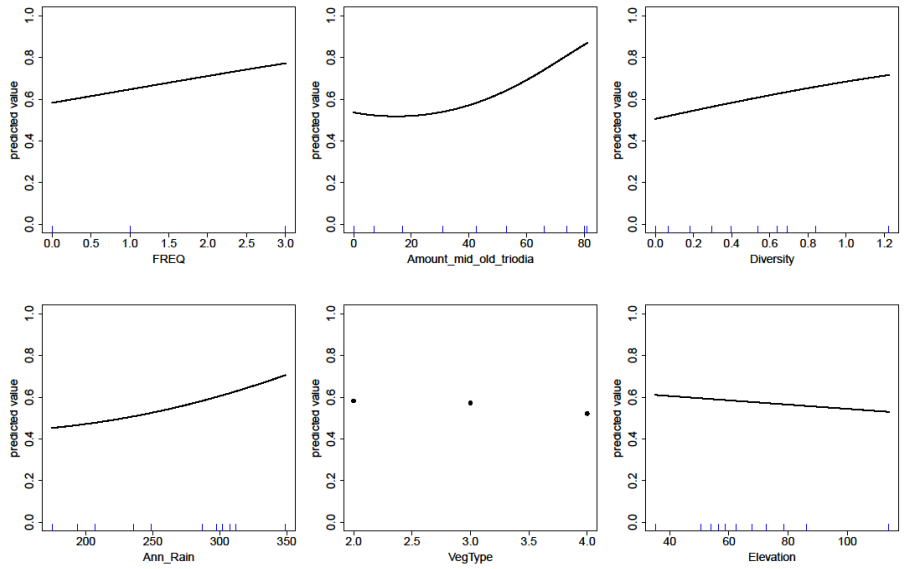
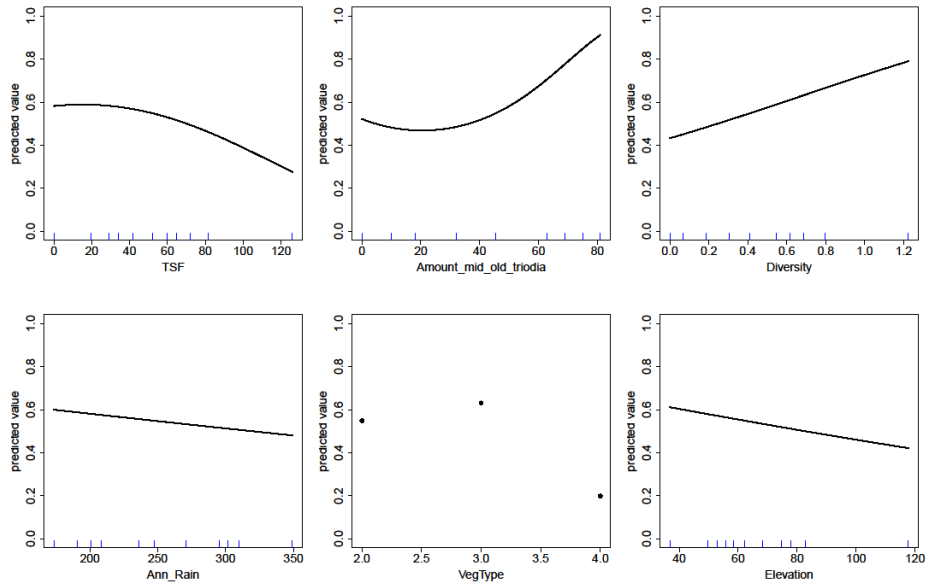
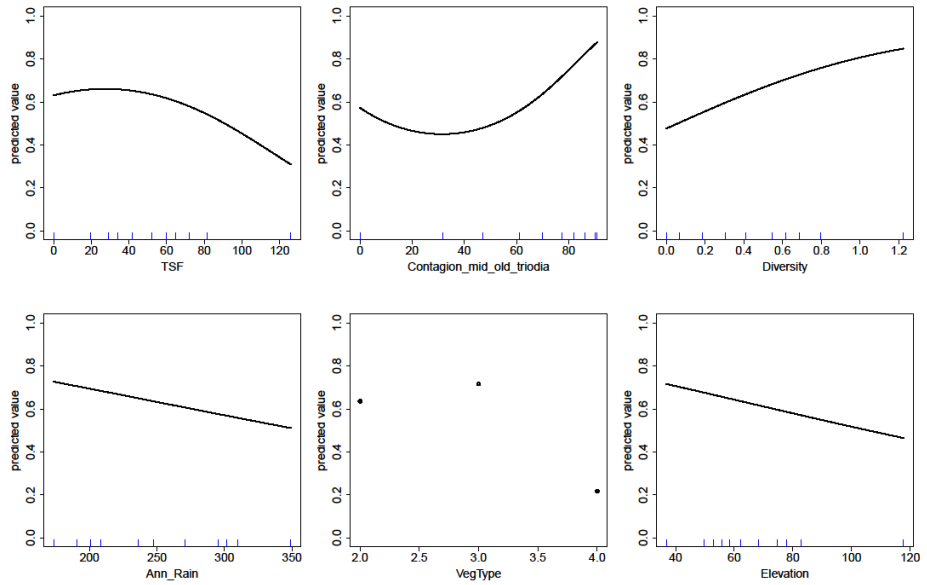


Figure S4.3.43: Response of *Menetia greyii* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

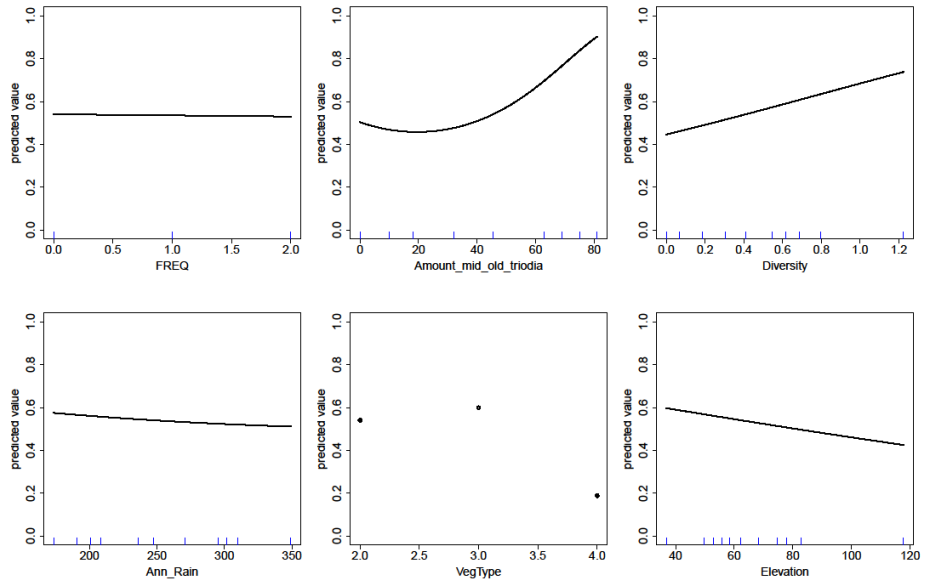
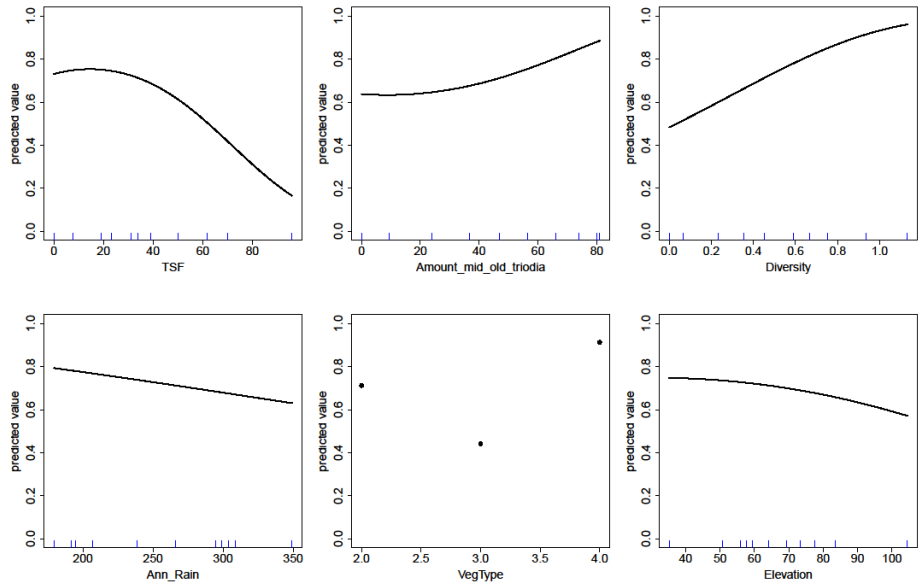
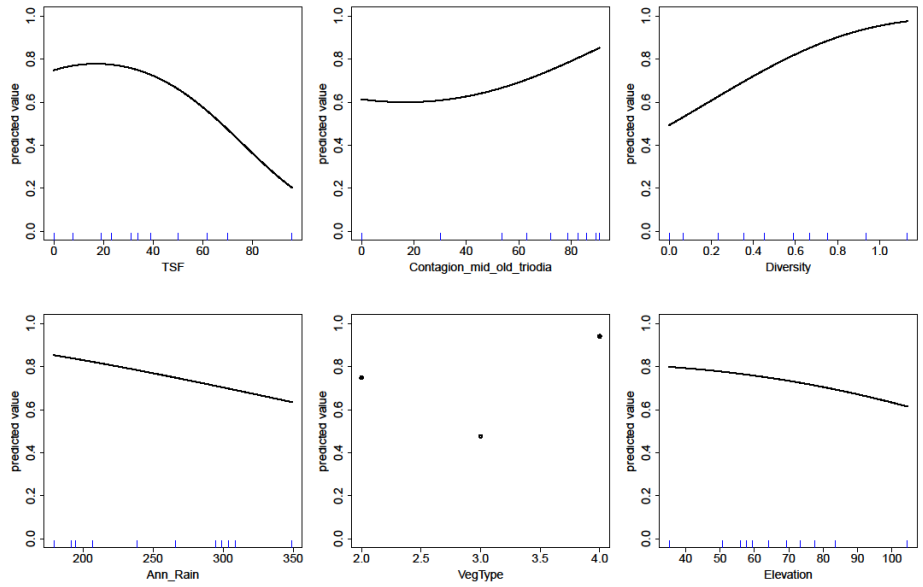


Figure S4.3.44: Response of *Morethia boulengeri* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

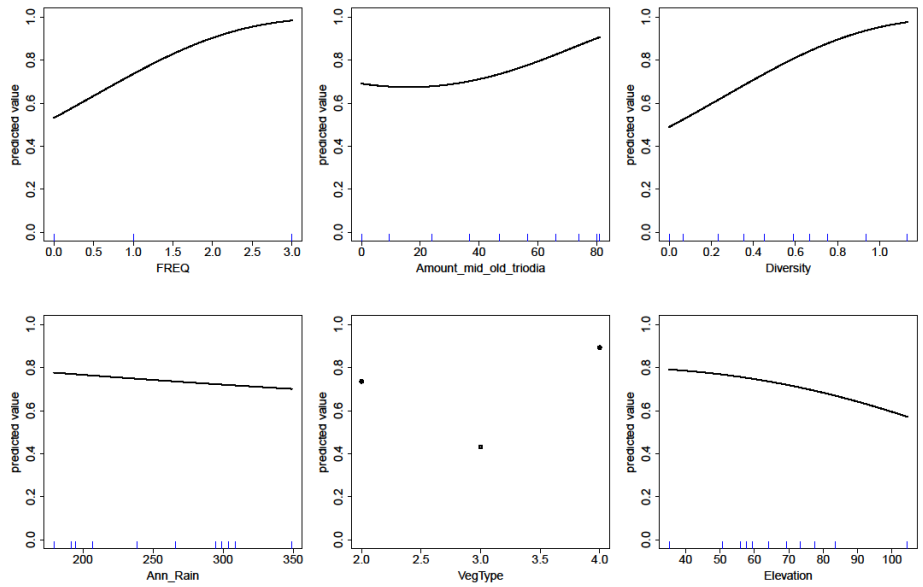
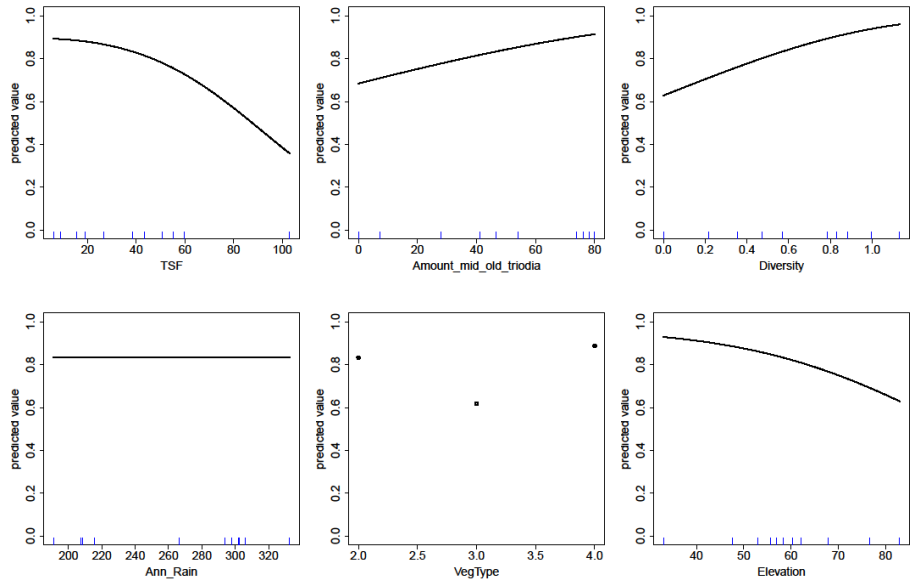
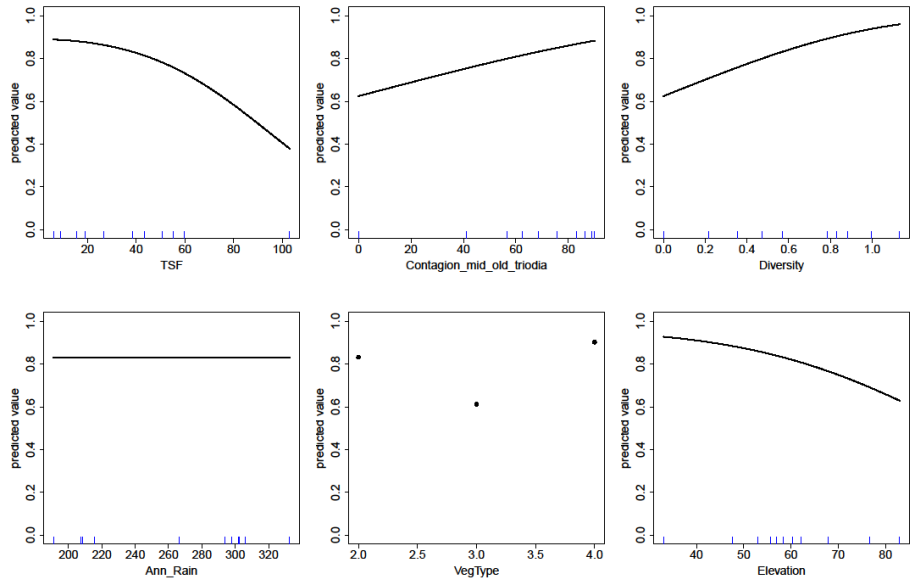


Figure S4.3.45: Response of *Morethia obscura* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

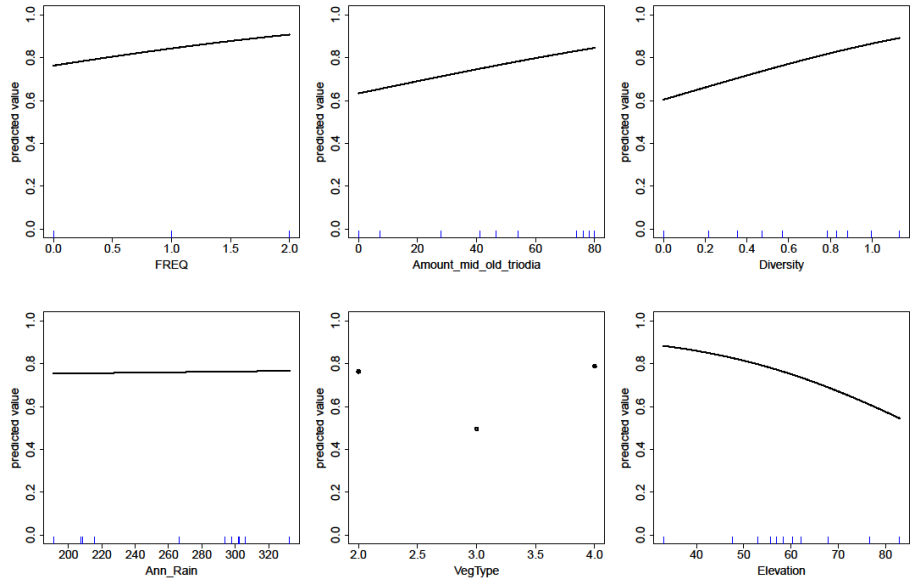
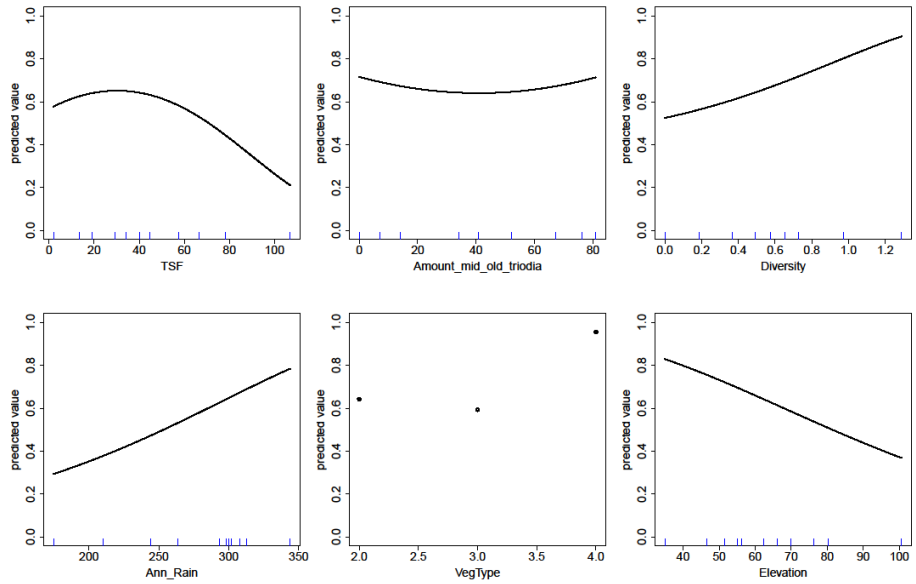
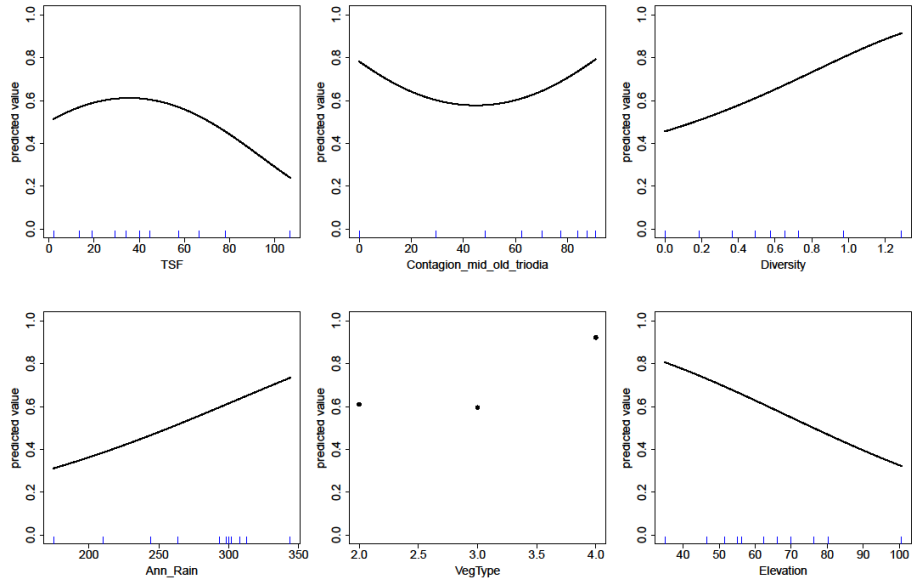


Figure S4.3.46: Response of *Tiliqua occipitalis* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

a)



b)



c)

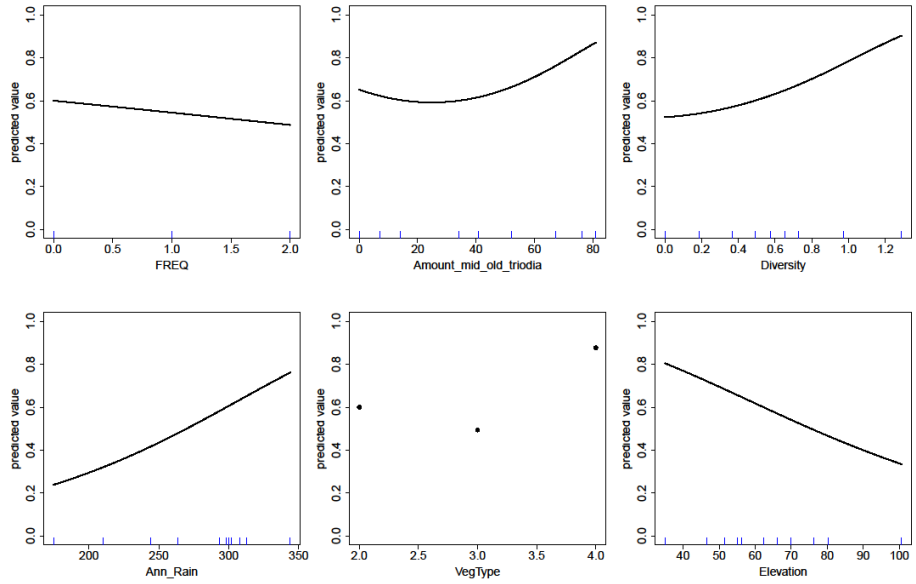


Figure S4.3.47: Response of *Tiliqua rugosa* to predictor variables in; a) M1; b) M2; c) M3 when all other variables are held at their median value.

Appendix S4.4 – Maps of predicted distributions of mallee reptiles by taxonomic group

Preface: Figures in this appendix are formatted as embedded PDF files and open externally for enhanced viewing

Blind snakes (Typhlopidae)

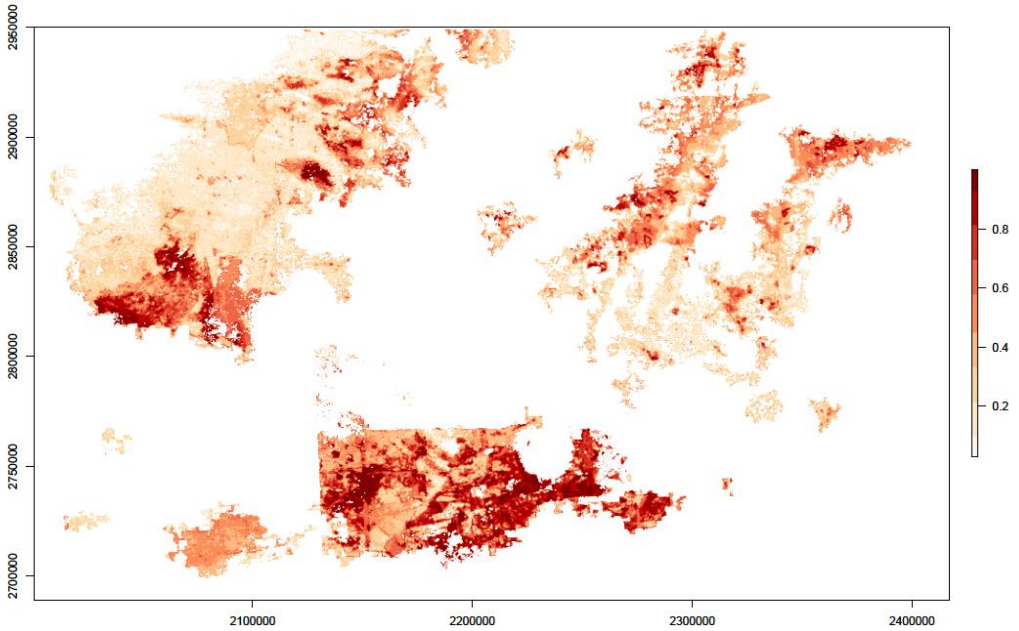


Figure S4.4.1: Relative likelihood of occurrence (cloglog function) of *Anilios bicolor* in relation to M1 predictor variables

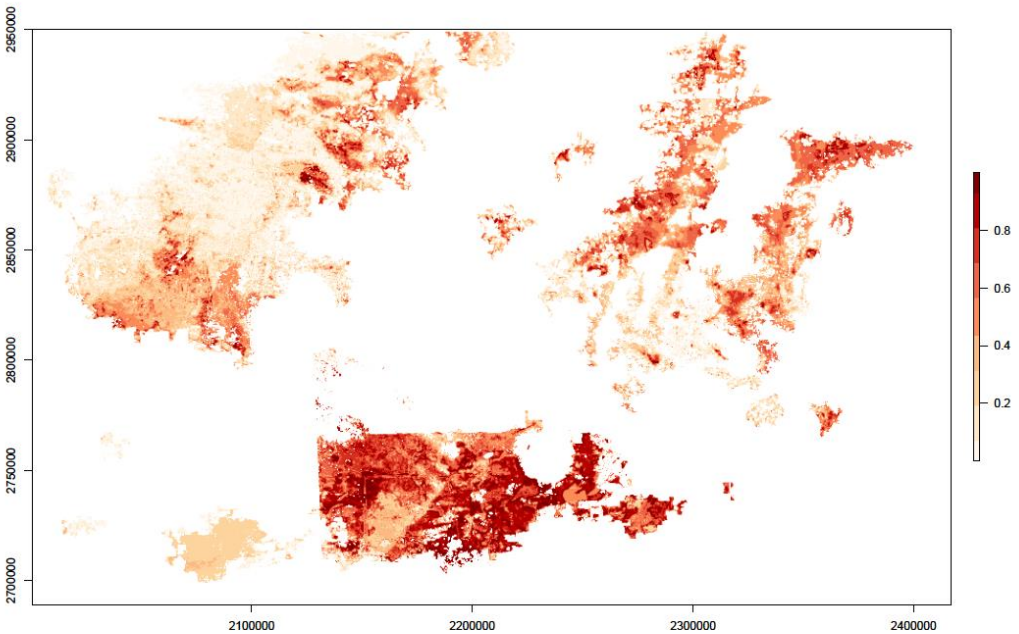


Figure S4.4.2: Relative likelihood of occurrence (cloglog function) of *Anilios bituberculatus* in relation to M1 predictor variables

Dragons (Agamidae)

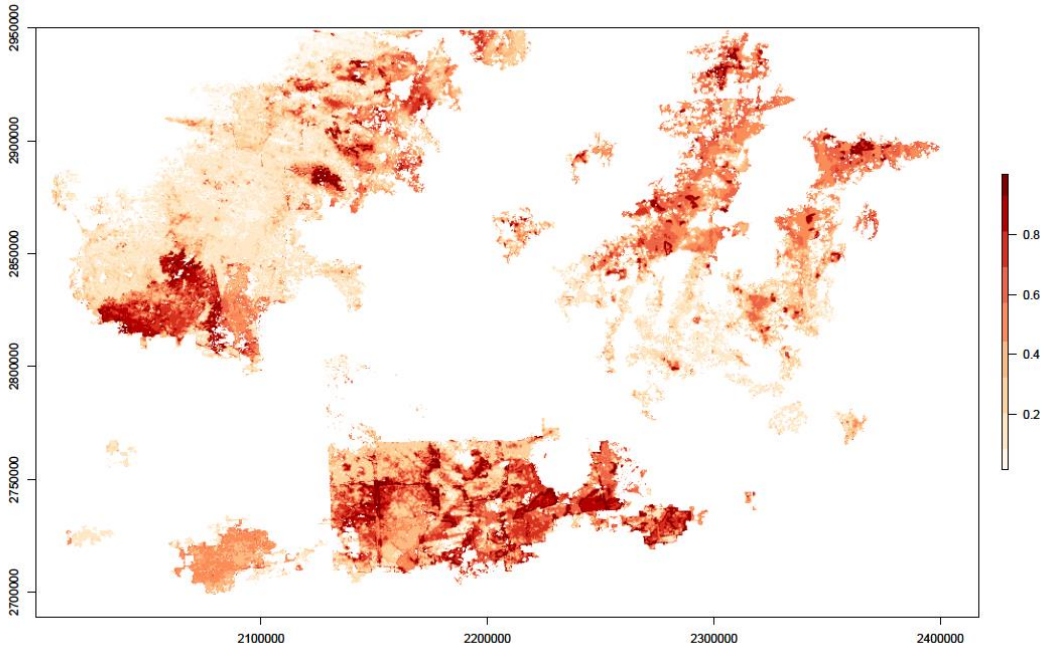


Figure S4.4.3: Relative likelihood of occurrence (cloglog function) of *Ctenophorus fordi* in relation to M1 predictor variables

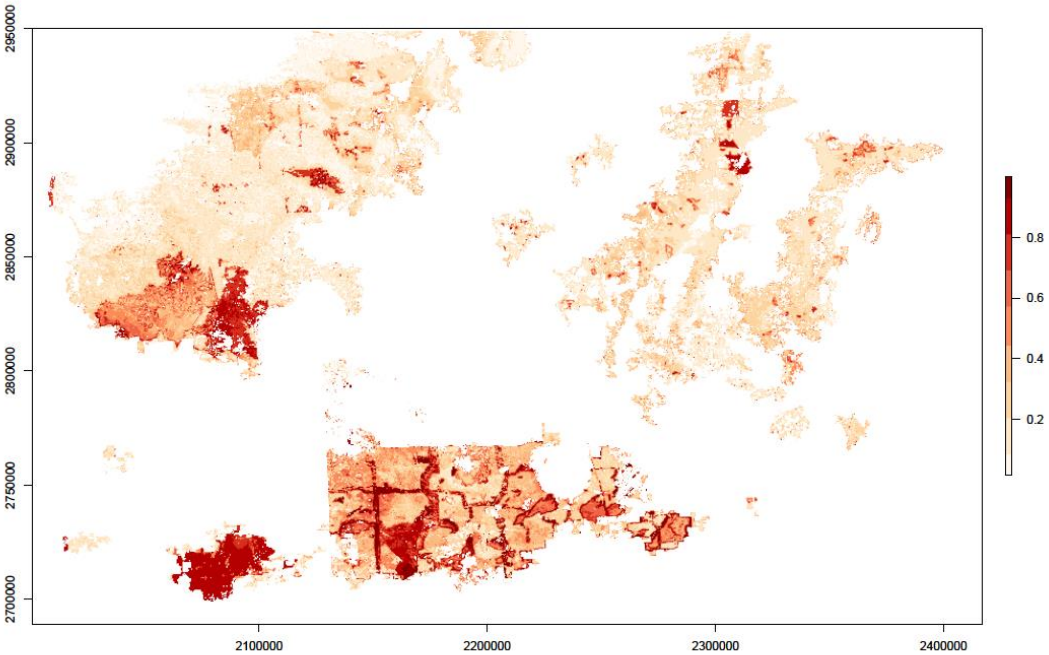


Figure S4.4.4: Relative likelihood of occurrence (cloglog function) of *Ctenophorus pictus* in relation to M1 predictor variables

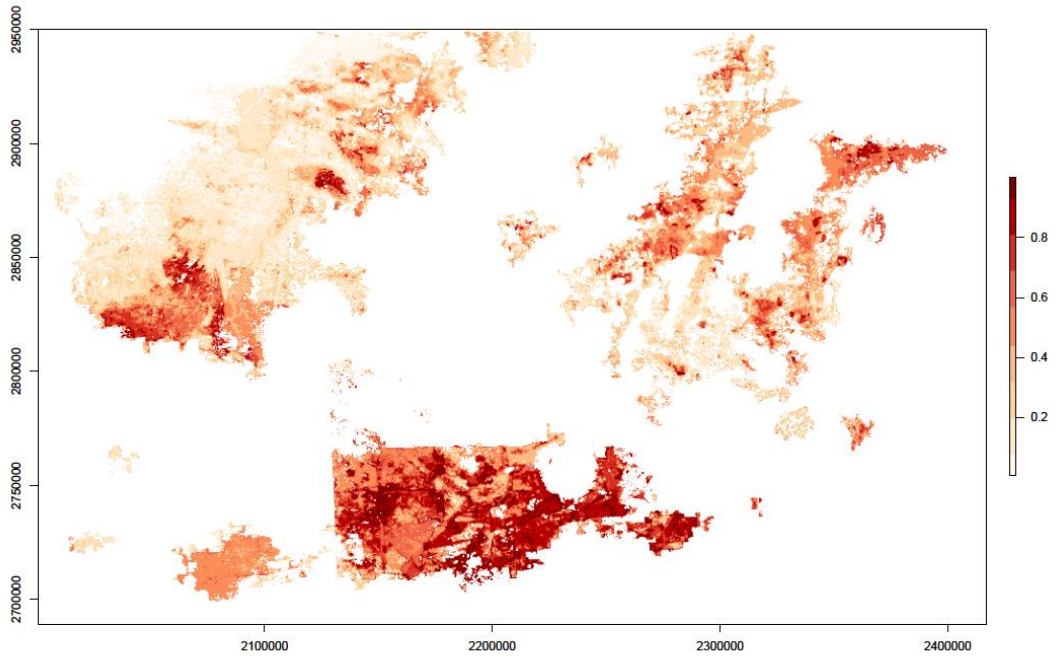


Figure S4.4.5: Relative likelihood of occurrence (cloglog function) of *Diporiphora nobbi* in relation to M1 predictor variables

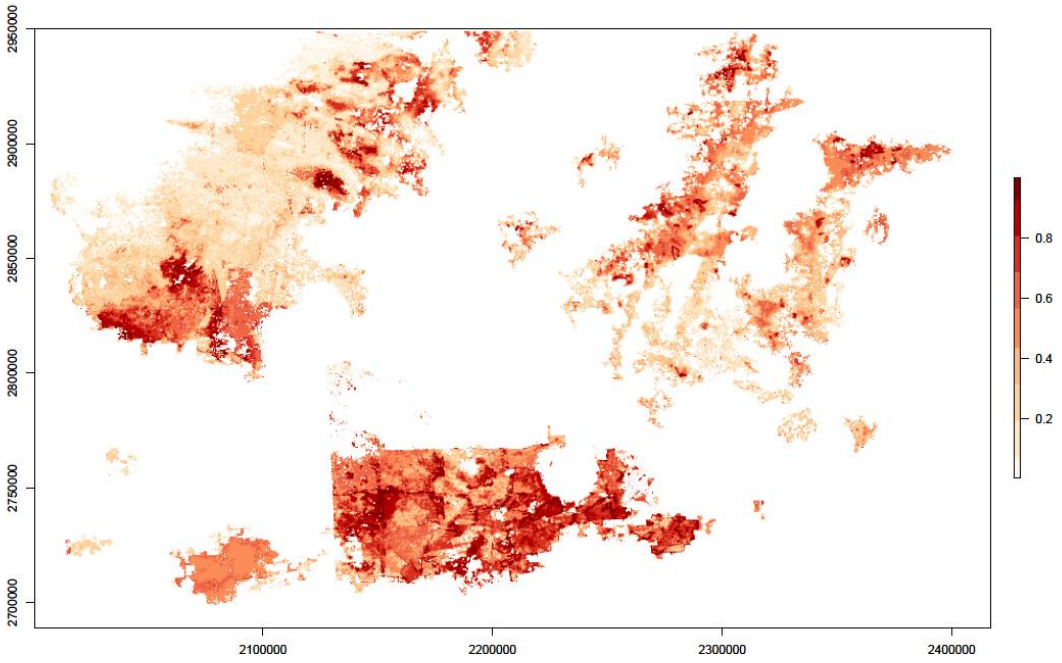


Figure S4.4.6: Relative likelihood of occurrence (cloglog function) of *Pogona vitticeps* in relation to M1 predictor variables

Elapid snakes (Elapidae)

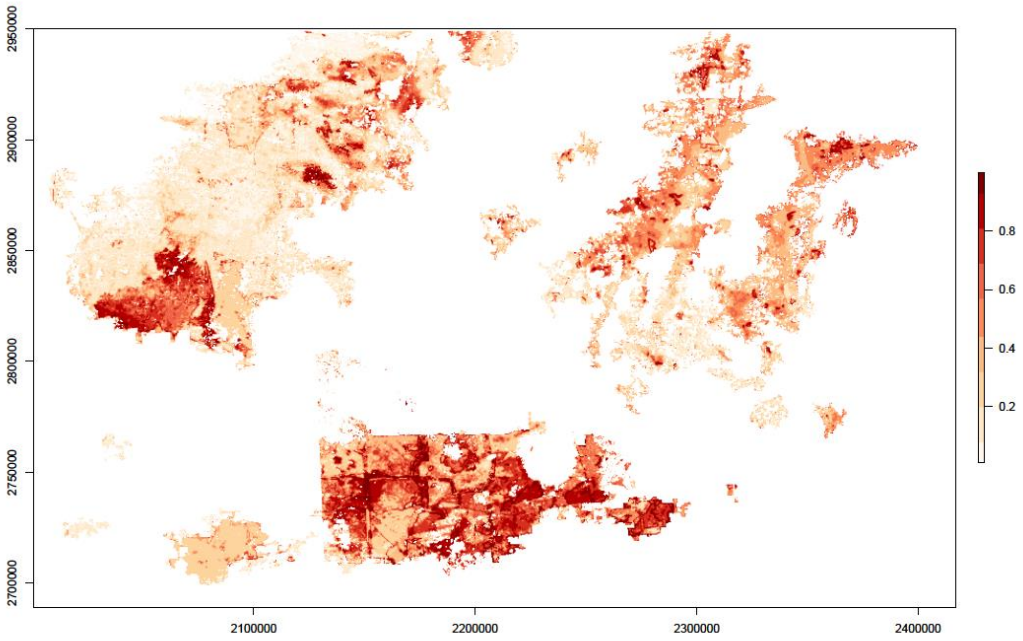


Figure S4.4.7: Relative likelihood of occurrence (cloglog function) of *Brachyurophis australis* in relation to M1 predictor variables

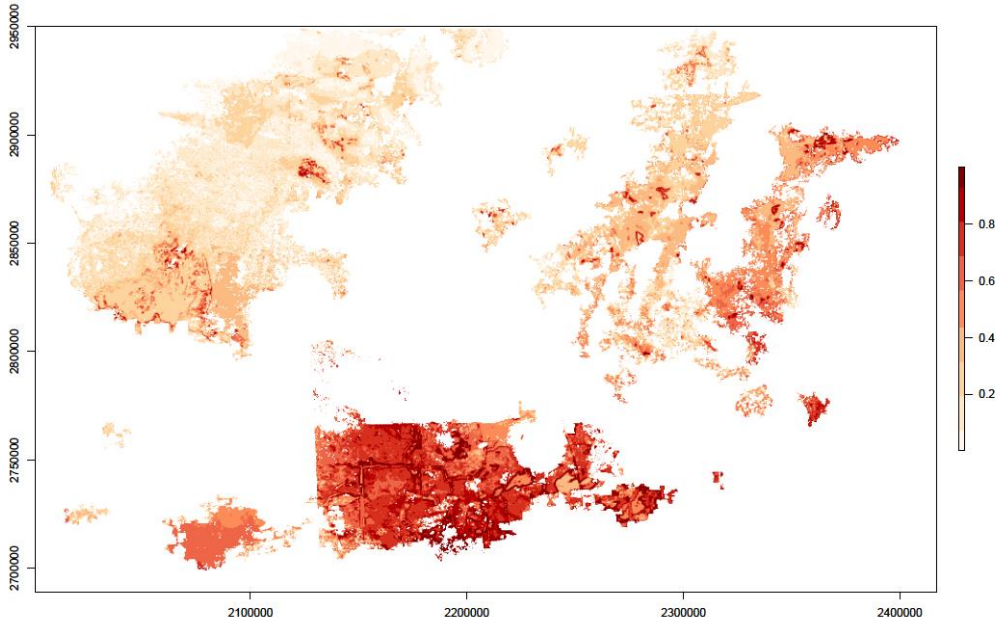


Figure S4.4.8: Relative likelihood of occurrence (cloglog function) of *Parasuta nigriceps* in relation to M1 predictor variables

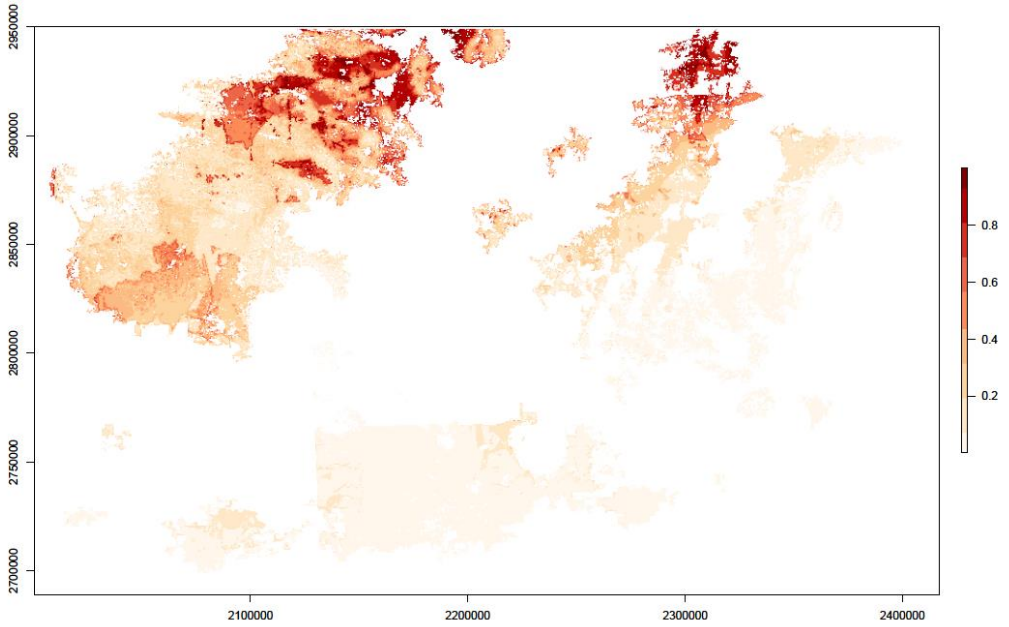


Figure S4.4.9: Relative likelihood of occurrence (cloglog function) of *Pseudonaja modesta* in relation to M1 predictor variables

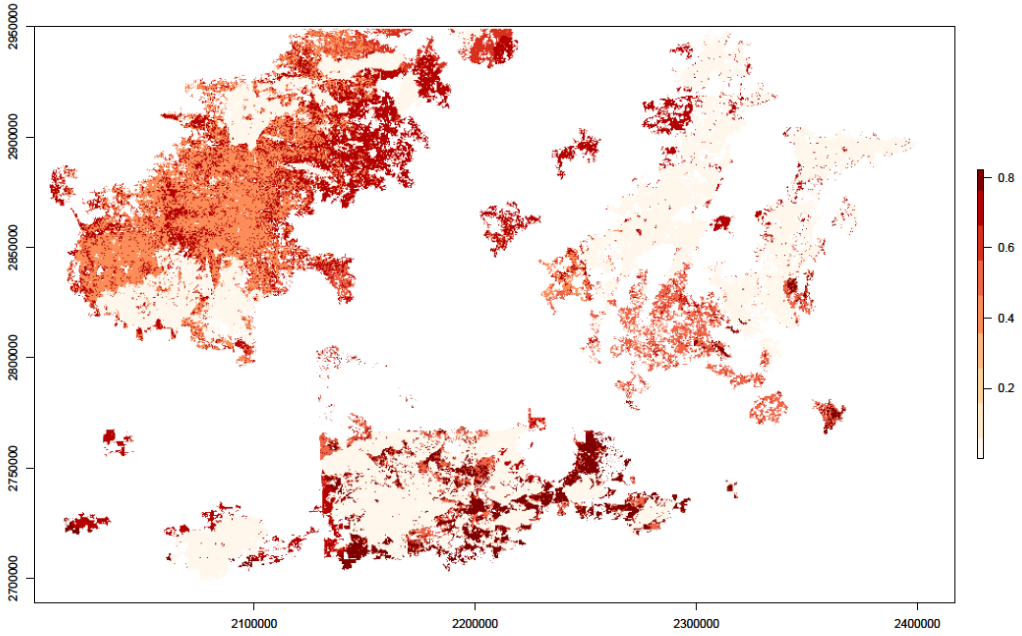


Figure S4.4.10: Relative likelihood of occurrence (cloglog function) of *Pseudonaja textilis* in relation to M3 predictor variables

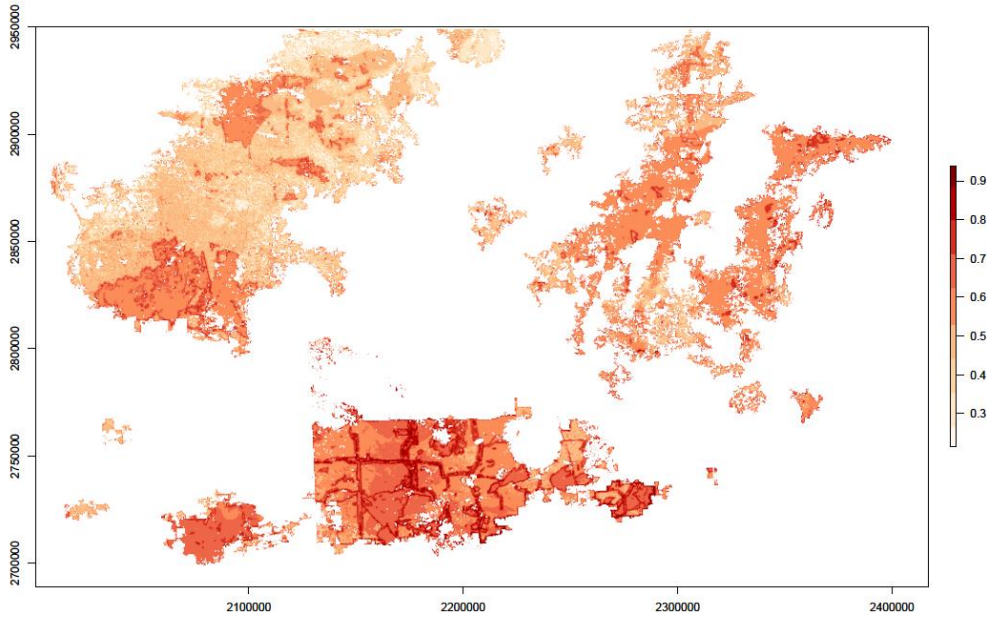


Figure S4.4.11: Relative likelihood of occurrence (cloglog function) of *Vermicella annulata* in relation to M1 predictor variables

Geckos (Carphodactylidae and Diplodactylidae)

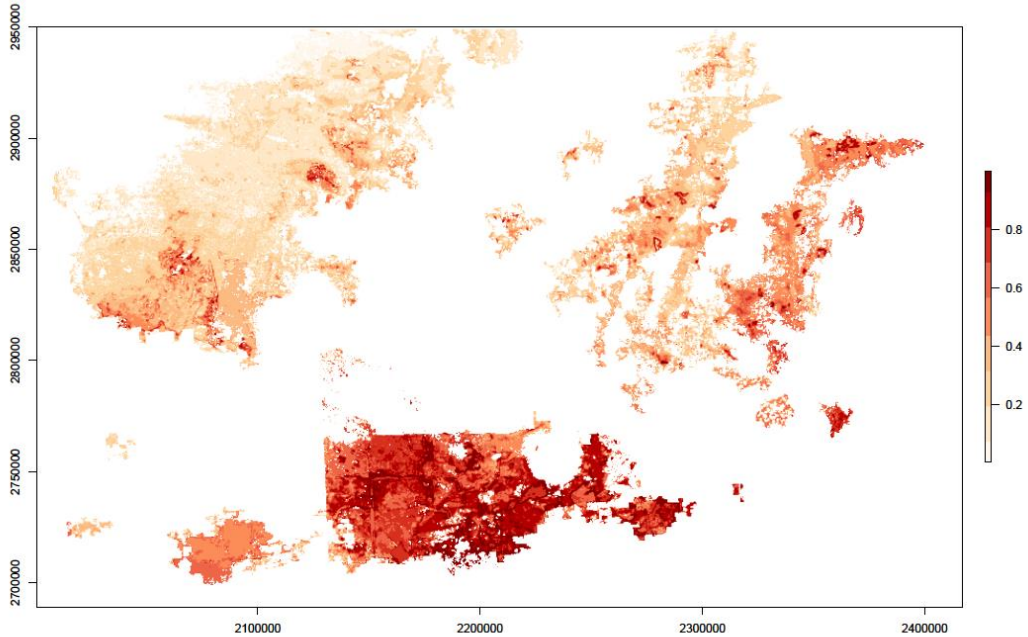


Figure S4.4.12: Relative likelihood of occurrence (cloglog function) of *Diplodactylus vittatus* in relation to M1 predictor variables

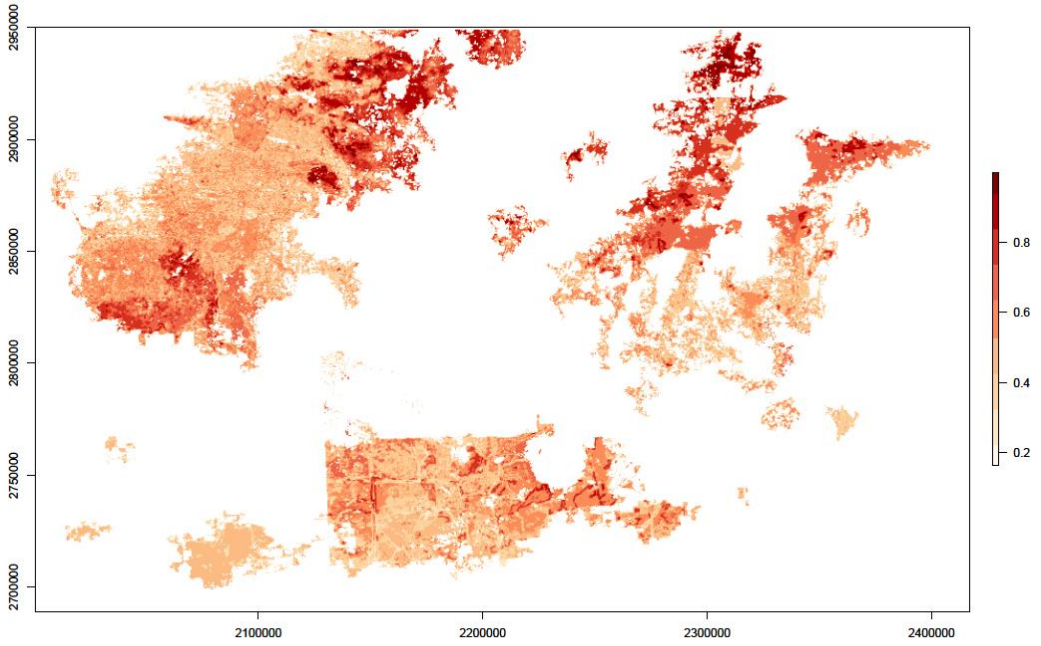


Figure S4.4.13: Relative likelihood of occurrence (cloglog function) of *Gehyra variegata* in relation to M1 predictor variables

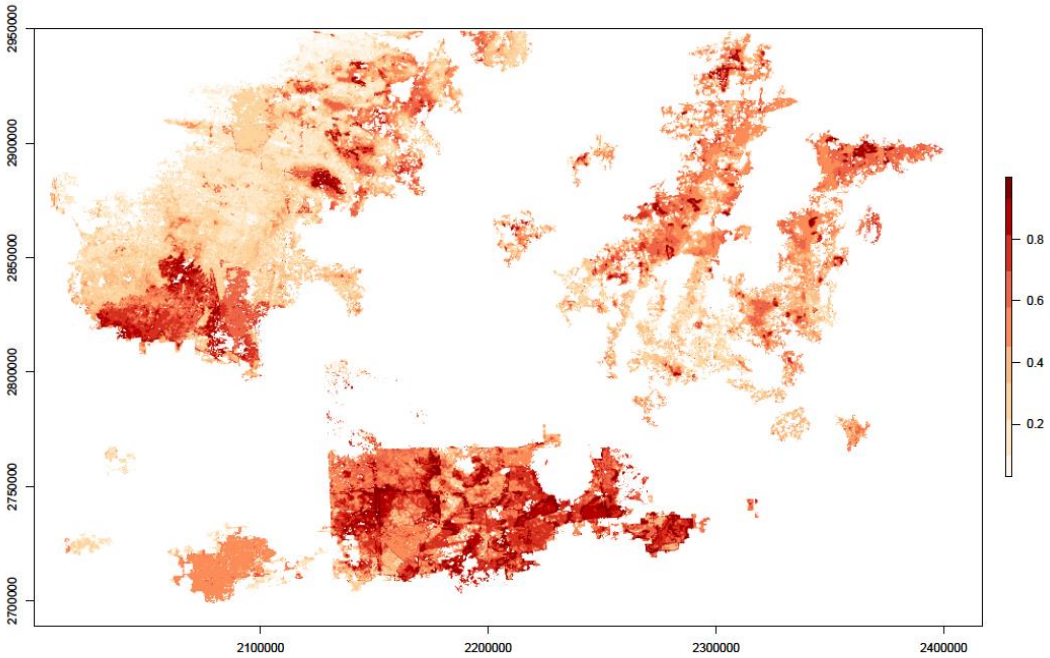


Figure S4.4.14: Relative likelihood of occurrence (cloglog function) of *Lucasium damaeum* in relation to M1 predictor variables

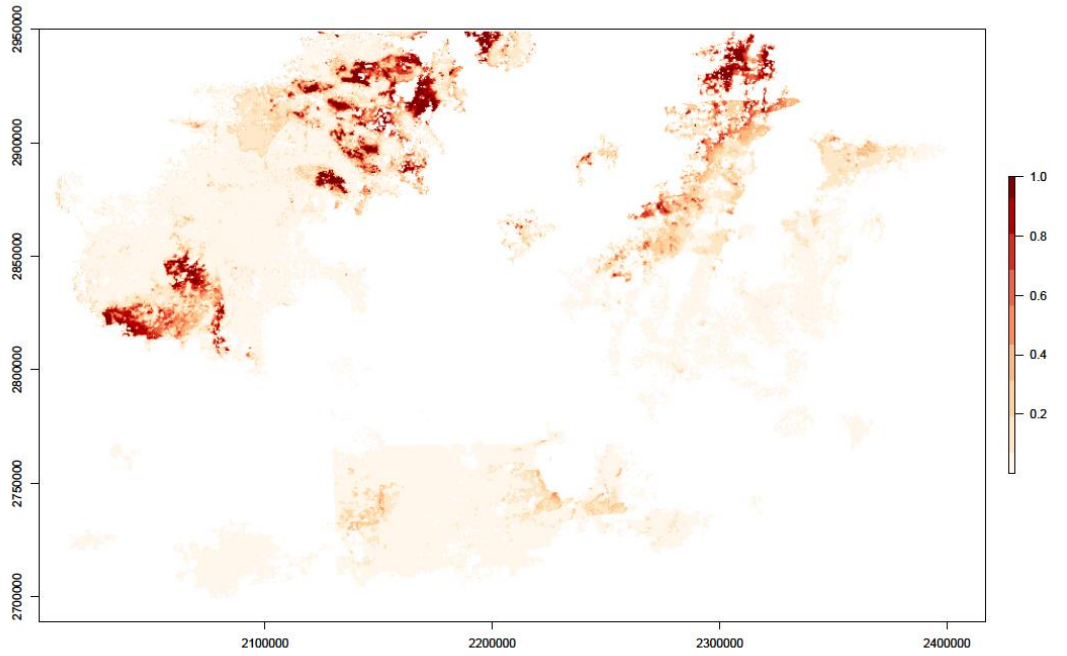


Figure S4.4.15: Relative likelihood of occurrence (cloglog function) of *Nephrurus levis* in relation to M1 predictor variables

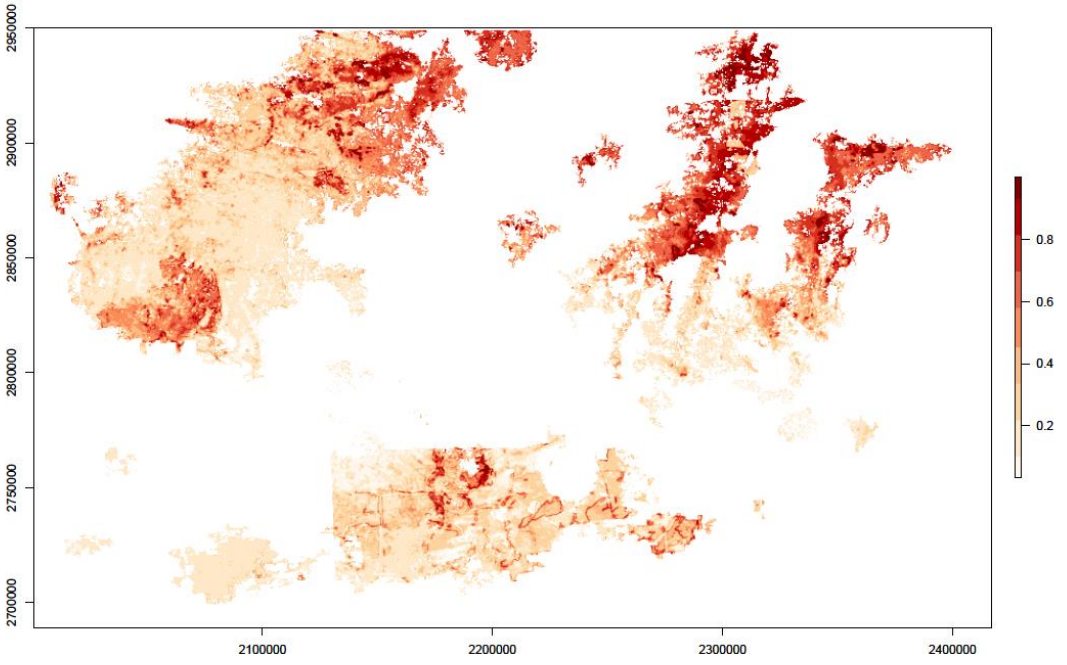


Figure S4.4.16: Relative likelihood of occurrence (cloglog function) of *Rhynchoedura ormsbyi* in relation to M1 predictor variables

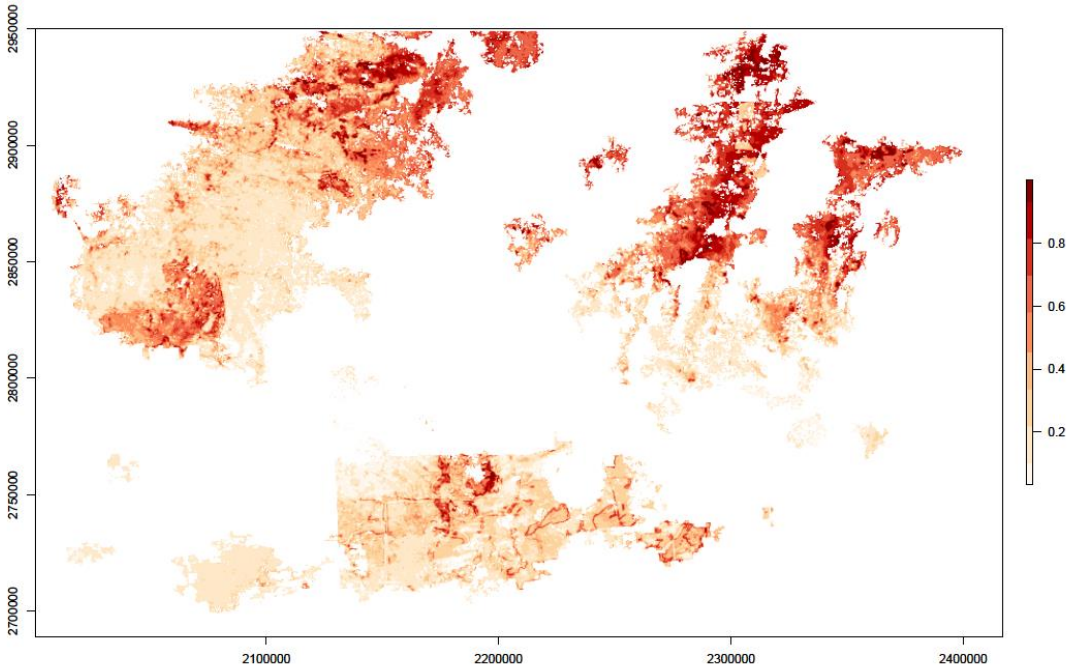


Figure S4.4.17: Relative likelihood of occurrence (cloglog function) of *Rhynchoedura ornata* in relation to M1 predictor variables

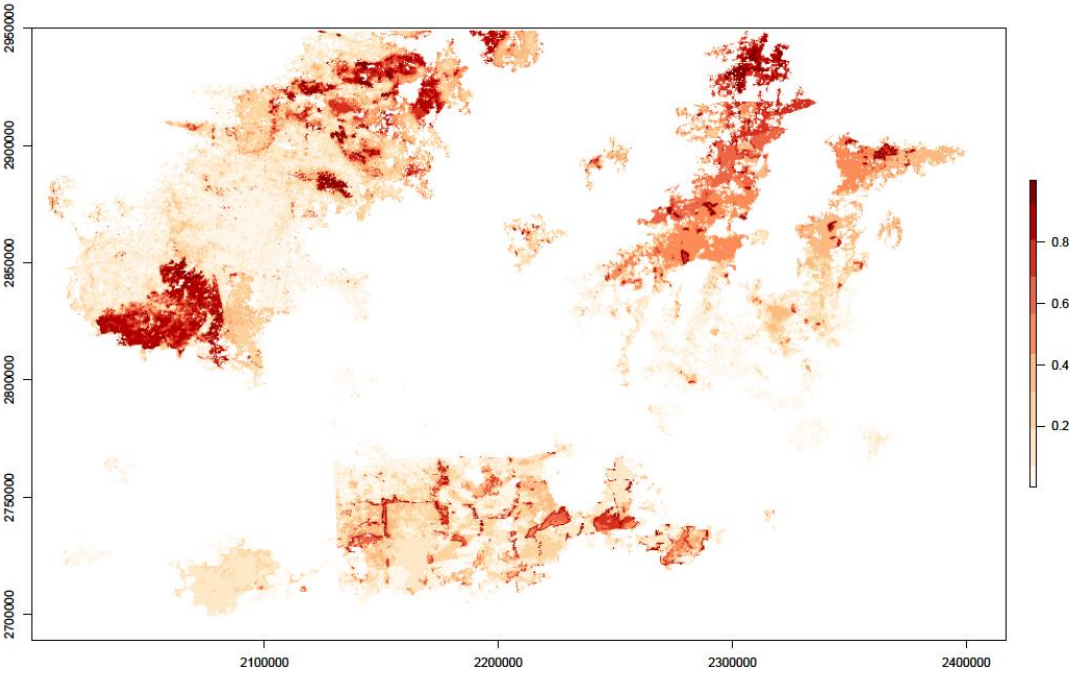


Figure S4.4.18: Relative likelihood of occurrence (cloglog function) of *Strophurus elderi* in relation to M1 predictor variables

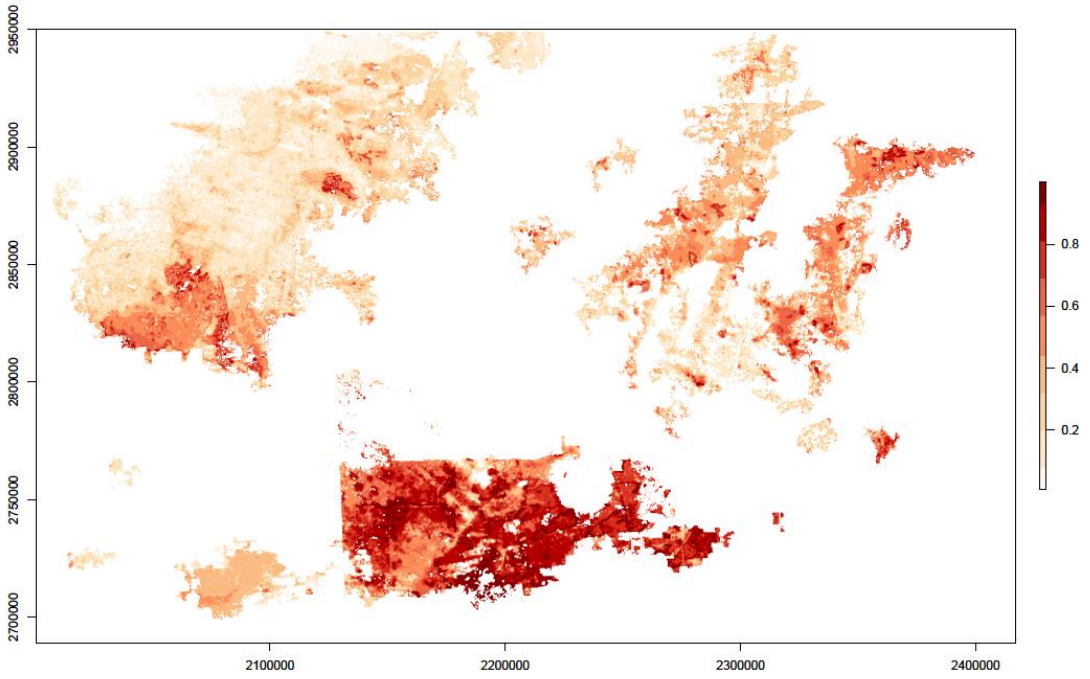


Figure S4.4.19: Relative likelihood of occurrence (cloglog function) of *Strophurus intermedius* in relation to M1 predictor variables

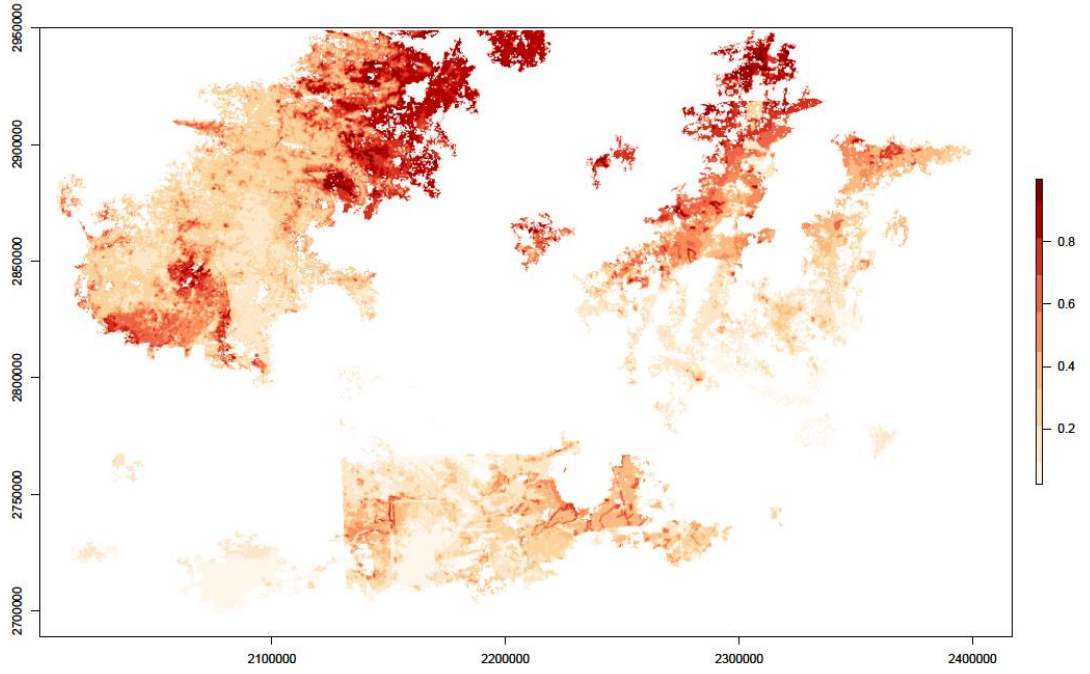


Figure S4.4.20: Relative likelihood of occurrence (cloglog function) of *Strophurus williamsi* in relation to M1 predictor variables

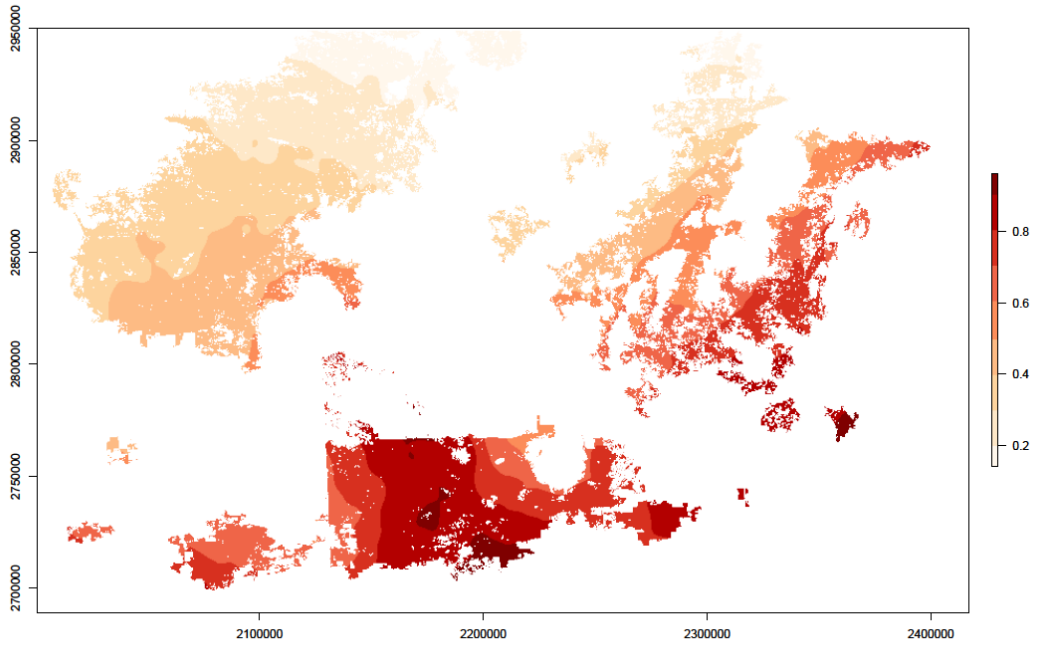


Figure S4.4.21: Relative likelihood of occurrence (cloglog function) of *Underwoodisaurus milii* in relation to M1 predictor variables

Goanna (Varanidae)

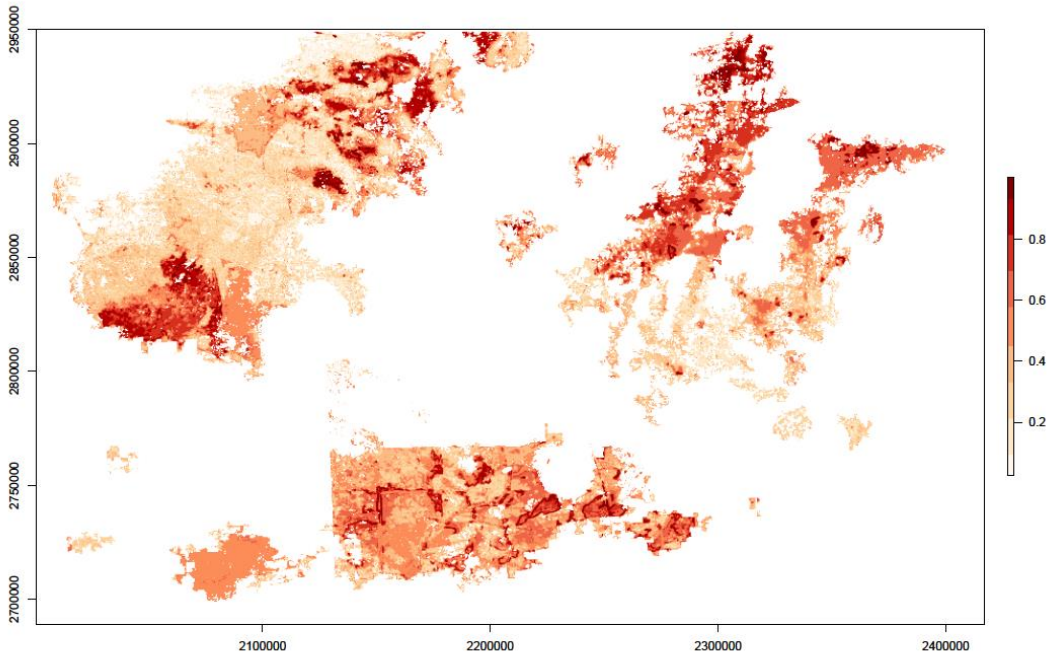


Figure S4.4.22: Relative likelihood of occurrence (cloglog function) of *Varanus gouldii* in relation to M1 predictor variables

Legless lizards (Pygopodidae)

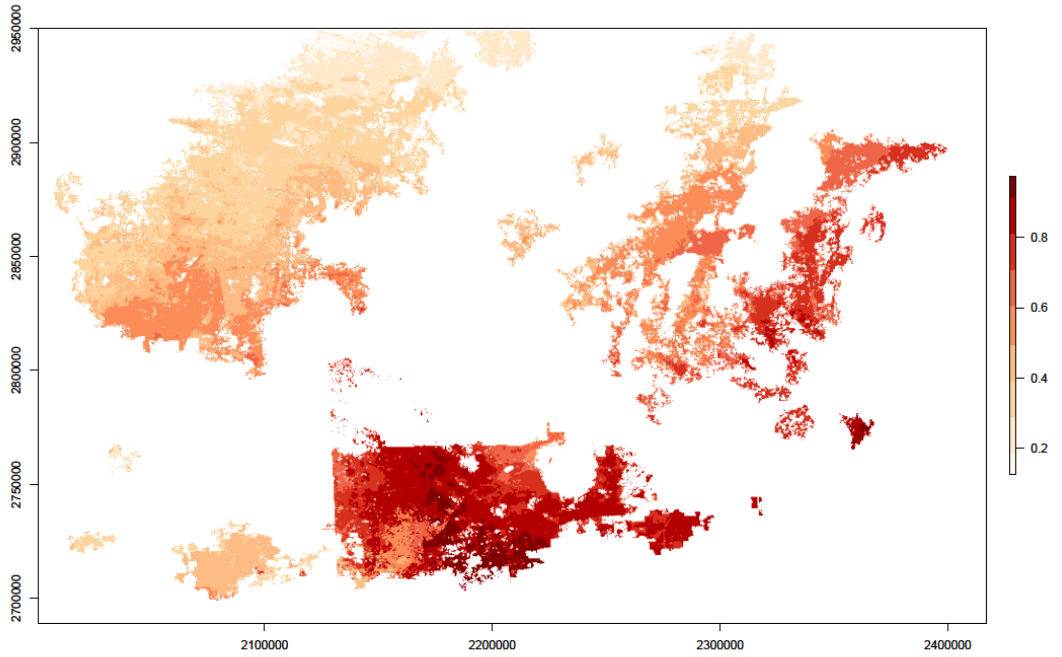


Figure S4.4.23: Relative likelihood of occurrence (cloglog function) of *Aprasia inaurita* in relation to M1 predictor variables

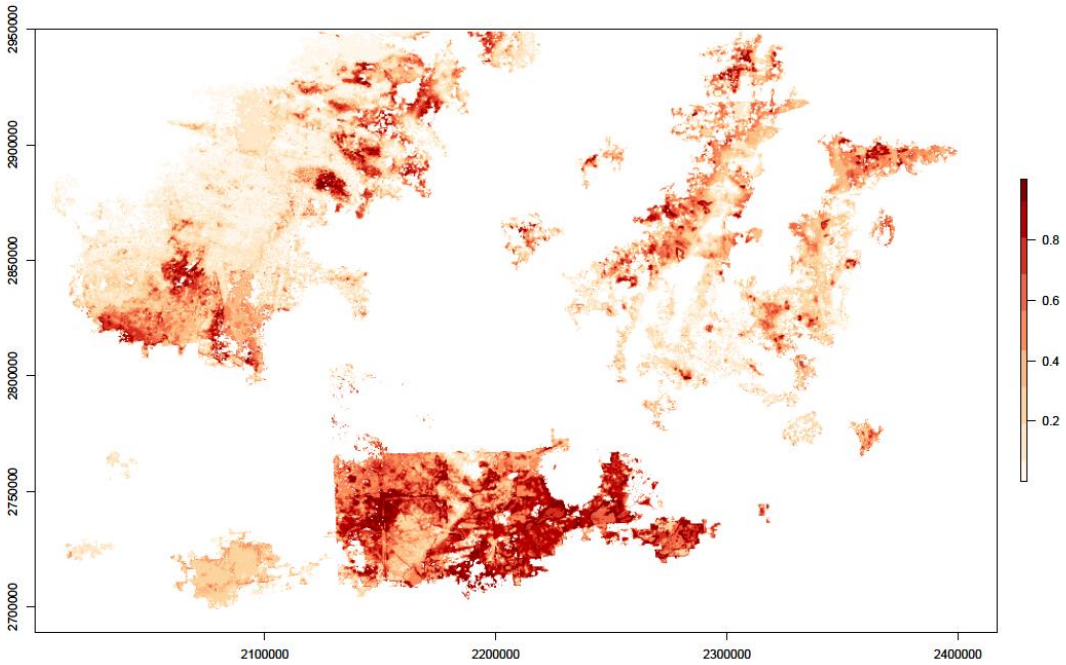


Figure S4.4.24: Relative likelihood of occurrence (cloglog function) of *Delma australis* in relation to M1 predictor variables

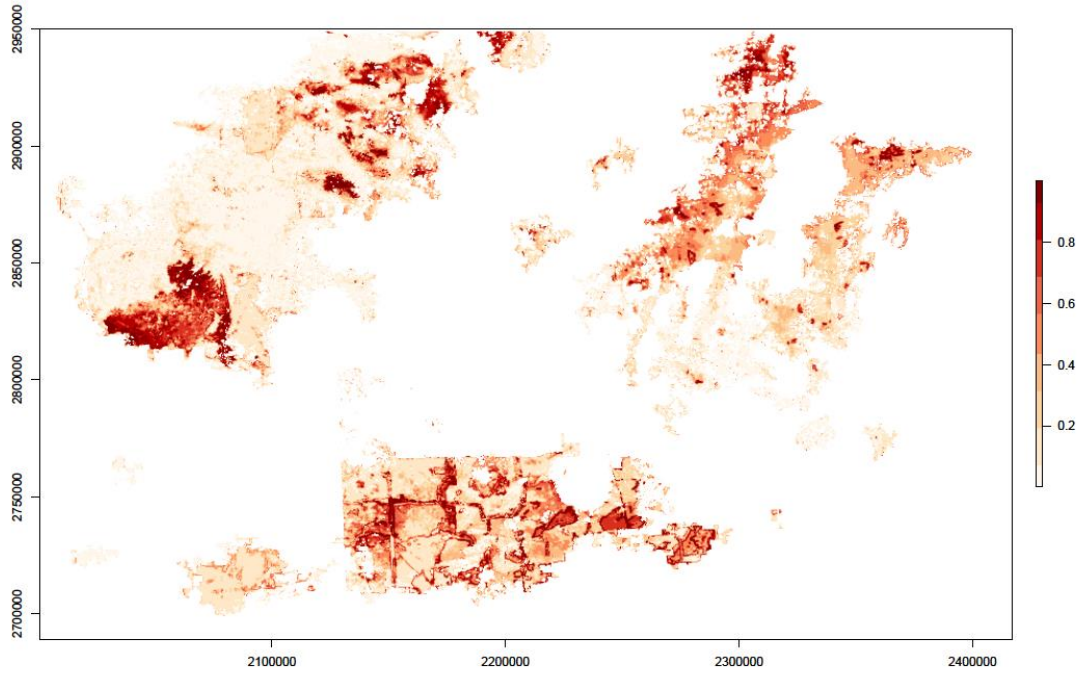


Figure S4.4.25: Relative likelihood of occurrence (cloglog function) of *Delma butleri* in relation to M1 predictor variables

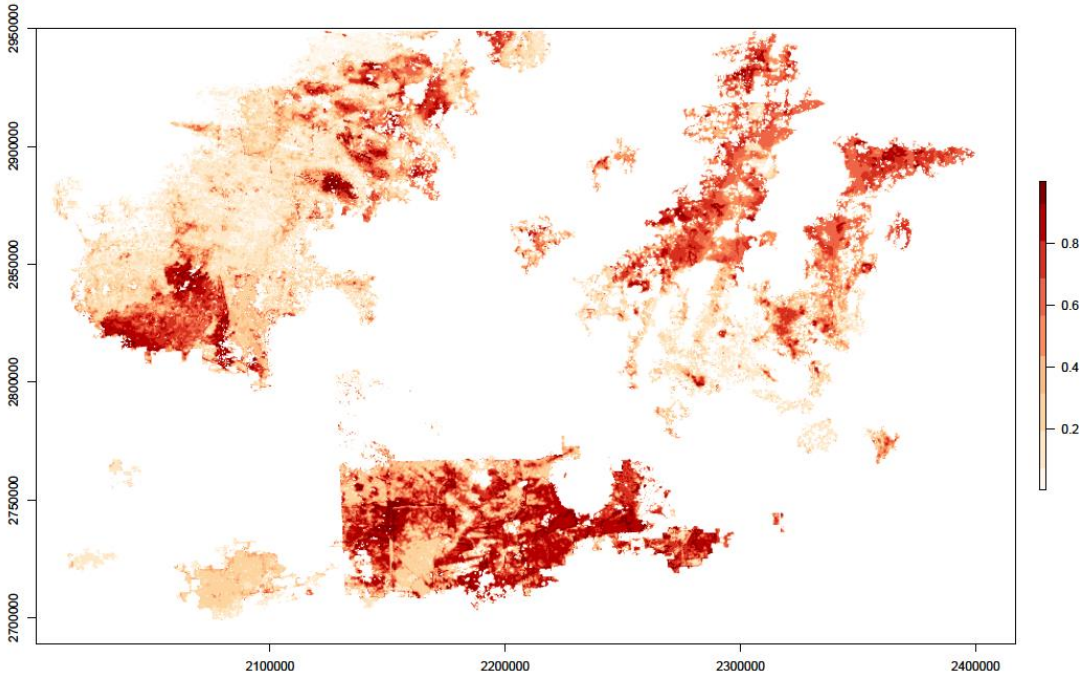


Figure S4.4.26: Relative likelihood of occurrence (cloglog function) of *Lialis burtonis* in relation to M1 predictor variables

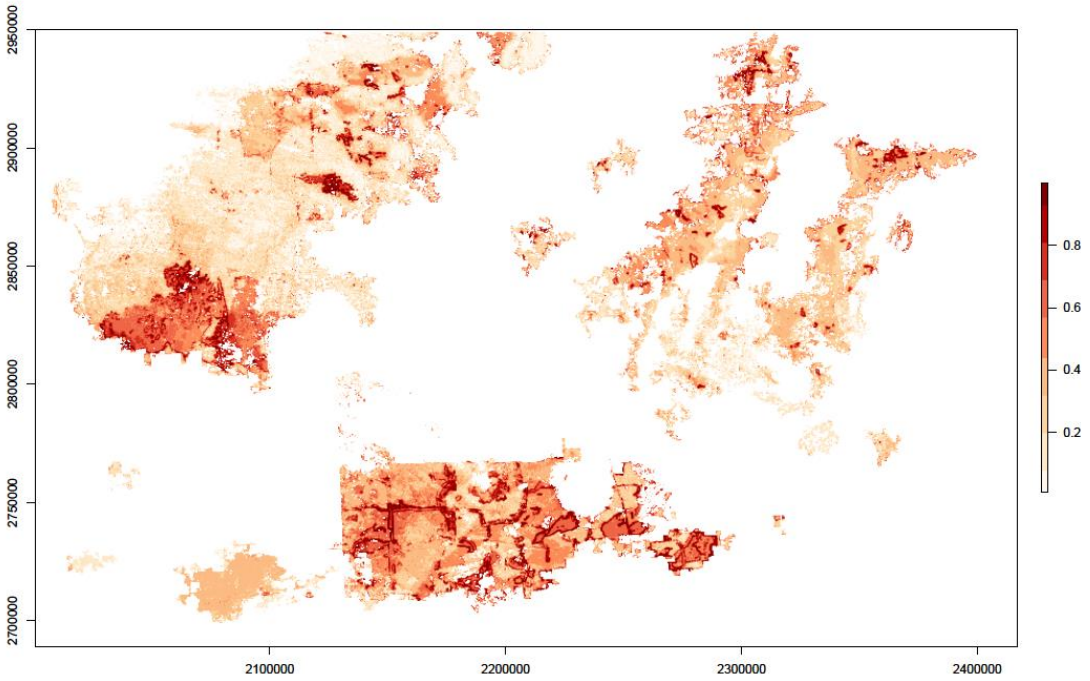


Figure S4.4.27: Relative likelihood of occurrence (cloglog function) of *Pygopus lepidopodus* in relation to M1 predictor variables

Skinks (Scincidae)

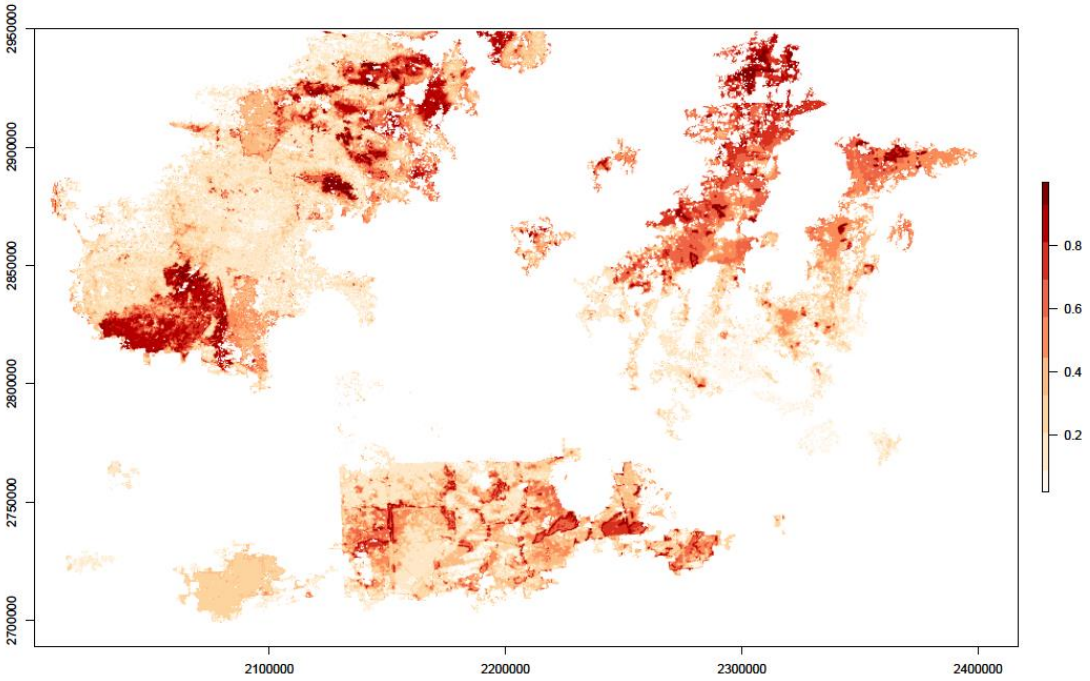


Figure S4.4.28: Relative likelihood of occurrence (cloglog function) of *Ctenotus atlas* in relation to M1 predictor variables

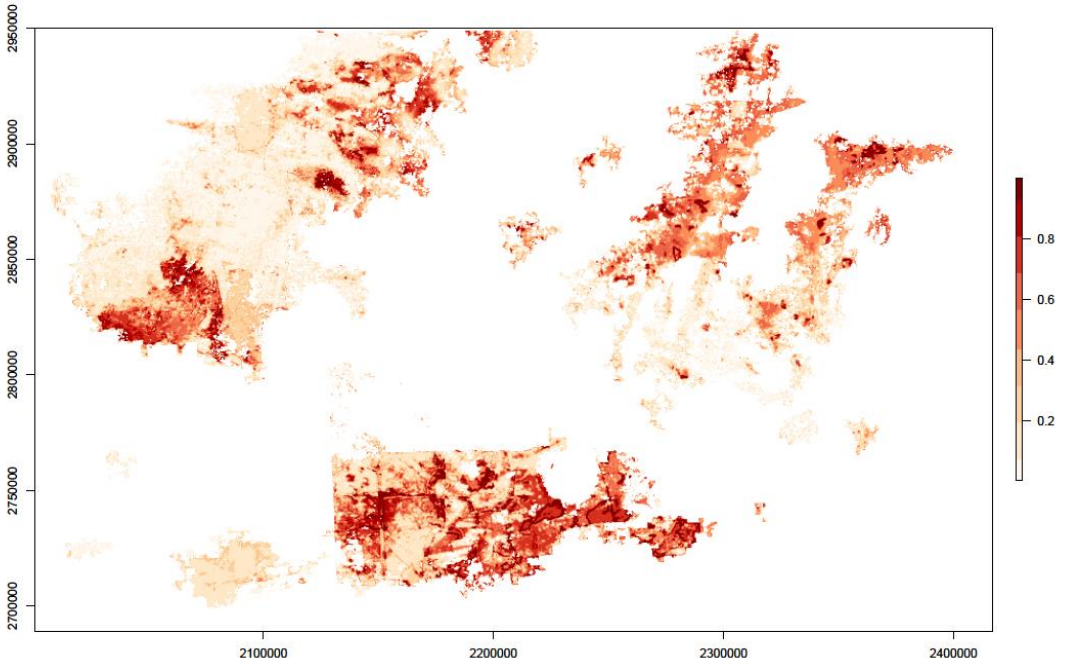


Figure S4.4.29: Relative likelihood of occurrence (cloglog function) of *Ctenotus brachyonyx* in relation to M1 predictor variables

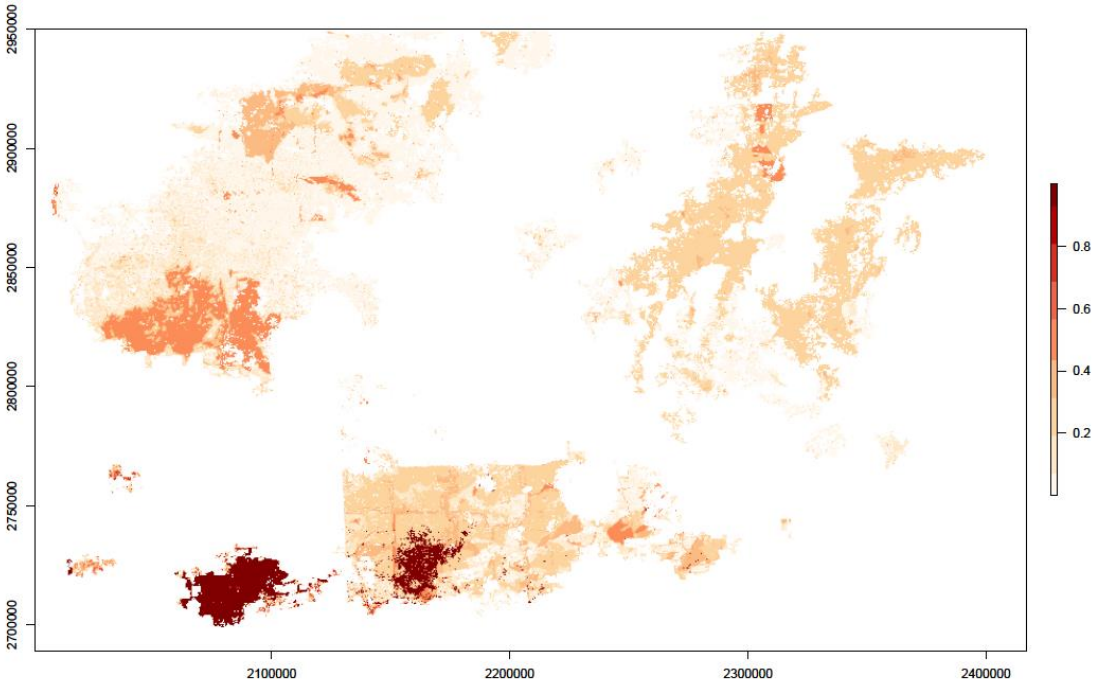


Figure S4.4.30: Relative likelihood of occurrence (cloglog function) of *Ctenotus brooksi iridis* in relation to M1 predictor variables

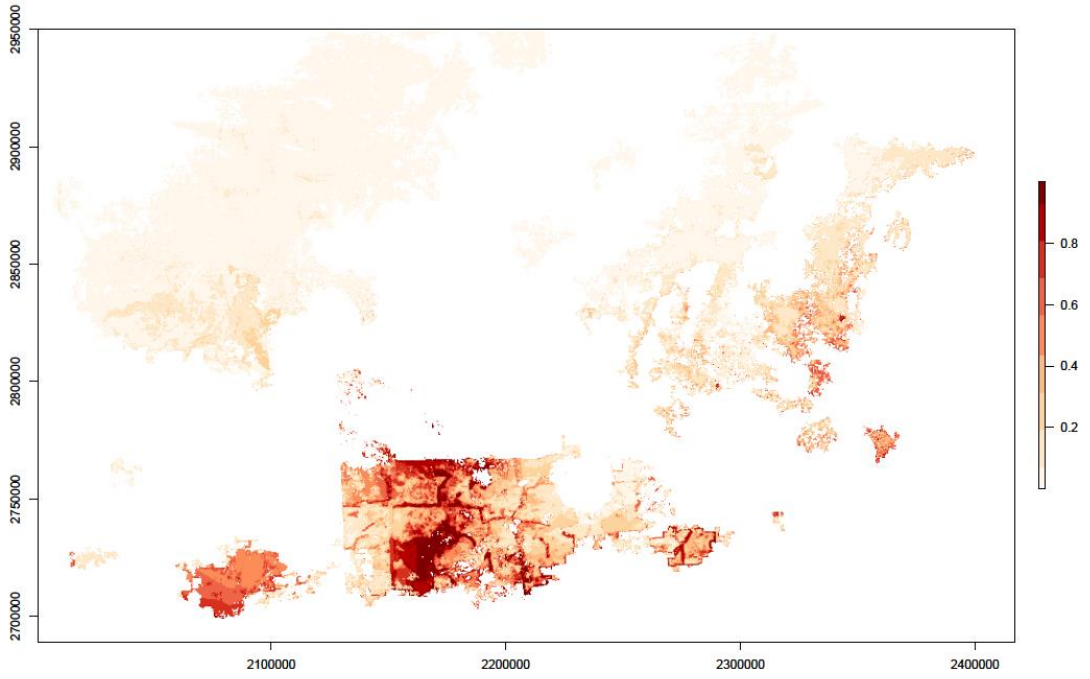


Figure S4.4.31: Relative likelihood of occurrence (cloglog function) of *Ctenotus orientalis* in relation to M1 predictor variables

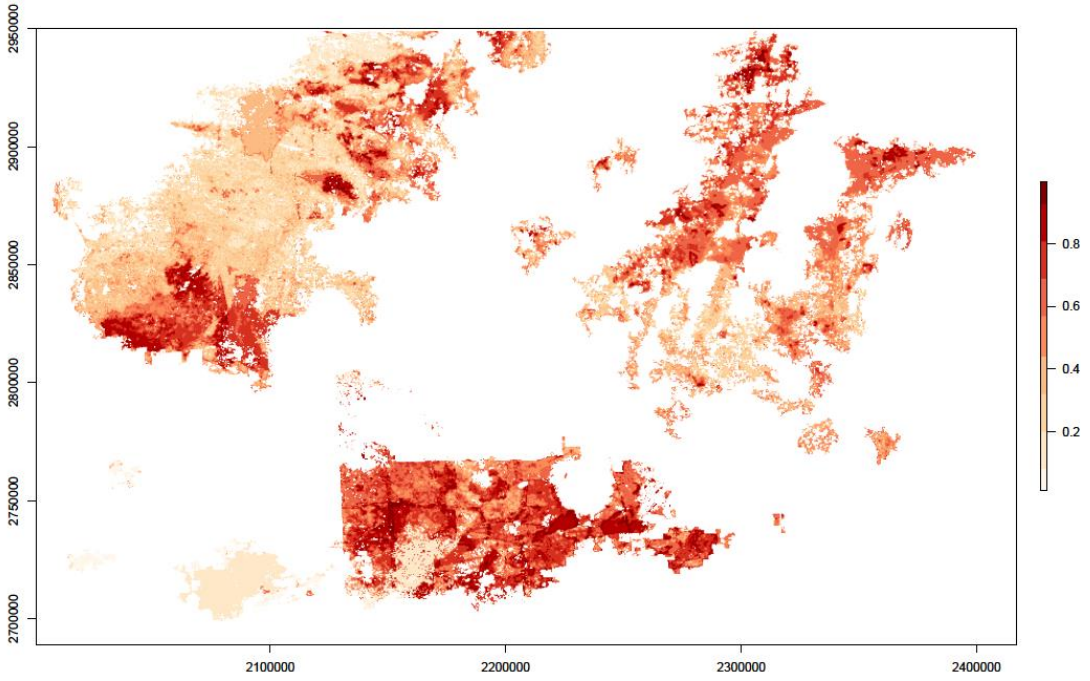


Figure S4.4.32: Relative likelihood of occurrence (cloglog function) of *Ctenotus regius* in relation to M1 predictor variables

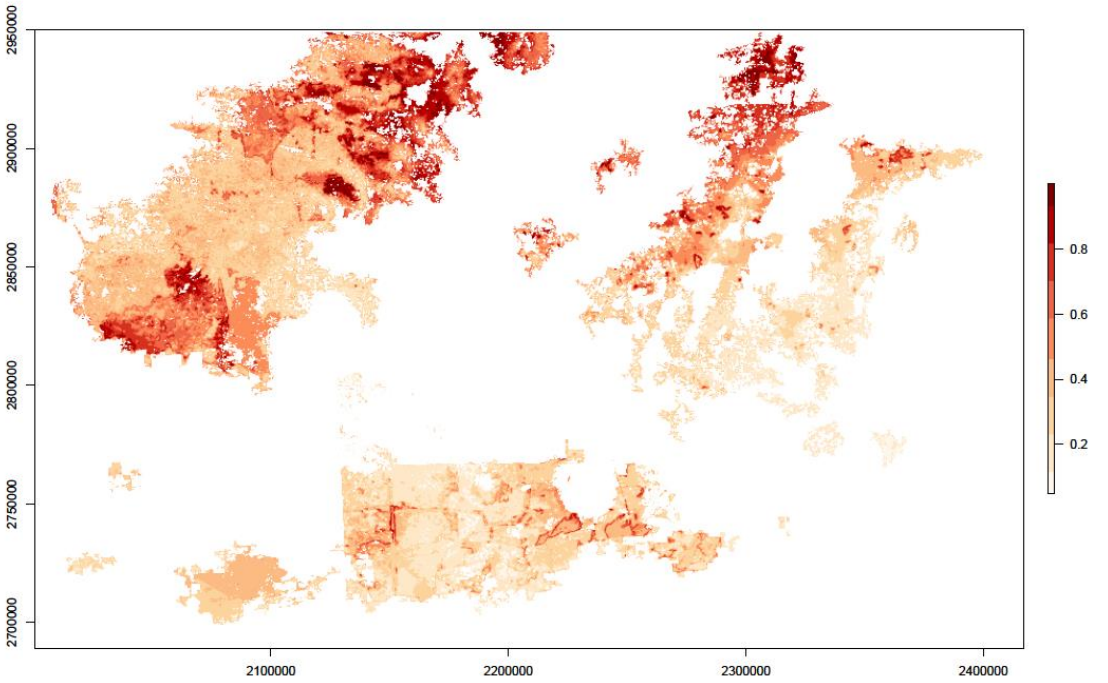


Figure S4.4.33: Relative likelihood of occurrence (cloglog function) of *Ctenotus schomburgkii* in relation to M1 predictor variables

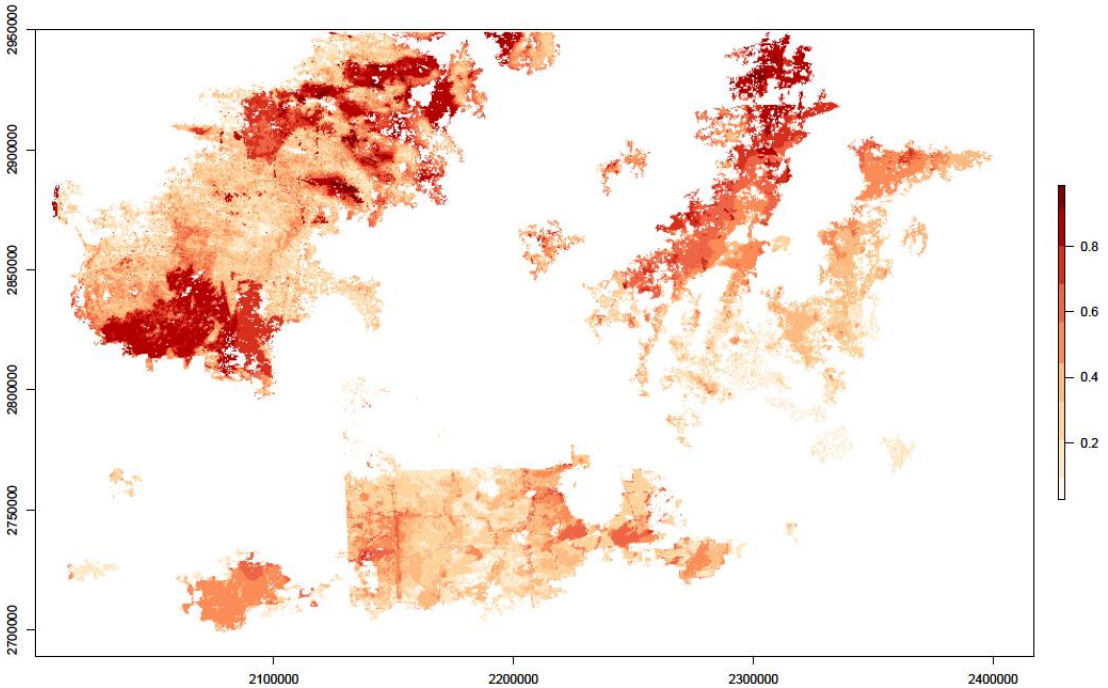


Figure S4.4.34: Relative likelihood of occurrence (cloglog function) of *Cyclodomorphus melanops* in relation to M1 predictor variables

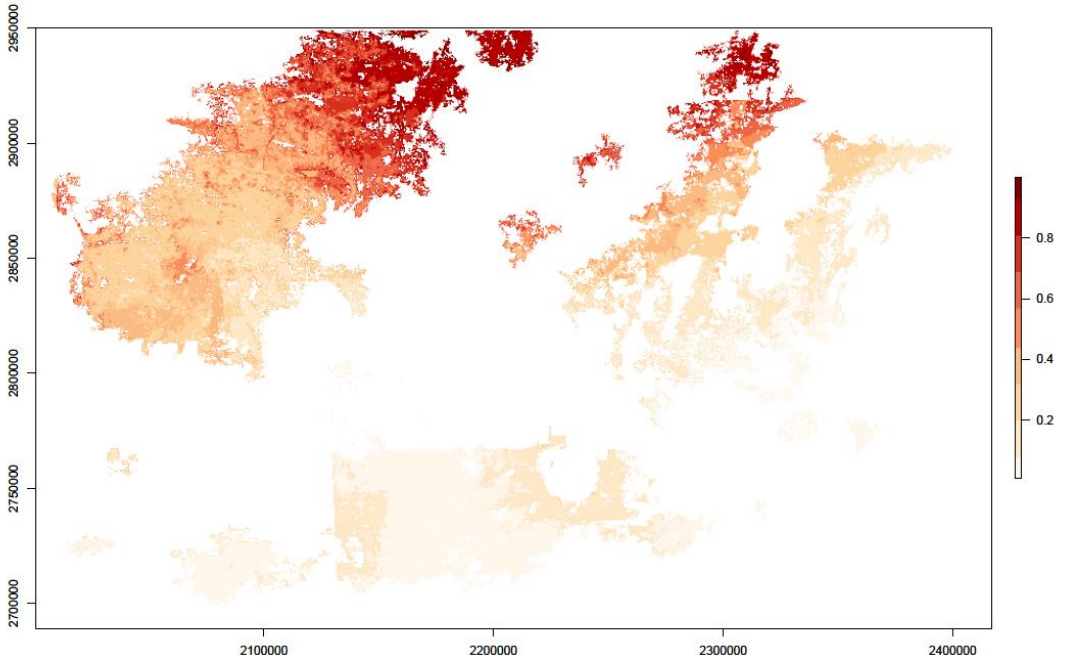


Figure S4.4.35: Relative likelihood of occurrence (cloglog function) of *Eremiascincus richardsonii* in relation to M1 predictor variables

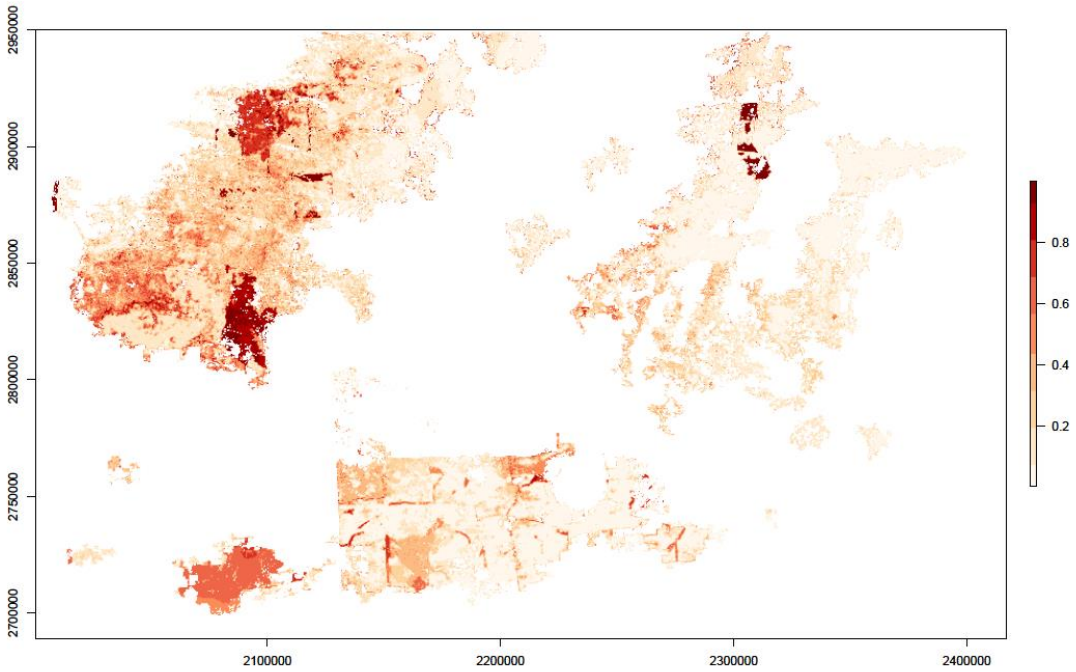


Figure S4.4.36: Relative likelihood of occurrence (cloglog function) of *Lerista aericeps* in relation to M1 predictor variables

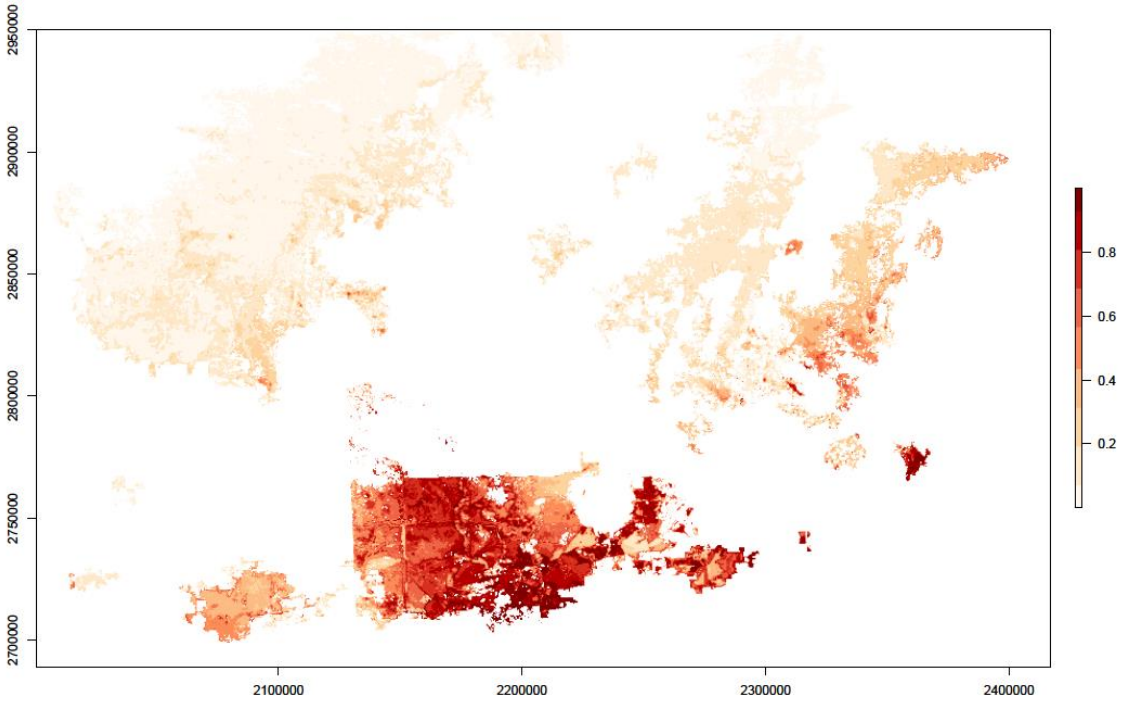


Figure S4.4.37: Relative likelihood of occurrence (cloglog function) of *Lerista bougainvillii* in relation to M1 predictor variables

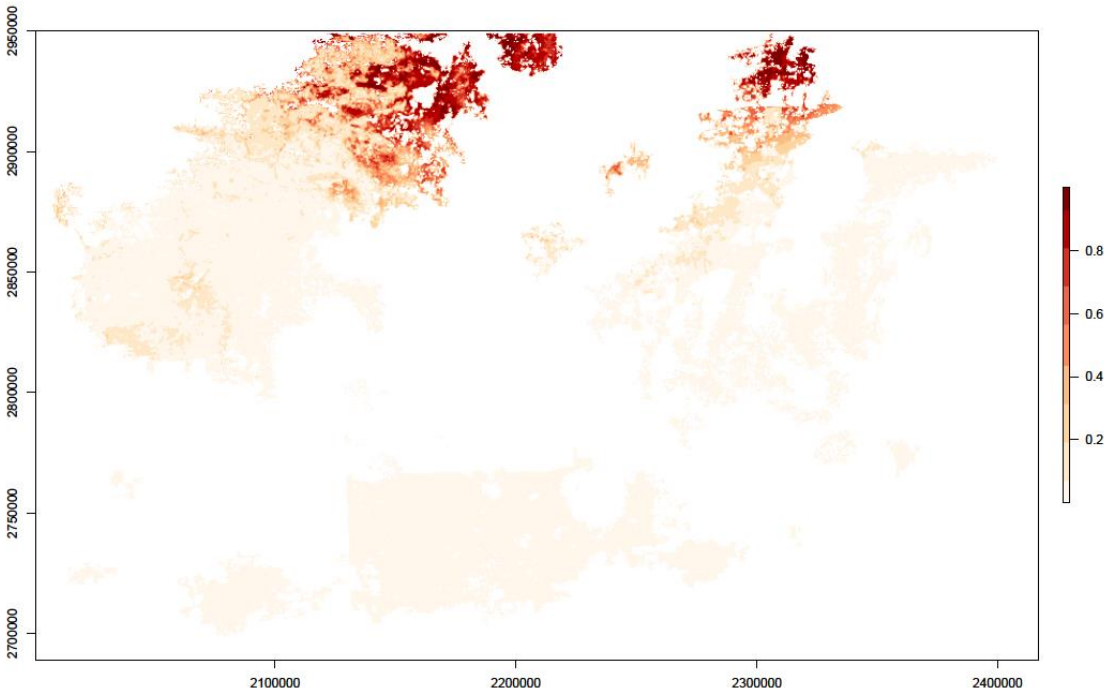


Figure S4.4.38: Relative likelihood of occurrence (cloglog function) of *Lerista labialis* in relation to M1 predictor variables

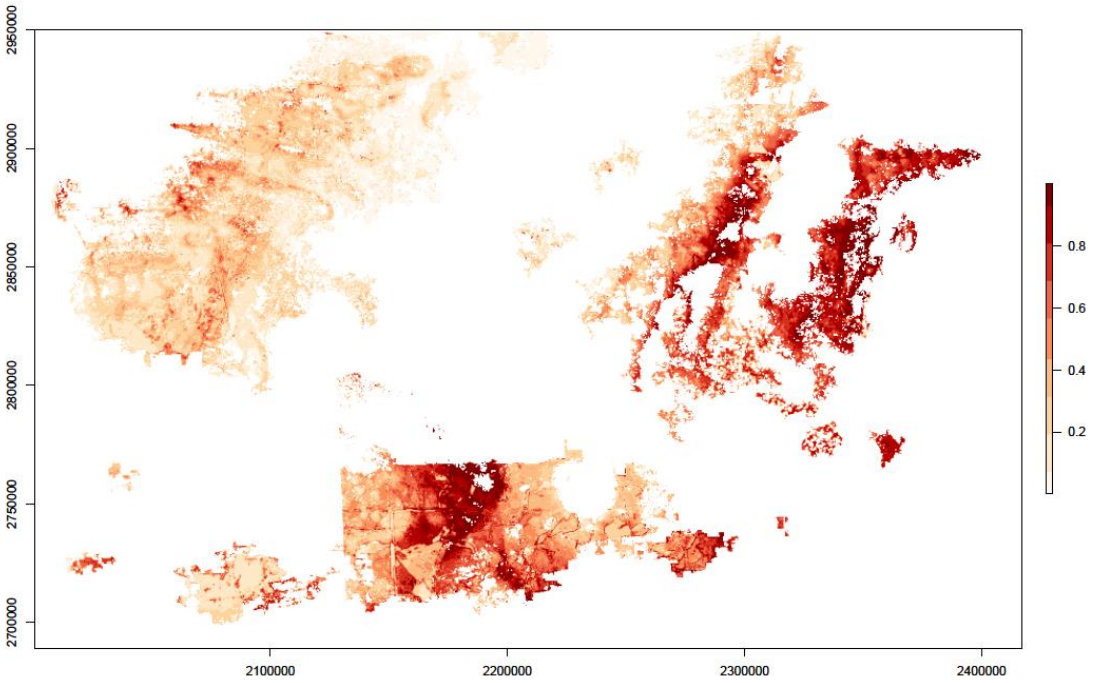


Figure S4.4.39: Relative likelihood of occurrence (cloglog function) of *Lerista muelleri* in relation to M1 predictor variables

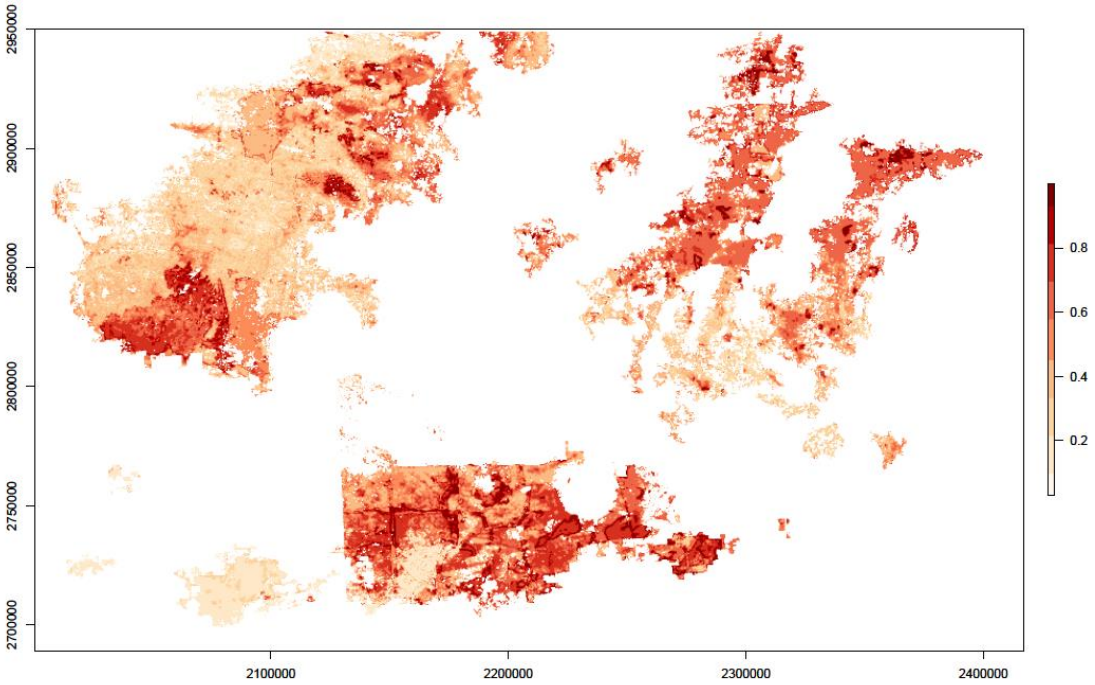


Figure S4.4.40: Relative likelihood of occurrence (cloglog function) of *Lerista punctatovittata* in relation to M1 predictor variables

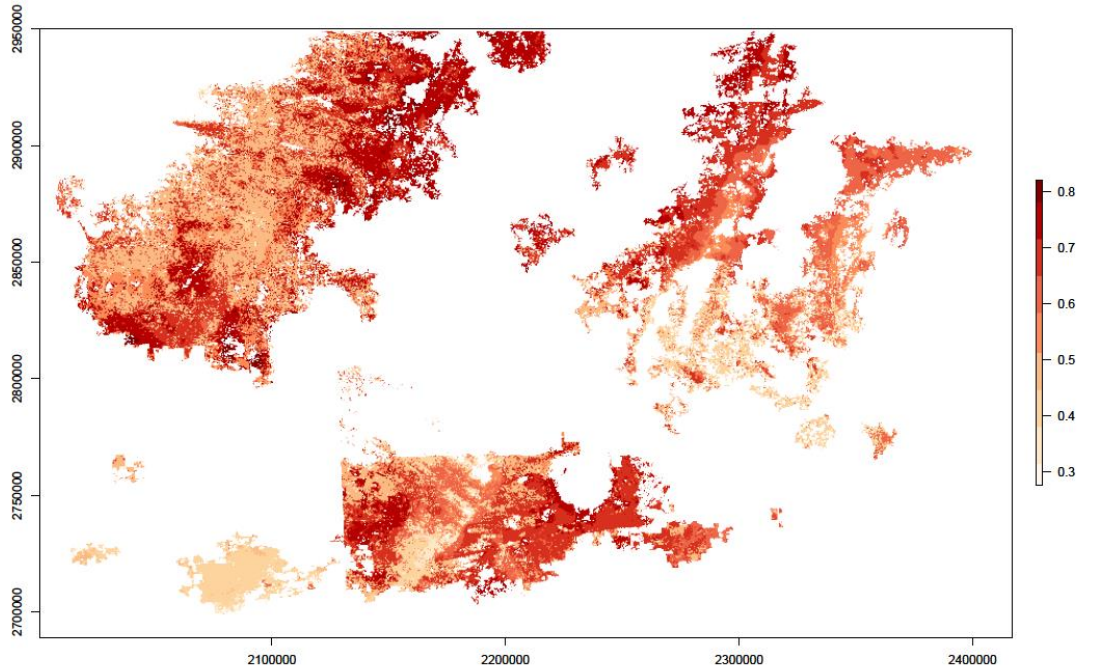


Figure S4.4.41: Relative likelihood of occurrence (cloglog function) of *Lerista xanthura* in relation to M1 predictor variables

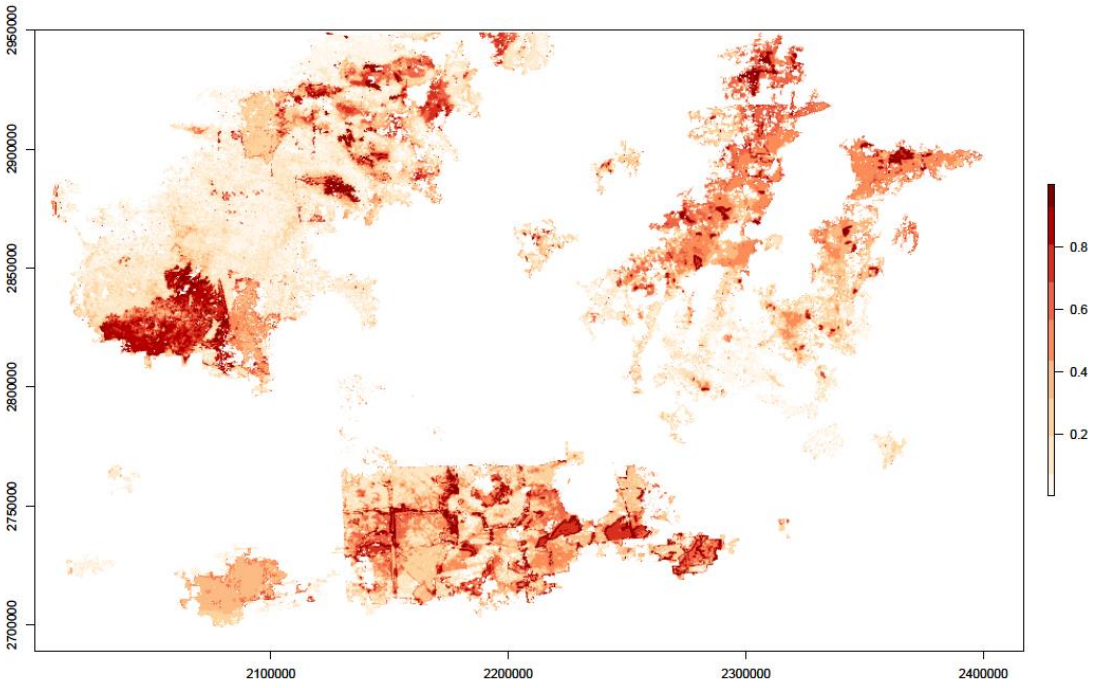


Figure S4.4.42: Relative likelihood of occurrence (cloglog function) of *Liopholis inornata* in relation to M1 predictor variables

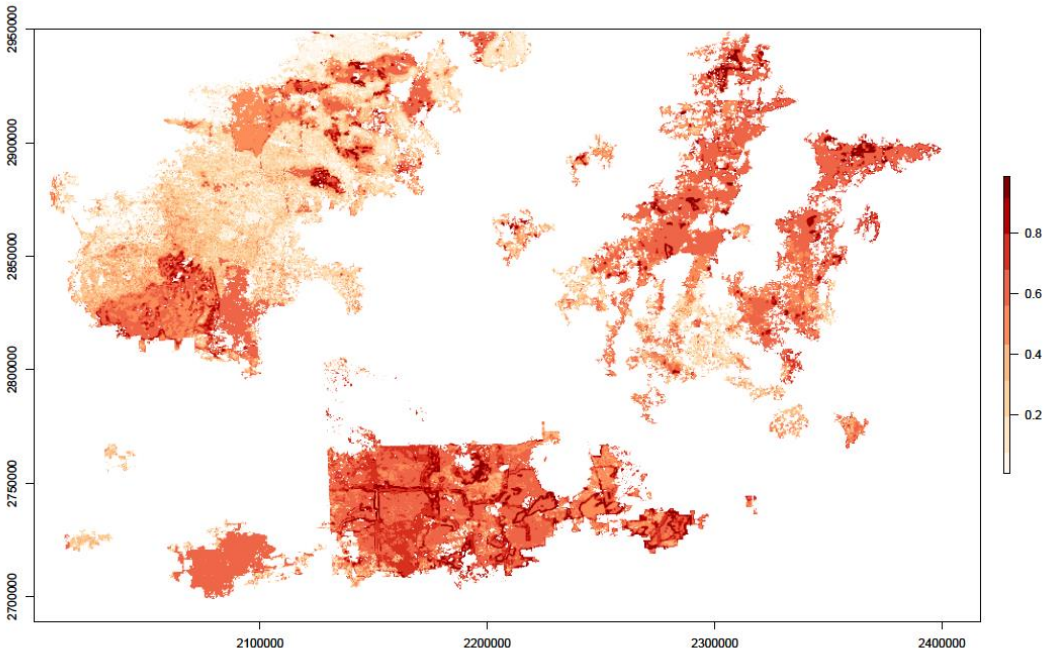


Figure S4.4.43: Relative likelihood of occurrence (cloglog function) of *Menetia greyii* in relation to M1 predictor variables

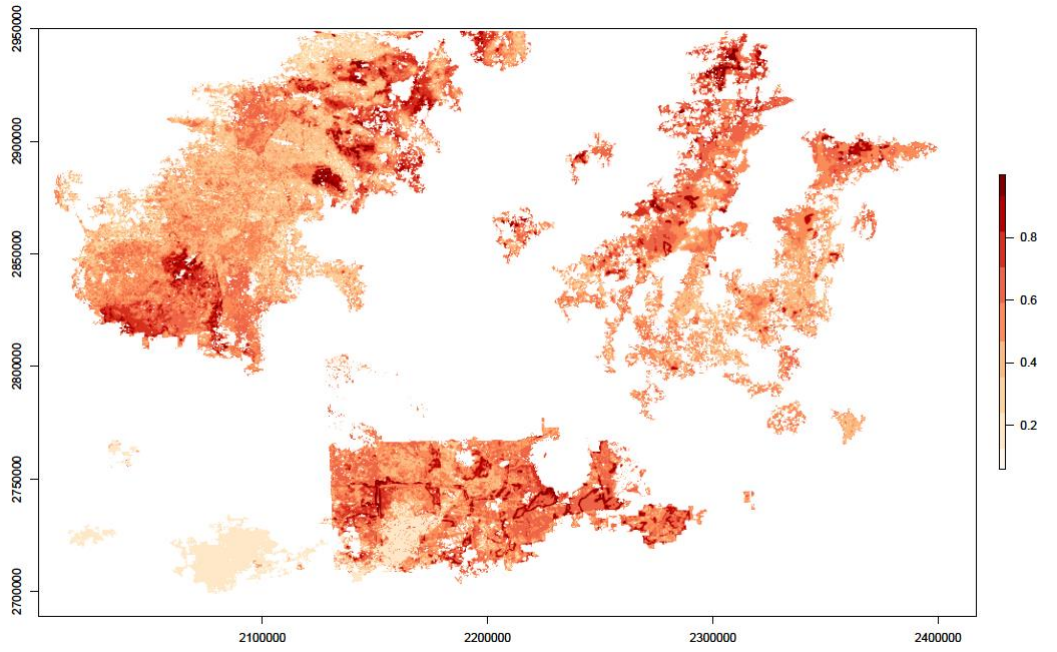


Figure S4.4.44: Relative likelihood of occurrence (cloglog function) of *Morethia boulengeri* in relation to M1 predictor variables

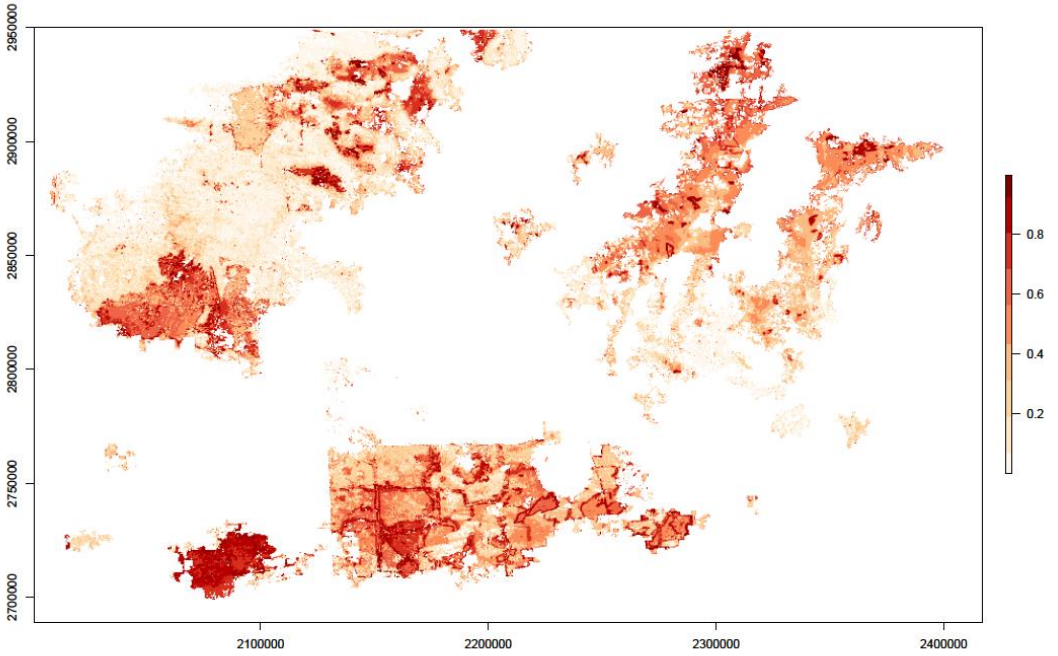


Figure S4.4.45: Relative likelihood of occurrence (cloglog function) of *Morethia obscura* in relation to M1 predictor variables

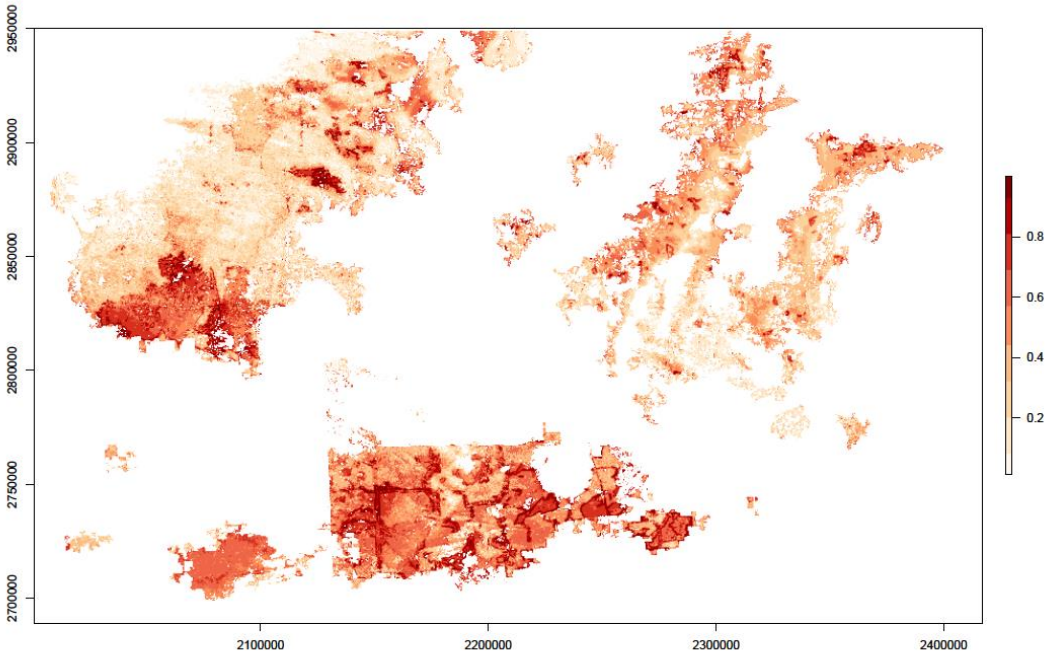


Figure S4.4.46: Relative likelihood of occurrence (cloglog function) of *Tiliqua occipitalis* in relation to M1 predictor variables

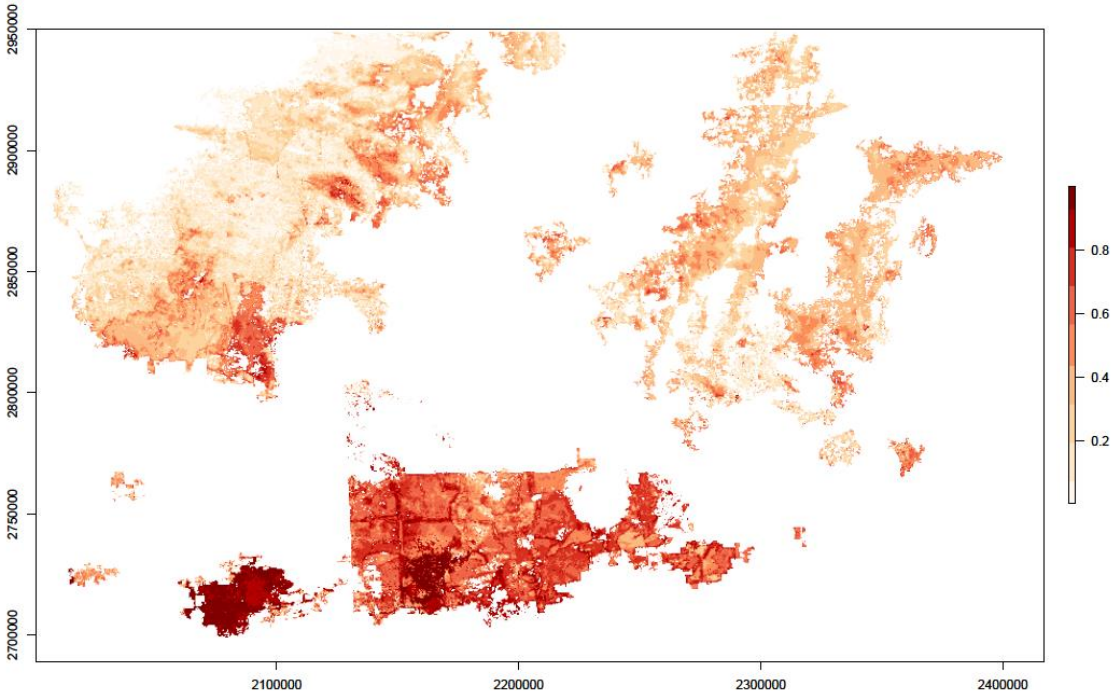


Figure S4.4.47: Relative likelihood of occurrence (cloglog function) of *Tiliqua rugosa* in relation to M1 predictor variables

Appendices – Chapter 5

Appendix S5.1 – Mammals and reptiles recorded during field surveys after the planned fire

Table S5.1.1: Species recorded during field surveys after the planned fire, and included in univariate GLMs to explore the influence of spatial patterns of fire on mammal and reptile populations

Taxonomic group	Family	Scientific name	Common name	Individual species model	Included in species richness model
Mammal	Canidae	*Vulpes vulpes*	Red fox	Y	N
Mammal	Burramyidae	*Cercartetus lepidus*	Little pygmy possum	Y	Y
Mammal	Dasyuridae	*Sminthopsis murina*	Common dunnart	Y	Y
Mammal	Muridae	*Mus musculus*	House mouse	Y	N
Mammal	Muridae	*Notomys mitchellii*	Mitchell's hopping mouse	N	Y
Reptile	Agamidae	*Ctenophorus fordi*	Mallee dragon	Y	Y
Reptile	Agamidae	*Ctenophorus pictus*	Painted dragon	N	Y
Reptile	Agamidae	*Diporiphora nobbi*	Nobbi dragon	Y	Y
Reptile	Agamidae	*Pogona vitticeps*	Central bearded dragon	Y	Y
Reptile	Diplodactylidae	*Diplodactylus vittatus*	Wood gecko	Y	Y
Reptile	Diplodactylidae	*Lucasium damaeum*	Beaded gecko	Y	Y
Reptile	Diplodactylidae	*Strophurus intermedius*	Southern spiny-tailed gecko	Y	Y
Reptile	Elapidae	*Brachyurophis australis*	Australian coral snake	Y	Y
Reptile	Elapidae	*Parasuta nigriceps*	Mitchell's short-tailed snake	N	Y
Reptile	Elapidae	*Vermicella annulata*	Bandy-bandy	N	Y
Reptile	Pygopodidae	*Aprasia inaurita*	Mallee worm-lizard	N	Y
Reptile	Pygopodidae	*Delma australis*	Marble-faced Delma	N	Y
Reptile	Pygopodidae	*Lialis burtonis*	Burton's snake-lizard	N	Y
Reptile	Scincidae	*Ctenotus brachyonyx*	Bar-Shouldered Ctenotus	Y	Y
Reptile	Scincidae	*Lerista bougainvillii*	Bougainville's skink	Y	Y
Reptile	Scincidae	*Lerista punctatovittata*	Eastern robust slider	N	Y
Reptile	Scincidae	*Liopholis inornata*	Desert skink	Y	Y
Reptile	Scincidae	*Menetia greyii*	Common dwarf skink	Y	Y

Reptile	Typhlopidae	*Anilios bicolor*	Dark-spined blind snake	N	Y

Appendix S5.2 – Model coefficients from GLMs

Table S5.2.1: Model coefficients ± standard error for univariate GLMs. Bold coefficients indicate where the 95% confidence intervals around that estimate did not overlap with zero. Grey cells indicate models where complete separation occurred, and reliable parameter estimates could not be calculated. sp. rich. = species richness

Species	Burn status	Distance	Amount of remnant veg.	LPI of remnant veg.	Aggregation of remnant veg.	Amount of *Triodia*
Cercartetus lepidus	0.34 ± 0.82	-0.72 ± 0.98	0.62 ± 0.83	0.63 ± 0.77	0.31 ± 0.85	-0.47 ± 1.07
Mus musculus	0.51 ± 0.34	-0.37 ± 0.37	-0.02 ± 0.35	-0.04 ± 0.35	0.42 ± 0.35	**0.56** ± **0.25**
Sminthopsis murina	0.93 ± 0.56	-0.86 ± 0.67	0.51 ± 0.54	0.94 ± 0.49	0.98 ± 0.60	-0.21 ± 0.61
Vulpes vulpes	0.21 ± 0.13	-0.14 ± 0.14	0.23 ± 0.13	0.21 ± 0.13	0.21 ± 0.13	0.08 ± 0.13
Small mammal sp. rich.	0.41 ± 0.58	-0.54 ± 0.66	0.29 ± 0.58	0.59 ± 0.54	0.40 ± 0.60	-0.10 ± 0.63
Brachyurophis australis	**0.70** ± **0.33**	-0.30 ± 0.35	0.48 ± 0.33	0.58 ± 0.31	0.63 ± 0.34	-0.17 ± 0.37
Ctenophorus fordi	0.51 ± 0.52	-0.14 ± 0.54	0.74 ± 0.53	**1.06** ± **0.47**	0.80 ± 0.56	0.17 ± 0.48
Ctenotus brachyonyx	0.79 ± 0.65	-1.40 ± 0.92	1.24 ± 0.66	1.13 ± 0.58	0.92 ± 0.69	0.40 ± 0.51
Diplodactylus vittatus	0.79 ± 0.65	-0.21 ± 0.67	0.72 ± 0.64	0.51 ± 0.60	0.66 ± 0.67	0.44 ± 0.50
Diporiphora nobbi	0.97 ± 0.56	-0.91 ± 0.67	**1.37** ± **0.57**	**1.29** ± **0.49**	1.12 ± 0.60	-1.07 ± 0.89
Lerista bougainvillii	0.13 ± 0.51	-0.73 ± 0.60	0.09 ± 0.51	0.09 ± 0.51	-0.06 ± 0.52	**-2.70** ± **1.09**
Liopholis inornata			**2.13** ± **0.91**	**1.93** ± **0.73**	4.69 ± 2.68	**0.99** ± **0.46**
Lucasium damaeum	0.46 ± 0.28	**-0.99** ± **0.36**	0.33 ± 0.28	0.37 ± 0.27	0.42 ± 0.29	-0.53 ± 0.38
Menetia greyii		0.40 ± 0.62	-1.46 ± 0.79	-1.67 ± 0.98	**-1.96** ± **0.88**	-2.74 ± 1.39
Pogona vitticeps	-0.02 ± 0.65	0.40 ± 0.62	0.009 ± 0.65	-0.18 ± 0.67	0.23 ± 0.65	-0.17 ± 0.71
Strophurus intermedius	**1.54** ± **0.51**	-1.02 ± 0.56	**1.01** ± **0.45**	0.81 ± 0.41	**1.61** ± **0.55**	0.39 ± 0.36
Reptile sp. rich.	**0.51** ± **0.17**	**-0.56** ± **0.20**	**0.47** ± **0.17**	**0.45** ± **0.16**	**0.52** ± **0.18**	-0.01 ± 0.18

Appendix S5.3 – Response plots of each species (or species richness) to the six fire variables

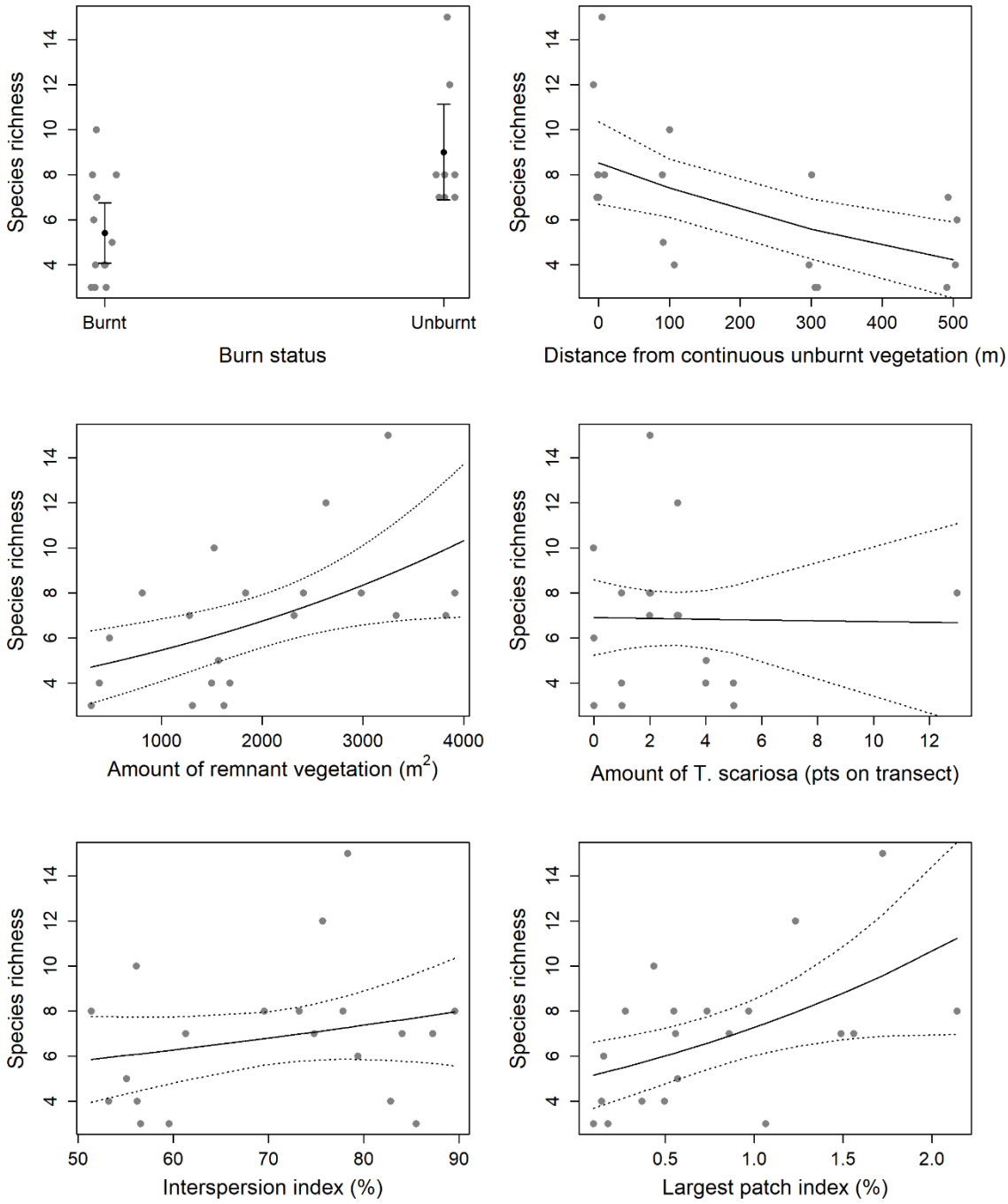


Figure S5.3.1: Predicted response of reptile species richness to the six fire variables used in univariate GLMs. Black lines (or dots, for the categorical variable of burn status) represent the predicted response of reptile species richness to each fire variable and dotted lines (or error bars, for the categorical variable of burn status) represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites.

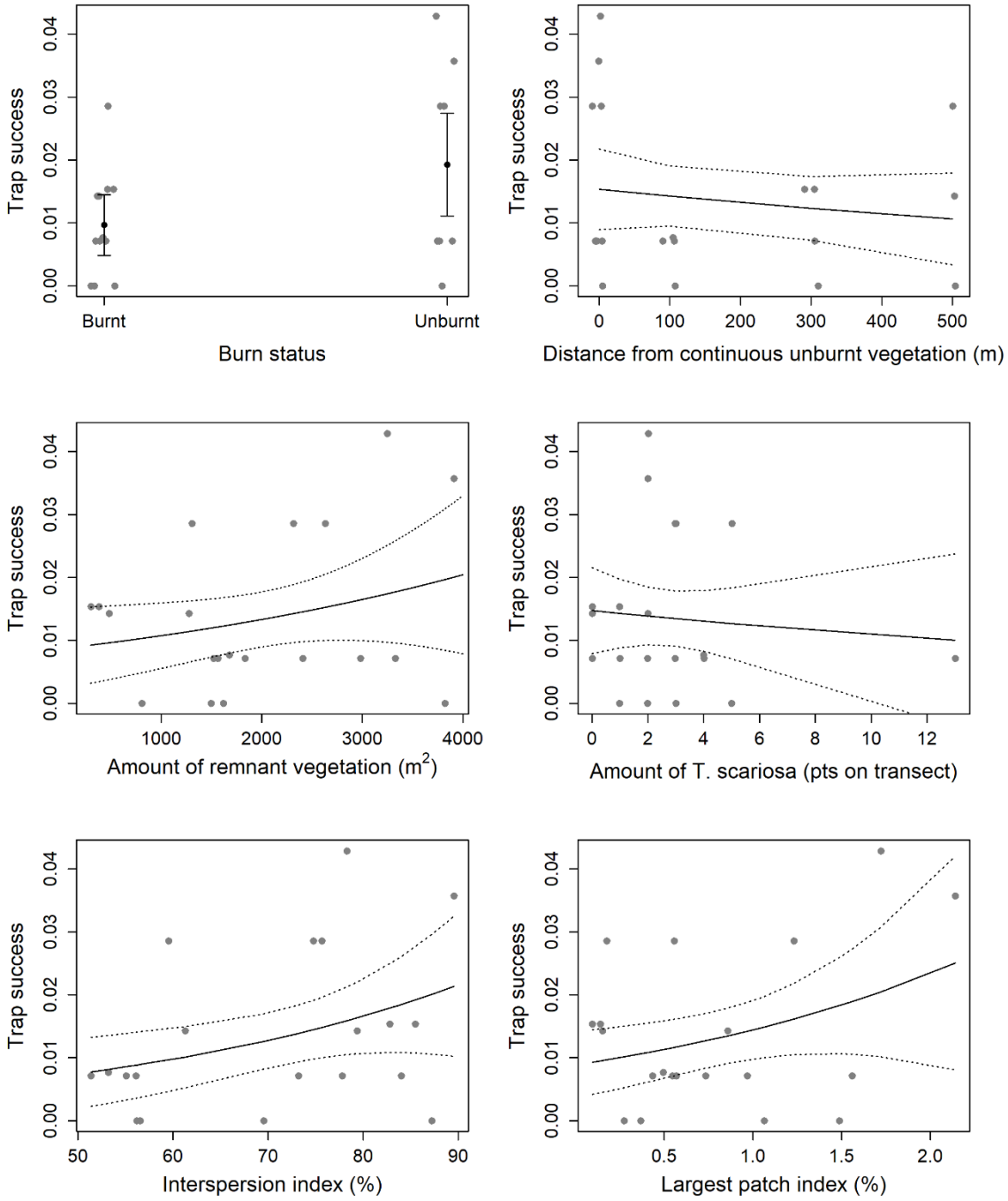


Figure S5.3.2: Predicted response of *Brachyurophis australis* capture rate (trap success) to the six fire variables used in univariate GLMs. Black lines (or dots, for the categorical variable of burn status) represent the predicted response of *B. australis* to each fire variable and dotted lines (or error bars, for the categorical variable of burn status) represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites.

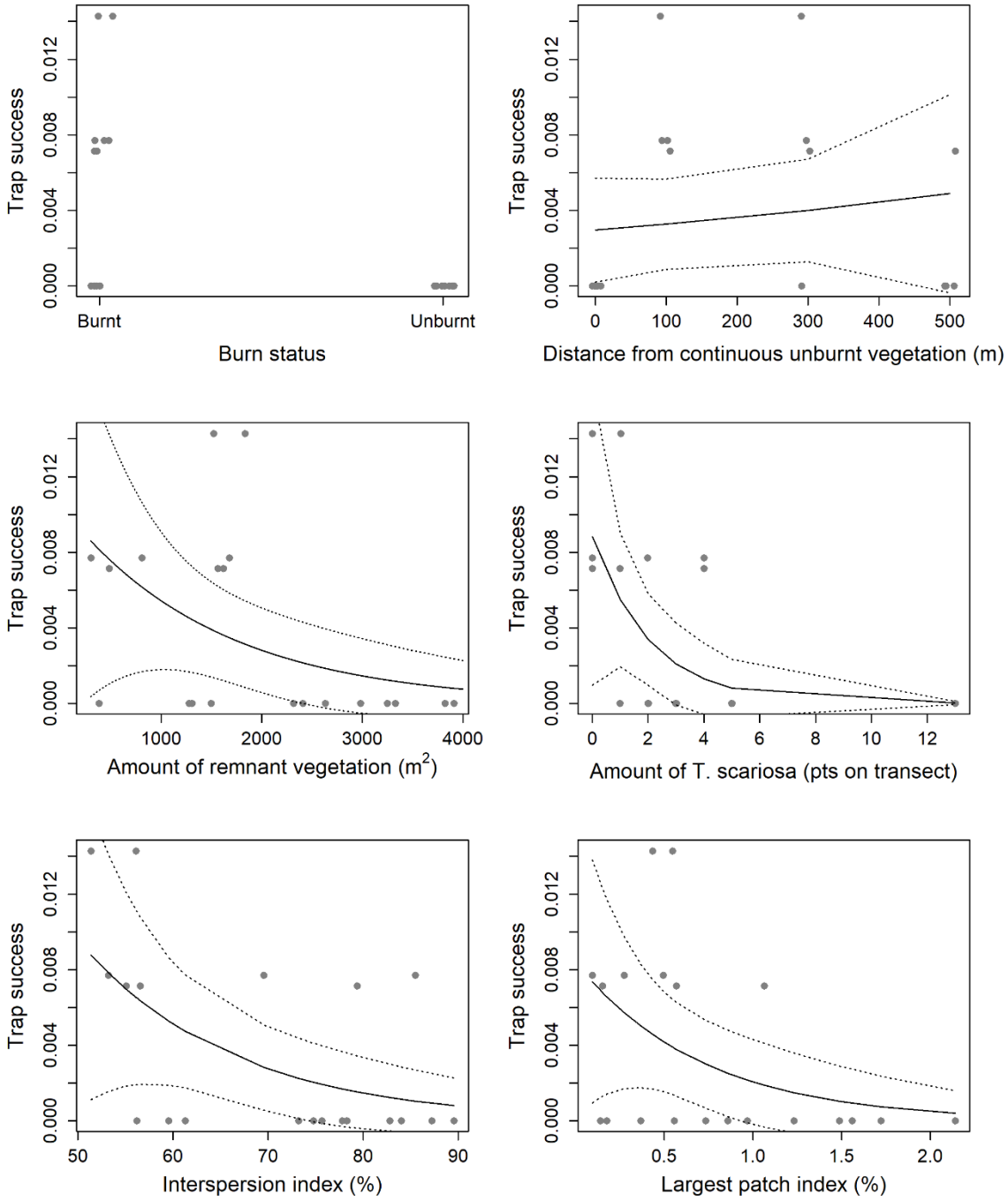


Figure S5.3.3: Predicted response of *Ctenotus brachyonyx* capture rate (trap success) to the six fire variables used in univariate GLMs. Black lines (or dots, for the categorical variable of burn status) represent the predicted response of *C. brachyonyx* to each fire variable and dotted lines (or error bars, for the categorical variable of burn status) represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites.

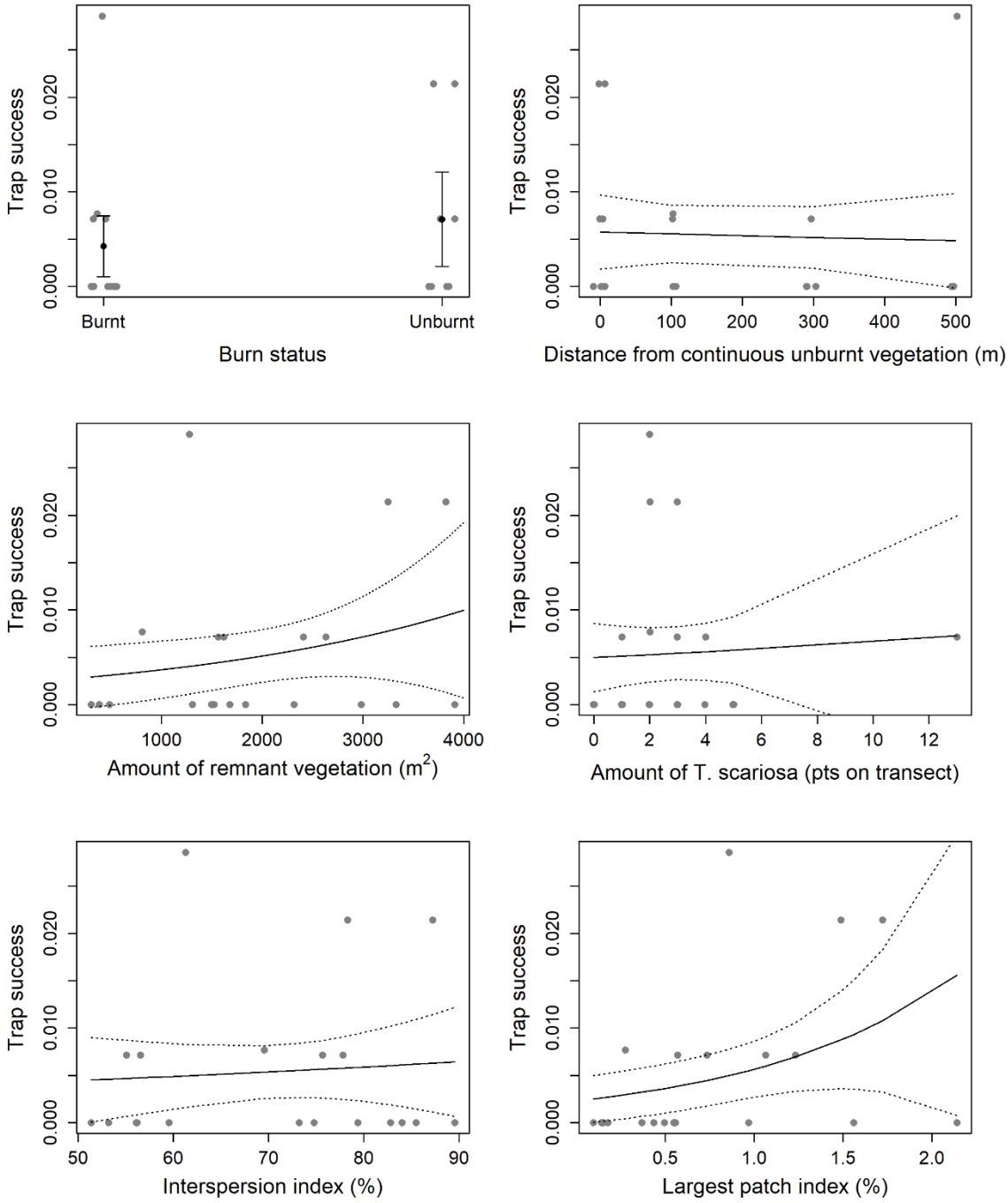


Figure S5.3.4: Predicted response of *Ctenophorus fordi* capture rate (trap success) to the six fire variables used in univariate GLMs. Black lines (or dots, for the categorical variable of burn status) represent the predicted response of *C. fordi* to each fire variable and dotted lines (or error bars, for the categorical variable of burn status) represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites.

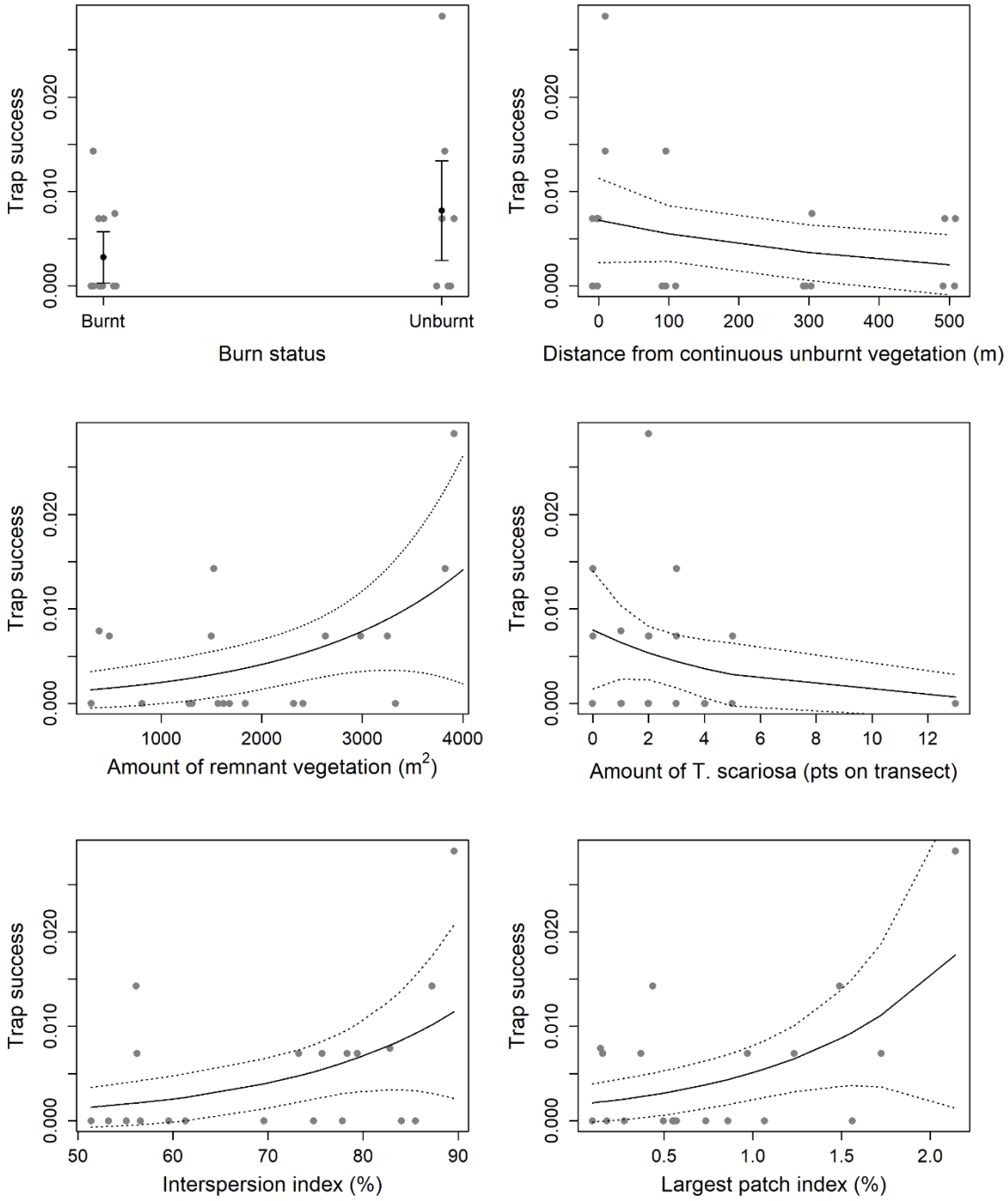


Figure S5.3.5: Predicted response of *Diporiphora nobbi* capture rate (trap success) to the six fire variables used in univariate GLMs. Black lines (or dots, for the categorical variable of burn status) represent the predicted response of *D. nobbi* to each fire variable and dotted lines (or error bars, for the categorical variable of burn status) represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites.

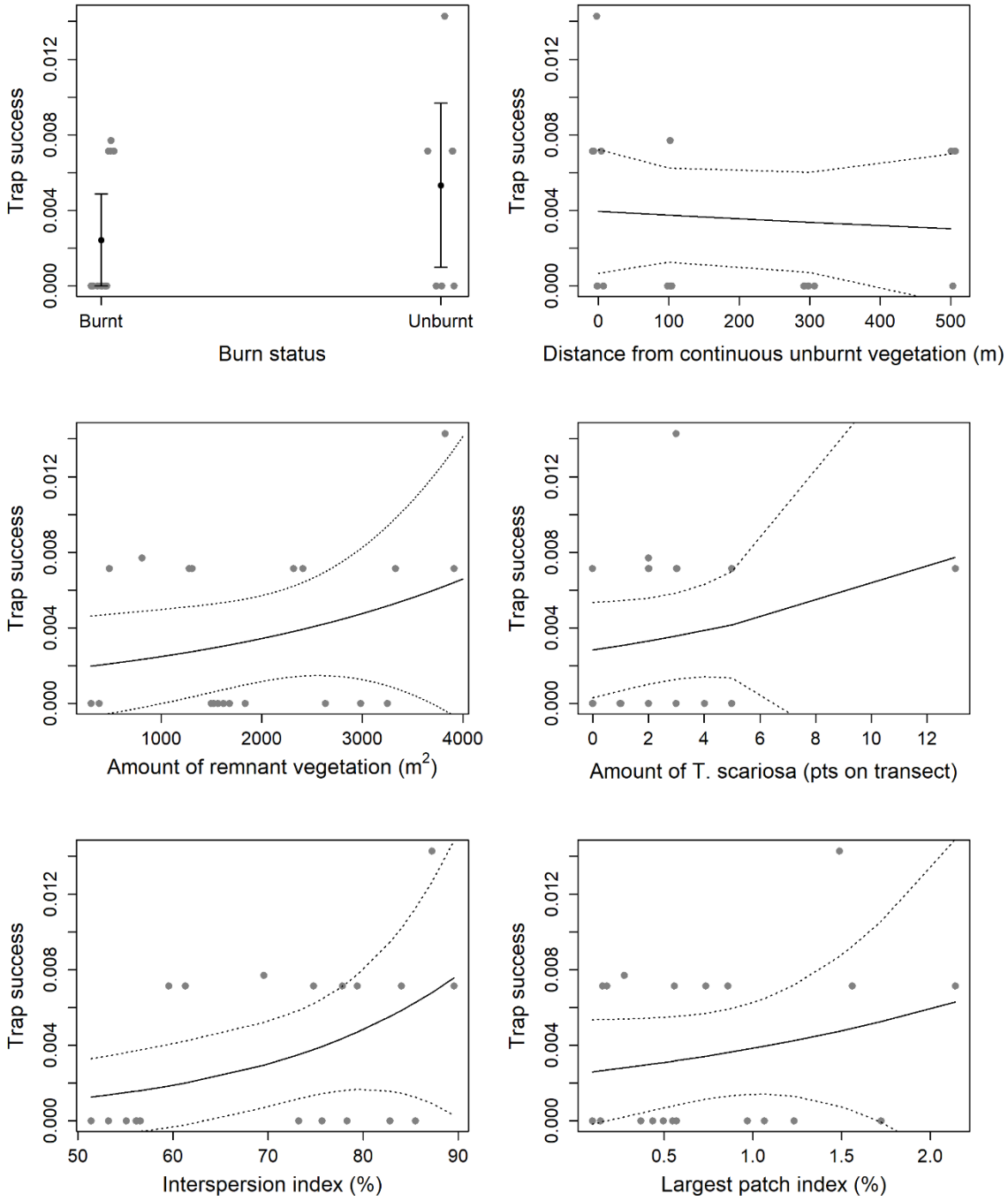


Figure S5.3.6: Predicted response of *Diplodactylus vittatus* capture rate (trap success) to the six fire variables used in univariate GLMs. Black lines (or dots, for the categorical variable of burn status) represent the predicted response of *D. vittatus* to each fire variable and dotted lines (or error bars, for the categorical variable of burn status) represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites.

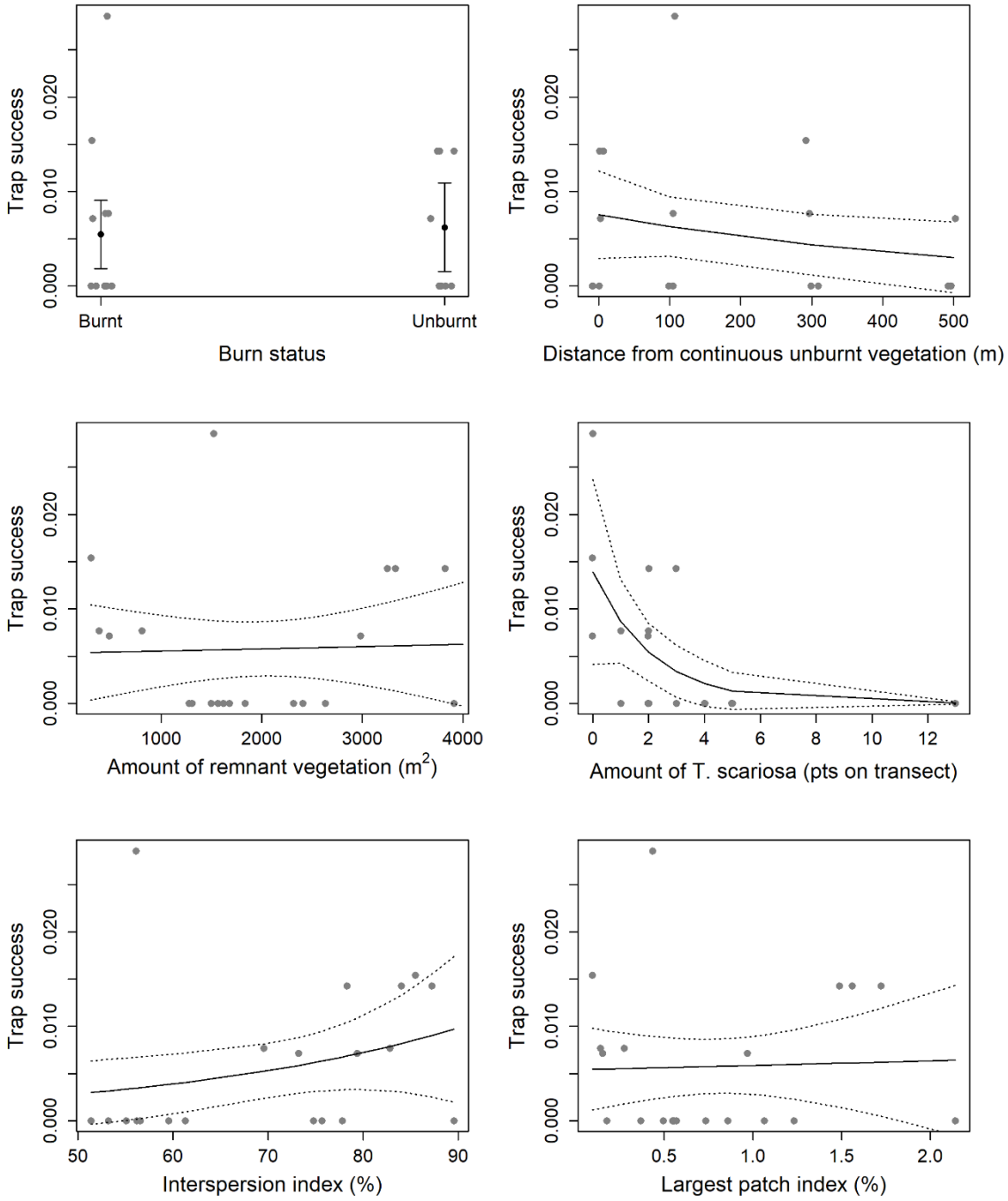


Figure S5.3.7: Predicted response of *Lerista bougainvillii* capture rate (trap success) to the six fire variables used in univariate GLMs. Black lines (or dots, for the categorical variable of burn status) represent the predicted response of *L. bougainvillii* to each fire variable and dotted lines (or error bars, for the categorical variable of burn status) represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites.

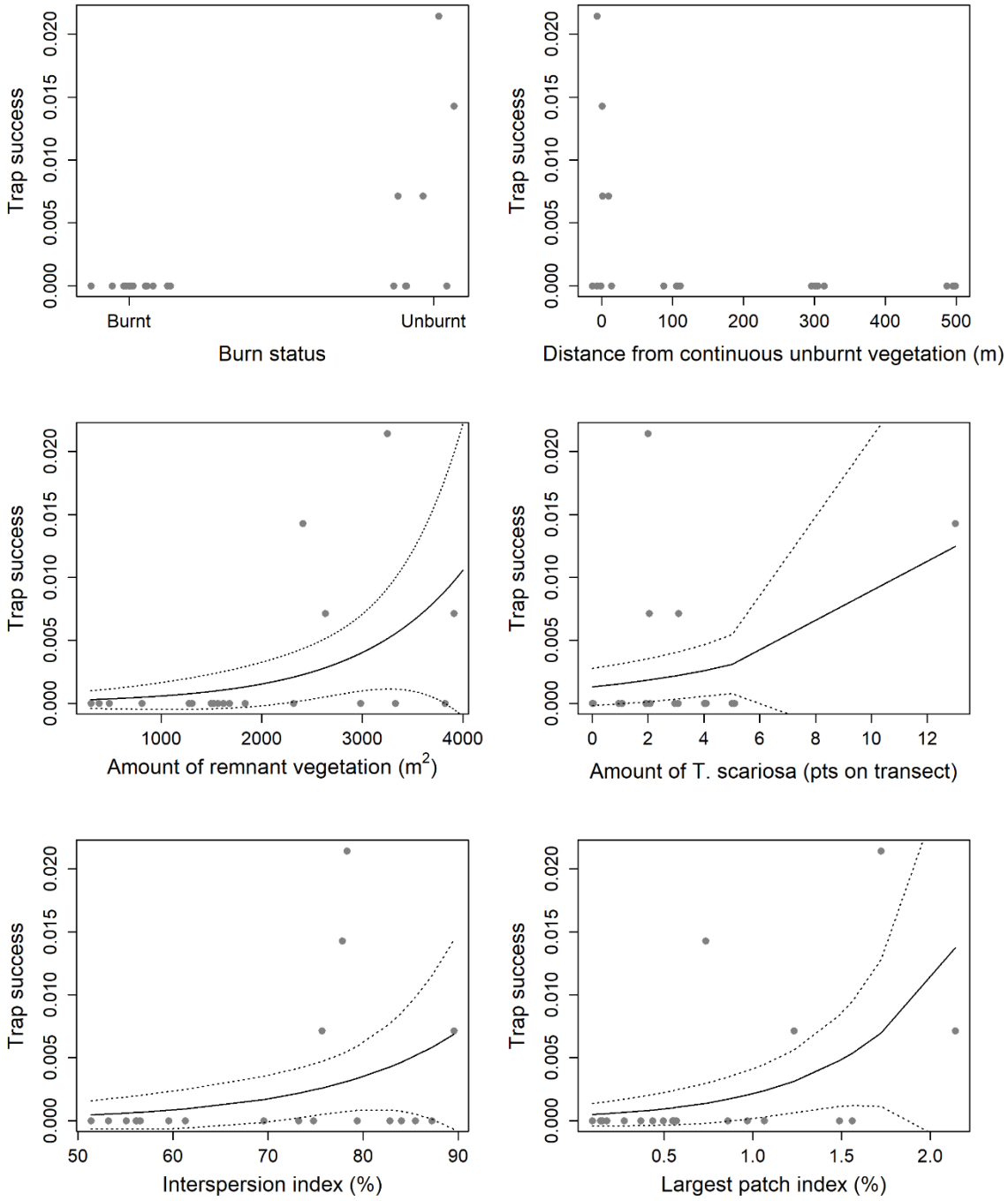


Figure S5.3.8: Predicted response of *Liopholis inornata* capture rate (trap success) to the six fire variables used in univariate GLMs. Black lines represent the predicted response of *D. vittatus* to each fire variable and dotted lines represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites. Predictions were not possible for burn status or distance, as complete separation of the predictor variables occurred, and reliable parameter estimates and predictions could not be computed.

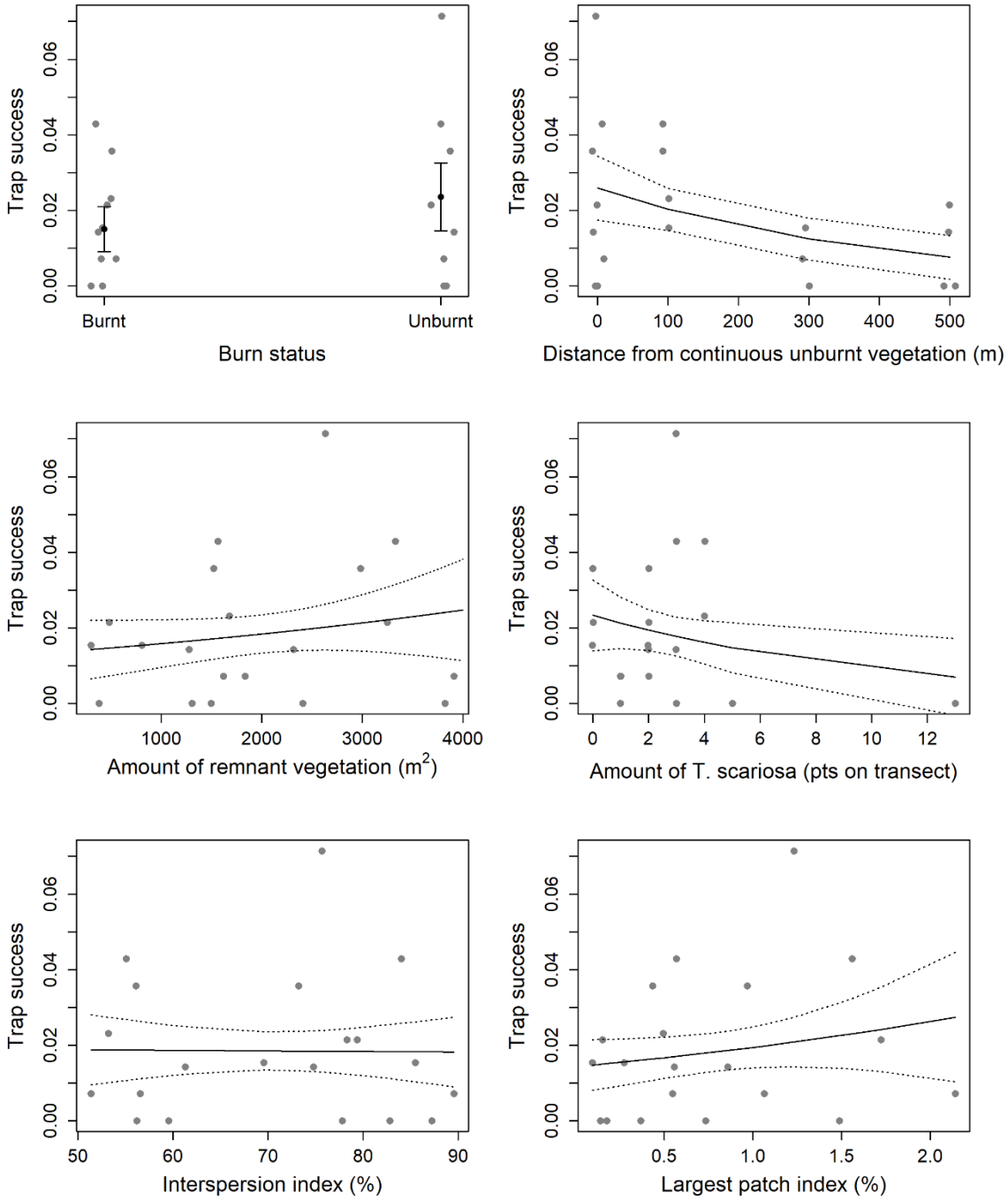


Figure S5.3.9: Predicted response of *Lucasium damaeum* capture rate (trap success) to the six fire variables used in univariate GLMs. Black lines (or dots, for the categorical variable of burn status) represent the predicted response of *L. damaeum* to each fire variable and dotted lines (or error bars, for the categorical variable of burn status) represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites.

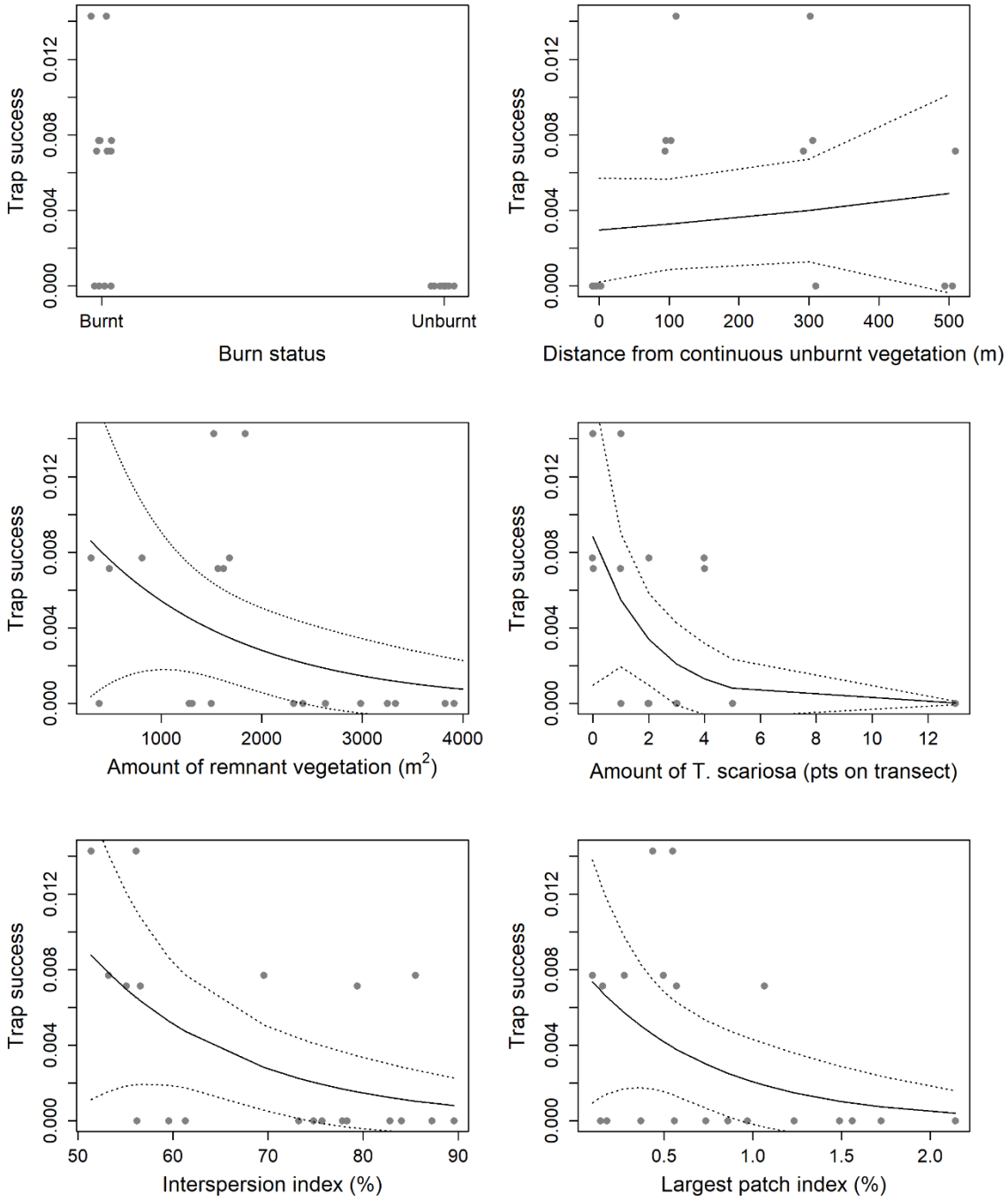


Figure S5.3.10: Predicted response of *Menetia greyii* capture rate (trap success) to the six fire variables used in univariate GLMs. Black lines represent the predicted response of *M. greyii* to each fire variable and dotted lines represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites. Predictions were not possible for burn status, as complete separation of the predictor variable occurred, and reliable parameter estimates and predictions could not be computed.

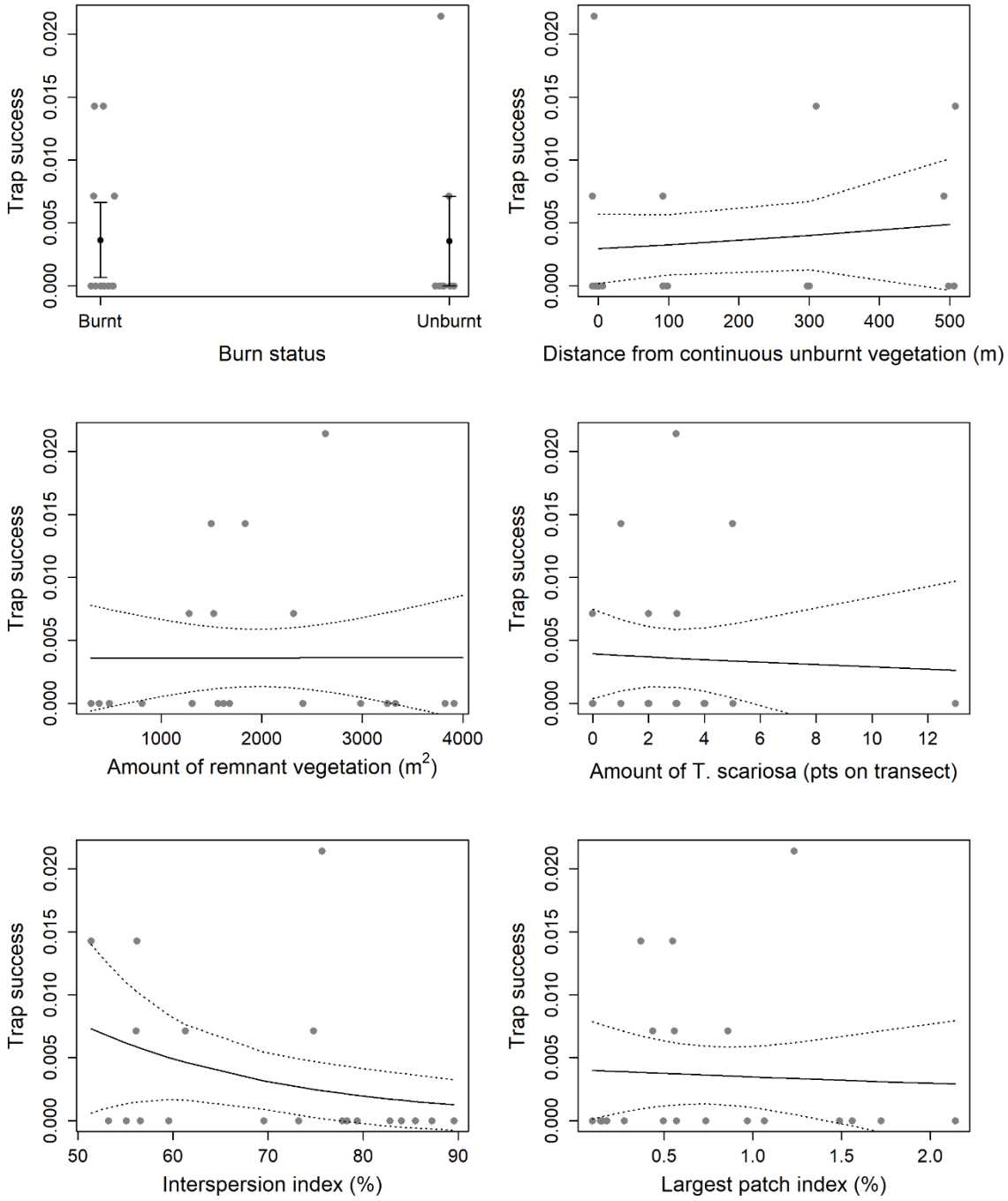


Figure S5.3.11: Predicted response of *Pogona vitticeps* capture rate (trap success) to the six fire variables used in univariate GLMs. Black lines (or dots, for the categorical variable of burn status) represent the predicted response of *P. vitticeps* to each fire variable and dotted lines (or error bars, for the categorical variable of burn status) represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites.

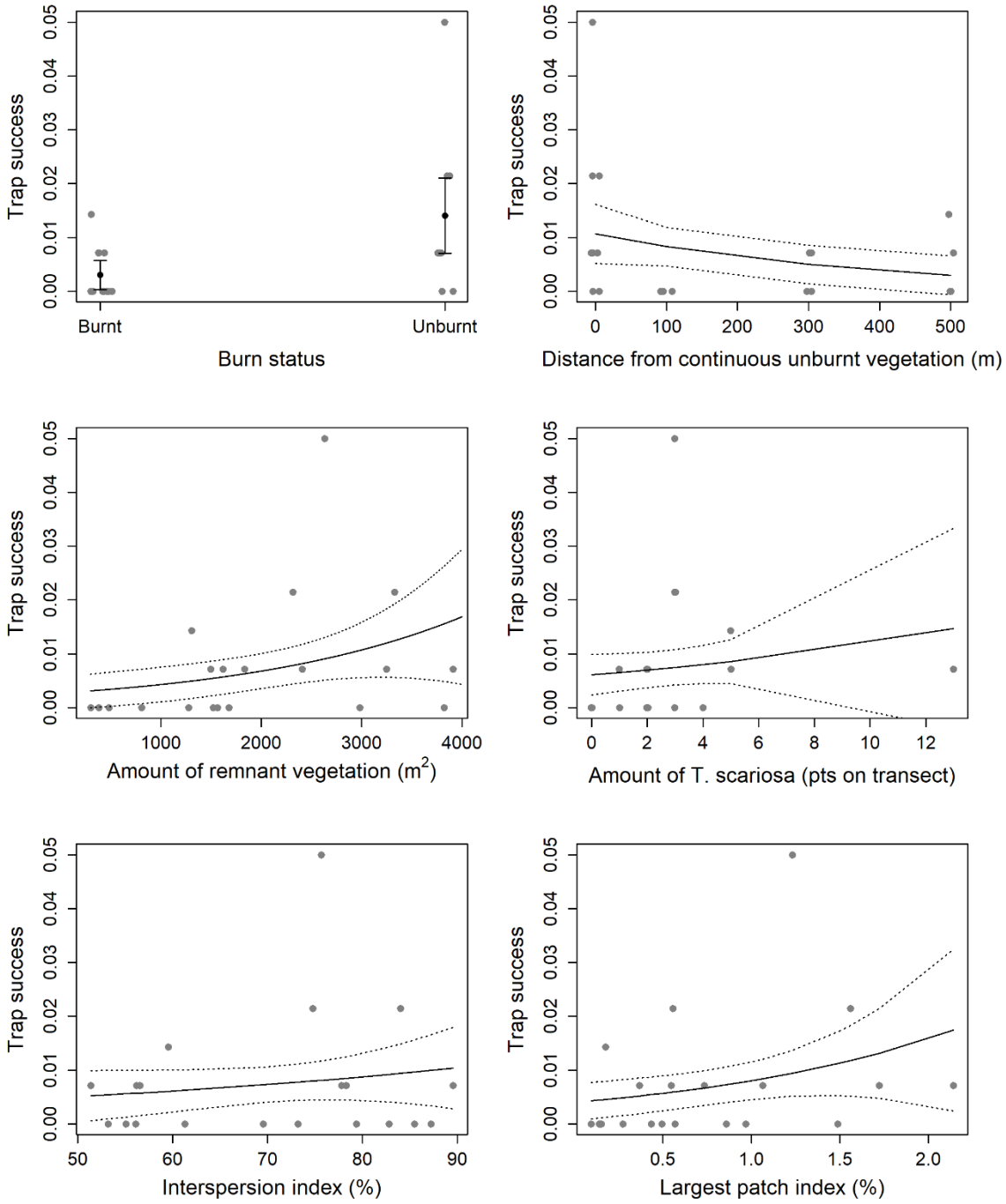


Figure S5.3.12: Predicted response of *Strophurus intermedius* capture rate (trap success) to the six fire variables used in univariate GLMs. Black lines (or dots, for the categorical variable of burn status) represent the predicted response of *S. intermedius* to each fire variable and dotted lines (or error bars, for the categorical variable of burn status) represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites.

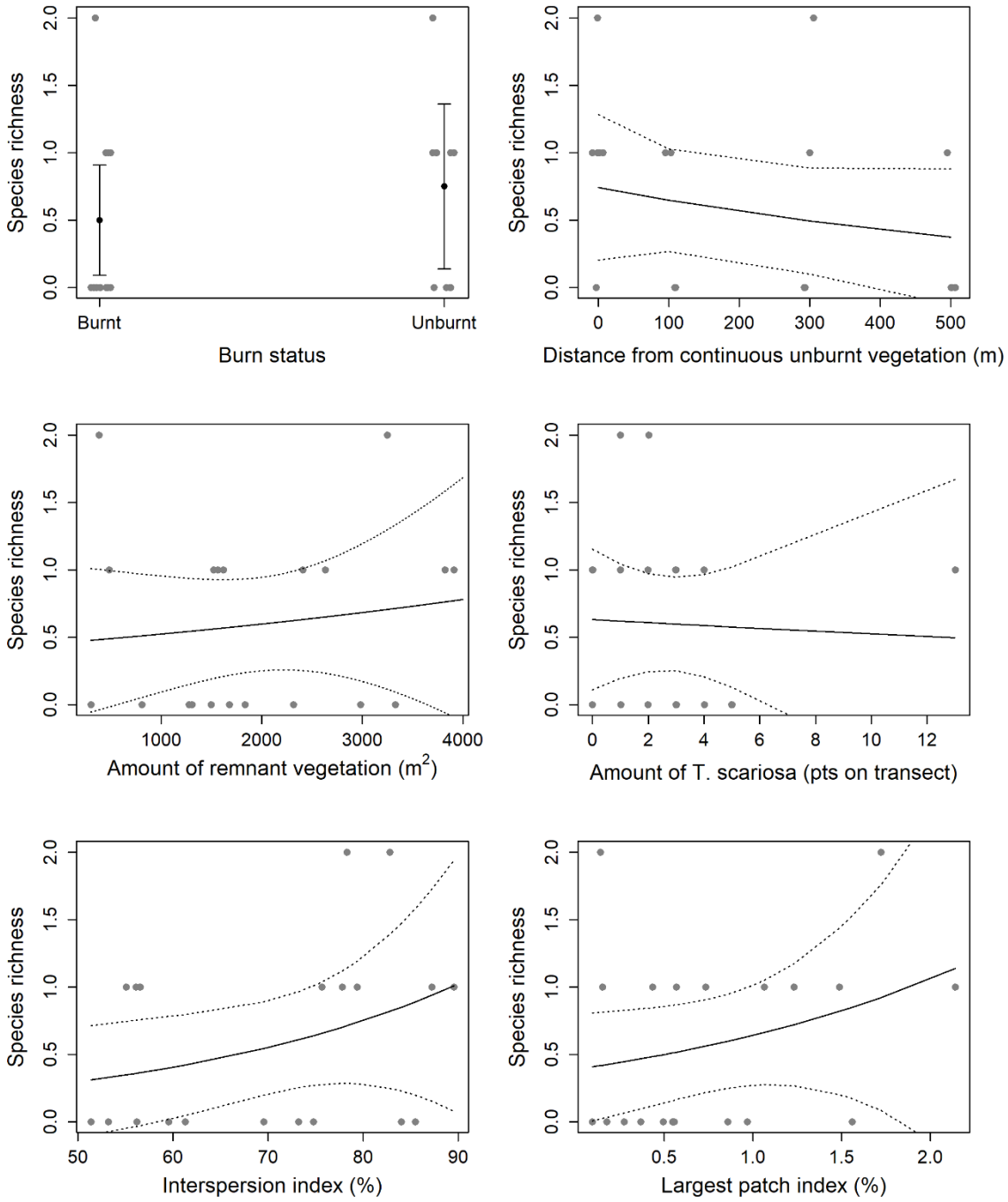


Figure S5.3.13: Predicted response of small mammal species richness to the six fire variables used in univariate GLMs. Black lines (or dots, for the categorical variable of burn status) represent the predicted response of small mammal species richness to each fire variable and dotted lines (or error bars, for the categorical variable of burn status) represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites.

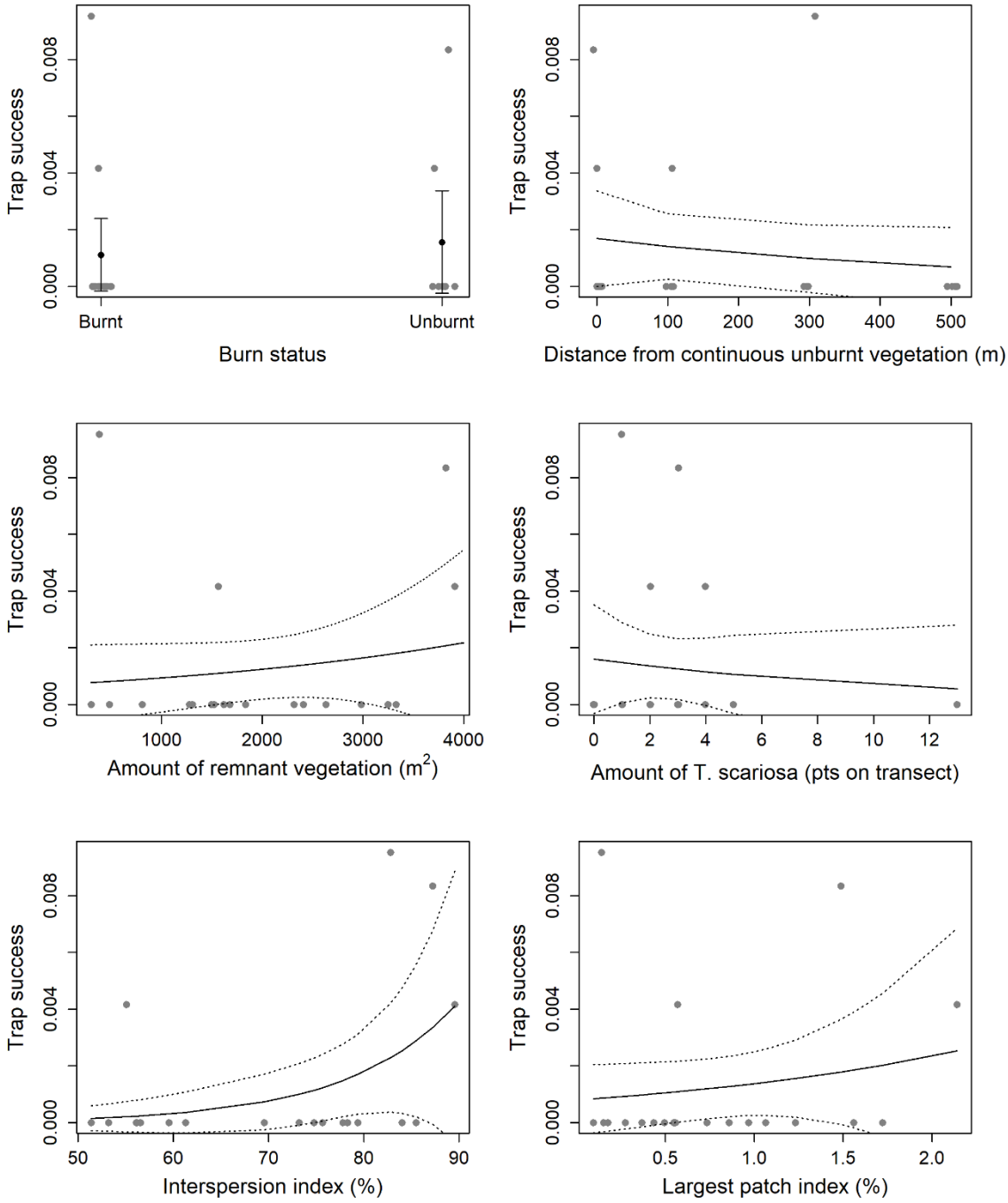


Figure S5.3.14: Predicted response of *Cercartetus lepidus* capture rate (trap success) to the six fire variables used in univariate GLMs. Black lines (or dots, for the categorical variable of burn status) represent the predicted response of *C. lepidus* to each fire variable and dotted lines (or error bars, for the categorical variable of burn status) represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites.

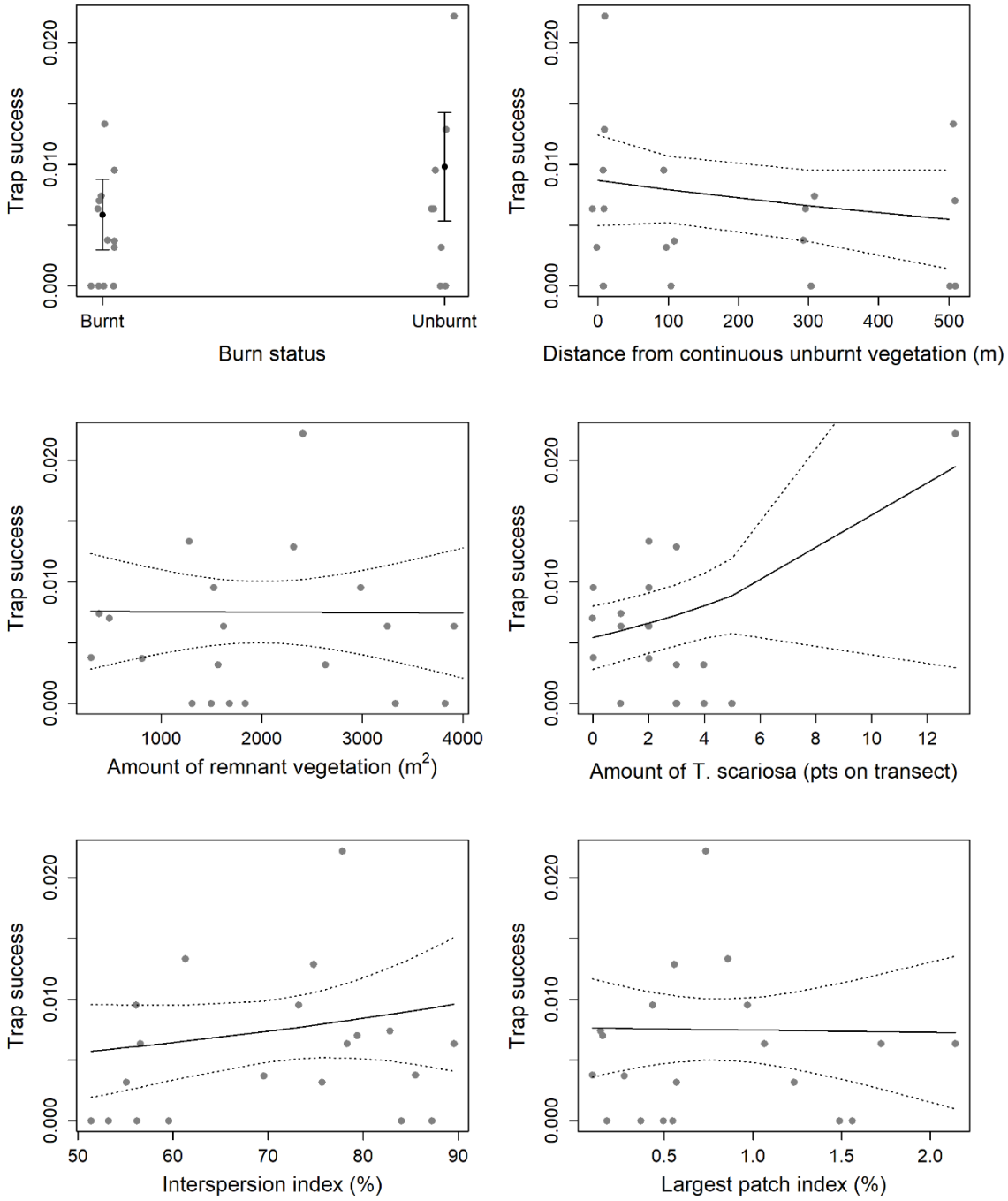


Figure S5.3.15: Predicted response of *Mus musculus* capture rate (trap success) to the six fire variables used in univariate GLMs. Black lines (or dots, for the categorical variable of burn status) represent the predicted response of *M. musculus* to each fire variable and dotted lines (or error bars, for the categorical variable of burn status) represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites.

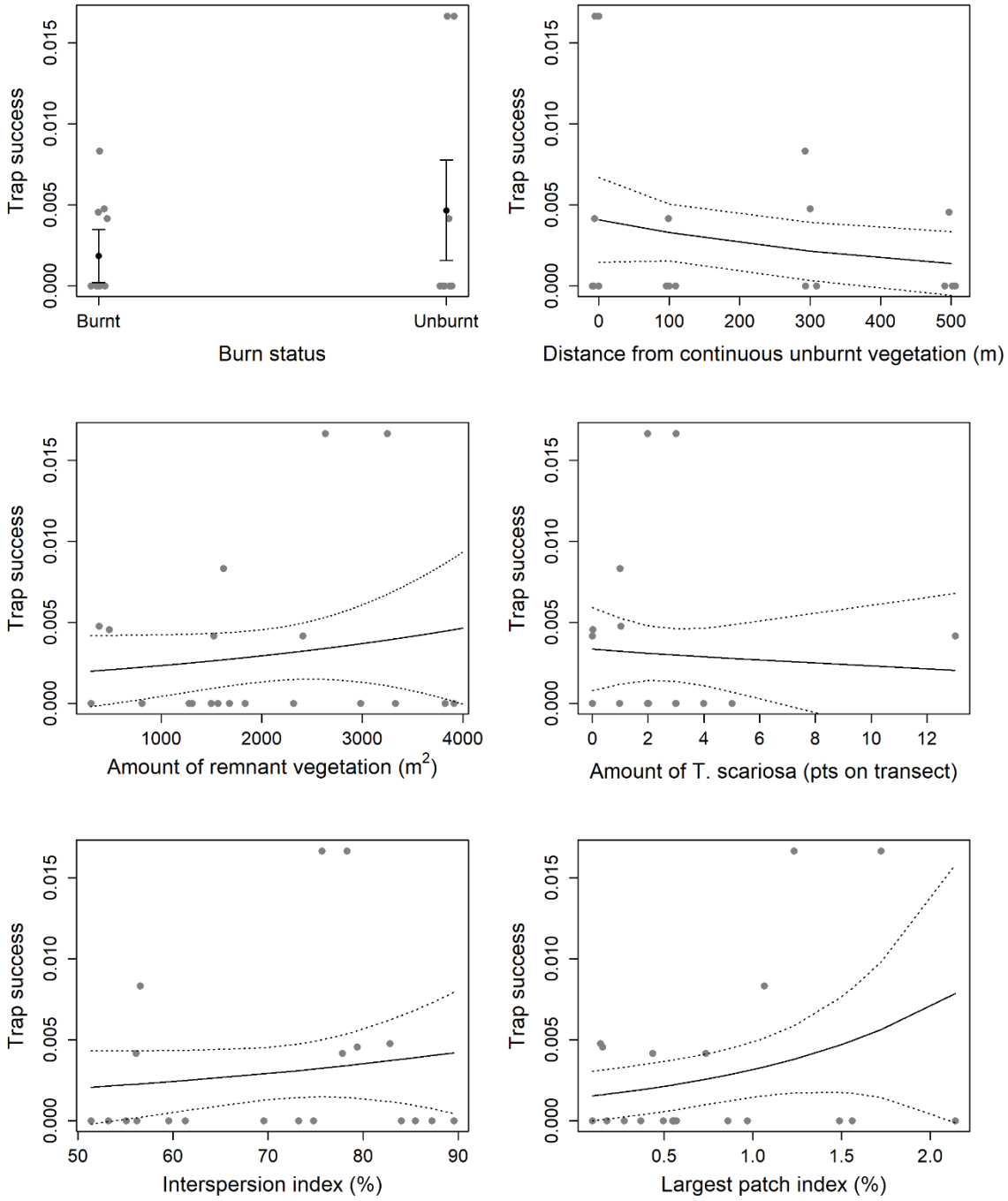


Figure S5.3.16: Predicted response of *Sminthopsis murina* capture rate (trap success) to the six fire variables used in univariate GLMs. Black lines (or dots, for the categorical variable of burn status) represent the predicted response of *S. murina* to each fire variable and dotted lines (or error bars, for the categorical variable of burn status) represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites.

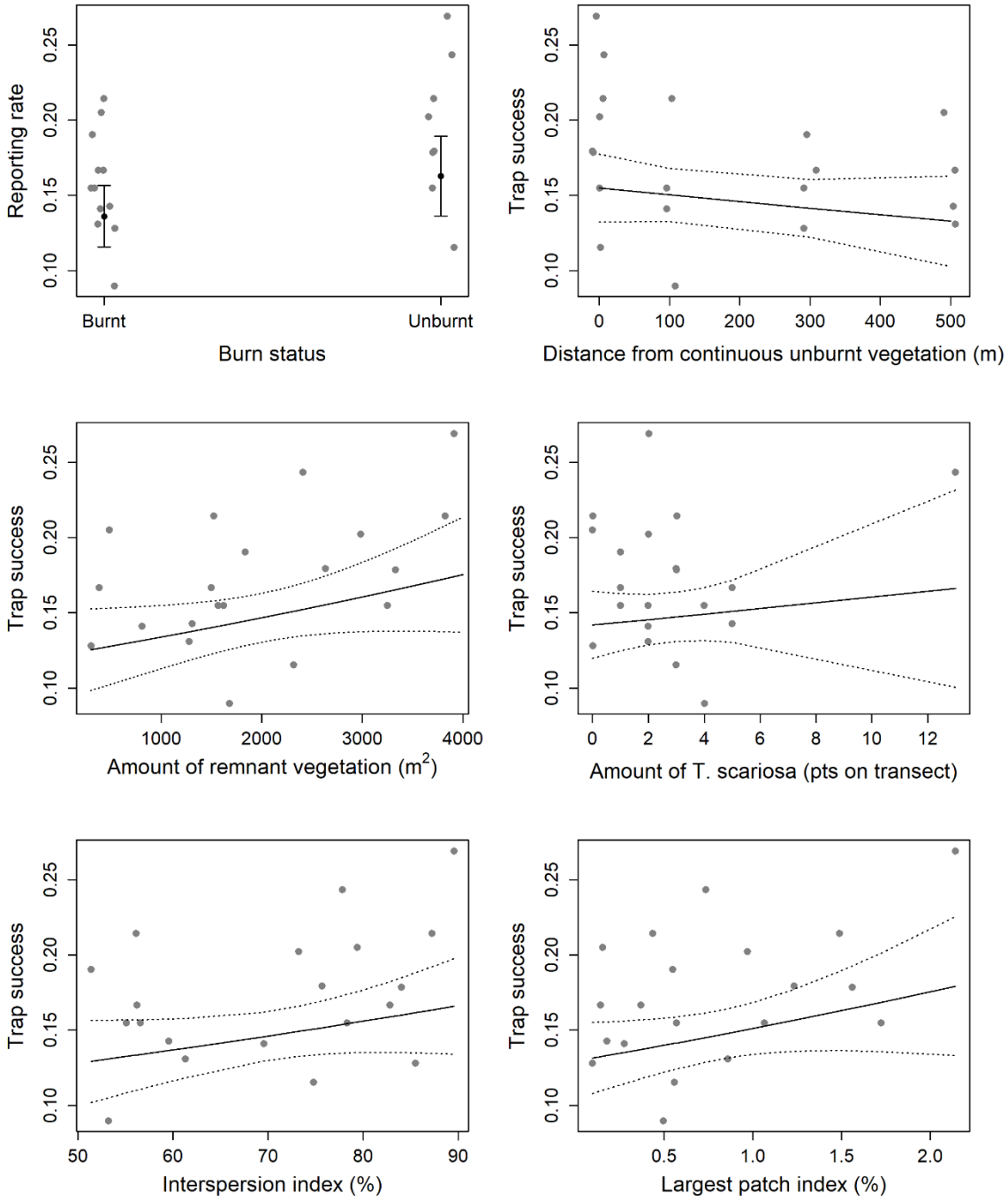


Figure S5.3.17: Predicted response of *Vulpes vulpes* capture rate (trap success) to the six fire variables used in univariate GLMs. Black lines (or dots, for the categorical variable of burn status) represent the predicted response of *V. vulpes* to each fire variable and dotted lines (or error bars, for the categorical variable of burn status) represent the 95% confidence intervals around that estimate. Grey dots show the raw data from the 20 survey sites.