Nestbox use and predicted distribution of the southern greater glider (*Petauroides volans*) following the 2019/2020 megafires

by

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Declaration Statement

I, Jenna Ridley, declare that this thesis, in fulfillment of the requirements of the award Master of Philosophy, in the Fenner School of Environment and Society, Australian National University is my own work unless otherwise referenced or acknowledged.

This document has not been submitted for qualifications at any other academic institution.

This thesis contains two data chapters which have been written as independent research articles. Chapter 1 was accepted for publication by Austral Ecology in July 2024.

The conceptual development, planning and execution of each chapter was my own responsibility and work as a post-graduate student of the Australian National University. I was guided and mentored in this from my supervisory panel.

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Statement of author contributions

Chapter 2: Australian megafires alter predicted occurrence of the southern greater glider (*Petauroides volans*)

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Contributions: JCHR and WB conceived and designed the analysis, JCHR collected the data, JCHR analysed the data, JCHR led the writing of the manuscript. All authors contributed to the drafts and gave final approval for publication.

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Publications and Presentations

Throughout this degree, I have contributed to the following:

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- Oral Presentation for the NSW National Parks Saving Threatened Species Program (2022)
- Oral Presentation for the Yass Men's Shed members (2021)

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Covid-19/La Niña Statement

During 2021, I had to suspend fieldwork for large periods of time due to Covid-19, border restrictions and government mandated lockdowns. This restricted my ability to continue fieldwork in late-2021 and increased the time pressure in early 2022 which led to delays in collecting data. The main issue with these delays was that they impacted the number of spotlighting surveys in Tallaganda that could be completed prior to nestbox deployment.

La Niña also significantly affected my fieldwork and increased the time taken to complete the monitoring due to ongoing issues with road access and poor weather conditions.

Abstract

Species losses and declines continue to occur across the globe and are predicted to increase exponentially due to anthropogenic climate change. South-eastern Australia has been identified as a future extinction hotspot for terrestrial mammal fauna. The southern greater glider (*Petauroides volans*), Australia's (and the world's) largest gliding marsupial, was recently uplisted from Vulnerable to Endangered under the *Environment Protection and Biodiversity Conservation Act 1999*. Southern greater gliders are fire-sensitive and severely threatened by the loss of hollow-bearing trees associated with logging and altered fire regimes. In 2019/2020, after an extended period of drought, south-eastern Australia experienced wildfires over an eight-month period. These wildfires, referred to here as the 2019/2020 megafires, were unprecedented in severity and extent, and affected large parts of southern greater glider habitat in Victoria and New South Wales.

Careful post-fire management is essential for species persistence. Landowners, conservationists, and researchers have implemented various conservation actions to protect species following fire. With a projected increase in the severity, intensity, and frequency of wildfires, understanding where southern greater gliders can persist across the landscape is essential for their management.

In Chapter 1, I aimed to determine where southern greater gliders were predicted to occur in the two years before and two years following the 2019/2020 megafires. Using a Maximum entropy (Maxent) species distribution model, I found that the predicted southern greater glider distribution remained stable throughout this period; however, there was a clear decline in marginal (moderately suitable) southern greater glider habitat. Forests at higher elevations (>750m) and that experienced lower fire severity and fewer fires demonstrated higher likelihoods of species occurrence.

Nestboxes are a common tool used to supplement hollow loss, yet successful implementation is often limited and difficult to quantify. In Chapter 2, I implemented a landscape-scale nestbox study in areas affected by the 2019/2020 megafires, with the aim of testing effectiveness of nestboxes as a post-fire recovery tool for the southern greater glider. Nestboxes were purpose built and installed at sites within Tallaganda National Park and State Conservation area, New South Wales and within the forests of the East Gippsland region, Victoria. I conducted spotlighting surveys at nestbox sites and control sites (without nestboxes) before and after nestbox installation. Southern greater gliders occupied nestboxes in both study areas. They were the dominant species identified occupying nest boxes and during spotlighting surveys in Tallaganda, however, were not the dominant species in East Gippsland. I

also found there were fewer southern greater gliders in areas of higher fire severity within Tallaganda. My work outlines the establishment of a long-term study that hopes to quantify the utility of nestboxes in supporting the post-fire recovery of this Endangered species.

Southern greater gliders continue to persist in forests affected by the 2019/2020 megafires. However, my research has revealed that increased fire severity and frequency decreases habitat suitability for this species. Nestboxes may provide an adequate recovery tool to assist population recovery, however, more research is required to understand the complex interactions that may limit post-fire recovery of southern greater gliders.

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1.0 CHAPTER 1

Introduction

Global species loss is an ongoing conservation challenge (Martin, 2019), with one-fifth of the world's vertebrates facing extinction (Davidson et al., 2017). Numerous threats are contributing to the ongoing decline of species, including human activities that exploit and alter natural environments (Ellis, 2019; Woinarski et al., 2015). As the global human population, and effects of human-induced climate change increase, species declines are predicted to accelerate (Woinarski et al., 2015).

Australia is recognised as having one of the highest rates of wildlife extinctions (Legge et al., 2023; Woinarski et al., 2015), with 100 Australian endemic species (both flora and fauna) listed as extinct or extinct in the wild since the invasion and subsequent colonisation of Australia in 1788 (Woinarski et al., 2019). Since colonisation and settlement, Australia's natural environments have been altered significantly. Habitat has been modified and cleared, fire regimes have been altered and several invasive flora, fauna and pathogens have been introduced (Woinarski et al., 2019). Australian mammals have suffered the greatest declines and extinctions (Woinarski et al., 2015). Australia has approximately 10% of the world's mammals, with ~90% of these mammals endemic to the continent (Cole et al., 1994). Of these mammals, 38 are confirmed extinct (AMTC, 2024), nine are listed as critically endangered, 40 are listed as endangered, and 58 are listed as vulnerable (Australian Government, 2021). Threats to mammal species are often numerous and compounding, with each species varying vastly in their susceptibility and response to each threat (Woinarski et al., 2015). Understanding the impacts of these key threats is crucial if we are to mitigate them and aid recovery of native Australian mammal populations.

Fire can be a major driver of species diversity (He et al., 2019), due to the inherent link between plant and fire cycles (Doerr and Santín, 2016). However, rapid increases in global human populations, the disruption and loss of traditional fire practices, and the impacts of climate change have resulted in alterations to the frequency and intensity of fires globally (Bowman et al., 2009; Goss et al., 2020; Pausas and Keeley, 2009; Wintle et al., 2020). These alterations have culminated in a global shift from “high-frequency, low-severity fire regimes” to “low-frequency, high-intensity crown fires” (Pausas and Keeley, 2009). Changes to fire regimes can increase pressure on ecosystems and species, and extensively modify landscapes. For example, frequent fires can alter the structure of ecosystems like Mountain Ash forest, where older trees are killed in high severity fires, and short fire intervals prevent the re-establishment of Mountain Ash dominated forest, and re-vegetation of non-ash species ensues (Lindenmayer et al., 2022a).

In the southern-hemisphere summer of 2019/2020, Australia experienced some of the most intense and severe fires in recent history (Wintle et al., 2020). These fires have been deemed Australia’s most costly fire season to date (Davey and Sarre, 2020; Wintle et al., 2020). From September 2019 to March 2020 almost 12.6 million hectares (126,000 km^2) of forest burned across Queensland, South Australia, New South Wales, Victoria and the Australian Capital Territory (Abram et al., 2021; Boer et al., 2020). The 2019/2020 megafires followed Australia’s hottest and driest year on record (Boer et al., 2020). These megafires were unprecedented in geographical scale, duration, and intensity (Wintle et al., 2020). In addition to the loss of human life and property, and the prolonged health and economic impacts, the 2019/2020 megafires significantly impacted native species, with 91 taxa recommended to be listed or uplisted on the threatened species list as a result of the fires (Abram et al., 2021; Legge et al., 2022).

Native mammals can survive the direct impact of fire; however, they may then be impacted by the post-fire reduction in food and shelter (Santos et al., 2022). Post-fire threats to species can exacerbate species declines, particularly if these threats existed pre fire and intensify afterwards (Ward et al.,2022). Fire-sensitive species, such as the greater glider (*Petauroides* spp.), were negatively affected by the 2019/2020 megafires, with 29% of their habitat burnt and 37% of this habitat burnt at a high severity (Ashman et al., 2021).

Greater gliders (*Petauroides* spp.) are hollow-dependent, nocturnal arboreal marsupials that occupy eucalypt-dominated forests in eastern Australia (Maloney and Harris, 2008; Tyndale-Biscoe and Smith, 1969a). Similar to the koala (*Phascolarctos cinereus*), greater gliders rely on eucalypt leaves (predominantly trees in the genus *Eucalyptus*) for food, although they can also feed on eucalypt buds and blossoms, and some other non-eucalypt plant species (Maloney and Harris, 2008). In 2016, greater gliders were listed as Threatened under the Victorian Flora and Fauna Guarantee Act 1988, and Vulnerable under the Commonwealth Environment Protection and Biodiversity Conservation Act 1999 (DCCEEW, 2021; DELWP, 2021) and the IUCN Red list (Burbidge and Woinarski, 2014).

Recent research has revealed that greater gliders in the northern and southern parts of their range are genetically distinct, and might include up to three separate species (McGregor et al., 2020). In 2022, the taxonomy was formalised to include a northern species (*Petauroides minor*) and a southern and central species (*Petauroides volans*) (Figure 1.1; Department of the Environment (2024a, 2024b)). The geographical boundaries of the northern and southern greater glider populations have not yet been established. We currently know that the southern greater glider (*Petauroides volans*), the largest of the gliding species, occupies parts of Queensland, New South Wales, the ACT and Victoria (Figure 1.1). Southern greater gliders were uplisted from Vulnerable to Endangered shortly after the 2019/2020 megafires (DCCEEW, 2022). This uplisting to Endangered was in part due to the reduction in population and habitat following the 2019/2020 megafires (DCCEEW, 2022).

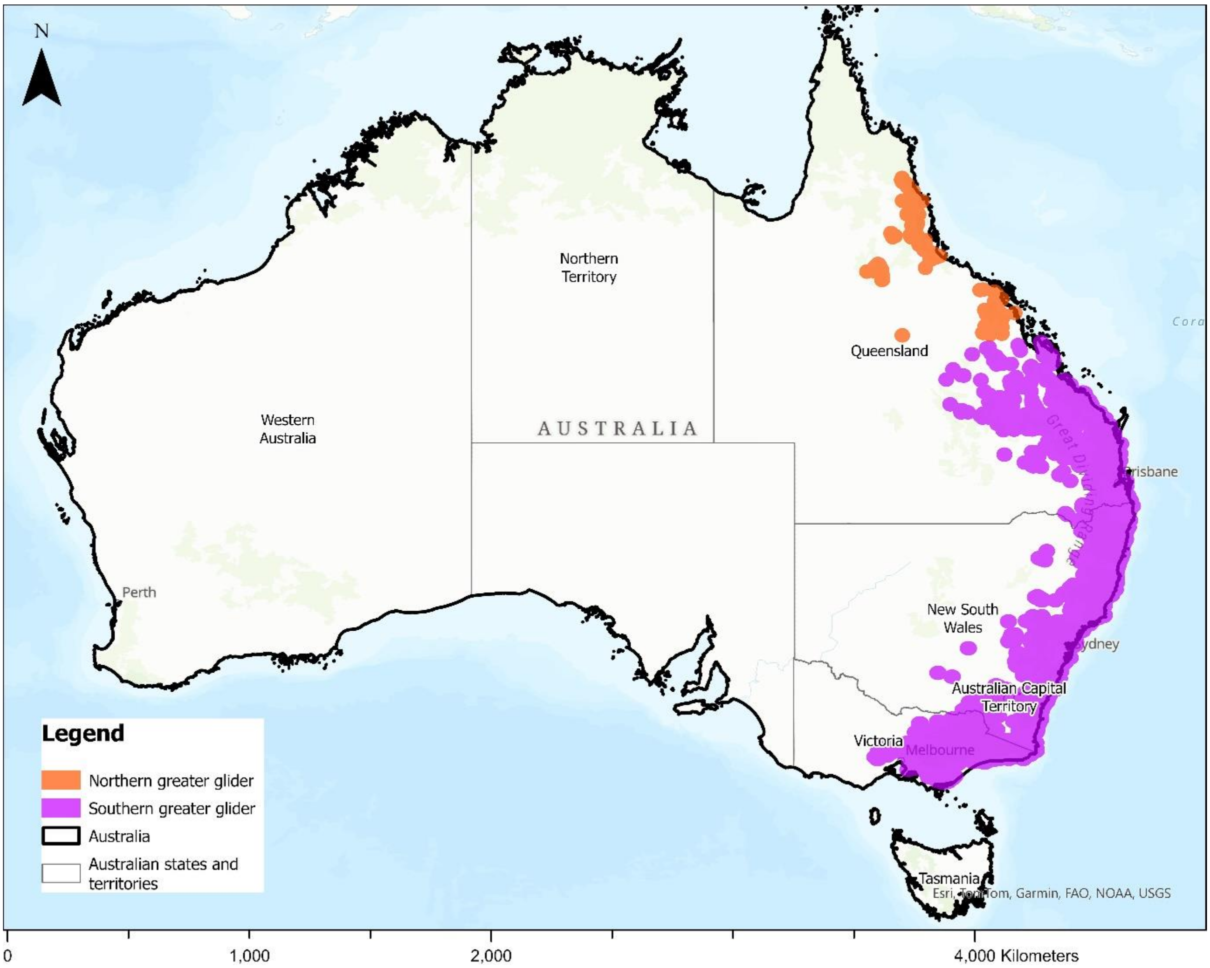

Figure 1.1: The estimated distribution of the southern (purple) and northern (orange) greater glider in eastern Australian. This map was made using information from species profiles (Department of the Environment, 2024a, 2024b) and individual occurrence records of each species from the Atlas of Living Australia (Atlas of Living Australia, 2024a, 2024b). Points were buffered by a 30km distance and smoothed to get an estimated distribution.

Southern greater glider populations have declined over the past two decades (Lindenmayer et al., 2022b). The reasons for these declines vary within and between populations. One of the most prominent threats is the loss of critical greater glider habitat from deforestation, logging, severe wildfire (May-Stubbles et al. 2022; McLean et al. 2018) and anthropogenic climate change (Ashman et al., 2021) that has been occurring for decades. This loss of habitat has had both direct effects, such as mortality, and indirect effects such as loss of food and hollows for shelter, reproduction, and safety from predators (Gibbons and Lindenmayer, 2002; Goldingay, 2012). Tree hollows provide shelter from temperature extremes and predation, and refuge for reproduction and caring for young (Lindenmayer et al., 1996). Southern greater gliders are highly vulnerable to heat stress, with a thermoneutral point of 20 degrees Celsius (Rübsamen et al., 1984). The increased temperatures expected due to anthropogenic climate change, therefore, are predicted to negatively impact the southern greater glider (Wagner et al., 2020).

Long-term studies in the central highlands in Victoria have shown that forest-age, elevation and the abundance of tree hollows all affect the ecology and survival of southern greater gliders (Lindenmayer et al., 2022b, 2024). In the Blue Mountains of NSW, southern greater gliders were once widespread; however, recent monitoring and research shows they are now rarely observed at lower elevations (Smith and Smith, 2018).

With many species in south-eastern Australia at risk of further declines and extinction (Cardillo et al., 2023), it will be critical to implement conservation initiatives to counteract these problems in remaining populations (Kuussaari et al., 2009). With ongoing anthropogenic climate change, an increase in fire frequency and severity is expected (Abram et al., 2021; Canadell et al., 2021; Fairman et al., 2015; Lindenmayer et al., 2023). These predictions suggest a lengthening of fire seasons and ignitions occurring earlier during the cooler months (Canadell et al., 2021), that will likely have negative impacts on southern greater glider populations (McLean et al. 2018).

In this thesis, I aim to quantify the impacts of the 2019/2020 megafires on southern greater gliders in East Gippsland, a region severely impacted by fire. I then outline the establishment of a long-term study to determine the effectiveness of purpose-built nestboxes in aiding the post-fire recovery of the species. I aim to close important knowledge gaps to improve conservation for southern greater gliders. My thesis is presented as two distinct but related chapters, as summarised below.

Chapter 2: Australian megafires alter predicted distribution of the southern greater glider (*Petauroides volans*)

Understanding where a threatened species may persist following large-scale fire is crucial for its ongoing management and conservation (Geary et al., 2022). The East Gippsland region of Victoria was once considered a stronghold for the southern greater glider. Populations in these areas, however, have declined over the last 20 years (DELWP, 2019). Approximately 1.5 million hectares of Victoria's public and private land was burnt during the 2019/2020 megafires (AIDR, 2020). East Gippsland forests, which are home to populations of the southern greater glider, were extensively affected by the 2019/2020 megafires (Atlas of Living Australia, 2024b; Victorian Biodiversity Atlas, 2023).

In this chapter, I outline the construction of a maximum entropy (Maxent) species distribution model using presence-only occurrence records of southern greater gliders across East Gippsland before and after the megafires. I sought to quantify the short-term impact of the 2019/2020 megafires on the occurrence of the southern greater glider in East Gippsland, Australia. I hypothesised that, following the megafires, (1) southern greater glider occurrence would be highest in unburnt areas and locations subject to lower-severity fire compared to more severely burnt places; and (2) southern greater glider occurrence would be greater at higher elevations than lower elevations.

Chapter 3: Establishing foundations: Designing a long-term experiment to evaluate whether nest boxes assist population recovery of an endangered species after fire

Several conservation management tools can be used after disturbance events to limit the effects of habitat loss (Watchorn et al., 2022). One of those tools, nestboxes, can be used as supplementary shelter for a range of species in many habitat types across Australia (Goldingay et al., 2018; Watchorn et al., 2022). It is unclear; however, whether nestboxes can assist recovery of hollow-dependent species such as the southern greater glider. Here, I outline the establishment of a long-term study to determine the utility of purpose-built nestboxes in aiding the recovery of southern greater gliders following the 2019/2020 megafires. I focus on the preliminary results from the baseline data and first round of monitoring post nestbox deployment. Specifically, I asked, (1) Do southern greater gliders use our purpose-built nestboxes?; (2) Did non-target species use nestboxes purpose-built for southern greater gliders?; (3) Did southern greater glider abundance increase in treatment sites (i.e., with nestboxes) compared to control sites (i.e., without nestboxes) post-treatment?; and (4) Does fire severity affect the abundance of southern greater gliders in the study area?

1.1 Implications of this research

Increasing knowledge of southern greater glider populations and their distribution following megafire can assist in the ongoing management of the species. Of the habitat in Victoria predicted to be suitable for southern greater gliders, 16% was affected by high-severity fire during the 2019/2020 megafires (DELWP, 2022).

My first chapter provides crucial information about where southern greater gliders are predicted to occur post fire. In developing this Maxent model, I have aimed to assist land managers in determining where highly suitable habitat may occur for the southern greater glider and therefore reveal priority areas for conservation management.

My second chapter demonstrates the value of a rigorous experimental design for assessing the impact of a common management tool on post-fire population recovery. Nestboxes can be useful, however, they can also have negative consequences. I aimed to fill gaps in our understanding of whether nestboxes can support the southern greater glider after fire through the provision of additional shelter sites. This will help managers assess the benefits of nestboxes for managing southern greater glider recovery following fire.

Disturbances such as the 2019/2020 megafires underline the importance of monitoring and research to help conserve threatened species. In this thesis, I have sought to contribute new information on two important populations of the southern greater glider and provide answers to practical questions for improving the species future conservation.

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2.0 CHAPTER 2

Australian megafires alter predicted distribution of the southern greater glider (*Petauroides volans*)

East Gippsland, April 2021. J Ridley

2.1 Abstract

Altered fire regimes and the increased frequency and/or severity of wildfires are significant issues for fire-sensitive species. In the Austral summer of 2019/2020, Australia experienced one of the worst fire seasons in recorded history (known as the 2019/2020 megafires), affecting 12.6 million hectares of south-eastern Australia. The Endangered southern greater glider (*Petauroides volans*) inhabits large areas impacted by the 2019/2020 megafires. With increasing fire severity, frequency, and intensity, there is a need to understand where in the landscape southern greater gliders are most impacted by fire, and where the species is most likely to persist in fire-affected landscapes. This is challenging given the lack of systematic data collected for this species. Maximum entropy modelling (Maxent) is a type of species distribution model with good predictive performance for presence-only records. We used Maxent to model the distribution of the southern greater glider in the two years prior (2017–2019) and two years post (2020 – 2022) megafire across the East Gippsland region of Victoria, an area heavily affected by the 2019/2020 megafires. Whilst we found that the predicted short-term area of occurrence of the southern greater glider remained relatively stable in the two years after megafire, a large area of marginal habitat was considerably reduced. We interpret this result as a potential contraction of suitable habitat where southern greater gliders may become isolated. Post fire, higher elevation areas and locations that experienced comparatively lower fire severity and a lower fire frequency had a higher predicted occurrence of the southern greater glider. Understanding the role of fire in species survival is essential for effective biodiversity conservation and management of species. More comprehensive and systematic monitoring in suitable and unsuitable areas will enhance our understanding of the distribution of the southern greater glider, including areas subject to wildfire.

Key words: Wildfire, presence-only modelling, fire-sensitive, conservation, monitoring

2.2 Introduction

Anthropogenic climate change has contributed to changing fire regimes, an increased frequency of extreme weather events, and more severe wildfires globally (Abram et al., 2021; Kemter et al., 2021; Lindenmayer et al., 2023; Williams et al., 2009). Wildfires have increased in size and frequency, to the extent that new terms such as ‘megafire’ (fires >10,000 ha) and ‘gigafire’ (fires > 100,000 ha) have been adopted (Linley et al., 2022). These larger and more frequently occurring wildfires can be of high severity (as defined in Keeley, 2009) and are burning over prolonged periods, lengthening fire seasons worldwide (Krawchuk et al., 2009; McLauchlan et al., 2020) and leaving fewer and smaller unburnt patches within the perimeters of fire scars (Mackey et al., 2021). Moreover, wildfires are impacting areas previously less prone to fire or that historically have remained unburned, altering vegetation communities, such as rainforest, that have evolved without fire (Ward et al., 2020).

Over the last few decades alone, Australia has experienced some of the most extreme wildfire events in the world (Cruz et al., 2012). The 2019/2020 megafires were unprecedented in length, frequency, extent, and severity (Kemter et al., 2021). These fires burnt over an eight-month period, impacting almost 12.6 million hectares of eastern Australia (Davey and Sarre, 2020; Wintle et al., 2020). They caused significant loss of life and property, and had major impacts on water quality, wildlife, health, tourism, and the economy (Davey and Sarre, 2020; Kemter et al., 2021). Leading up to the 2019/2020 megafires, Australia experienced a prolonged period of drought (between 2017 and 2019) (Nguyen et al., 2021) with overall moisture levels at all-time lows (Kemter et al., 2021) and extreme maximum annual mean temperatures peaking in 2019. These megafires affected an estimated 21% of Australia’s threatened vertebrates, with an estimated one billion animals killed (Hughes et al., 2020; Nolan et al., 2021).

The increasing frequency of extreme wildfires can exacerbate existing pressures on threatened species and ecosystems (Canadell et al., 2021; Geary et al., 2022; Ward et al., 2020). Species respond differently to wildfire depending on their life-history traits, food, and shelter needs (McLean et al., 2018). For individuals that survive the direct impacts of wildfire, threats can increase afterwards due to the reduction of shelter and food resources or an increased risk of being depredated (Doherty et al., 2022). For hollow-dependent arboreal species, impacts of fire on hollow-bearing trees are also likely to create future resource challenges (Banks et al., 2011).

The southern greater glider (*Petauroides volans*) is an arboreal folivore dependent upon hollow-bearing trees for shelter and reproduction (Lindenmayer et al., 1990). Southern greater glider numbers can decline after a single fire (McLean et al., 2018), in response to exposure to high temperatures (DELWP, 2019), and due to the loss of hollow-bearing trees (Lindenmayer et al., 2012, 2020). During the 2019/2020 megafires, approximately 29% (5.3 million ha) of all greater glider habitat was burnt (Ashman et al., 2021). With megafires expected to increase in frequency and severity, understanding the factors affecting post-fire species distributions will be crucial in preventing further declines and extinction (Ward et al., 2020).

Successful species management requires ongoing monitoring (Legge et al., 2018). Effective monitoring assists in identifying population declines early, and gives ecological insight into the response of populations to disturbance (Legge et al., 2018). However, Legge et al., (2018) and Scheele et al., (2019) showed between 21% and 46% of Australia's threatened vertebrates received no monitoring at all, and many of the monitoring programs in place were underfunded or ineffective. The southern greater glider is among those species that lack ongoing systematic monitoring across much of their range. Given a lack of robust data from long-term monitoring, we used presence-only maximum entropy modelling (Maxent), a type of species distribution modelling that enables species occurrence to be estimated from presence-only data collected with differing methods, even when absence data are lacking (Elith* et al., 2006; Phillips et al., 2004, 2006; Yackulic et al., 2013). We used presence-only observations of southern greater gliders from three separate datasets to model the species occurrence two years prior to, and two years post, the 2019/2020 megafires.

We sought to quantify the impact of the 2019/2020 megafires on the short-term occurrence of the southern greater glider in East Gippsland. Due to the sensitivity of southern greater gliders to fire (McLean et al., 2018) and heat (Rübsamen et al., 1984), and previous studies showing a link between glider abundance and elevation (Lindenmayer et al., 2022b; Wagner et al., 2020), we hypothesised that following the 2019/2020 megafires: (1) southern greater glider occurrence would be higher in unburnt areas and locations subject to lower-severity fire compared to more severely burnt places; and (2) southern greater glider occurrence would be greater at higher elevations than lower elevations. By examining these questions, we hoped to provide insight into the potential drivers of southern greater glider occurrence and how this may change before and after fire. This will assist conservation practitioners in their survey and monitoring targets.

2.3 Method

Study Area

Our study was located within the East Gippsland region of Victoria (~2.2 million hectares) on Gunaikurnai, Monero (Ngarigo), Yuin, and Bidwell Country (Figure 2.1). East Gippsland has a temperate climate that varies across an elevational gradient from sea level to 1970 metres (DELWP, 2023; DELWP, 2024b; Victorian Resources Online, 2023). The region has an annual average rainfall of between 800 – 1200mm, falling primarily within winter and spring and experiences snowfall at elevations >1500m in winter (Victorian Resources Online, 2023). Elevation and proximity to the coast influence the major vegetation communities present across the region.

The forests in East Gippsland are managed for several uses. Over half of the region is state forest (~52%) managed by the Department of Environment, Land, Water, and Planning (DELWP) and has been logged for more than a century (East Gippsland CMA, 2023). However, a recent change in Victorian State policy ended native forest logging in January 2024 (DEECA, 2023). Many areas of native forest where logging is excluded are managed by Parks Victoria.

The 2019/2020 megafires affected over 1 million hectares of East Gippsland's forest and water catchment areas (AIDR, 2020) - approximately 56% of the region (East Gippsland CMA, 2023). The southern greater glider has been detected over a large proportion of this burnt area in the past (~1500 occurrence records between 1970 to 2010) (Atlas of Living Australia, 2024).

We chose this study region for the following reasons: 1. It contains extensive forested areas affected by the 2019/2020 megafires; 2. It supports populations of southern greater gliders (Atlas of Living Australia, 2024; Victorian Biodiversity Atlas, 2023); 3. Pre-fire and post-fire observation data were available for the southern greater glider, in addition to extensive fire mapping (Figure 2.1); and 4. There have been ongoing concerns for the conservation status of the southern greater glider within this region (DELWP, 2019).

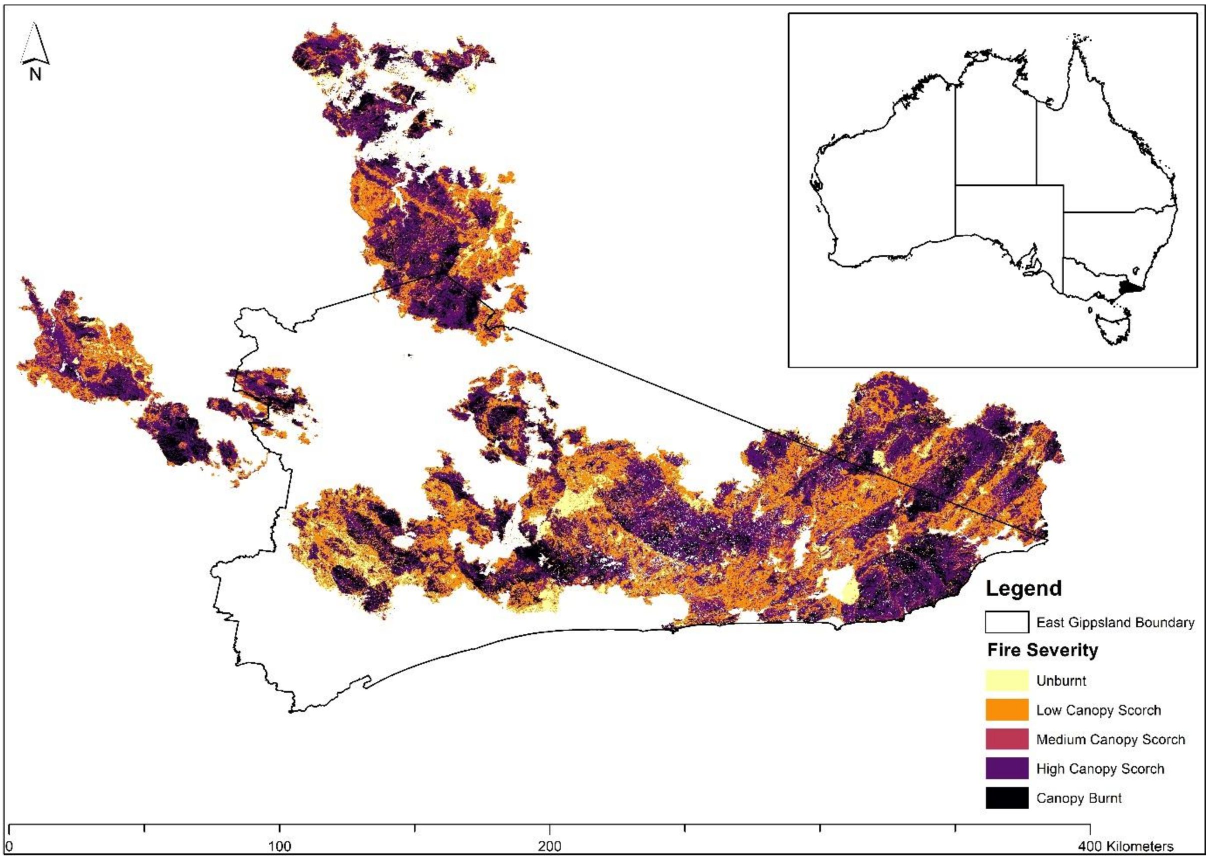

Figure 2.1. Map of the East Gippsland region (black outline) with the 2019/2020 megafire scar extent overlayed. The inset map shows Australia's state boundaries and East Gippsland as the filled in section in the bottom right corner of Victoria. The mapped fire severity areas are shown in the legend. The white (uncoloured) areas have no data, are unclassified data or are not part of the fire severity extent. This map was generated in ArcPro (Esri, 2022) using the 2019 2020 fire severity layer (DEECA, 2022b).

Occurrence records for the Southern Greater Glider

We collated southern greater glider location records from the Victorian Biodiversity Atlas (VBA), Goongerah Environment Centre (GECO) surveys, and unpublished surveys by the Australian National University (ANU) (Table 2.1). These ANU surveys used both single and double-observer spotlighting methods (Cripps et al., 2021). We downloaded data from the VBA on the 20th February 2023 (Victorian Biodiversity Atlas, 2023). The southern greater glider is typically silent, and only very occasionally calls with a sound that is difficult to distinguish from the common ringtail possum (*Pseudocheirus peregrinus*) (D.B Lindenmayer, personal observation, 2023). Therefore, we removed observations documented as heard (not seen) from the dataset. We also removed records if the individual count was 0 or ‘blank’, the individual was found dead, or estimated location accuracy was greater than 100m. We removed VBA records that were not verified in the VBA database.

We categorised location records as pre fire (20th November 2017 to 20th November 2019), during (21st November 2019 to 28th February 2020), or post fire (01st March 2020 to 01st March 2022). We excluded records in the ‘during’ category from all analyses, due to uncertainty about fire status during this period. We converted the spatial projection of all occurrence records to GDA 1994 to ensure they were the same projection as the environmental data used in each of the models (EPSG: 4283).

Table 2.1: The databases used to compile southern greater glider location records and the survey methods used to record these observations with the number of records from each source shown in brackets.

Database/Surveyor	Pre-Fire Records	Post-Fire Records	Survey Method
Goongerah Environment Centre	✓ (124)	×	Spotlighting on foot
Victorian Biodiversity Atlas	✓ (484)	✓ (578)	Spotlighting Stag Watching Thermal Imaging Active searches Trapping (Camera/Cage)
ANU Surveys	×	✓ April to Dec 2021 Jan to Feb 2022 (50)	Spotlighting on foot

Maxent modelling

To generate predicted species occurrence based on records of southern greater gliders across East Gippsland, we fitted Maxent models using the 'SDMtune' (Vignali *et al.*, 2020) and 'dismo' (Hijmans et al., 2023) packages in R (R Core Team, 2023) (Table 2.1; Appendix 2.1). Maxent modelling is one of the most common species distribution modelling tools used in conservation (Inman et al., 2021) and is suitable for almost all presence-only datasets (Phillips et al., 2006). The method has been successfully employed in studies of other marsupials such as the mahogany glider (*Petaurus gracilis*) (Chang et al., 2022) and the northern quoll (*Dasyurus hallucatus*) (Molloy et al., 2017).

Spatial Thinning

Southern greater gliders typically occupy small home ranges of approximately 1 – 4ha (Comport et al., 1996; Pope et al., 2004). In keeping with this and the spatial scale used in previous studies (Wagner et al., 2020), we used a 250m resolution for all spatial data in our modelling. We spatially thinned all presence data at a 250m resolution to ensure that southern greater glider presence was recorded a maximum of once per raster cell for each study period (Chang et al., 2022).

Bias correction (Target Background Group Method)

Sampling bias can influence presence-only species distribution models (Barber et al., 2022), because records are often not derived from systematic survey efforts (Phillips et al., 2009).To control for sampling bias, we used the target background group method (Barber et al., 2022; Chang et al., 2022; Phillips et al., 2009). We downloaded occurrence data for all species of gliders and possums in East Gippsland that were collected using similar survey methods applied for the southern greater glider (see Appendix 2.1 for list of species). We cleaned these data using the same record removal criteria as applied to the southern greater glider.

We spatially thinned all target background group data separately for the pre-fire and post-fire periods at a 250m resolution to ensure that arboreal species presence was recorded a maximum of once per raster cell (Appendix 2.2). We used these records to create pre- and post-fire two-dimensional kernel density plots (Calenge, 2006) and converted both plots to rasters to use for bias correction (Table 2.1; Appendix 2.1).

We generated 2000 background points using the pre- and post-fire kernel density rasters as the bias layers (Phillips et al., 2009; Vollering et al., 2019). These background points allowed us to compare the environmental factors at these points to the southern greater glider occurrence locations. Southern greater glider occurrence records are true presence records, whereas the background records are randomly selected and therefore treated as non-occurrence records in this study.

Predictor dataset

All spatial data are freely available from spatial data share (https://datashare.maps.vic.gov.au/). We clipped all variables to the same geographic area (extent = 147.1903 xmin, 149.8932 xmax, -37.91049 ymin, -36.61237 ymax) and spatial resolution (cell size = 0.0022405, ~250m) (Esri, 2022; R Core Team, 2023).

We chose six predictor variables (Table 2.2) to model the distribution of southern greater gliders before and after fire in East Gippsland. Three of these variables were related to fire; fire frequency, fire severity, and time since last fire (Table 2.2). Haslem et al. (2012) found that frequent fires within a 45-year period reduced the density of dead hollow-bearing trees and recommended that future studies examining hollow-dependent fauna include both fire frequency and time since fire variables. Recent evidence also suggests that areas which experience high severity fire would be less suitable for the southern greater glider (Campbell-Jones et al., 2022; Lindenmayer et al., 2020; May-Stubbles et al., 2022). The fire severity layer was produced for the 2019/2020 megafires (see description of categories in (DEECA, 2022b; Table 2.2). Fire frequency and time since last burn were restricted to 1980 onwards because spatial fire layers prior to 1980 were not collected using the same methods and could confound our data. We modified the time since last burnt dataset to quantify the number of fires from 1980 to 2019 for pre fire and from 1980 to 2022 for post fire (Table 2.2).

Our other three predictor variables were elevation, Ecological Vegetation Class (EVC), and land tenure (Table 2.2). Elevation can be an important predictor variable for southern greater gliders, with elevations over 700m providing highly suitable habitat up to a point at which fewer suitable forest types become dominant (Lindenmayer et al., 2022b; Wagner et al., 2020). Elevation is correlated with numerous climate variables including temperature and rainfall (Lindenmayer et al., 2022b; Wagner et al., 2020). We therefore used elevation as a predictor variable, with the expectation it would also act as a proxy for other climatic variables. We used Ecological Vegetation Classes which are mapped across all of Victoria as a standard unit for classifying vegetation (DEECA, 2018, 2022a). Given the southern greater glider is sensitive to the effects of logging (Lindenmayer and Sato, 2018), we used land tenure as a covariate to incorporate management variables, contrasting areas managed to exclude logging (national parks or conservation reserves) with areas that have a history of logging.

Table 2.2: Information on each of the predictor variables used in the presence-only maxent modelling of southern greater gliders before and after the 2019/2020 megafires. We retrieved all original raster files from https://datashare.maps.vic.gov.au. The columns mean the following: data is the type of data, year is when the data were collected, units is the unit of measurement in which the dataset was presented, modifications are what were done after downloading from the source.

Predictor Variables	Data	Year	Units	Modifications	Source
Elevation	Continuous	2008 - 2016	Metres	None	(DELWP, 2024b)
Ecological Vegetation Class	Categorical	2002 - 2005	Class	None	(DEECA, 2018)
Fire Frequency	Continuous	1980 - 2022	Number	Records 1980 onwards. Modified to count all fires not just the most recent	(DELWP, 2024a)
Fire Severity	Continuous	2020	Number	Combined no data (0) and unclassified (1) into the same grouping	(DEECA, 2022b)
Time since last burnt	Continuous	1980 - 2022	Year	Records 1980 onwards	(DELWP, 2024a)
Land Management	Categorical	2008 - 2019	Category	None	(DEECA, 2024)

	(1) Other Public Land; (2) National Park; (3) State Forest; (4) Unknown; (5) Other Conservation Reserves; (6) Plantation

Model training and selection

We trained each model using 700 iterations specifying the ‘maxent’ method in the ‘train’ function from SDM tune package (Vignali et al., 2020). We compared Maxent models using Akaike’s Information Criterion corrected for small sample sizes (AICc) (Burnham and Anderson 2002), assuming models ΔAICc < 2 to be candidates for best-fit models. We determined the best-fit model for before and after fire using a stepwise method where we removed the variable with the lowest permutational importance each time until there were only two variables remaining (Appendix 2.3; Appendix 2.4). Low permutational importance indicates a lower contribution to the model overall, whereas variable percent contribution is heuristically defined and can change within the model each time it is run (Phillips, 2017). We used Area under Curve (AUC) to determines the accuracy of our models, where 0.8 to 0.9 is considered excellent. We determined the best-fit model for both pre- and post-fire periods.

Model evaluation

Previous studies have used a 0.5 presence/absence threshold (Bean et al., 2012; Chang et al., 2022) to predict the potential distribution and a threshold of 0.75 and 0.9 to predict core habitat of a species (Nelli et al., 2022). We adhered to the 0.5 presence absence probability threshold followed in other studies to present our overall models. We also ran these models at the 0.75 and 0.9 thresholds. For individual predictor variables, we were able to increase the presence/absence threshold and have stated this when done. We did this to determine which predictor variables changed, and potentially drove suitability before and after fire. We used the term “suitability” to explain the model outputs of southern greater glider occurrence on a scale of suitable (1) to unsuitable (0). A higher suitability score indicated an increase in predicted occurrence.

2.4 Results

Sampling Bias

We used the kernel density estimate to correct for sampling bias (Figure 2.2), which represent the survey effort for arboreal marsupial species during the pre-fire and post-fire time periods (Figure 2.2). The highest density sampling was in the central eastern parts of the study area (Figure 2.2). There was more survey effort in the far eastern section post fire.

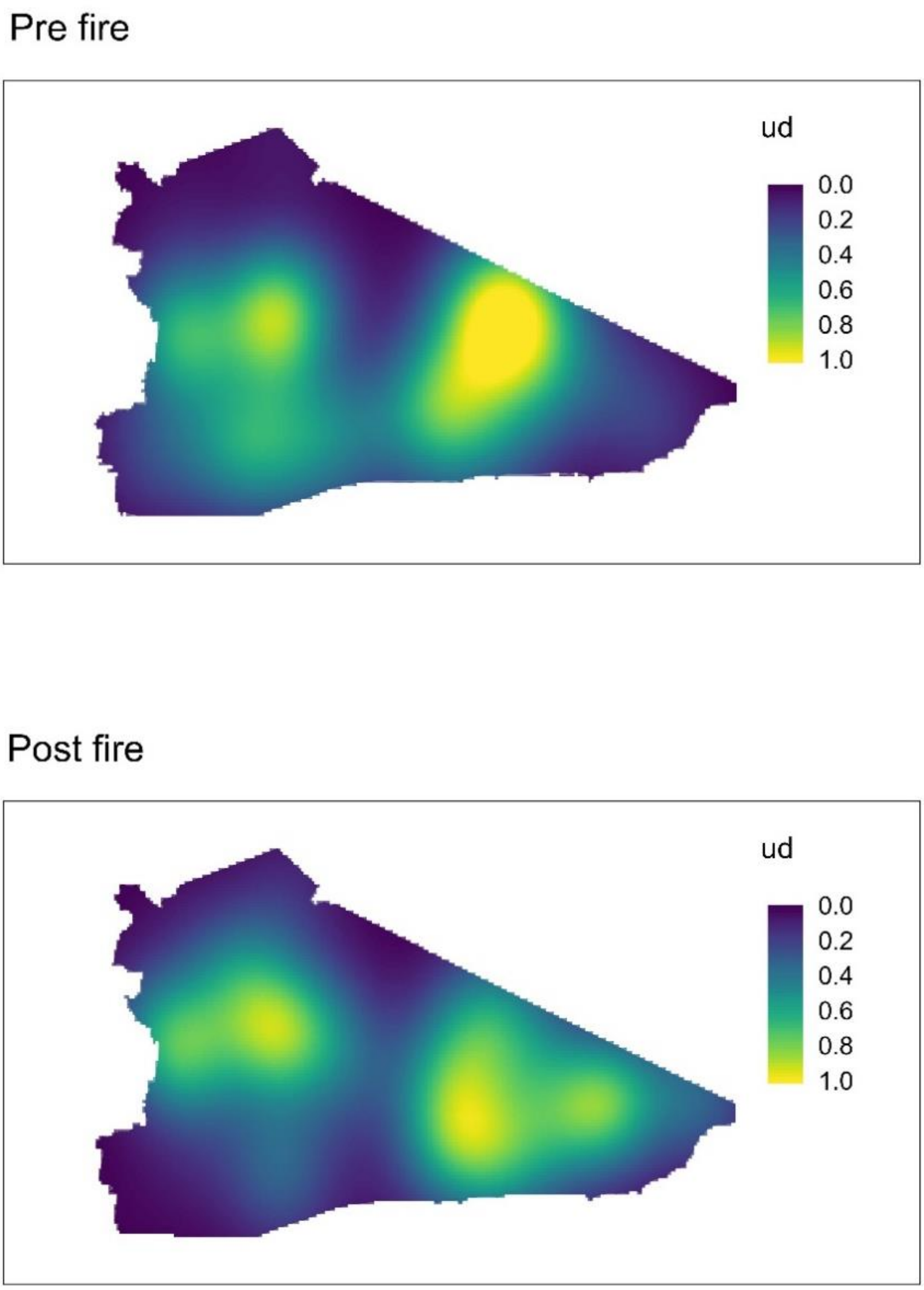

Figure 2.2: Kernel density estimate (KDE) of possum and glider species using the target background group (TGB) method as a proxy for survey effort pre (left) and post fire (right) across the study region. We used records of possums and gliders as described in the Methods section and Appendix 2.1 and 2.2. The legend shows a colour gradient of lower density (dark blue) to higher density (yellow) using the ud (utilisation distribution) scale which gives the probability distribution of the probability density that a species would be found in a particular location.

Pre-fire maxent modelling

The best-fit model for the pre-fire data contained all five predictor variables: elevation, land management, Ecological Vegetation Class, fire frequency, and time since fire (ordered from highest to lowest permutational importance; AUC= 0.89; Figure 2.3, Appendix 2.3). Permutational importance indicated that elevation was the most important variable in the model (45.44%; Figure 2.3), with time since fire being the least important (3.82%; Figure 2.3). Ecological Vegetation Class followed by elevation had the highest percent contributions to the model (Figure 2.3).

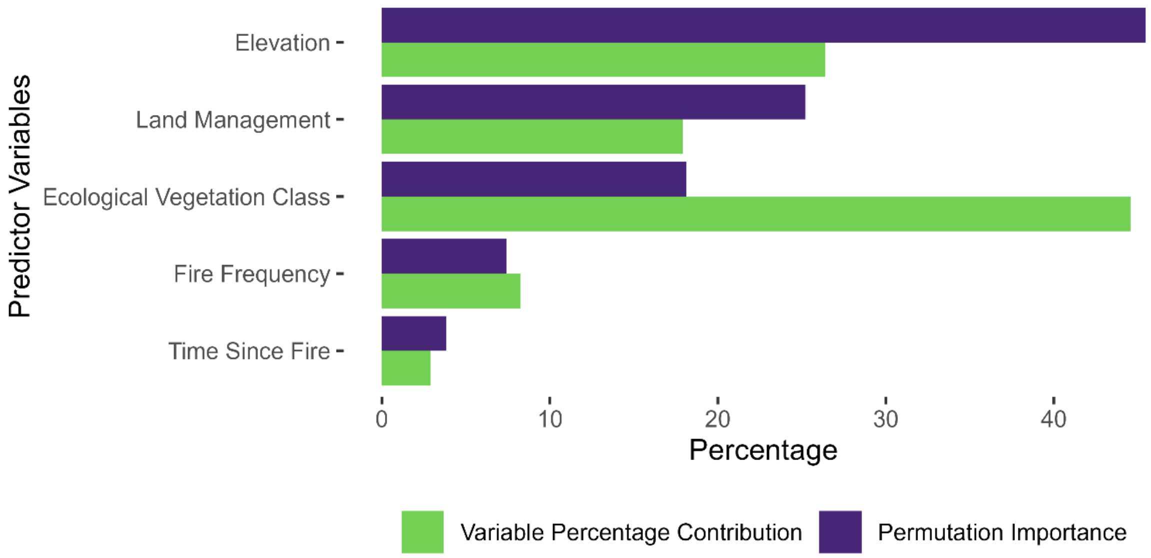

Figure 2.3: The permutation importance (purple) and variable percentage contribution (green) for each of the predictor variables in the best fitting pre-fire model.

Before the 2019/2020 megafires, only a small part of East Gippsland was predicted to be suitable for the southern greater glider (Figure 2.4). These suitable areas are primarily in the central and northern part of the region, with an additional thick band of mid to high suitable area along the coast (southern edge of map) (Figure 2.4). These hotspots aligned with concentrations of survey effort according to our sampling bias co-variate (Figure 2.2; Figure 2.4). The areas of distribution predicted for the species in our study area pre fire were 5.74% (0.5 suitability), 2.60% (0.75 suitability), and 1.20% (0.9 suitability) (Appendix 2.5).

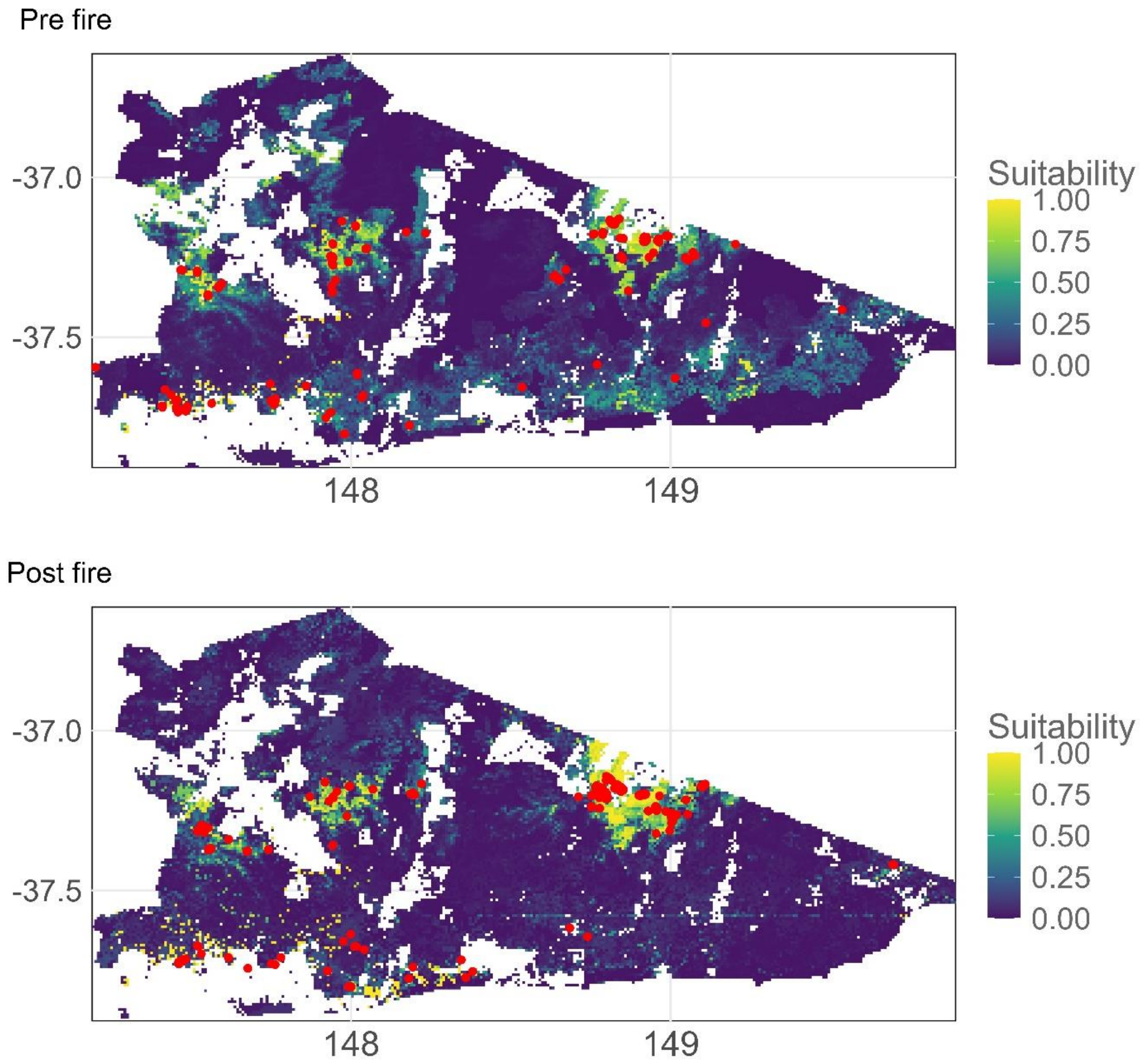

Figure 2.4: Predicted distribution for the southern greater glider before (Top; pre fire) and after (Bottom; post fire) the 2019/2020 megafires. Predicted distribution has a suitability score of 0 (dark blue) indicating unsuitable areas to 1 (yellow) indicating suitable areas. The red points are southern greater glider presence records, and the white areas are NA's and have no values attributed to them. The pre-fire model includes the five predictor variables from the best-fitting model: elevation, land management, ecological vegetation class, fire frequency and time since fire. The post-fire model includes the six predictor variables from the best-fitting model; elevation, ecological vegetation class, fire severity, fire frequency, time since fire and land management.

Post-fire maxent modelling

The best-fit model for the post-fire data contained all six predictor variables; elevation, Ecological Vegetation Class, fire severity, fire frequency, time since fire and land management (ordered from highest to lowest permutational importance) (AUC = 0.93, Figure 2.5; Appendix 2.4). Elevation had the highest permutational importance to the model (54.1%; Figure 2.5 & 2.6). Of the fire variables, fire severity was slightly (9.50%) more important than fire frequency (7.80%) and time since fire (7.30%).

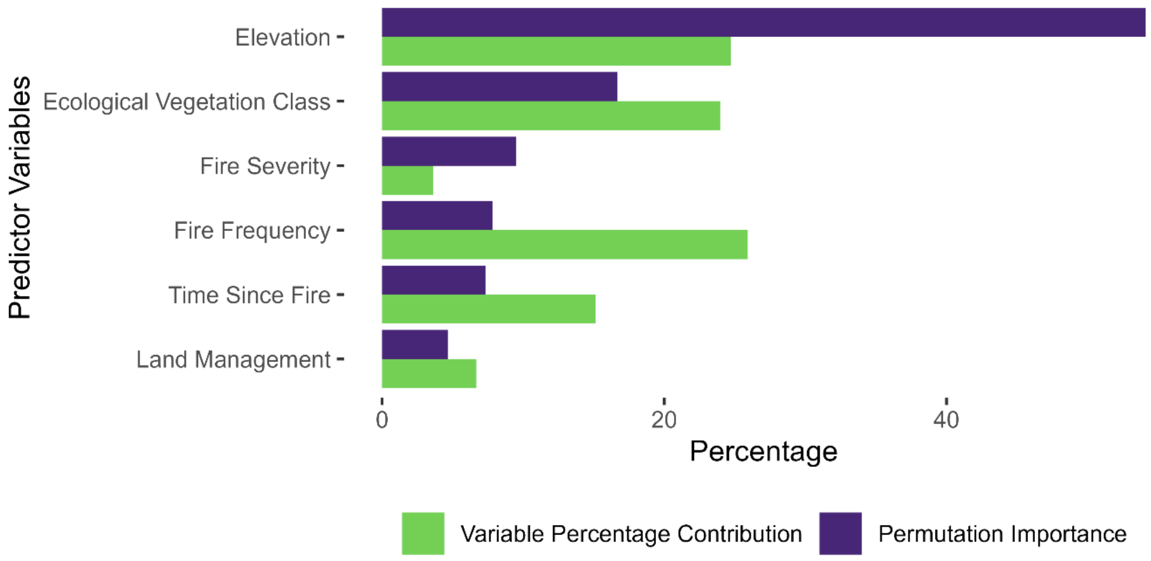

Figure 2.5: The permutation importance (purple) and variable percent contribution (green) of predictor variables for the best-fitting post-fire model.

Using the 0.5 suitability threshold, the post-fire elevational range was quite different (range: 889.40m to 986.40m) to the pre-fire model (range: 0.20m to 1358.95m). However, suitability increases at ~750m elevation before and after fire and decreases again at ~1300m (Figure 2.6). Before fire, the minimum elevation at 0.75 suitability remains at 0.20m, however, increases to above 869m at 0.9 suitability (pre fire). Whereas post fire, the elevational range is narrow above 0.5 suitability (Figure 2.6).

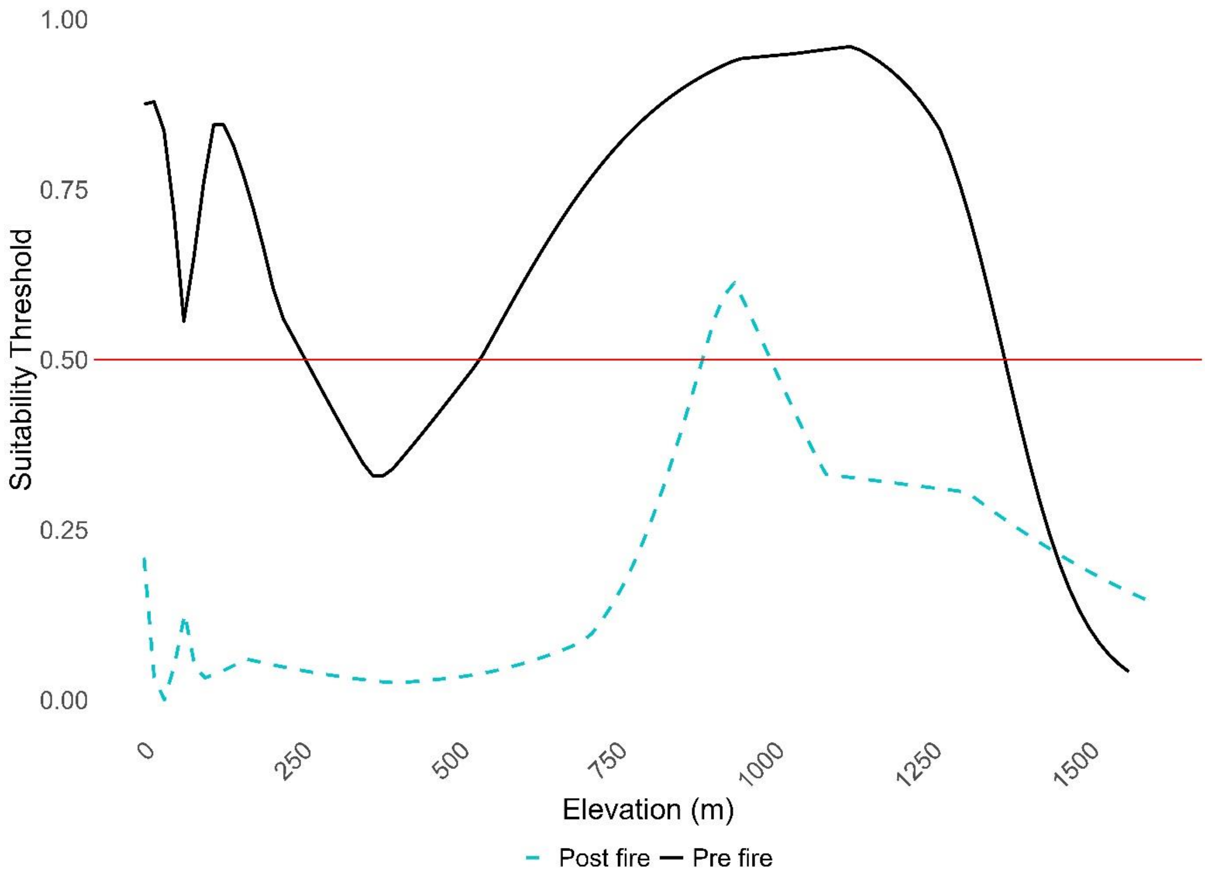

Figure 2.6: The elevational range across suitability thresholds (with 0.5 a solid red line) showing where southern greater gliders are more likely to occur before (Black solid line) and after (Blue dashed line) fire.

In the post-fire models, southern greater gliders were not predicted to occur within any areas of high severity or canopy burn (category 5; second highest fire severity category or category 6; the highest fire severity category) or in areas that had been burnt recently at suitability thresholds >0.5. Overall, the predicted areas of occurrence post-fire were slightly higher than pre-fire at 6.06% (0.5 suitability), 3.85% (0.75 suitability) and 2.66% (0.9 suitability) of the region (Figure 2.4; Appendix 2.6). Despite the overall area, the band of mid to high suitable habitat highlighted in the pre-fire model had zero to low suitability in the post-fire model.

Pre- and post-fire comparison

All variables, except for fire severity (which could only be used post fire), were predictors in the best-fitting models both pre and post fire. In both pre- and post-fire models, the most suitable areas (>0.5 suitability) were predicted to be in state forest. Elevation was permutationally more important in predicting species occurrence post fire than pre fire (pre fire 45.44%, Figure 2.3; post fire 54.08%, Figure 2.5). The time since fire variable showed that pre fire, suitability at 0.5, 0.75 and 0.9 were all in areas burnt five or more years ago. The most suitable areas (>0.5 suitability) post fire were predicted to be in areas >32 years (0.5) since last burnt, to > 40 years (0.75) since last burnt. We estimated suitable habitat remained similar following the 2019/2020 megafires with an overall 1.46% increase in predicted area of occurrence of southern greater gliders from before to after the 2019/2020 megafires. The most suitable (suitability = 1) areas after the megafires were concentrated in the western and northern parts of the study area, with coastal areas displaying low suitability (Figure 2.4). There were 70 Ecological Vegetation Classes within the 0.5 suitable predicted post-fire area, however, the majority of these made up less than five per cent of the total area. The following seven vegetation classes made up higher than 5% of suitable area respectively: Wet Forest (29%), Montane Dry Woodland (14.7%), Damp Forest (10.84%), Tableland Damp Forest (8.30%), Shrubby Dry Forest (6.90%), Montane Damp Forest (5.60%) and Lowland Herb-rich Forest (5.50%) (Appendix 2.7). Notably, a thick band of predicted marginally suitable habitat along the coast of East Gippsland in the pre-fire model was significantly reduced in the post-fire model (Figure 2.4).

2.5 Discussion

Under projected climate change scenarios, megafires are predicted to be a more frequent occurrence across Australia (Canadell et al., 2021), posing a significant threat to biodiversity (Ward et al., 2020). It is difficult to assess the impact of disturbance events such as megafires without systematic records before and after a disturbance event (Lindenmayer et al., 2013, 2022a). We found that the amount of predicted area of southern greater glider occurrence remained relatively stable in the two years following the 2019/2020 megafires across East Gippsland. However, we also found that the areas of higher predicted suitability became more concentrated and isolated, with an extensive loss of marginal or connecting habitat. Our Maxent modelling has allowed us to gain a better understanding of the variables that influence where southern greater gliders are likely to occur in the landscape following megafire. Specifically, the importance and influence of key covariates included in our models and how they changed between pre and post fire.

Estimates of direct mortality suffered by mammal species during megafire are limited (but see Jolly et al., 2022). However, high-severity fires are more likely to lead to animal mortality due to the increased heat, smoke, and chance of injury (Chia et al., 2015; McLean et al., 2018; Santos et al., 2022). Unsurprisingly, we found low levels of predicted southern greater glider occurrence within areas of high severity fire and canopy consumption, which are the highest categories of fire severity. Data on numbers of southern greater gliders that may have suffered direct mortality in East Gippsland are unavailable. However, our results suggest that direct mortality is a plausible explanation for the lack of animals predicted to be in areas that suffered canopy burns.

Severe fire can consume the canopy of *Eucalyptus*, reducing food availability for folivorous species (Banks et al., 2011), and removing structural cover and tree hollows (Chia et al., 2015; Lindenmayer et al., 2012). Koalas (*Phascolarctos cinereus*) have been found to use burnt trees three months after fire, however, the persistence of unburnt patches in the interim period was essential for their survival (Matthews et al., 2007). Food availability in unburnt areas may be of particular importance for the southern greater glider which feeds almost exclusively on *Eucalyptus* leaves (DCCEEW, 2022). Southern greater gliders can consume epicormic growth on eucalypts (Gopalan, 2022); however, until epicormic regrowth occurs, they are unlikely to survive in severely burnt areas (Smith and Smith, 2022). Understanding species nutritional requirements, and the link between food availability and occurrence is a critical factor in determining species persistence (Moore et al., 2004), particularly because southern greater gliders are likely to be more susceptible to heat effects (Rübsamen et al., 1984) than other species.

The southern greater glider is thought to be relatively sedentary compared to other arboreal marsupials (Visintin et al., 2021). For example, mountain brushtail possums (*Trichosurus cunninghami*) have been found to move from burnt to unburnt refugia after fire (Banks et al., 2011). We know little about rates of direct mortality of southern greater gliders under different fire severities. For those that do survive, we have a limited understanding of their movements post fire, however there appears to be no strong difference between home ranges in burnt and unburnt forest (Miritis et al., 2024). If southern greater gliders were able to persist in nearby unburnt areas, it may be possible for them to move into and increase their numbers in burnt areas as time since fire increases. However, this requires there to be suitable habitat and connectivity between suitable areas. More research is needed to determine how individual southern greater gliders might move or recolonise in response to wildfires. Although, recent work suggests with strong home range fidelity it is likely they are persisting in the areas they occupied prior to fire (Miritis et al., 2024).

Fire frequency and time since fire were equally important for predicting southern greater glider occurrence post fire (Figure 2.5), whereas fire frequency was more important than time since last burnt in the pre-fire model (Figure 2.3, Figure 2.4). We found that southern greater gliders were associated with areas that were long unburnt, and areas burnt between one and three times since 1980 before fire. In the pre-fire model, they were not predicted to be in areas which had been burnt more than three times. Post fire, southern greater gliders were predicted to be in areas that hadn’t been burnt at all. This supports our hypothesis that the species would be associated with areas subject to less severe and frequent fires. Important to note, is that this association could also be due to different burn frequencies for different forest types, which includes differences in land tenure. For example, logged forest has been found to burn at higher severities (Lindenmayer et al., 2022c). Fire frequency has been understudied in the context of the southern greater glider; however, it may be an important predictor of their abundance (McLean et al., 2018). As megafires and the frequency of fire increases across forest landscapes, it will be important to investigate the effects of repeated fire on the southern greater glider and other fire-sensitive species across the region. We recommend that burning operations in the form of hazard reduction or prescribed burns should be kept at a low intensity and should avoid canopy scorch.

We found that southern greater gliders before and after megafire were predicted to occur at elevations greater than ~850m (Figure 2.6). This is consistent with previous research showing there is a greater likelihood of southern greater glider occurrence at higher elevations where forest type is appropriate (Kavanagh et al., 1995; Smith and Smith, 2020; Wagner et al., 2020). Before the 2019/2020 megafires, areas of low elevation (~0.20m) were considered suitable and southern greater gliders were found in these areas. However, as we increased the suitability threshold (>0.9) areas under ~850m were no longer predicted to be suitable. As shown in Figure 2.6, there is an elevational peak in suitability between ~850 to 1250m, whereafter suitability declines.

The change in suitable elevation could be an effect of fire behaviour and a response to burnt areas, rather than elevation. Or the 2019/2020 megafires may have disproportionately affected low elevation areas (<750m) which may explain the increased species occurrence at higher elevations post fire. Alternatively, our findings could be driven, at least partly, by access and survey effort. As demonstrated by the kernel density estimate, survey effort was not consistent across the region before or after fire (Figure 2.2). In some areas, accessing sites can be a challenge due to road conditions or lack of roads through large expanses of forest which then go unmonitored.

Most of the southern greater glider distribution predicted by our analysis was within state forest, and outside of protected areas, a pattern that also was detected by Wagner et al., (2020). There has been more survey effort in Victorian State Forests due to the increased motivation by conservationists to conduct surveys in these areas for enhanced forest protection. Although correcting for bias will reduce this effect, we cannot change where data have primarily been collected. Most of the state forests where the species is predicted to occur remained unburnt after the 2019/2020 megafires or were situated along the edges of the megafire scar. Therefore, it is not surprising we found more suitable area predicted to be within state forest (Figure 2.3). To continue preserving southern greater glider habitat, the end of native forest logging in State Forest is a promising step forward for the conservation of this species.

Study Limitations

Despite our attempts to account for sampling bias, we acknowledge that there are still implications for our results. The kernel density results show there were more records/survey effort after fire than before (Figure 2.2) which could be due to many reasons. Disturbances such as wildfire create community concern for threatened species, sparking fluxes of post-fire monitoring projects and survey efforts to determine where species persist. This could explain why there is a relatively similar predicted area occurrence before and after fire. Timber harvesting regulations in Victoria were supposed to preclude timber harvesting in East Gippsland if southern greater glider numbers were observed above a certain threshold (DELWP, 2022). Therefore, more effort was concentrated in state forest to boost records relative to other land tenures. If records were concentrated in these areas and surveys were predefined to these areas, then this could influence the results pre and post fire. The Target Group Background Method, which is considered the most robust to deal with sampling bias, still assumes each species has equal detection probability (Yackulic et al., 2013). This may not hold true due to differences in the mobility of different species of arboreal marsupials. These factors are important to consider for future survey work and species distribution modelling. Understanding where and when surveys have been conducted can be used to guide a more cohesive and systematic survey effort in future. The results from this work can be utilised in developing ongoing monitoring programs, particularly in predicted high suitability areas. With the contraction of suitable habitat in a few years, it will be important to prevent future disturbance.

Models are only as effective as their inputs and previous studies have shown that different input data can change Maxent estimates (Crates et al., 2022). Expanding our work by adding more southern greater glider records, or systematic surveys before and after significant fire events across a broader area would help to increase model predictive capabilities. Using a finer-scale resolution for specific areas may also be useful to guide future research efforts and conservation planning. Focusing on suitable areas with a finer resolution may also allow modelling with additional or alternate predictor variables. For example, additional fire severity layers focussed on smaller areas could assess the impacts of fire frequency and severity on southern greater gliders over a longer time frame.

2.6 Conclusion

We found the predicted distribution of southern greater gliders experienced habitat isolation and contraction after the 2019/2020 megafires. This included a loss of marginal habitat in the south of the region burnt during the megafires. These results are a serious concern for this Endangered species. Our results indicate that high severity fire will lead to a reduction in how suitable habitat is for the southern greater glider. This will likely create further pressure on a species already subject to a range of ongoing threatening processes, such as logging, land clearing, and climate change. Our results provide a better understanding of the environmental factors that influence the distribution of this Endangered marsupial post fire. With the projected rise in temperatures and increased frequency of severe fire, it is crucial to prioritise conservation management of the predicted high-suitability areas. This includes ongoing and comprehensive monitoring to understand how best to manage the southern greater glider and similar species in post-megafire conditions.

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2.9 Supplementary Information

Appendix 2.1: List of the arboreal species in East Gippsland used for the target group background sampling bias kernel density estimations (Background Species Records).

Number	Scientific Name	Common Name	Threatened Status
1	*Acrobates spp.*	Feathertail Glider (Combined narrow-toed and broad-toed)	Least Concern
2	*Cercartetus nanus*	Eastern Pygmy-possum	Least Concern
3	*Petaurus australis*	Yellow-bellied Glider	Least Concern
4	*Petaurus breviceps*	Sugar Glider	Least Concern
5	*Pseudocheirus peregrinus*	Common Ringtail Possum	Least Concern
6	*Trichosurus spp.*	Brushtail Possum (Combined common and mountain)	Least Concern

Appendix 2.2: Count of southern greater glider and background species records before and after spatial thinning at a 250m resolution.

	Before thinning	After thinning
Southern greater glider (pre)	445	192
Southern greater glider (post)	484	208
Background animals (pre)	823	466
Background animals (post)	911	531

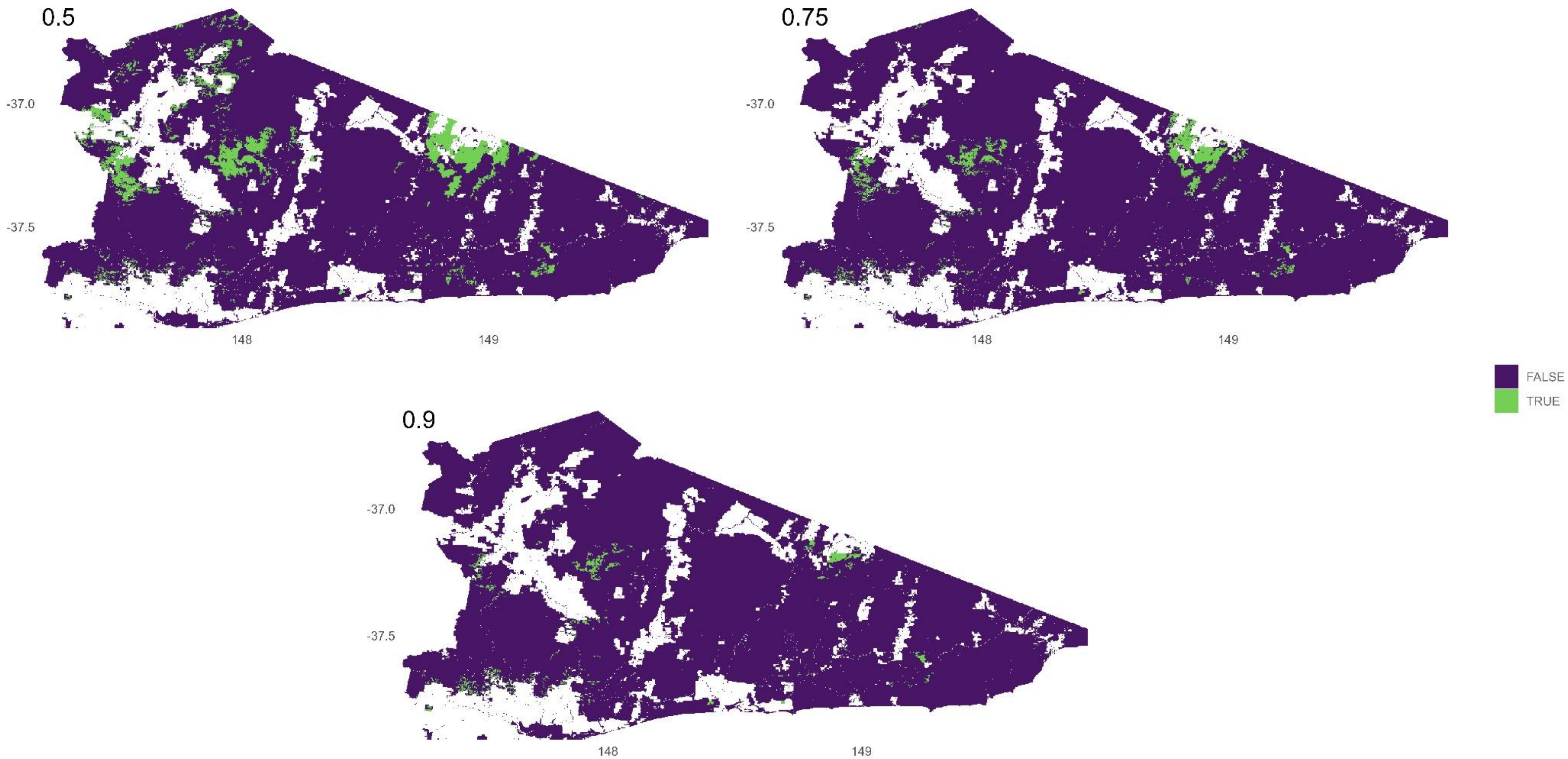
Appendix 2.3: Evaluation criterions ordered by ΔAICc for each of the pre-fire models using the AICc (Akaike's Information Criterion), AUC (area under curve) and tss (Total sum of squares). Land management (Land Mgmt) and ecological vegetation class (EVC) are both categorical variables. * Final model used for species distribution modelling.

Variables (#)	Elevation	EVC	Fire Frequency	Land Mgmt	Last burnt	ΔAICc	AICc	AUC	tss
5*	✓	✓	✓	✓	✓	0	4281.02	0.8883	0.6193
4	✓	✓	✓	✓		58.71	4339.72	0.8885	0.6116
3	✓	✓		✓		100.63	4381.65	0.8848	0.6041
2	✓			✓		262.81	4543.83	0.8387	0.5498

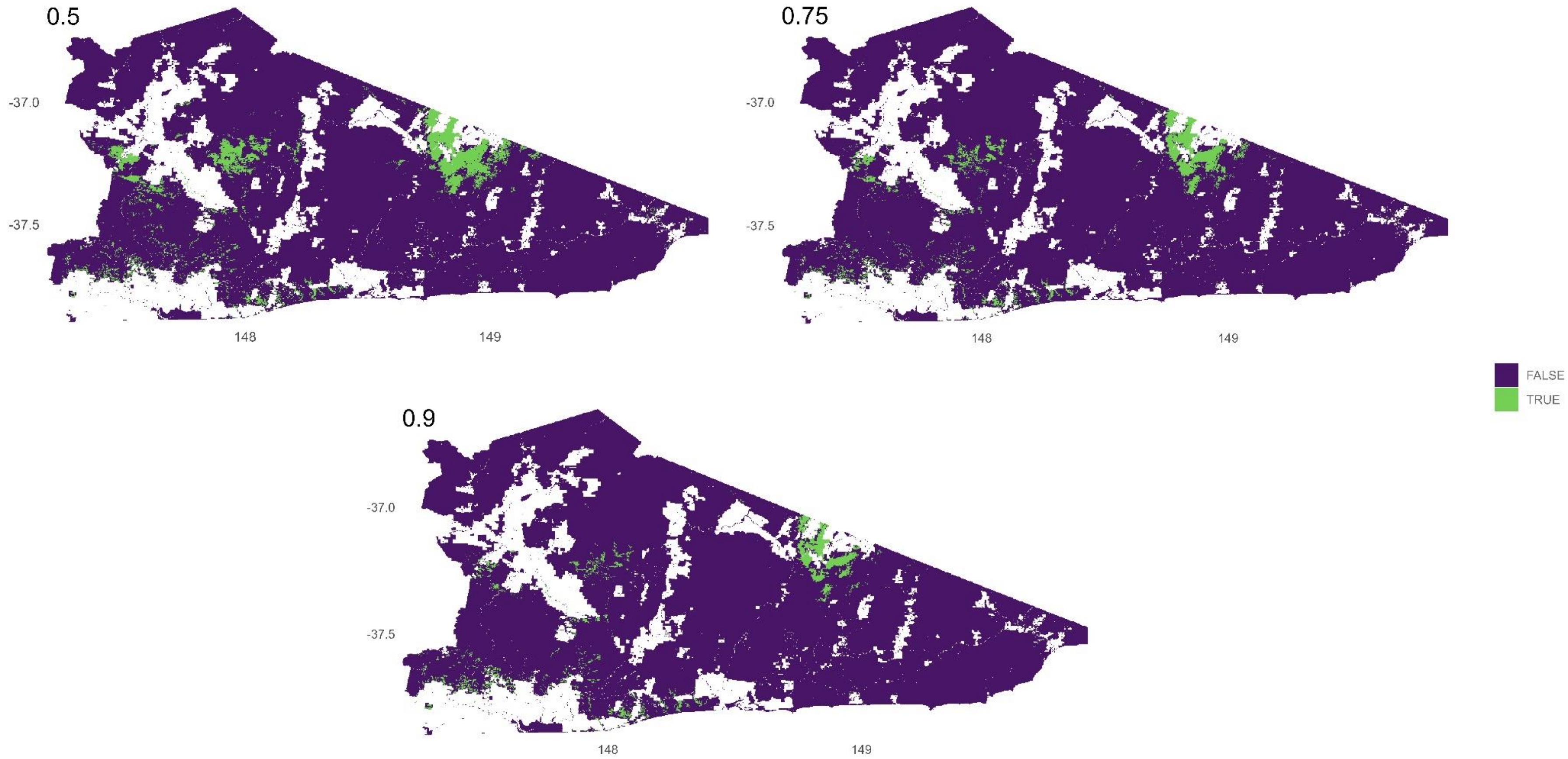
Appendix 2.4: Evaluation criterions ordered by ΔAICc for each of the post-fire models using the AICc (Akaike's Information Criterion), AUC (area under curve) and tss (Total sum of squares). Including the training and testing AUC for the best fitting (ΔAICc = 0) model. Land Mgmt and EVC are always categorical variables. * Final model used for species distribution modelling.

Variables (#)	Elevation	EVC	Fire Frequency	Fire Severity	Last Burnt	Land Mgmt	ΔAICc	AICc	AUC	tss
6 *	✓	✓	✓	✓	✓	✓	0	4858.99	0.9275	0.7320
2	✓	✓					188.18	5047.17	0.9076	0.6934
3	✓	✓	✓				273.33	5132.32	0.9258	0.7350
4	✓	✓	✓	✓			309.25	5168.23	0.9269	0.7303
5	✓	✓	✓	✓	✓		312.16	5171.14	0.9299	0.7273

Appendix 2.5: Map of predicted southern greater glider presence and absence before the 2019/2020 megafires using a 0.5, 0.75 and 0.9 suitability threshold. True (green) is suitable and false (purple) is unsuitable. The white area does not have values and are NA's.



Appendix 2.6: Map of predicted southern greater glider presence and absence after the 2019/2020 megafires using a 0.5, 0.75 and 0.9 suitability threshold. True (green) is suitable and false (purple) is unsuitable. The white area does not have values and are NA's.



Appendix 2.7: Ecological Vegetation Classes (EVC) bioregion and description. The descriptions are copied from Bioregions and EVC Benchmarks (DEECA, 2022a).

EVC	Name	Bioregion/s	Description
36	Montane Dry Woodland	Uplands	Occurs on dry and exposed mid to upper slopes at montane elevations, often extending onto ridges in high rainfall areas. Effective rainfall may be enhanced by cold air drainage, low cloud cover and in some cases, persistent snow cover. Open forest to open woodland to 15 m tall, according to altitude, precipitation and frequency of fire. The shrubby understorey is an unusual combination of species, which tolerate exposed montane conditions and species which enjoy the increased moisture available from cloud and fog drip.
30	Wet Forest	Lowlands, Uplands	Grows on fertile, well-drained loamy soils on a range of geologies and elevation levels. It is largely restricted to protected sites in gullies and on southern aspects of hills and mountains where rainfall is high and cloud cover at ground level is frequent. Characterised by a tall eucalypt overstorey to 30 m tall with scattered understorey trees over a tall broad-leaved shrubby understorey and a moist, shaded, fern-rich ground layer that is usually dominated by tree-ferns.

35	Tableland Damp Forest	Uplands	Tall eucalypt forest to 30 m tall with an understorey dominated by a tall, diverse shrub layer of broad-leaved shrubs and heaths with sometimes a scattering of tree-ferns. The ground layer is dominated by a variety of herbs and graminoids. Occurs on topographically protected south-facing slopes and occasionally in gully heads on northern or western slopes in low relief montane areas.
29	Damp Forest	Lowlands, Uplands	Grows on a wide range of geologies on well-developed generally colluvial soils on a variety of aspects. Dominated by a tall eucalypt tree layer to 30 m tall over a medium to large dense shrub layer of broad-leaved species typical of wet forest mixed with elements from dry forest types. The ground layer includes herbs and grasses as well as a variety of moisture-dependent ferns.
38	Montane Damp Forest	Uplands	Grows on a wide range of geologies on well-developed generally colluvial soils on a variety of aspects at montane elevations. Dominated by a tall eucalypt tree layer to 30 m tall over a medium to tall dense shrub layer of broad-leaved species typical of wet forest mixed with elements from dry forest types. The ground layer includes herbs and grasses as well as a variety of moisture-dependent ferns.

21	Shrubby Dry Forest	Lowlands, Uplands	Occurs on a range of geologies on exposed aspects such as ridgelines and medium to steep upper slopes, often in high rainfall areas and on shallow infertile soils. Open forest to 25 m tall characterised by the diversity and variability of the eucalypts. The understorey often lacks a secondary tree layer but contains a well-developed medium to low shrub layer. The ground layer is often very sparse with tussock-forming graminoids being the dominant life form.
877	Lowland Herb-rich Forest	Lowlands, Uplands	Primarily occurs in gullies and along lower slopes close to minor drainage lines. Soils are reasonably fertile colluvial loams. Open eucalypt forest to 20 m tall with a range of medium shrubs conspicuous in the mid-stratum. The ground layer is dominated by a dense cover of grasses and a high diversity of tussock-forming graminoids and herbs.

3.0 CHAPTER 3

Establishing foundations: Designing a long-term experiment to evaluate whether nestboxes assist population recovery of an endangered species after fire

Tallaganda National Park, Photo Credit: M Cobden

3.1 Abstract

The loss of hollow-bearing trees drives population declines for hollow-dependent species. Disturbances such as wildfire can exacerbate these declines. Artificial structures, like nestboxes, are a commonly used management tool to offset losses of hollow-bearing trees. However, the effectiveness of nestboxes as a conservation strategy is rarely tested within an experimental framework. The southern greater glider (*Petauroides volans)* was recently uplisted to Endangered, in part due to direct mortality and habitat degradation associated with the 2019/2020 megafires. To test whether purpose-built nestboxes could assist the post-fire recovery of the southern greater glider, we established a long-term landscape-scale experiment. Purpose-built nestboxes (n= 234) were installed at 68 sites in East Gippsland (north-eastern Victoria) and Tallaganda (southern New South Wales) in areas burnt during 2019/2020. Each nestbox site was matched with a control (no nestboxes) site and camera traps were installed at a subset of nestboxes in both study areas. Here, we present the experimental design and, using data from the first two years of the study, evaluated whether; 1) southern greater gliders or 2) other non-target species used purpose-built nestboxes, 3) southern greater glider numbers were higher in areas with nestboxes compared to those without, before treatment or after the first survey round, and (4) fire severity impacted southern greater glider abundance in our study areas. Southern greater gliders were observed using nestboxes in both regions, with substantially more observations in Tallaganda. We fitted generalised linear mixed models to the spotlighting data. No significant difference was found in southern greater glider abundance between treatment and control sites. We found more southern greater gliders in areas of lower fire severity than high fire severity in Tallaganda. Our study demonstrates the potential of nestboxes to assist in the recovery of southern greater gliders following disturbance. Given the slow natural breeding rate of southern greater gliders, ongoing monitoring is required to determine whether nestboxes are an effective management tool for the species in the long term.

3.2 Introduction

Hollow-bearing trees play a critical role as habitat for cavity-dependent wildlife in many forest ecosystems (Gibbons and Lindenmayer, 2002; Goldingay, 2009, 2012; Lindenmayer et al., 1991; Saunders et al., 1982; van der Hoek et al., 2017). Tree hollows are a valuable habitat resource for fauna, providing warmth, shelter and safety from predators, and are critical to sustaining viable populations (Gibbons and Lindenmayer, 2002; Goldingay, 2012). In Australia, hollows that serve these functions can take between 120 to 250 years to develop in eucalypt trees, and, once lost are difficult to replace (Gibbons and Lindenmayer, 2002; Koch et al., 2008; Lindenmayer et al., 1993; Wormington et al., 2003).

Many Australian ecosystems have experienced declines in the abundance of tree hollows (Lindenmayer et al., 2018) stemming from major disturbances such as logging (Gibbons et al., 2010) and wildfire (Banks et al., 2011; Lindenmayer et al., 1990; Lindenmayer and Sato, 2018). Fire can directly reduce hollow abundance through hollow bearing trees being consumed or killed (Lindenmayer et al., 1993, 2012), but conversely, fire can be a hollow-forming process (Haslem et al., 2012). In areas burnt by moderate or high severity fire, losses of hollow bearing trees may be especially pronounced compared to areas affected by low severity fires (Lindenmayer et al., 2012; May-Stubbles et al., 2022; Wagner et al., 2024).

Artificial cavities provide a fast alternative to the slow, natural development of tree hollows, and augmenting habitat with nestboxes is a common management approach to assist hollow-dependent species (Beyer and Goldingay, 2006; Cowan et al., 2021; Goldingay and Stevens, 2009; Watchorn et al., 2022). Using nestboxes has proven to be successful for numerous threatened species conservation projects. For example, nestbox provisioning was identified as a key conservation action for the critically endangered Leadbeater’s possum (*Gymnobelideus leadbeateri*) (Harley, 2023). Nestboxes were also successfully used by the endangered swift parrot (*Lathamus discolor*) (Stojanovic et al., 2019).

Although nestboxes are frequently used to support a range of threatened, hollow-dependent species (Goldingay et al., 2020), few studies have used rigorous and reliable experimental designs to accurately measure the long-term impacts of nestboxes on population trajectories (Cowan et al., 2021; Macak, 2020). There have also been several nestbox projects with unexpected results. For example, a study of Victoria's Mountain Ash (*Eucalyptus regnans*) forests, found the majority of nestboxes (68.8%) were not used or occupied by any arboreal marsupials (Lindenmayer et al., 2009). Moreover, limitations and unintended outcomes from nestbox provision can occur, including uptake by non-target species (Lindenmayer et al., 2009), thermal stress (Flaquer et al., 2014), mortality (Catry et al., 2011; Cowan et al., 2021) or lack of use by the targeted species (Lindenmayer et al., 2017).

Extreme drought conditions in 2018 and 2019 (Boer et al., 2020), preceded an eight-month period of unprecedented megafires across south-eastern Australia (Davey and Sarre, 2020). These megafires, referred to as the 2019/2020 megafires, burnt ~8.9 million hectares of native forest (Boer et al., 2020; Davey and Sarre, 2020). The loss of critical habitat features was amongst the threats identified as a concern for 60 species after the 2019/2020 megafires (Geary et al., 2022). Post fire, there was concern about the conservation of several species (Ward et al., 2020).

Southern greater gliders (*Petauroides volans*) are hollow-dependent nocturnal arboreal marsupials that occupy eucalypt-dominated forests in eastern Australia (DELWP, 2019; Maloney and Harris, 2008). Population declines have been observed across their range (DELWP, 2019) and the 2019/2020 megafires further reduced these populations (DCCEEW, 2022), prompting their uplisting from Vulnerable to Endangered (DCCEEW, 2022).

Here we describe the establishment of a long-term experiment to assess the efficacy of nestbox installation as a tool to support the southern greater glider in recently burnt habitat. Nestboxes are sometimes recommended as a conservation action for arboreal marsupial species in areas affected by fire (Geary et al., 2022; Miritis et al., 2024).

Using a combination of baseline data and the first round of survey data collected post deployment, we asked the following questions:

1. Do southern greater gliders use our purpose-built nestboxes?
2. Did non-target species use nestboxes purpose-built for southern greater gliders?
3. Did southern greater glider abundance increase in treatment sites (i.e., with nestboxes) compared to control sites (i.e., without nestboxes) post-treatment?

4. Does fire severity affect the abundance of southern greater gliders in the study area?

We expected southern greater gliders to use our purpose-built nestboxes based on their successful use of a similarly designed prototype deployed in unburnt East Gippsland forests (pers. obs. D Liepa). We postulated that the modified design with a smaller entrance size, would exclude large competitors, such as brushtail possums (Goldingay et al., 2020; Le Roux et al., 2016). We expected that high fire severity would affect southern greater glider abundance negatively, based on the findings of other post-fire studies (Green et al., 2024; May-Stubbles et al., 2022). Given that southern greater gliders have high site fidelity, relatively small home ranges, and are slow to reproduce (Henry, 1984; Miritis et al., 2024; Pope et al., 2004), we did not expect to see an increase in southern greater gliders in areas with nestboxes in the short time span of our study.

3.3 Methods

Study Region

We conducted this study in eucalypt-dominated forests in two study areas within the 2019/2020 megafire footprint (Figure 3.1). We established one study area in the East Gippsland Shire (hereafter called East Gippsland), which is the furthest east region of Victoria, and located on the lands of the Bidwell, Yuin, Gunnaikurnai and Monaro (Ngarigo) First Nations people. Our study was situated in a small part of the wider East Gippsland region around the Errinundra Plateau in the East Gippsland Uplands Bioregion (DEECA, 2022), near the town of Bendoc (37.1468° S, 148.8816°), approximately 10km south of the NSW border (Figure 3.1). The overstorey in these forests is dominated by shrubby dry, damp, and wet forest and receives an average rainfall between 800 – 1200mm, increasing to 1400mm in some of the areas around the Errinundra plateau (DEECA, 2022; VicFlora, 2023). The key overstorey species commonly include a mix of stringybark, gum, and ash-type eucalypt species with a broad-leaved shrubby understorey mix, particularly in the wetter areas (DEECA, 2022). The East Gippsland population of southern greater gliders has declined substantially in recent years and persists at relatively low densities (DELWP, 2019).

Our second study area was at Tallaganda National Park and State Conservation Area (hereafter called Tallaganda) on the lands of the Ngarigo and Walbanga First Nations people, located 30km southeast of Canberra (35.2802° S, 149.1310° E) (Figure 3.1; NSW National Parks and Wildlife Service, 2011). Tallaganda is on the Great Dividing Range of eastern Australia and has mixed vegetation types, with both moist and dry sclerophyll forest (NSW National Parks and Wildlife Service, 2011). Dominant overstorey species include *E. fastigata* (Brown Barrel), *E. radiata* (Narrow-leaved peppermint) and *E. viminalis* (Ribbon gum), with an understorey dominated by *Acacia* spp. (NSW National Parks and Wildlife Service, 2011). The elevational range in Tallaganda is 700mm to 1400mm (NSW National Parks and Wildlife Service, 2011) and the average annual rainfall is typically ~900mm (Belcher et al., 2007). Detection rates of southern greater gliders in Tallaganda are typically high (Goldingay and Daly, 1998), and population density is also considered to be high (NSW National Parks and Wildlife Service, 2011).

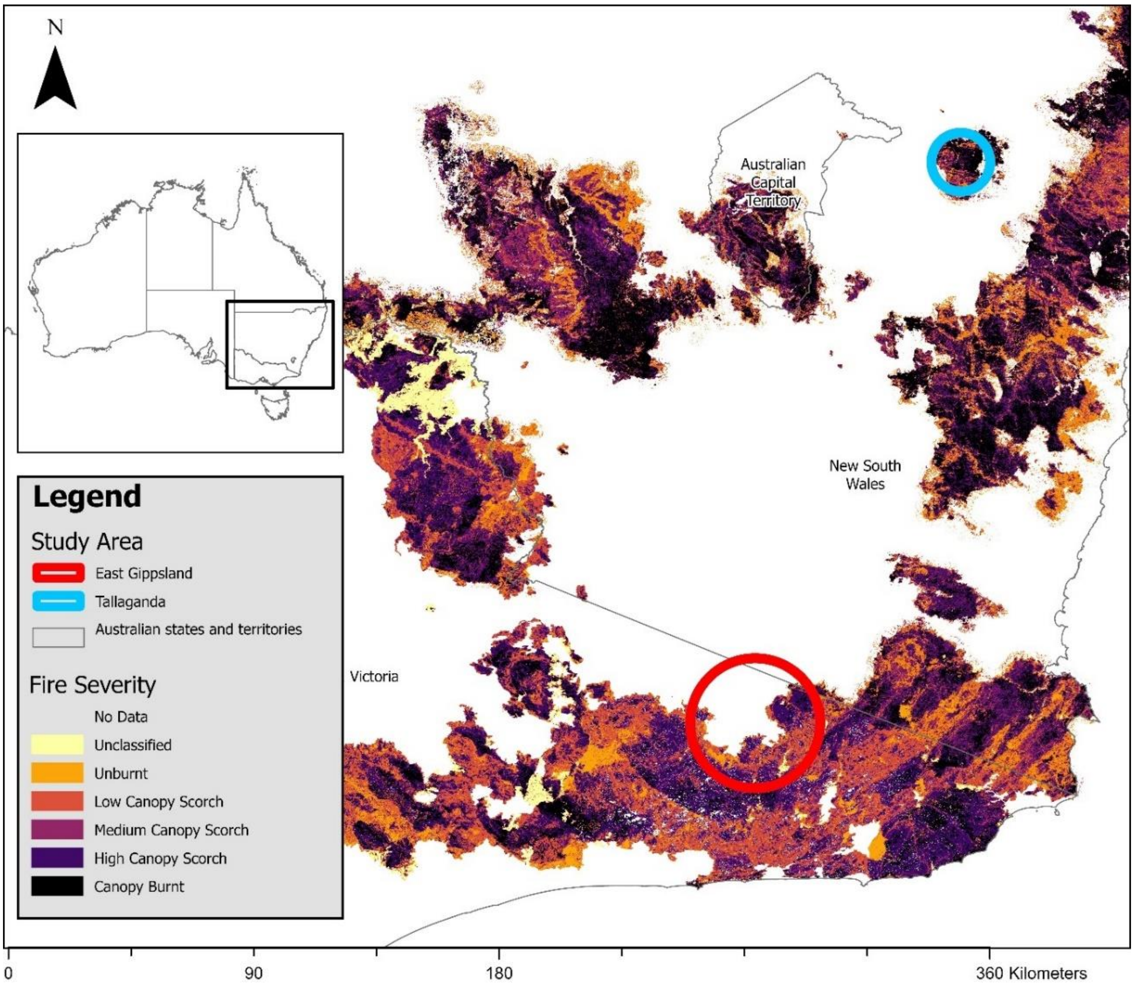


Figure 3.1: Areas targeted for studies of the southern greater glider (East Gippsland = red; Tallaganda = blue), with the 2019/2020 megafire scar displaying the discrete fire severity classes (DCCEEW, 2020; DEECA, 2022b). The inset map of Australia has a square around the highlighted study region in the main map. The study area circles represent the area covered by all the individual study sites in each area.

Pre-site selection surveys

To guide site selection, we compiled historical locations of southern greater gliders (Atlas of Living Australia, 2024b; Gabarov, 2015; Goongerah Environment Centre and Fauna and Flora Research Collective, 2016; Hill and Hanson, 2016; Lincoln and Hill, 2016; Victorian Biodiversity Atlas, 2023), and post-fire southern greater glider observations (pers. obs. L Durkin/Arthur Rylah Institute). We then confirmed locations where southern greater gliders persisted post megafire, using spotlighting surveys throughout East Gippsland (Appendix 3.1) and Tallaganda (Appendix 3.2). Surveys began one hour after sunset, and we identified all arboreal marsupials (Appendix 3.3) and recorded the distance and bearing to the observed animal(s). Where southern greater gliders were confirmed as present, we selected our individual study sites within 1.5km of these locations, according to the additional stratified criteria outlined below.

Study sites

In East Gippsland, we selected 40 sites, consisting of 20 matched pairs. We matched one treatment site (i.e. nestbox; n = 20) to one control site (n = 20) based on site and environmental attributes (Figure 3.2, Appendix 3.4). All sites were systematically selected and ground truthed. Due to a lack of southern greater glider observations in areas burnt in the 2019/2020 megafires, we located some sites within a 2019 or 2021 prescribed burn scar. In Tallaganda, we selected 30 sites within the 2019/2020 megafire burn extent (Figure 3.2, Appendix 3.5). Due to the comparatively smaller extent of the burn area in Tallaganda we had a 2:1 ratio, with two nestbox sites matched to one control site. All sites in both study areas were >1km apart to reduce the likelihood of the same southern greater glider utilising multiple sites.

We paired sites based on Ecological Vegetation Class (EVC) (DCCEEW, 2010; DEECA, 2018), fire severity (DCCEEW, 2020; DEECA, 2022b), and forest age (DEECA, 2023) obtained from field assessments and spatial layers. We selected sites in areas that were not part of logging plans in the next five years, and which supported trees >30 years old. We created matched pairs of sites and then, using a random number generator, allocated one site in each pair as a treatment site and the other as a control site.

We established 300m long off-track transects with flagging tape and reflective markers at each site. To minimise edge effects and road disturbance, the start of each site was >25m from a road or track. In some areas, we had to establish the transect parallel to a road. Where this occurred, all transects and nestboxes were a minimum of 50m from the road/track.

We collected environmental data during site establishment. This included measuring the percentage of the site that was burnt (percentage burnt) and whether canopy foliage was present or absent (canopy burnt).

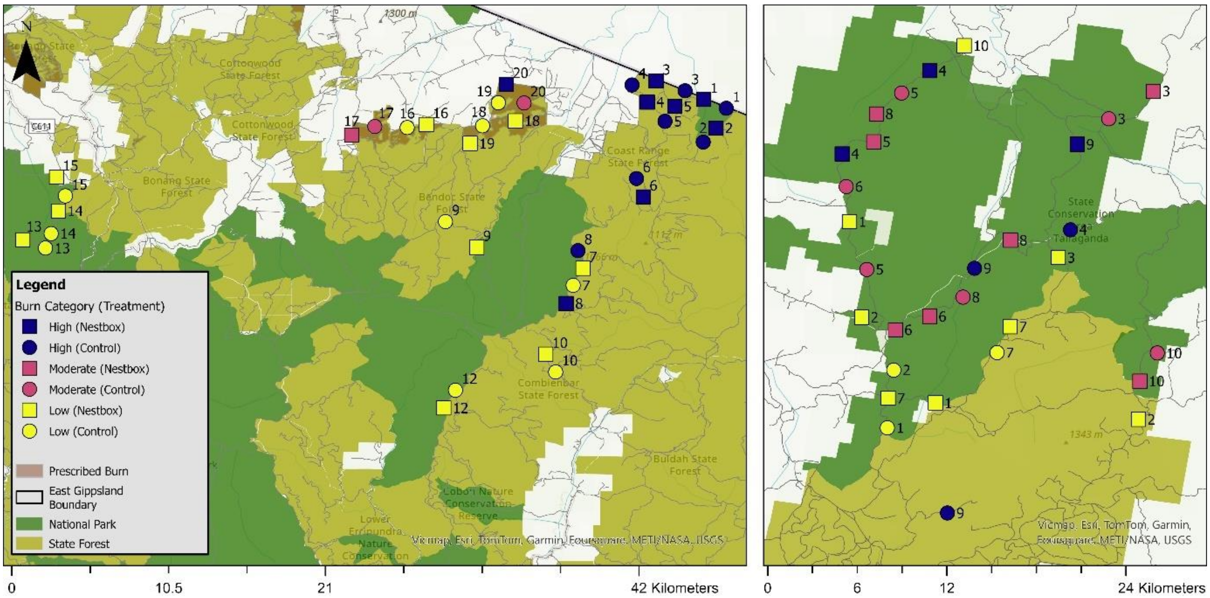


Figure 3.2: Nestbox (square) and control (circle) sites across (Left) East Gippsland and (Right) Tallaganda. We used burn percentage to assign sites to one of three fire severity classes: low (<35%; yellow), moderate (36 to 70%; pink) and high (>71%; purple). The numbers represent the matched pairs or triples.

Spotlight surveys

In East Gippsland, we surveyed all treatment and control sites prior to nestbox deployment (Round 1; 04/01/22 to 27/03/2022; 40 sites). In Tallaganda, we surveyed 10 nestbox sites prior to or within two days of nestbox deployment. However, due to constraints because of Covid-19 and weather, we were unable to survey 10 nestbox sites prior to nestbox installation.

Our spotlighting surveys followed the double-observer survey method (see Cripps et al., 2021 for full methodology). We walked at a pace of approximately 10min/100m (i.e. 30 minutes for 300m), and the second observer began surveying a transect 15 minutes after the first observer had started. The double observer method assumes animals do not move between observer 1 and observer 2’s survey periods, and that animals seen independently by both observers are not double counted. We recorded the colour morph, position in the tree, the observer’s location (using Avenza maps, Avenza Systems Inc, 2017) and the bearing and distance to the animal from the observer’s location. The bearing was recorded using a compass and the distance using a laser range finder. Immediately after each survey, both observers used this information to identify each southern greater glider along the transect and ensure that individuals were not double counted. This allowed us to be confident in counting ‘independent’ sightings. However, this was more challenging to implement in Tallaganda than in East Gippsland due to the high-density of ‘black and white’ morph individuals on a single transect. Due to inconsistencies between survey rounds, and the independent detection challenges, we chose not to analyse the double observer data using distance sampling in this study. More accurate data from future survey rounds will enable the use of distance sampling, utilising the double-observer method to its full potential.

Post-deployment surveys

We conducted spotlighting surveys on the transects in East Gippsland almost one year post-nestbox deployment using the same methodology as pre-deployment surveys (Round 2; 16/01/2023 to 6/02/2023; 36 sites). During nestbox deployment, we removed two sites because of accessibility constraints, and we were unable to access an additional two sites in the post-deployment survey round. This left 36 East Gippsland sites for analysis (18 nestbox, 18 control sites). We completed spotlight surveys at the Tallaganda sites in late October/early November of 2022, approximately 6 to 7 months post-nestbox deployment. We surveyed all 30 sites using the same methodology as pre-deployment surveys and used all 30 sites in the analysis.

Survey indices

From our field surveys, we calculated the highest count of independent southern greater gliders seen per survey, using the most southern greater gliders seen by either observer per survey. Dependent young (in pouch or on back) were not counted as independent from their mother.

Nestbox deployment

We installed purpose-built nestboxes targeted for southern greater gliders (Howard et al., 2022). Southern greater gliders are heat sensitive (Rübsamen et al., 1984; Wagner et al., 2020; Youngentob et al., 2021). Nestboxes have been shown to be hotter than natural hollows and may contribute to heat stress (Rowland et al., 2017). The nestbox design also incorporated a relatively small entrance size suitable for entry by southern greater gliders but which preclude access to larger arboreal marsupials (Goldingay et al., 2020; Le Roux et al., 2016) but large enough to potentially be less appealing to very small arboreal marsupials (Beyer and Goldingay, 2006).

We installed six nestboxes at each treatment site; two nestboxes each within 50m of the 50m,150m and 250m (Appendix 3.6). We deployed one nestbox per tree. We used tree-climbers to install nestboxes with drills and screws above 8m on the south-eastern aspect of mature eucalypt trees (Goldingay, 2015; Larson et al., 2018) which ideally had a trunk width exceeding the width of the nestbox. We deployed 120 nest boxes (6 per transect; 20 transects) in Tallaganda between the 21st of March and the 2nd of April 2022 and 114 nestboxes (6 per transect; 19 transects) in East Gippsland between the 28th of March 2022 and 12th of May 2022.

Camera trapping

We installed 54 Reconyx IR cameras (HP2X Professional) with a 900-1200mm focus on a subset of sites in East Gippsland (14 sites, 28 cameras) and Tallaganda (13 sites, 26 cameras). At each site, we installed one camera on one nestbox at the 50m point along a transect and one camera on one nestbox at the 250m point along a transect. Previous work has found that southern greater gliders may experience increased competition from larger species of arboreal marsupials at habitat edges (Youngentob et al., 2012). Therefore, our placement of nestboxes may allow us to investigate this potential competition issue in the future. We set cameras to ‘night only’ to conserve battery life, on the trigger setting, with two pictures per trigger and no delay. We installed cameras during nestbox installation in East Gippsland, and between 12th to the 16th of July 2022 at Tallaganda, approximately three to four months after nestbox installation.

Image coding

To sort and code our images, we used Reconyx Map view Professional (Reconyx, 2021) and Timelapse (Greenberg, 2023). We discarded day and blank images. We sorted images into ‘mammal or ‘no mammal’ and discarded those with no mammal present. We coded each individual image with species, behaviour (on tree, on nestbox, looking at/in nestbox, inside nestbox, entering or exiting nestbox), colour morph (if southern greater glider), age (adult/juvenile/juvenile on back of adult). We then sub-categorised the behaviour images into how the nestbox was being used: “use nestbox” (in, entering, exiting), “interact with nestbox” (looking at/in, on nestbox) and “no interaction with nestbox” (on tree, jumping/gliding). We used one image as a single detection but used presence per day as our main index of use. Only one southern greater glider was counted per day unless there were multiple individuals in one image, or there were characteristics that enabled differentiation e.g. different colour morphs. Upon reviewing the data, we found that seven cameras in East Gippsland were set up with half the nestbox entrance outside the field of view. We chose to include these data in our analysis, however, the set up may have impacted identification of some species, as less of the animal was visible. In Tallaganda, there were six cameras where the SD card filled up prior to the retrieval date and this likely impacted the amount of data collected.

Statistical analysis

Camera Monitoring [Question 1 and 2]

We used the CamtrapR program (Niedballa et al., 2024) and the packages dplyr (Wickham et al., 2023) and ggplot2 (Wickham, 2016) to create summary statistics of the camera trapping data in R (R Core Team, 2023). We investigated the occupancy of nestboxes by target and non-target species. We were unable to gather data on the time period between nestbox installation and use by southern greater gliders at Tallaganda given the ~3-month delay in camera deployment at that location. We calculated the number of trap nights and filtered the image datasets based on the start and end dates, removing any problem periods. Problem periods included times where the camera had stopped taking images or had been moved significantly from the original field of view.

Spotlight Surveys [Question 3 and 4]

We fitted generalized linear mixed models, implemented in the glmmTMB package (Brooks et al., 2024) in R, to test for the effects of nestbox installation, and fire severity on the abundance of southern greater gliders. A Poisson error distribution was assumed. We used *highest count* of southern greater gliders as the response variable and analysed Tallaganda and East Gippsland data separately.. We used *treatment* (nestboxes or control) to define the category of each site and *round* (pre or post) to define whether the survey was completed prior to (pre; round 1) or after (post; round 2) nestbox deployment. The effect of *burn percentage* (%), a continuous variable recorded in the field to represent the percentage burnt at each site was tested. Both *site ID* and *rep* (the matched sites) were included as random intercept factors to allow for dependence between repeated surveys at the same site. Model performance was assessed by inspecting the histograms of residuals and checking for overdispersion and zero inflation using the ‘performance’ package in R (Lüdecke et al., 2021).

For Tallaganda, we used a corrected *round* due to not all surveys being completed prior to nestbox installation. Therefore, all surveys conducted before or within two days of nestbox deployment were counted as pre- and any conducted afterwards were counted as post-deployment surveys.

3.4 Results

Pre-Site Selection Surveys

We found a marked difference in abundance of southern greater gliders between East Gippsland and Tallaganda. Overall, we detected 34 individual southern greater gliders in ~50km of surveys in East Gippsland (Appendix 3.1) and 84 individual southern greater gliders in ~ 8km in Tallaganda (Appendix 3.2). There were many historical records of southern greater gliders in East Gippsland, however, southern greater gliders were found in very few of these areas. The difference in distances traversed was also due to a much larger area surveyed within East Gippsland and the addition of some driving transects.

Camera trapping at nestboxes

Question 1: Did southern greater gliders use purpose-built nestboxes?

East Gippsland
We deployed cameras for 4,983 trap nights in round 1, and 8,756 in round 2 of surveys giving a total of 13,739 trap nights. Feathertail gliders (*Acrobates pygmaeus*) were detected the most, followed by mountain brushtail possum, southern greater glider, and inland sugar glider (*Petaurus notatus*) (Table 3.1). Overall, six species of arboreal marsupials were detected, plus Antechinus and bats, which were not identified to species level.

Cameras detected southern greater gliders at nine of 28 individual nestboxes (32%), five of 14 sites in East Gippsland. The shortest period for detecting a southern greater glider at a nestbox after installation was 10 days. We did not observe definitive use of the nestbox (i.e. entering, exiting or inside) until 307 days after installation (Figure 3.3; Appendix 3.7). Two of the nine cameras at nestboxes (22%) with southern greater gliders did not have the entrance in full view, which may have resulted in some aspects of behaviour or nestbox occupancy being missed. Only one camera detected nestbox occupancy (entering/exiting/inside) by a southern greater glider (Appendix 3.7). This nestbox was seen occupied by the inland sugar glider prior to the southern greater glider observation (Appendix 3.7).

Table 3.1. The relative frequency (number of images/total trap nights) and percentage of trap nights each fauna species was detected at each study site during camera trapping. There were 13,739 trap nights in East Gippsland and 2,350 trap nights in Tallaganda. The numbers in bold are the highest for each column.

		East Gippsland		Tallaganda	
Scientific Name	**Common Name**	**Relative Frequency**	**Trap nights detected (%)**	**Relative Frequency**	**Trap nights detected (%)**
Antechinus spp.	Antechinus spp.	0.0028	0.12	0.002	0.04
Unknown	Bat spp.	0.0081	0.22	0	0
Trichosurus vulpecula	Common Brushtail Possum	0	0	0.002	0.04
Acrobates pygmaeus	Feathertail Glider	**0.4885**	**10.39**	0.032	1.49
Petaurus notatus	Inland Sugar Glider	0.1091	1.66	0.009	0.34
Trichosurus cunninghami	Mountain Brushtail Possum	0.1197	0.37	0	0
Petauroides volans	Southern Greater Glider	0.1091	0.72	**3.031**	**22.85**

Petaurus australis	Yellow-bellied Glider	0.0023	0.08	0	0

Figure 3.2: (Left) Camera installed facing a nestbox in East Gippsland (Photo Credit: M Cobden), (Right) Southern greater glider inside purpose-built nestbox, while another greater glider looks toward it from the tree.

Tallaganda

We deployed cameras at nestboxes for a total of 2,350 trap nights. During this time, some cameras collected data for a short period (<30 days, n = 5). Five cameras did not detect any arboreal marsupials. We observed repeated use of nestboxes by the southern greater glider, with the relative frequency from camera images being 3.03, far higher than any other species (Table 3.1). Initial observations of nestbox occupancy were during camera deployment in July 2022 (~3 months after nestbox deployment). Cameras detected southern greater gliders at 16 of 26 nestboxes (61.5%), which was 12 of 13 sites. We confirmed that southern greater gliders were using (entering/exiting/inside) nine of the 26 monitored nestboxes across the study period of ~six months (37.5%; Appendix 3.8).

We observed southern greater glider offspring on camera at two nestboxes (Figure 3.4). They appeared to be on the back of the mother and spent time moving in and out of the nestbox. At one site (03B), the offspring was seen on camera for 31 days over a 47-day period (From 25/09/2022 to 11/11/2022) and 14 days over a 34-day period (From 31/10/2022 to 3/12/2022) at the second site (07A). We occasionally observed two similar sized southern greater gliders entering and exiting nestboxes.

At two nestboxes, we observed two adults on several occasions in August-September and October-November.

Figure 3.4: Southern greater glider female adult and offspring at nestbox sites in Tallaganda. (a) Mother and offspring, and (b) offspring climbing the tree outside of the nestbox.

Question 2: Did other species occupy purpose-built nestboxes?

East Gippsland

Inland sugar gliders occupied eight nestboxes, and a mountain brushtail possum (*Trichosurus cunninghami*) occupied one nestbox (Figure 3.5). We observed inland sugar gliders using two of the eight nestboxes for 21 (23/10/2022 to 21/03 2023) and 46 days (30/11/2022 to 09/08/2023) (Appendix 3.7). Both nestboxes were along the same transect (Site 17NB; Appendix 3.7). We observed inland sugar gliders with their offspring in nestboxes (Figure 3.5) and bringing vegetation into the nestboxes. We also detected a mountain brushtail possum using a nestbox (06NB_250m) from 19/12/2022 to 12/09/2023. We observed several bats on camera, although it is unknown if they used nestboxes to roost.

Tallaganda

At both Tallaganda and East Gippsland (Figure 3.5), we interpreted feathertail gliders as ‘using’ the nestbox, observing them going in and out of different nestboxes. This species’ movements were more rapid than those of southern greater gliders, exiting soon after they seemed to enter. We assumed, therefore, that they were not occupying these nestboxes

Figure 3.5: Mammal species detected at nestboxes in East Gippsland. A) *Antechinus* spp. inspecting nestbox, B) Bat spp.in a nestbox entrance, C) Feathertail glider in a nestbox entrance, D) Mountain Brushtail Possum exiting nestbox, E) Inland sugar glider and dependent young inside a nestbox, F) Yellow-bellied glider on top of nestbox.

Spotlighting surveys

East Gippsland

We observed a total of 29 southern greater gliders across both spotlighting survey rounds in East Gippsland (i.e. 36 sites, surveyed twice each). We detected southern greater gliders in 27.8% of pre-deployment surveys and 22.2% of post-deployment surveys. The highest number of southern greater gliders detected at one survey site was six (site 16C).

Tallaganda

We detected 245 southern greater gliders across 60 surveys (i.e. 30 sites surveyed twice each). We detected southern greater gliders in 90% of pre-deployment surveys and 93.3% of post-deployment surveys; the highest number detected was 14 (1 survey during post-deployment surveys; site 02B). The mean number of southern greater gliders observed per transect was four.

Question 3: Did southern greater glider abundance increase in treatment sites (i.e., with nestboxes) compared to control sites (i.e., without nestboxes) post treatment?

There was no significant interactive effect of treatment or control on southern greater glider abundance in East Gippsland or Tallaganda (Table 3.2). Mean southern greater glider abundance was predicted to be higher at nestbox sites and post treatment in both East Gippsland and Tallaganda (Appendix 3.9). However, this result was not significant and indicates that there was no significant increase in southern greater glider abundance in treatment sites post deployment during this time (Table 3.2; Appendix 3.9).

Table 3.2: East Gippsland and Tallaganda generalised linear mixed model results using highest count as the response variable. The estimate, standard error, z value and p value are shown for each variable in each model. Neither models were zero-inflated, with the ratio of observed and predicted zeros within the tolerance range (Lüdecke et al., 2021). Significant values are shown by an * and in bold.

	East Gippsland				Tallaganda			
	Estimate	*Std. error*	*z value*	*P-value*	*Estimate*	*Std. error*	*z value*	*P-value*
(Intercept)	-0.8710	0.6495	-1.341	0.18	1.6645	0.2864	5.811	**6.2e-09 *** **
Treatment	-1.4224	0.7726	-1.841	0.066	0.2084	0.2551	0.817	0.4141
Survey round	-0.4055	0.4564	-0.888	0.37	-0.3567	0.2845	-1.254	0.2100
Burn Severity	-0.0006	0.0086	-0.076	0.9398	-0.0132	0.0040	-3.284	**0.001** **
Treatment: Round (Pre)	0.4055	0.9354	0.434	0.6647	0.1015	0.3676	0.276	0.7825

Question 4: Does fire severity affect the abundance of southern greater gliders in the study area?

We found no significant effect of burn percentage in East Gippsland, however, in Tallaganda there were significantly fewer southern greater gliders in areas with higher burn percentage than those with a lower burn percentage (Table 3.2; Figure 3.6).

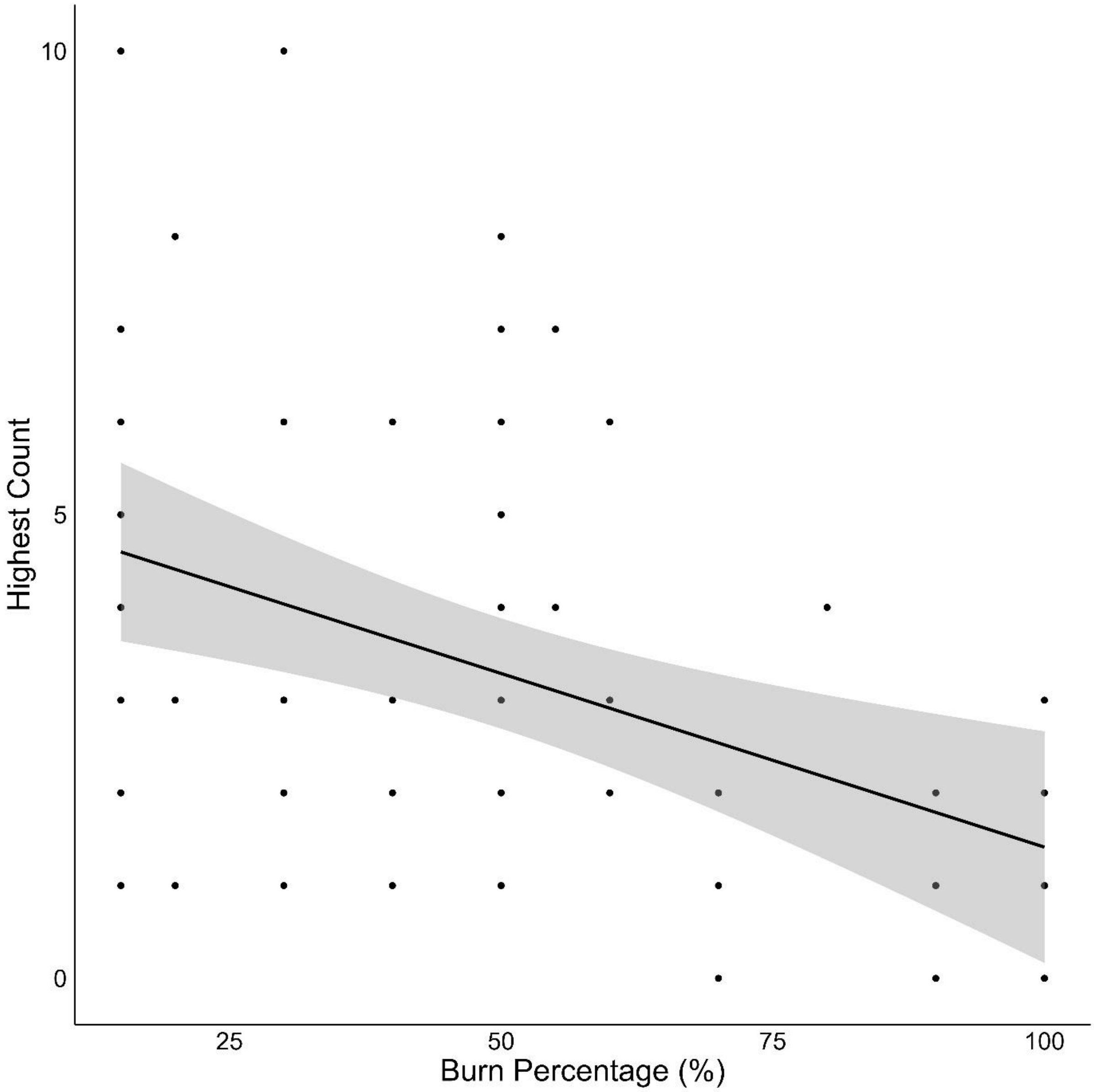


Figure 3.6: Abundance of the southern greater glider in relation to burn percentage (%) at survey transects in Tallaganda using both pre- and post-treatment survey data. Points represent the number of southern greater gliders detected at different burn percentages. The black line is the fitted response from a generalised linear mixed model with 95% confidence intervals. Created using ggplot2 in R (Wickham, 2016).

Species detection rates using different monitoring methods

We detected southern greater gliders in spotlighting surveys and through camera trapping (Table 3.3). In Tallaganda, southern greater gliders were the most common species detected using both methods (Table 3.3). In East Gippsland, detection of our target species was similar, however per site, spotlighting yielded better detection rates (Table 3.3). We detected *Antechinus* spp. only in Tallaganda and East Gippsland on camera, whereas the common ringtail possum was detected only during spotlighting surveys. The feathertail glider was detected more frequently by camera trapping than spotlighting in both areas, particularly in East Gippsland where they were detected at 100% of camera sites, but only 8% of spotlighting sites (Table 3.3).

Table 3.3: The percentage of sites where each species was detected using combined data from two rounds of spotlighting surveys and camera trapping at nestbox sites. There are 26 cameras, 13 sites used in Tallaganda and 28 cameras, 14 sites used for East Gippsland.

	East Gippsland		Tallaganda	
	Spotlighting	Camera Monitoring	Spotlighting	Camera Monitoring
Antechinus	0	35.71	0	7.7
Bat	0	42.86	0	0
Common Brushtail Possum	41.67	0	36.67	7.7
Common Ringtail Possum	8.33	0.00	2	0
Feathertail Glider	8.33	100	10	53.85
Inland Sugar Glider	8.33	85.71	3.33	23.08
Mountain Brushtail Possum	33.33	42.86	6.67	0
Southern Greater Glider	41.67	57.14	96.67	69.23
Yellow-bellied Glider	2.78	21.43	0	0

3.5 Discussion

We document the establishment of an experimental study to test the long-term efficacy of purpose-built nestboxes on the post-fire recovery of two southern greater glider populations. Our study revealed that the two populations were vastly different in the number of southern greater gliders detected. However, in both study areas, southern greater gliders were observed occupying purpose-built nestboxes. We found no significant difference in numbers of southern greater gliders between treatment and control sites prior to nestbox (treatment) installation. We did not find an increase in southern greater gliders in areas with nestboxes compared to those without post-treatment, most likely due to the short duration of our study.

Nestbox use by southern greater gliders

Southern greater gliders were observed using purpose-built nestboxes in both study areas. As expected, southern greater glider densities differed markedly between our two study sites, with a substantially higher abundance in Tallaganda. We observed higher nestbox use (i.e. definitive occupation) by southern greater gliders in Tallaganda (9 of 26 sites) compared to East Gippsland (1 of 28 sites). However, we observed southern greater gliders investigating several nestboxes at sites in both regions, which may indicate occupation that was not captured by the cameras. We monitored only a subset of nestboxes with cameras, and therefore our observations of nestbox occupation are limited to these sites.

Differences in hollow abundance or availability may also explain the differences in nestbox uptake between study areas. Wagner et al. (2024) found that hollow occurrence and abundance decreased in areas burnt at high severity in East Gippsland. However, there were similar numbers of sites in high severity areas in both East Gippsland and Tallaganda. More in-depth hollow counts at all sites will aid future work in determining whether hollow abundance is a driver of these differences in uptake. Alternatively, nestbox uptake could be related to differences in animal behaviour. At two nestboxes in Tallaganda we observed southern greater gliders with dependent young (Figure 3.4). Female southern greater gliders may be more likely to utilise nestboxes during the breeding season.

We documented adult (similar sized) southern greater gliders sharing nestboxes, as reported in earlier studies (Henry, 1984; Kehl and Lindenmayer et al. 2004). The social organisation of arboreal marsupials is split into three types; colonial, solitary or pair (Lindenmayer, 1997). Southern greater gliders have primarily been recorded as a solitary species, although early studies have suggested that arboreal marsupials alter their social organisation depending upon resource availability (Lindenmayer, 1997). It is also possible that the two similar sized individuals were parent and a previous years' offspring, or a mating pair (Comport et al., 1996). Understanding these behaviours and how they affect population dynamics will be an important consideration for future studies of the species, particularly if these changes are from the effects of wildfire.

Nestbox use by non-target species

Competition for purpose-built nestboxes by other species can affect occupancy rates of target species (Beyer and Goldingay, 2006). For example, Superb parrots (*Polytelis swainsonii*) did not occupy any of the 587 nestboxes deployed in an offset project in inland NSW, which was thought to be due, in part, to occupation by common or invasive species (Lindenmayer et al., 2017). Both our study areas are home to diverse assemblages of arboreal marsupials (Appendix 3.3), most of which are hollow dependent and may compete with the southern greater glider for this essential resource. Common ringtail possums (*Pseudocheirus peregrinus*) use similar sized hollows as southern greater gliders; however, they are not solely reliant on hollows for shelter (Goldingay, 2012; Lindenmayer et al., 2008). Common species such as brushtail possums (*Trichosurus* spp.) often occupy nestboxes targeted for other hollow-dependent species (Le Roux et al., 2016; Lindenmayer et al., 2017).

We observed occupancy by non-target species primarily at nestboxes in East Gippsland. We did not observe occupancy of nestboxes by non-target species in Tallaganda. This could be a result of the differences in numbers of southern greater gliders between the two populations. The population at Tallaganda seems to be significantly denser than in East Gippsland, and southern greater gliders were the most common species observed (Table 3.3). This could have resulted in southern greater gliders quickly saturating the nestboxes in Tallaganda, therefore, not allowing other species to take up residence. Inland sugar gliders were the most common nestbox inhabitants in East Gippsland, occupying approximately 29% of monitored nestboxes. However, compared to other studies where non-target species are the most commonly observed occupants (Lindenmayer et al., 2016), we observed low levels of non-target species uptake. The low rate of non-target species uptake may be attributed to the design of our nestboxes, which were constructed specifically for the southern greater glider. We built nestboxes with a small entrance to exclude competitors such as brushtail possums, however, this design was unlikely to exclude smaller species such as inland sugar gliders (Beyer and Goldingay 2006). Ongoing camera monitoring will assist in determining whether non-target species continue to occupy the nestboxes and if this occupancy has an adverse effect on southern greater gliders.

Nestboxes supporting recovery

We did not find an increase in southern greater gliders numbers in areas with nestboxes compared to those without during the first round of post-treatment monitoring (Table 3.2). This was expected due to the low fecundity (DCCEEW, 2022), high site fidelity, and small home ranges (Miritis et al., 2024) of the southern greater glider. A prolonged period of monitoring will be needed to determine if the deployment of nestboxes leads to an increase in abundance in the medium to long-term

Fire severity and glider abundance

To the best of our collective knowledge, our study is the first to document recurring use of nestboxes in burnt habitat by the southern greater glider. Southern greater gliders were found in sites from low to high burn percentage in both study areas. This is a promising indication that nestboxes may support recovery of fire-sensitive species, particularly with projected increases in the occurrence and severity of wildfires (McKenzie et al., 2004). However, we did find that fire severity influenced southern greater glider occurrence in Tallaganda, with fewer animals in areas with a higher burn percentage (Figure 3.6). This result is consistent with other studies of southern greater gliders (Green et al., 2024; May-Stubbles et al., 2022; Wagner et al., 2024). We did not find the same relationship between burn percentage and southern greater glider numbers in East Gippsland. This may be attributed to the fewer records of southern greater gliders and a reduced number of sites supporting the species. The direct impact of fire leading to mortality of individual southern greater gliders is challenging to mitigate, therefore, proactive measures to reduce the risk of increased fire severity in future events is important (Collins et al., 2021).

3.6 Conclusion

Our study documented the use of purpose-built nestboxes by southern greater gliders in recently burnt habitat. This finding highlights the potential for nestboxes to be a successful management tool to supplement natural hollows after disturbance. We also found that nestboxes were used by several non-target species and there was a large difference in nestbox use by southern greater gliders in Tallaganda compared to East Gippsland. However, it is unknown whether nestbox use by non-target species is of concern. Future spotlighting surveys should ensure the double-observer distance sampling method is continued correctly. Camera set up may also need modification to minimise problem periods. Long-term monitoring at these sites will be imperative to assess southern greater glider population changes in response to nestboxes and confirm whether nestboxes are an effective management tool for this endangered species.

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3.9 Supplementary Materials

Appendix 3.1: The number of spotlighting surveys conducted in East Gippsland between April and December 2021, the transect length, and number of individual southern greater gliders detected. Survey method and number of observers per survey differed. Surveys were conducted on foot (except for those marked with * which were driven) along tracks and roads. Some surveys were on foot through the forest. These surveys were conducted within the Bendoc, Bonang, Goongerah and West Cann area of East Gippsland, Victoria.

Transect Length (m)	Number of Surveys	Number of Individuals
120	1	0
200	46	8
220	1	0
270	1	1
300	15	4
350	1	0
380	1	0
400	1	0
450	1	0
470	1	3
500	38	3
550	3	3
600	1	0
700	1	2
710	1	0
850	1	0
1000	2	1
1100	1*	0
1150	2	3
1300	1*	4
1400	1*	0
2000	1	2
49.97km	122	34

Appendix 3.2: The number of spotlighting surveys conducted in Tallaganda National Park between October and December 2021, the transect length and the number of individual southern greater gliders detected. Surveys were conducted on foot along tracks and roads. Number of observers differed for each survey.

Transect Length (m)	Number of Surveys	Number of Individuals
200	2	5
220	1	14
250	1	2
300	3	10
350	1	9
400	2	5
500	9	38
550	1	1
7.87km	20	84

Appendix 3.3: List of arboreal mammal species that inhabit Tallaganda and East Gippsland, their scientific name, threatened status, and which study area their range overlaps with (EG = East Gippsland, T = Tallaganda). Threatened status is from the Commonwealth Environment Protection and Biodiversity Conservation Act 1999 (DCCEEW, 2021).

Common Name	Scientific Name	Threatened Status	Study Area
Mountain brushtail possum	*Trichosorus cunninghami*	Not Listed	Both
Common Brushtail Possum	*Trichosurus vulpecula*	Not Listed	Both
Common Ringtail Possum	*Pseudocheirus peregrinus*	Not Listed	Both
Southern Greater Glider	*Petauroides volans*	Endangered	Both

Sugar Glider	*Petaurus breviceps*	Not Listed	Both
Feathertail Glider	*Acrobates* spp. (Broad-toed or Narrow-toed)	Not Listed	Both
Yellow-bellied glider	*Petaurus australis*	Vulnerable	EG
Eastern Pygmy Possum	*Cercartetus nanus*	Not Listed	T
Antechinus	*Antechinus* spp. (Multiple species)	Not Listed	Both

Appendix 3.4: East Gippsland spotlighting transects, treatment (nestbox or control), direction of transect (PE = perpendicular to road or PA = parallel to road), the burn percentage (%), canopy burn (field measure; yes - canopy at the site was burnt, no – no canopy scorch visible), severity class according to the satellite fire mapping (DELWP, 2022), land tenure, and Ecological Vegetation Class (Only the majority class is recorded here). To determine the fire severity class from the fire mapping, each transect was given a 50m buffer, and the class that covered the majority of the area was selected. * Not used in analysis. Site 11NB and Site 11C were removed entirely. Sites 16 to 20 were within the prescribed burn areas.

Site	Treatment	Direction of transect	Burn (%)	Canopy burn	Severity class	Tenure	Ecological Vegetation Class
01C	Control	PE	50	Yes	High	Flora & Fauna Reserve	Shrubby Dry Forest
01NB	Nestbox	PE	80	Yes	High	Flora & Fauna Reserve	Montane Dry Woodland
02C	Control	PE	100	Yes	High	Flora & Fauna Reserve	Shrubby Dry Forest
02NB	Nestbox	PE	100	Yes	High	Flora & Fauna Reserve	Shrubby Dry Forest
03C	Control	PE	100	Yes	High	State Forest	Damp Forest
03NB	Nestbox	PA	100	Yes	High	State Forest	Damp Forest
04C	Control	PE	100	Yes	High	State Forest	Montane Dry Woodland
04NB	Nestbox	PE	100	Yes	High	State Forest	Montane Dry Woodland
05C	Control	PE	100	Yes	High	State Forest	Shrubby Dry Forest
05NB	Nestbox	PE	80	Yes	High	State Forest	Shrubby Dry Forest
06C	Control	PE	95	Yes	High	State Forest	Wet Forest
06NB	Nestbox	PE	80	Yes	High	State Forest	Damp Forest
07C	Control	PE	15	Yes	Low	National Park	Wet Forest
07NB	Nestbox	PE	10	Yes	Low	National Park	Damp Forest
08C	Control	PE	100	Yes	High	National Park	Damp Forest

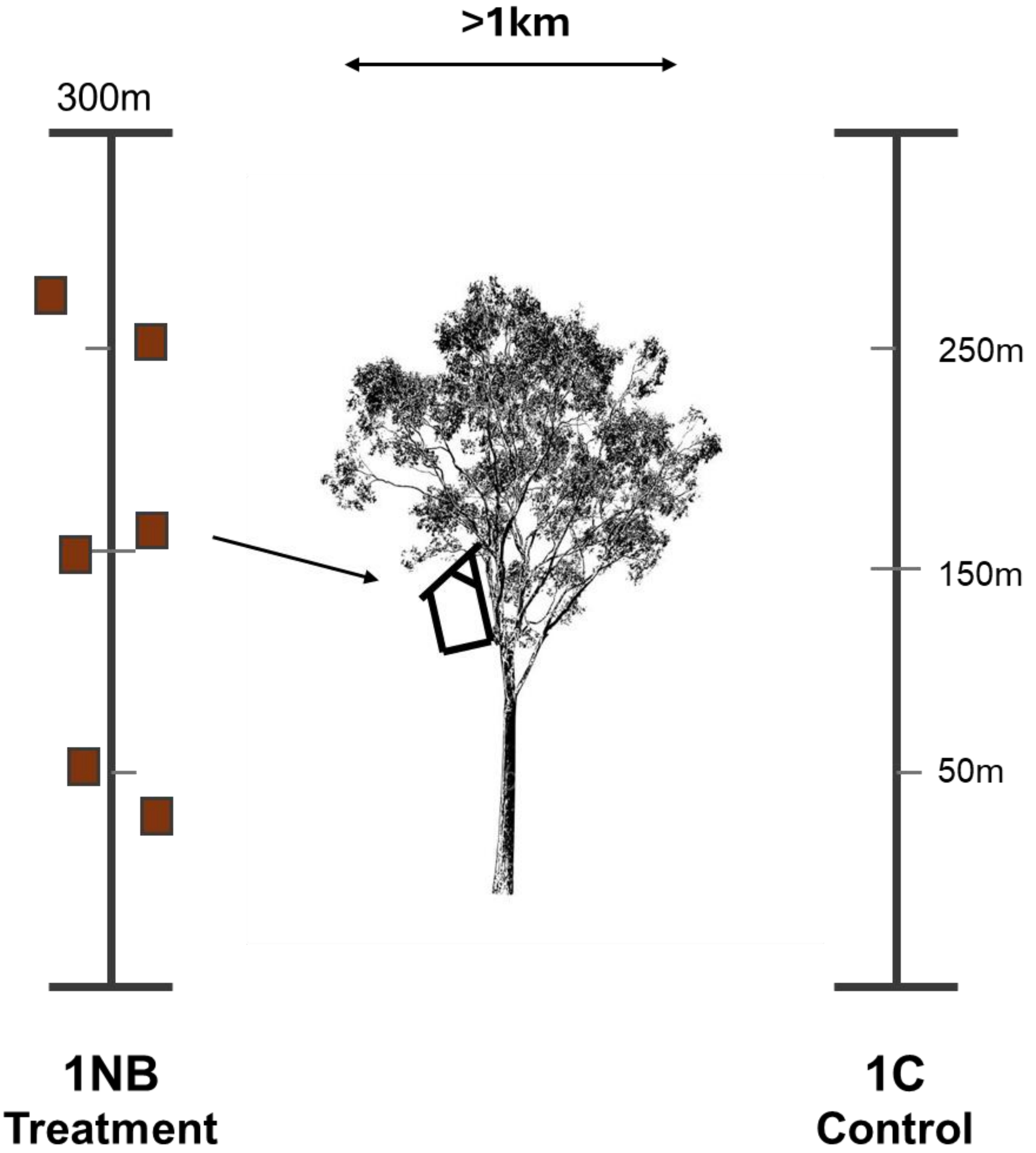
08NB	Nestbox	PE	95	Yes	High	National Park	Damp Forest
09C	Control	PA	20	No	No Data	State Forest	Wet Forest
09NB	Nestbox	PE	20	No	No Data	National Park	Wet Forest
10C	Control	PA	20	No	Low	State Forest	Damp Forest
10NB	Nestbox	PA	30	No	Low	State Forest	Damp Forest
12C*	Control	PA	70	Yes	Low/No Data	National Park	Shrubby Dry Forest
12NB*	Nestbox	PA	100	Yes	Low	National Park	Shrubby Dry Forest
13C	Control	PE	40	No	Low	National Park	Damp Forest
13NB	Nestbox	PE	40	No	Low	National Park	Wet Forest
14C	Control	PE	40	No	Low	National Park	Wet Forest
14NB	Nestbox	PE	30	No	Low	National Park	Wet Forest
15C	Control	PE	20	No	Low	National Park	Wet Forest
15NB	Nestbox	PE	35	No	Low	National Park	Wet Forest
16C	Control	PA	10	No	No Data	State Forest	Montane Grassy Woodland
16NB	Nestbox	PA	20	No	No Data	State Forest	Montane Grassy Woodland
17C	Control	PE	50	No	No Data	State Forest	Montane Grassy Woodland
17NB	Nestbox	PE	50	No	No Data	State Forest	Montane Dry Woodland
18C	Control	PA	20	No	No Data	State Forest	Montane Dry Woodland
18NB	Nestbox	PE	10	No	No Data	State Forest	Montane Dry Woodland
19C	Control	PE	30	Yes	No Data	State Forest	Tableland Damp Forest
19NB	Nestbox	PA	10	No	No Data	State Forest	Tableland Damp Forest
20C	Control	PE	50	No	No Data	State Forest	Montane Dry Woodland
20NB	Nestbox	PE	80	Yes	No Data	State Forest	Montane Dry Woodland

Appendix 3.5: Tallaganda spotlighting transects, treatment (nestbox or control), the burn percentage (%), canopy burn (field measure; yes - canopy at the site was burnt, no – no canopy scorch visible), severity class according to fire mapping (DCCEEW, 2020), land tenure, and vegetation class (Only the majority class is recorded here). To determine the fire severity class from the fire mapping, each transect was given a 50m buffer, and the class that covered the majority of the area was selected.

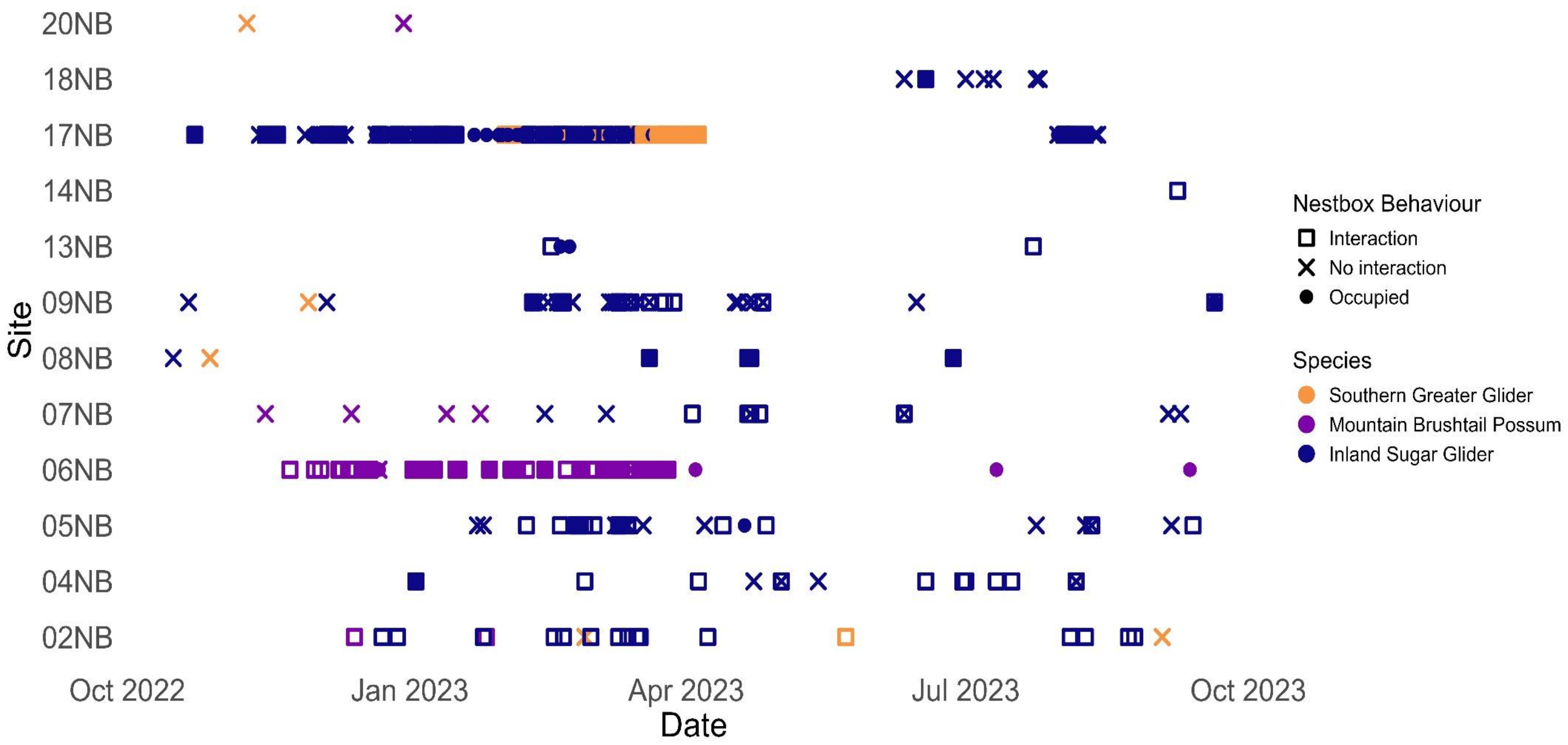
Site	Treatment	Burn (%)	Canopy Burn	Severity Class	Tenure	Vegetation Class
01A	Nestbox	15	No	Low	National Park	Tableland and escarpment Moist herb/fern grass forest
01B	Nestbox	30	No	Moderate	National Park	Tableland and escarpment Moist herb/fern grass forest
01C	Control	20	Yes	Low	National Park	Tableland and escarpment Moist herb/fern grass forest
02A	Nestbox	15	No	Moderate	National Park	Eastern Tablelands Acacia/Herb/Grass Forest
02B	Nestbox	30	No	High	State Forest	Eastern Tablelands Acacia/Herb/Grass Forest
02C	Control	20	No	High	National Park	Eastern Tablelands Acacia/Herb/Grass Forest
03A	Nestbox	30	No	Low	National Park	Eastern Tableland Fern/Herb/Grass Moist Forest
03B	Nestbox	55	No	Moderate	National Park	Eastern Tableland Fern/Herb/Grass Moist Forest
03C	Control	50	Yes	Extreme	National Park	Eastern Tableland Fern/Herb/Grass Moist Forest
04A	Nestbox	100	Yes	High	National Park	Eastern Tableland Fern/Herb/Grass Moist Forest
04B	Nestbox	90	Yes	Extreme	National Park	Eastern Tableland Fern/Herb/Grass Moist Forest
04C	Control	100	Yes	High	National Park	Eastern Tableland Fern/Herb/Grass Moist Forest
05A	Nestbox	50	Yes	Moderate	National Park	Eastern Tableland Fern/Herb/Grass Moist Forest
05B	Nestbox	50	Yes	Moderate	National Park	Eastern Tableland Fern/Herb/Grass Moist Forest
05C	Control	40	Yes	Moderate	National Park	Eastern Tableland Fern/Herb/Grass Moist Forest
06A	Nestbox	60	Yes	High	National Park	Eastern Tablelands Acacia/Herb/Grass Forest

06B	Nestbox	60	Yes	High	National Park	Tableland and escarpment Moist herb/fern grass forest
06C	Control	70	Yes	Moderate	National Park	Tableland and escarpment Moist herb/fern grass forest
07A	Nestbox	15	No	Moderate	National Park	Eastern Tableland and Escarpment Shrub/Fern Dry Forest
07B	Nestbox	15	No	Low	National Park	Eastern Tableland and Escarpment Shrub/Fern Dry Forest
07C	Control	15	Yes	High	National Park	Tableland and escarpment Moist herb/fern grass forest
08A	Nestbox	50	Yes	Moderate	National Park	Eastern Tableland Fern/Herb/Grass Moist Forest
08B	Nestbox	50	Yes	Extreme	National Park	Tableland and escarpment Moist herb/fern grass forest
08C	Control	40	Yes	High	National Park	Eastern Tablelands Acacia/Herb/Grass Forest
09A	Nestbox	90	Yes	High	National Park	Tableland and escarpment Moist herb/fern grass forest
09B	Nestbox	100	Yes	Extreme	National Park	Eastern Tableland and Escarpment Shrub/Fern Dry Forest
09C	Control	80	Yes	High	State Forest	Eastern Tableland Fern/Herb/Grass Moist Forest
10A	Nestbox	30	No	Moderate	National Park	Eastern Tablelands Dry Shrub Forest
10B	Nestbox	50	Yes	High	National Park	Tableland and escarpment Moist herb/fern grass forest
10C	Control	70	Yes	High	National Park	Eastern Tablelands Dry Shrub Forest

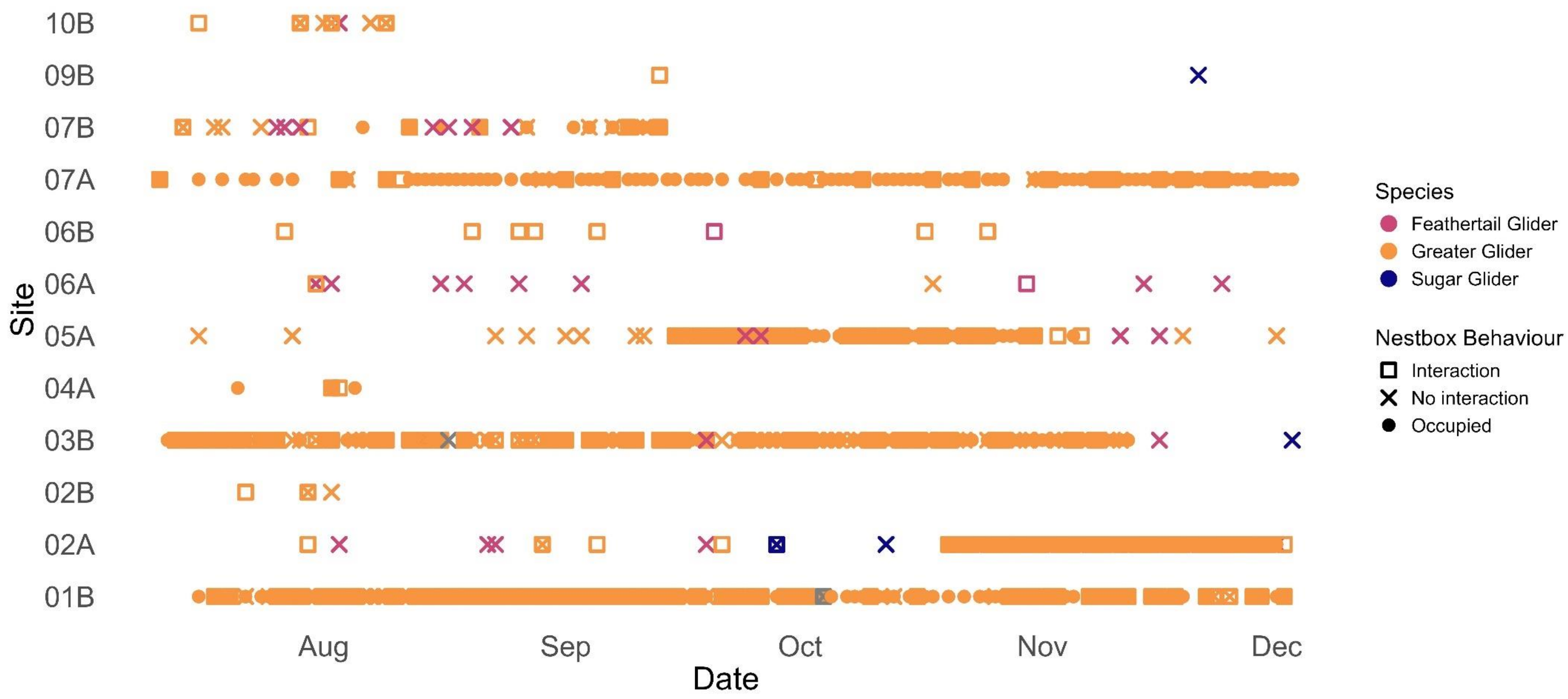
Appendix 3.6: The design and layout of each study site, (1NB) nestbox site and (1C) control site. Both nestbox and control sites were marked out with flagging and reflective tape in the same way.



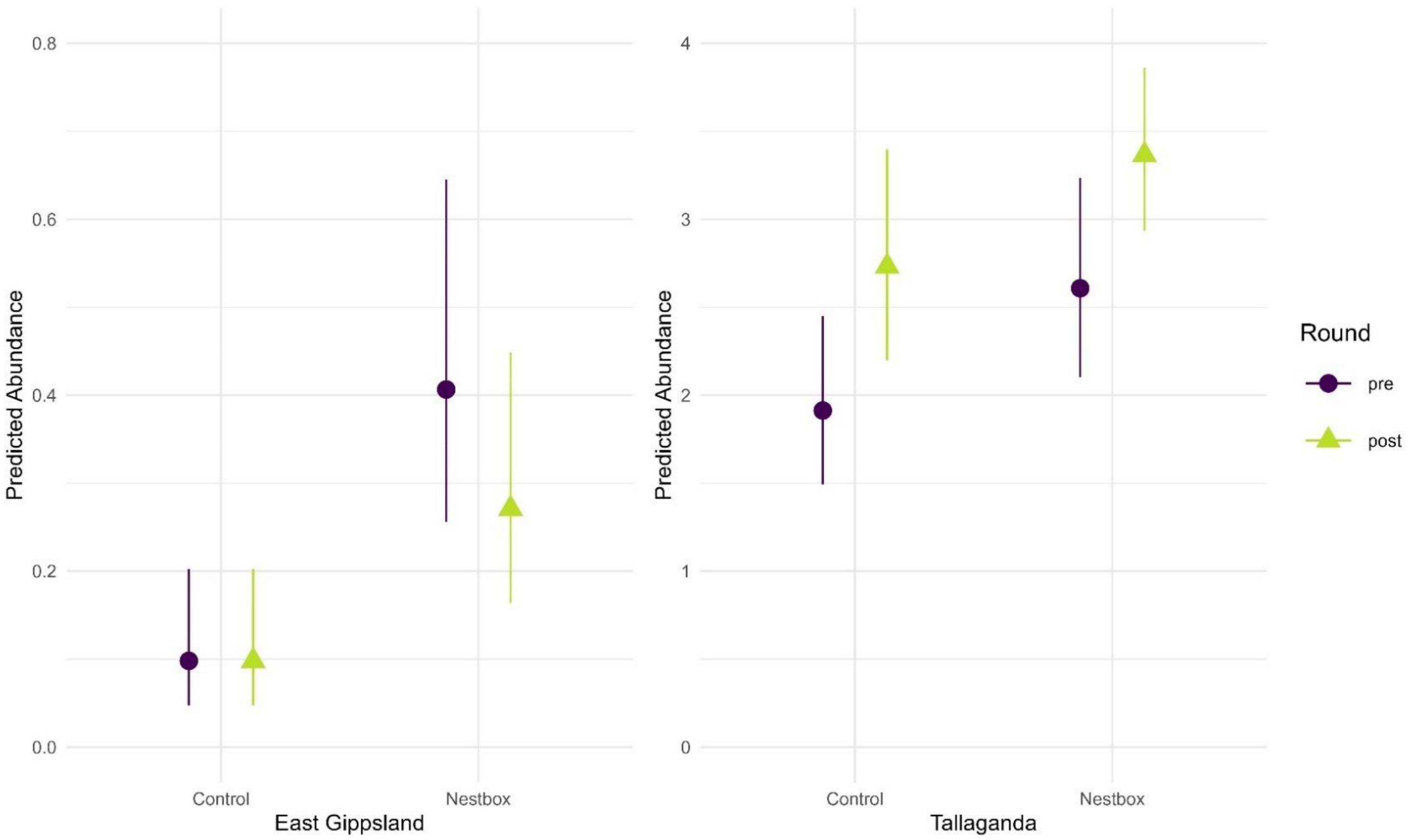
Appendix 3.7: Camera detections of southern greater gliders (orange), inland sugar gliders (blue) and mountain brushtail possums (purple) at nestbox sites over time in East Gippsland, Victoria. Each site displays the combined species detections from the two nestbox cameras per site. Animal behaviour at the nestbox is split into no interaction (cross), interaction (square) or use (filled circle). Feathertail gliders were excluded from this figure for clarity.



Appendix 3.8: Camera detections of southern greater gliders (orange), inland sugar gliders (blue) and feathertail gliders (pink) at nestbox sites over time in Tallaganda, New South Wales. Each site displays the combined species detections from the two nestbox cameras per site. Animal behaviour at the nestbox is split into no interaction (cross), interaction (square) or use (filled circle).



Appendix 3.9: The predicted mean abundance of southern greater gliders at nestbox and control sites before (pre) and after (post) nestbox deployment. Predicted means and standard error bars are shown for each study site, generated from the generalised linear mixed model results presented in Table 3.2.



4.0 CHAPTER 4

Synthesis and Conclusions

The impacts of megafire

The research presented in this thesis highlights threats to the persistence of populations of the southern greater glider, particularly in response to an increase in the frequency of megafires as projected under future climate change (Abram et al., 2021; Canadell et al., 2021; Fairman et al., 2015; Lindenmayer et al., 2023). My research showed that southern greater gliders were negatively affected by the 2019/2020 megafires in East Gippsland, Victoria. The maxent modelling (Chapter 1) predicted a reduction in marginal, or moderately suitable, habitat in the two years following megafire. This represents a likely retraction of the species to higher elevations, unburnt forest, or areas where fire was comparatively less severe or has been less frequent. My results from Tallaganda (Chapter 2) showed that fewer southern greater gliders were detected in areas burnt at a high severity, a finding consistent with other studies (Green et al., 2024; May-Stubbles et al., 2022; Wagner et al., 2024). These findings were confirmed anecdotally during my field surveys. I invested a significant amount of time and effort searching for southern greater gliders across East Gippsland within, and surrounding, areas affected by the 2019/2020 megafires (Chapter 2). Initially, I used a randomised design to guide my survey effort, however, after very few detections of the species, I used historical records to guide my survey effort. I spent approximately 43 nights spotlighting in East Gippsland during 2021. It appears that individuals are now absent from many areas of their previous distribution within East Gippsland. This is concerning, most likely indicating a reduction in the range of the species, as previously observed within the region (DELWP, 2019), elsewhere (Lindenmayer et al., 2011; Smith and Smith, 2018), and as shown in the post-fire maxent model in Chapter 1.

Installing nestboxes following megafire

My research demonstrated that purpose-built nestboxes show potential to assist southern greater glider recovery post fire. I observed southern greater gliders using nestboxes in East Gippsland and Tallaganda shortly after nestbox deployment. Southern greater gliders depend on tree hollows for shelter and reproduction

(Comport et al., 1996; Tyndale-Biscoe and Smith, 1969). An increase in fire frequency (Salmona et al., 2018) and fire severity (May-Stubbles et al., 2022) can reduce hollow abundance. There is potential, therefore, for nestboxes to provide supplementary shelter to replace lost hollows following megafires. However, caution should be taken in interpreting these results as I found no evidence that southern greater glider occupancy or abundance increased in sites with nestboxes compared to those without nestboxes. This result was expected, due to the short time period of this study, and the slow breeding rate of southern greater gliders (Tyndale-Biscoe and Smith, 1969). It will likely take several years to determine, with statistical rigour, positive effects of nestbox deployment (if any) on the species' populations.

Long-term monitoring of nestboxes

The construction and implementation of purpose-built nestboxes can require substantial financial investment (Howard et al., 2022; McKenney and Lindenmayer, 1994) and time commitment. Nestboxes and their matched control sites will continue to be monitored through spotlighting surveys, camera trapping, and other novel technologies. This may also include installing cameras at nestboxes that were not monitored in this study. Ongoing monitoring at these sites will allow an evaluation of nestbox effectiveness for the management of southern greater gliders in the medium and long term.

Informed nestbox implementation

Prioritising where nestboxes can be used to supplement hollows is critical, particularly because providing artificial habitat across large areas is unrealistic due to cost. Purpose-built nestboxes are expensive to build and often challenging to install. Additionally, there are costs associated with replacing boxes as materials deteriorate (Lindenmayer et al., 2009; Quin et al., 2020). To maximise uptake, nestboxes should be installed in areas where southern greater gliders currently live or within close proximity, assuming the addition of nestboxes provides a benefit. Southern greater gliders are slow to disperse into new areas, and have small home ranges (Miritis et al., 2024). Installing nestboxes close to known southern greater glider locations allows individuals to find the nestboxes and expand their range into nearby areas.

My findings suggest that nestbox uptake was higher in areas with more individuals (Chapter 2). However, further research is required to understand the drivers of nestbox use in both densely populated (e.g. Tallaganda) and less-densely populated (e.g. East Gippsland) areas. For example, it is important to determine whether high nestbox uptake is primarily driven by limited hollow availability, or if nestboxes simply contribute to a range of shelter options without affecting survival. Thus, habitat assessments, tree hollow surveys, and further exploration into hollow preferences (Hofman et al., 2022; Vinson et al., 2021) would complement this research and inform nestbox installation.

Areas impacted by higher fire severities are likely to require more artificial habitat due to the loss of hollow-bearing trees. Nestboxes could be placed in these areas to supplement natural hollows and enhance the quality of otherwise marginal habitat. However, nestboxes in unsuitable habitat may create ecological traps (as defined in Meer et al., 2014) or remain unused, yielding few conservation benefits. Therefore, care must be taken to ensure that installing nestboxes does not inadvertently draw individuals into areas where they may not have the resources to survive. For example, nestboxes should not be implemented in areas of extreme canopy scorch shortly after fire in East Gippsland. No southern greater gliders have been found, nor have been predicted to occur in these areas (Chapter 2, Chapter 3), so installing nestboxes there would be a waste of effort and resources.

In addition to identifying appropriate areas to deploy nestboxes, the design of the nestboxes is critical to their success. Therefore, purpose-building nestboxes for the target species is recommended, as reported in other studies (Goldingay et al., 2015; Le Roux et al., 2016). The design used in this study was effective for southern greater gliders. More research into design features will help confirm their long-term success.

Broader conservation of the southern greater glider

The sizes and densities of southern greater glider populations varies across their range, as highlighted by my work on the East Gippsland and Tallaganda populations. Different populations may, therefore, exhibit differing responses to threats. The effects of anthropogenic climate change will continue to threaten southern greater gliders. Rising temperatures, altered fire regimes, more megafires and ongoing habitat destruction (e.g. logging of native forests) may push this species towards extinction. Therefore, incorporating the recovery and restoration of burnt habitat into management strategies is crucial for the conservation of southern greater gliders.

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