Supplementary refuge as a tool for recovering small mammals and reptiles after fire

by

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Candidate's Declaration

This thesis contains no material which has been accepted for the award of any other degree or diploma in any university. To the best of the author's knowledge, it contains no material previously published or written by another person, except where due reference is made in the text.

1 February, 2025

Heather Burns

Date:

Acknowledgements

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Abstract

Global fire regimes are becoming more frequent, severe, and extensive. Within Australia, this trend was demonstrated by the unprecedented scale and ecological impact of the 2019-2020 Black Summer fires. Small mammals and reptiles are of particular conservation concern due to their reliance on ground-level vegetation and woody debris, and increased vulnerability to predation in burnt landscapes. Despite the importance of post-fire conservation for these species, there are relatively few evidence-based management interventions available to land managers looking to actively conserve native fauna after fire. In the following chapters I aim to address this knowledge gap by systematically mapping the literature to identify potential post-fire management actions (Chapter 2), and conducting a field-based experiment to test the effects of supplementary coarse woody debris (CWD) on small mammal and reptile populations (Chapters 3-5).

In Chapter 2 I used a systematic mapping protocol to identify the range of conservation-focused management interventions that have been used in disturbed landscapes globally. I aimed to identify which management interventions have been used in disturbed landscapes to date and whether those interventions could be suitable in response to other types of disturbance, specifically fire. Interventions were most commonly used to address invasive or overabundant species, agriculture, and production forestry. Coarse woody debris manipulation, predator control and revegetation were the most commonly studied interventions. However, understanding suitable management responses to fire represents a significant knowledge gap and only three studies assessed management interventions in post-fire landscapes.

In Chapters 3 and 4 I experimentally tested the effects of supplementary CWD on small mammal and reptile populations in areas burnt by the 2020 Orroral Valley Fire in Namadgi National Park, ACT. In a field-based study I established 20 experimental blocks with three treatments: (1) structure only (CWD piles); (2) predation only (strips of chicken wire); and (3) structure + predation (CWD piles covered in chicken wire). A combination of active and visual searches were used to survey reptile populations at each treatment for two consecutive summer seasons, and wildlife cameras were used to continuously monitor small mammal activity for 15 months. Woody debris treatments (i.e., structure and structure + predation) supported significantly higher abundances of reptiles compared to the control. Small mammal abundance and richness was not significantly greater at any treatment compared with the control, and instead increased most significantly with increasing levels of shrub cover.

In Chapter 5 I examined the thermal properties of supplementary CWD piles and compared them to the thermal properties of natural refugia in a post-fire landscape. Temperature loggers were used to monitor temperatures within CWD piles, hollow logs and rocky outcrops across

seven months. Temperatures within CWD piles were significantly less extreme than ambient maximum and minimum temperatures.

In Chapter 6 I synthesise the findings of Chapters 2-5 to answer the question "what should we do after the next fire?". In this chapter I identify the key lessons learned across my research that can help land managers make more informed decisions after future fires. Overall, supplementary CWD was proven to support a higher abundance of reptiles in areas burnt at high and low fire severity across our study. CWD piles without chicken wire (i.e., structure only treatment) were the most cost-effective treatment tested, and the simplicity of their construction provides an opportunity to involve citizen scientists and community members interested in post-fire recovery efforts. CWD piles were not successful at increasing small mammal richness or abundance in burnt areas in their current design. In this chapter, we suggest potential design alterations that may make this supplementary refuge more suitable for small mammals.

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Chapter 1: Introduction

Fire has a long history of shaping terrestrial landscapes, with many ecosystems adapted to fire intervals ranging from a few years to decades (Pausas and Keeley, 2009). A changing climate is expected to result in more frequent and severe fires (Bradstock, 2010; Jones *et al.*, 2020; Canadell *et al.*, 2021), which may interrupt the natural sequences of regeneration and disturbance and threaten native vegetation communities (Duivenvoorden *et al.*, 2024; Enright *et al.*, 2015).

These direct effects of fire on vegetation communities may have follow-on effects for small mammals and reptiles, which face a range of challenges in post-fire landscapes. For example, the response of ground cover and shrubby vegetation to fire affects the abundance and distribution for a range of small mammal species (Arthur *et al.*, 2012; Di Stefano *et al.*, 2011; Dixon *et al.*, 2019). Small mammals and reptiles also rely on coarse woody debris (CWD) and hollows for nesting, the availability of which can be affected by fire frequency and severity (Bassett *et al.*, 2015a; Collins *et al.*, 2012; Stares *et al.*, 2018). As fire affects the ability of vegetation communities to recover from fire, the availability of these resources may become more limited. Additionally, native prey species can face additional pressure from introduced predators such as foxes (*Vulpes vulpes*) and feral cats (*Felis catus*) after fire (Hradsky *et al.*, 2017a; McGregor *et al.*, 2017).

As landscapes are burnt at higher severity (Abram *et al.*, 2021b), and above their historical frequency (Canadell *et al.*, 2021), new management interventions may be necessary to conserve native small mammals and reptiles. In the aftermath of fire land managers are often under pressure to implement management and recovery strategies, but very little research exists to support this decision-making. Although solutions such as nest boxes are well-researched as a mitigation strategy to offset the loss of mature hollow-bearing trees, there are no well-established solutions to conserve small mammals and reptiles in burnt landscapes.

In Australia, the need for more post-fire conservation strategies came into sharp focus after the 2019-2020 megafires. My thesis will focus specifically on post-fire recovery efforts after the Orroral Valley Fire, which burnt over 80% of Namadgi National Park in the southern region of the Australian Capital Territory in 2020. In the following chapters, I will address the following research aims:

1. Conduct a review of the literature to identify (a) existing post-fire conservation strategies; and (b) management strategies used to conserve small mammals and reptiles in other disturbed landscapes.
2. Experimentally test the effects of supplementary CWD piles on small mammal and reptile richness and abundance in burnt areas

3. Compare the thermal properties of supplementary CWD and natural refugia to assess the suitability of this supplementary refuge to meet the thermal needs of species in our study area.

This thesis is structured in six chapters: an introduction chapter, four main chapters comprised of manuscripts intended for peer review and publication, and a conclusion chapter (see Figure 1-1). Chapter 2 uses systematic mapping methods to identify interventions that have previously been used in disturbed landscapes and determine which interventions, if any, may be suitable for application in burnt landscapes. The results from this chapter illustrated a significant knowledge gap in post-fire interventions aimed at conserving small terrestrial vertebrates and highlighted potential solutions. Chapters 3 and 4 illustrate the effects of supplementary coarse woody debris, an experimental post-fire intervention, on small mammal and reptile populations after the Orroral Valley Fire. Chapter 5 further examines the supplementary coarse woody debris piles, focusing on their thermal properties in comparison to the thermal properties of natural habitat. In the conclusion chapter, I draw upon key points from my systematic map and experimental studies to answer the question "What should we do after the next fire?". The chapter is designed as a resource for land managers who want to undertake conservation-focused management actions in burnt landscapes and concludes with directions for future research.

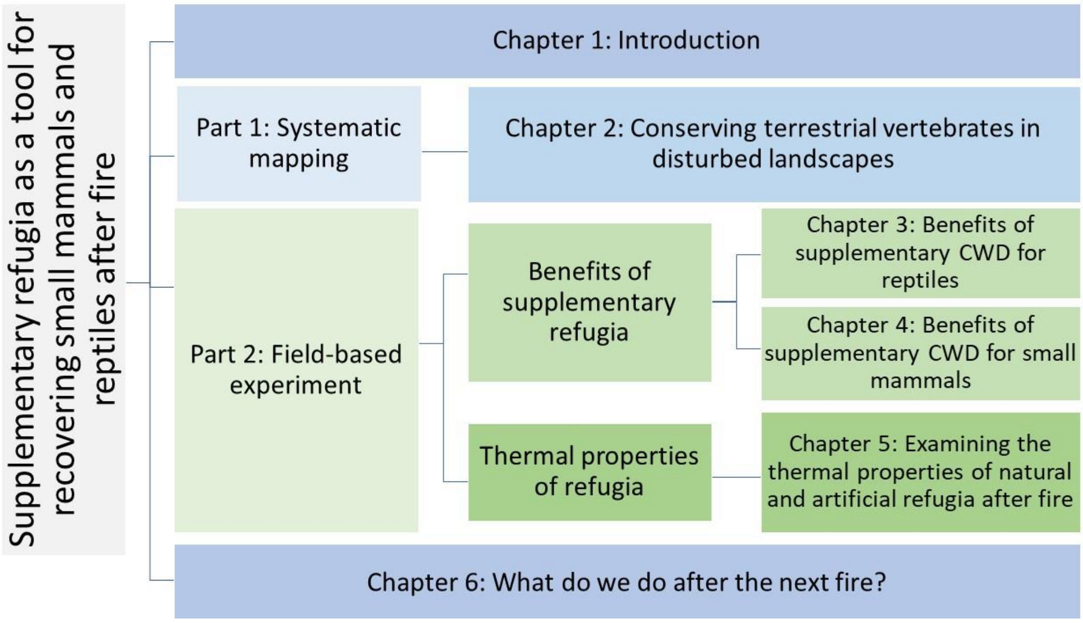

Figure 1-1: A diagram illustrating the structure of my thesis, including a summary of each chapter.

Chapter 2: Conserving terrestrial vertebrates in disturbed landscapes: a systematic mapping approach

Statement of Contribution

This thesis is submitted as a Thesis by Compilation in accordance with

https://policies.anu.edu.au/ppl/document/ANUP_003405

I declare that the research presented in this Thesis represents original work that I carried out during my candidature at the Australian National University, except for contributions to multi-author papers incorporated in the Thesis where my contributions are specified in this Statement of Contribution.

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Philip Gibbons ______________ ______________________ 15/07/2024 _____

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Foreword

After fire, terrestrial vertebrate populations face many barriers to recovery and survival. These include: increased predation, decreased habitat complexity at ground level (the regrowth of which can be hindered by overabundant herbivores), isolation of unburnt patches that limit dispersal and connectivity, limitation of food resources, and potential loss of key structural elements (i.e. hollow bearing trees and coarse woody debris). These barriers are not unique to post-fire environments, and can be present after numerous types of anthropogenic or natural disturbances. An initial scoping of the literature indicated that the body of work focused on post-fire management interventions was quite limited, particularly for a few of the barriers mentioned above. Therefore, understanding the interventions that have been used in other contexts may provide valuable insight for land managers looking to mitigate similar barriers after fire.

This chapter aims to help answer two questions: (1) what strategies have previously been used to conserve small mammals and reptiles after fire; and (2) what can we learn from management of other disturbed landscapes that could potentially apply to post-fire conservation of small vertebrates. Given the broad nature of these questions, I used a systematic mapping protocol to determine the scope of the literature and identify trends in the conservation of terrestrial vertebrates in disturbed landscapes. I used an a priori set of inclusion criteria to narrow the results to interventions that could be implemented immediately after fire, but allowed studies of other disturbed landscapes to be included as an example of what could be applied to post-fire landscapes in future. I used metadata collected from the set of relevant studies to assess trends in focal species, intervention type, disturbance type, and various combinations of these variables across the literature. The results from this chapter were used to inform my experimental design in Chapters 3 and 4.

This chapter has been submitted as a manuscript titled ‘Conserving terrestrial vertebrates in disturbed landscapes: a systematic mapping approach’ to *Biological Conservation* and is awaiting reviewer feedback.

2.1 Abstract

Terrestrial landscapes face a range of disturbances, including land clearing for agriculture or plantation forestry, wildfire, introduced or overabundant species, and habitat loss to commercial and residential development. These disturbances threaten vertebrate population viability through key habitat and refugia loss, decreased habitat complexity, and an increased risk of predation. To maximise conservation outcomes in these landscapes, land managers require more information on best-practice responses. Using a systematic mapping approach, we aimed to identify the following: (1) which management interventions have been used in disturbed landscapes to date; (2) whether interventions that have been used in one context may be suitable in response to a different type of disturbance; and (3) whether knowledge gaps exist in under-studied disturbances or management interventions. Our search string produced 13,697 results, of which 69 were included at full-text and used in our metanalysis. We found that invasive or overabundant species, agriculture, and production forestry were the most commonly studied disturbances in the literature. In response to disturbance, coarse woody debris manipulation, predator control and revegetation were the most commonly studied interventions. We also found that small mammals and reptiles were studied with a greater frequency than amphibians and medium to large mammals. Understanding suitable management responses to fire represents a significant knowledge gap and only three studies examined post-fire landscapes. Additionally, only four of 69 studies reported costs associated with implementing their management interventions.

2.2 Introduction

Terrestrial ground-dwelling vertebrates (hereafter ground-dwelling vertebrates) can be exposed to a range of natural and anthropogenic disturbances which threaten population viability. Examples include land clearing for agriculture or plantation forestry, wildfire, introduced or overabundant species, or habitat loss to commercial and residential development (Scott *et al.*, 2006; Lunney and Leary, 1988; Tolley *et al.*, 2016; Ribeiro *et al.*, 2009). After these disturbances, terrestrial vertebrate populations face many challenges to their recovery and survival, including decreased habitat complexity at ground level, potential loss of key habitat or refugia (i.e., hollow bearing trees and coarse woody debris), and an increased risk of predation (Hradsky *et al.*, 2017a; Griffiths and Brook, 2014; Webb and Shine, 2000; Cordier *et al.*, 2021; Doherty *et al.*, 2020).

Ground-dwelling vertebrates rely on varying levels of vegetation cover from shrubs, ground-level vegetation, and leaf litter. These habitat components provide shelter and food resources, facilitate movement in the landscape, and provide protection from predation (Whitfield *et al.*, 2014; Garden *et al.*, 2007; Castellano and Valone, 2006; Holland and Bennett, 2007; Stokes, 1995). Holland and Bennett (2007) found that native Australian species required a broad range of vegetation conditions to be successful, ranging from shrub-dominated landscapes to open canopy forest, with house mice (*Mus musculus*) being the only species that preferred disturbed landscapes. When ground cover is sparse after disturbance such as fire, small mammal abundance decreases compared to unburnt sites, particularly for species that depend on dense vegetation (Griffiths and Brook, 2014). Although ground cover may regenerate quickly in temperate climates, small mammals remain vulnerable in the immediate post-fire period. Reptiles also respond to vegetation structure in disturbed landscapes, with recorded abundances increasing as canopy cover and leaf litter increased in grazed pastures (Pulsford *et al.*, 2017).

In addition to vegetation, many species rely on a range of structural refuges for nesting, raising young, and for protection from predators. These refuges can include burrows, rocky outcrops, hollows and coarse woody debris (Steen *et al.*, 2010; Bhattacharyya *et al.*, 2015; Loeb, 1999; Bull, 2002). Disturbances such as the establishment of timber plantations and land clearing for agriculture can alter the availability of hollows and coarse woody debris (Cunningham *et al.*, 2007) by removing the large trees that generate such habitat (McCarthy and Bailey, 1994). Severe wildfire can also reduce the occurrence of 10-30 cm diameter coarse woody debris (Bassett *et al.*, 2015a). Even prescribed burns, which are usually lower intensity than wildfires, can affect the availability of trees and logs suitable for nesting (Flanagan-Moodie *et al.*, 2018; Collins *et al.*, 2012; Holland *et al.*, 2017).

Disturbed landscapes can expose small mammals and reptiles to increased predation risk, as some predators hunt more effectively in these simplified landscapes (McGregor *et al.*, 2015; McDonald *et al.*, 2020; Davies *et al.*, 2020). In areas of Australia and New Zealand where introduced predators such as foxes (*Vulpes vulpes*) and cats (*Felis catus)* occur, this problem can be particularly pronounced (Hradsky *et al.*, 2017b; Hradsky, 2020; Doherty *et al.*, 2017; Frank *et al.*, 2014; Murphy *et al.*, 2019; Glen *et al.*, 2019; Hoare *et al.*, 2007). Graham *et al.* (2012) found that cats and foxes were more active in agricultural landscapes compared to remnant forest. Feral cats in Australia have also been observed to select areas recently disturbed by wildfires for hunting (McGregor *et al.*, 2017; McGregor *et al.*, 2014).

Given the prevalence of disturbances such as wildfire, plantation establishments, agriculture, livestock grazing and land clearing (Abram *et al.*, 2021b; Bradstock, 2010; Halofsky *et al.*, 2020; Jones *et al.*, 2020; Cordier *et al.*, 2021; Doherty *et al.*, 2020), it is crucial to mitigate their impacts on target species. To successfully implement mitigation strategies, land managers must be aware of the interventions or actions available to them. While these options are clear-cut for some frequently studied scenarios, we hypothesise that existing literature may provide insights for the management of under-studied species or disturbances. That is, responses to one disturbance may have the potential to be successful in response to a range of many others. Using a systematic map approach, we aim to identify: (1) which disturbances and threats to ground-dwelling vertebrates have been studied; (2) which management interventions have been used in disturbed landscapes to date; and (3) whether knowledge gaps exist in under-studied disturbances or management interventions.

2.3 Methods

Systematic mapping, similar to systematic reviews, aim to reduce bias and increase repeatability. Where systematic reviews aim to answer a specific question, systematic maps address broader scale questions and aim to identify trends and gaps across a field of study (James *et al.*, 2016). Systematic mapping follows a similar methodology to systematic review, with thorough scoping and the use of inclusion criteria to screen studies (James *et al.*, 2016). Rather than analysing effect sizes and reported results, systematic mapping collects and summarises categorical metadata to identify broader trends. Given the broad scope of our aims, this study is highly suited to systematic mapping.

2.3.1 Study selection

Our systematic map (James *et al.*, 2016) focused on reviewing the literature and identifying interventions that aimed to mitigate the negative effects of disturbance on small ground-dwelling vertebrate populations. We elected to focus on non-volant terrestrial vertebrates because a range of interventions for arboreal mammals, birds, and bats has already been well-established in the literature (Lambrechts *et al.*, 2010; Rueegger, 2016; Goldingay *et al.*, 2018). We focused on management interventions that aimed to mitigate common threats to ground-dwelling vertebrates after disturbance. Common threats include increased predation risk, loss of ground layer or shrubby vegetation, and loss of structural habitat elements such as coarse woody debris and ground-level hollows. We assessed interventions that were deployed after disturbances where the disturbance led to substantive habitat loss (e.g., wildfire, clearcutting, land clearing for agriculture). Interventions such as biomass reduction (via thinning, controlled burns, etc.) that are not suitable or realistic to employ immediately after substantial disturbances were excluded. A complete list of criteria for our systematic map, as well as relevant definitions and descriptions, can be found in the inclusion criteria list in Table S2-1.

To accommodate the breadth of our aims, and to support our decision to not focus on a single type of intervention, we used a search string that was broad enough to capture a range of relevant results (see supplementary material). We included grey literature (e.g., government reports, thesis, non-peer reviewed material) in the search and limited our results to literature written in English. The final search was conducted using Scopus on October 14, 2021 and returned 13, 697 results.

Results were then iteratively screened at the title, abstract, and full-text levels against pre-determined inclusion criteria (Table S2-1) using Covidence (Veritas Health Innovation, 2022). At the title and abstract level, studies were allowed to progress if they did not clearly violate any of the stated inclusion criteria. At full-text screening, a study was only included in the final analysis if it met all of the stated inclusion criteria. When a study did not provide enough detail in the title or abstract to accurately decide inclusion or exclusion, it was screened at full-text to avoid the possibility of falsely excluding relevant works.

2.3.2 Metadata collection

After the final selection of studies was made at the full-text stage, a range of metadata were extracted from each study (supplementary database). These metadata categories, including definitions for terms used, were determined as part of the a-priori methodology (see section 2.6.3). Where studies collected data on multiple vertebrate groups or interventions, each vertebrate*intervention combination was included in a separate line for analysis. For example, a study that examined the effects of predator exclusion on small mammals and salamanders would have two lines of data to record results for each vertebrate group. Once metadata were extracted, categories were coded and categorised to simplify analysis. We used the IUCN Threats Classification Scheme (IUCN, 2019) to categorise disturbance types (see Cowan *et al.* (2021)).

Disturbances, and their effects on ecological systems, are complex and difficult to categorise. For example, cattle grazing simultaneously removes and simplifies vegetation and provides beneficial conditions for introduced predators. Rather than simply recording disturbances and interventions, we added a third category to describe the threats to ground-dwelling vertebrate populations that each intervention was seeking to mitigate after a given disturbance.

We collected metadata on aspects of interventions that could be used by land managers to assess intervention viability. These metadata included whether studies reported any secondary costs or benefits to non-target species or ecosystem dynamics resulting from interventions, initial monetary or labour costs of implementing each intervention, ongoing monetary or labour costs of each intervention, and public perceptions of an intervention.

We did not collect data on the ‘success’ or ‘failure’ of each intervention. As stated, our systematic map aims to understand the breadth of the field of knowledge rather than answer a specific quantitative question. The broad range of interventions tested and methods used in our systematic map does not lend itself to a meta-analysis, and therefore any attempt to quantify the study outcomes would be a simple count of intervention success or failure. Such vote-counting is discouraged in established systematic map methodologies (James *et al.*, 2016).

2.3.3 Metadata synthesis

We grouped our analysis into six broad categories. These were: (1) geographic distribution; (2) vertebrate groups; (3) vegetation types; (4) disturbances and threats; (5) interventions; and (6) management considerations. Analysing disturbances, threats and interventions addressed the first and second aims of our study. We analysed vertebrate groups and vegetation types to identify understudied areas and potential knowledge gaps. We used summary statistics such as counts and frequencies to create bar graphs and other visual representations of our results.

We aimed to analyse each metadata category in a way that minimised unnecessary repetition of data. In the analysis of geographic distribution, vegetation, and management consideration each study was counted once. This is because while each study may examine multiple types of vertebrates or interventions, they were all conducted in a single country within one vegetation type. Disturbance type was analysed in a similar way, except for two studies that examined two types of disturbance and were counted twice. Management consideration data were analysed as a binomial yes/no entry for each study. Vertebrate group data were analysed so that each vertebrate group in a study was represented once, even if it was studied in multiple interventions. Stand-alone intervention data were analysed so that each intervention in a study was represented once, even if it was studied across multiple species. Analysis of various interactions between vertebrate groups, disturbance, threats, and interventions included all available data extracted during metadata collection.

2.4 Results

Our search string found 13,697 results on Scopus, which were sequentially assessed at the title, abstract, and full-text level to determine suitability for inclusion in our review. As our study sought to map the breadth of a subject, rather than narrow down on a specific topic, our review had a broad search string and returned a large number of potential results. Many of the results returned were clearly not related to our review, and 12,992 (95%) of our initial search results were excluded at the title stage. Sixty-nine studies were found to be suitable at full-text and were included in the analysis presented above (see Appendix 1).

2.4.1 Geographic distribution

Most studies took place in North America or Oceania. Australia accounted for nearly half of all studies, with 32 of the 69 studies. The United States was the location for 17 studies, five studies were conducted in New Zealand and four studies took place in Canada. Two studies were conducted in both the United Kingdom and Spain, while one study was conducted in each of the following countries: Argentina, Bolivia, France, Ghana, Guinea, Malaysia and Mauritius.

2.4.2 Vertebrate groups

Reptiles were the most commonly studied vertebrate group, with 41 studies including reptiles in their results. Thirty-one studies included small mammals, while nine studies focussed on amphibians and eight studies on medium or large mammals.

Table 2-1: Descriptions and definitions of the levels within each of the six metadata categories used for analysis in this chapter.

Metadata category	Levels
Geographic distribution	Country or countries where the study took place
Vertebrate group	Reptile Amphibian Small mammal (35 g – 5.5 kg) Medium/large mammal (> 5.5 kg)
Vegetation types	Forest Savanna Shrubland Grassland Rocky areas Desert Artificial – Terrestrial Marine coastal
Disturbances and threats	Invasive or overabundant species Pastoral land use Production forestry Agriculture Mining Biological resource use Residential development Keystone species decline Fire
Interventions	Predator control- baiting, shooting, trapping Predator exclusion- providing a physical barrier Herbivore exclusion- providing a physical barrier, complete destocking Herbivore control- partial destocking, reduced grazing intensity, baiting, shooting, trapping Coarse woody debris manipulation- adding CWD, creating CWD piles, CWD retention Revegetation- intentional replanting or reseeding Artificial refuge- burrows, corrugated metal, etc.
Management considerations	Secondary impacts to non-target species or ecosystem dynamics Cost of implementation Ongoing costs Public perceptions

2.4.3 Vegetation types

Forests were the most common habitat type, with 43 studies taking place in forested environments. Grasslands and artificial terrestrial environments were the next most common with six studies each. Four studies were conducted in marine coastal habitats, three studies in rocky areas and savanna habitats, two studies were conducted in shrubland and one study was conducted in a desert.

2.4.4 Disturbances and threats

Invasive or overabundant species was the most frequently studied disturbance (21 studies; Figure 2-1). Pastoral land use was examined in 13 studies, production forestry in 12 studies, agriculture in 9 studies and mining in 8 studies. Biological resource use was examined in three studies, residential development in two studies, and keystone species decline in one study. Fire was examined in two studies, both focused on mitigating the impacts of prescribed rather than wildfires.

Predation was the most commonly mitigated threat in response to invasive or overabundant species, followed by vegetation removal/simplification (Figure S1). The threat of limited refuge availability was not mitigated after disturbance by invasive or overabundant species. Vegetation simplification was the most commonly mitigated threat in response to livestock farming and ranching, followed by predation and limited refuge availability (Figure S2). Limited refuge availability was the only threat mitigated in production forestry plantations, as well as areas disturbed by keystone species decline and biological resource use (Figure S3). Vegetation removal and simplification was the most commonly mitigated threat in response to agriculture, followed by predation and limited refuge availability (Figure S4). Limited refuge availability and vegetation removal/simplification were the only threats mitigated after mining disturbances (Figure S5). Predation and vegetation removal were the only threats mitigated after fire (Figure S6).

Threats posed to reptiles from predation, limited refuge availability, and vegetation removal and simplification were addressed by a similar number of studies across our review (Figure 2-2). Vegetation removal and simplification was the most commonly mitigated threat for small mammals, followed by predation and limited refuge availability (Figure 2-3). For amphibians, limited refuge availability was mitigated three times more often than vegetation removal and six times more often than predation (Figure S7).

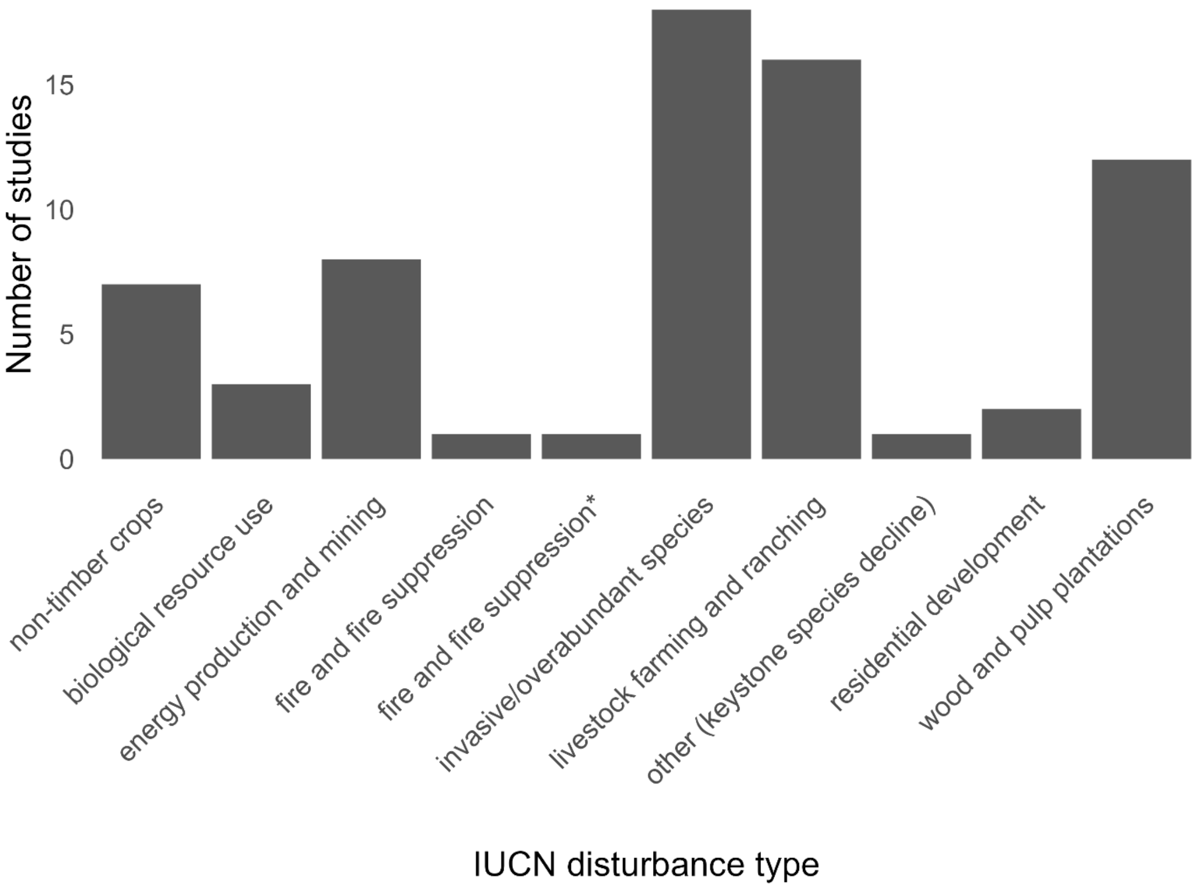


Figure 2-1: Frequency of studies within our review across nine disturbance types: (1) agriculture; (2) biological resource use; (3) energy production and mining; (4) fire; (5) invasive/overabundant species; (6) livestock farming and ranching; (7) keystone species decline; (8) residential development; and (9) wood and pulp plantations.

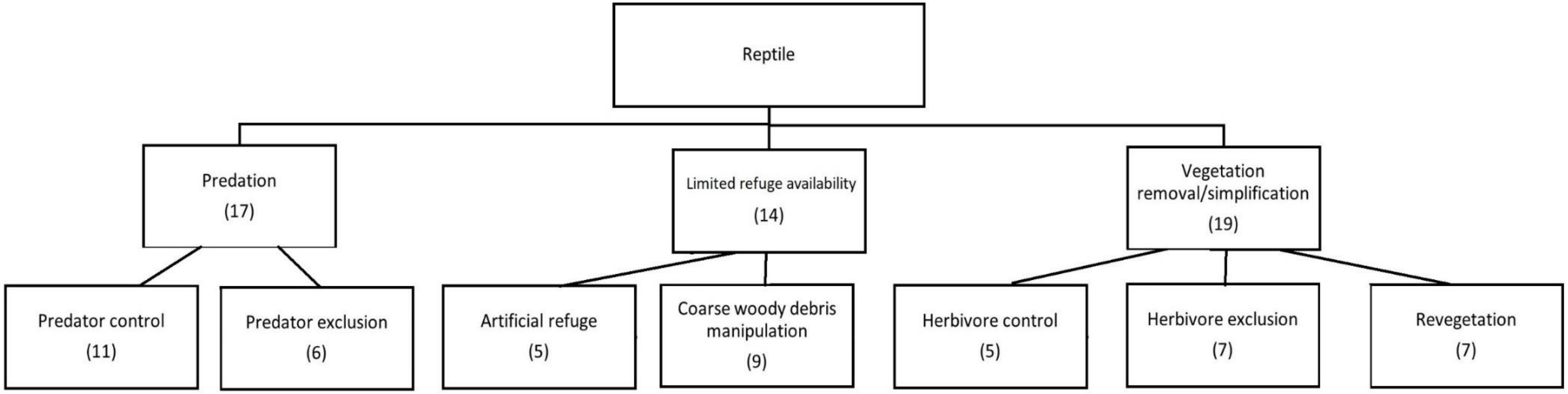


Figure 2-2: This flow chart represents the distribution of (1) studies in our review aimed to mitigate various threats to reptile populations; and (2) the subsequent interventions tested in response to those threats. Numbers in parentheses represent the number of studies in our review in the relevant category.

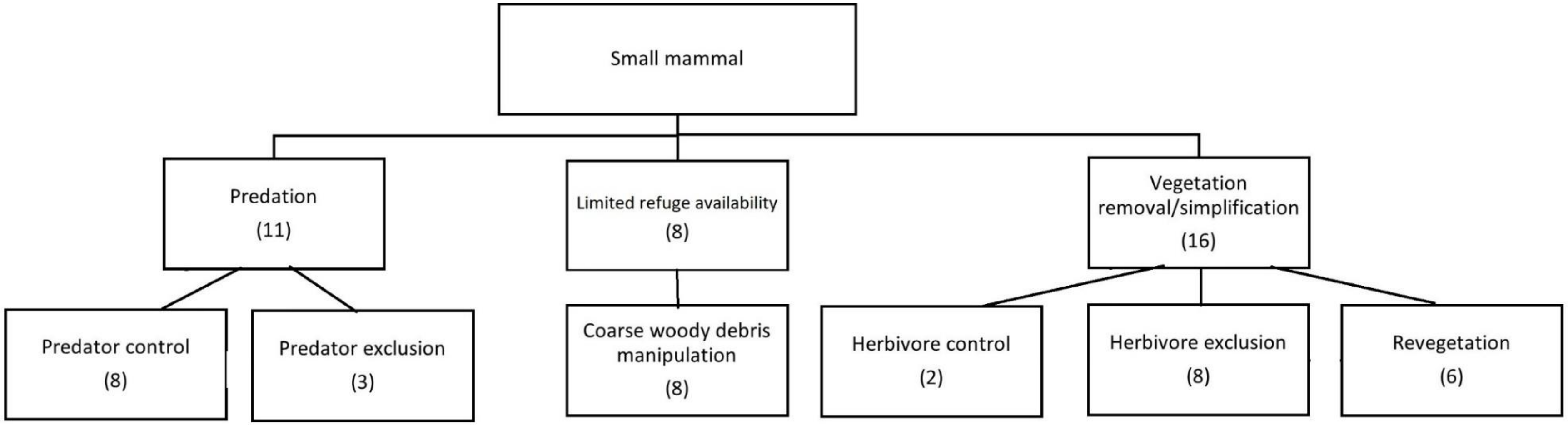


Figure 2-3: This flow chart represents the distribution of (1) studies in our review aimed to mitigate various threats to small mammal populations; and (2) the subsequent interventions tested in response to those threats. Numbers in parentheses represent the number of studies in our review in the relevant category.

2.4.5 Interventions

We identified seven interventions employed in disturbed landscapes (Figure 2-4). In this review, we will refer to lethal predator control methods (e.g., trapping, baiting, shooting) as predator control, and physical predator control (e.g., fencing) as predator exclusion (Table 2-1). Of the seven interventions, coarse woody debris manipulation was the most frequently tested with 18 studies (Figure 2-4). Predator control was used in 16 studies, while predator exclusion was used in 7 studies. Revegetation was used in 14 studies. Herbivore exclusion was used more frequently than herbivore control (12 and 6 studies, respectively). Artificial refuge was included in five studies.

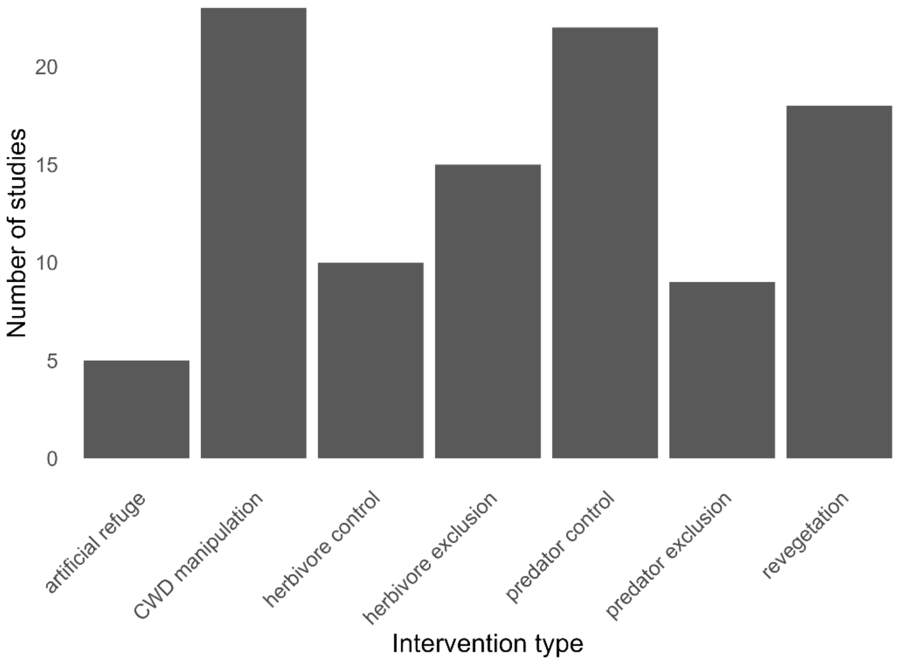


Figure 2-4: The frequency of studies experimentally testing seven management interventions in our review: (1) artificial refuge; (2) CWD manipulation; (3) herbivore control; (4) herbivore exclusion; (5) predator control; (6) predator exclusion; and (7) revegetation.

When interventions were assessed by vertebrate group, predator control was the most common intervention used to conserve reptiles (Figure 2-5). Coarse woody debris manipulation, predator control, and herbivore exclusion were the most common interventions used to conserve small mammals. Coarse woody debris manipulation was the most common intervention used to conserve amphibians in a terrestrial environment. Revegetation was the most common intervention used to conserve medium/large mammals.

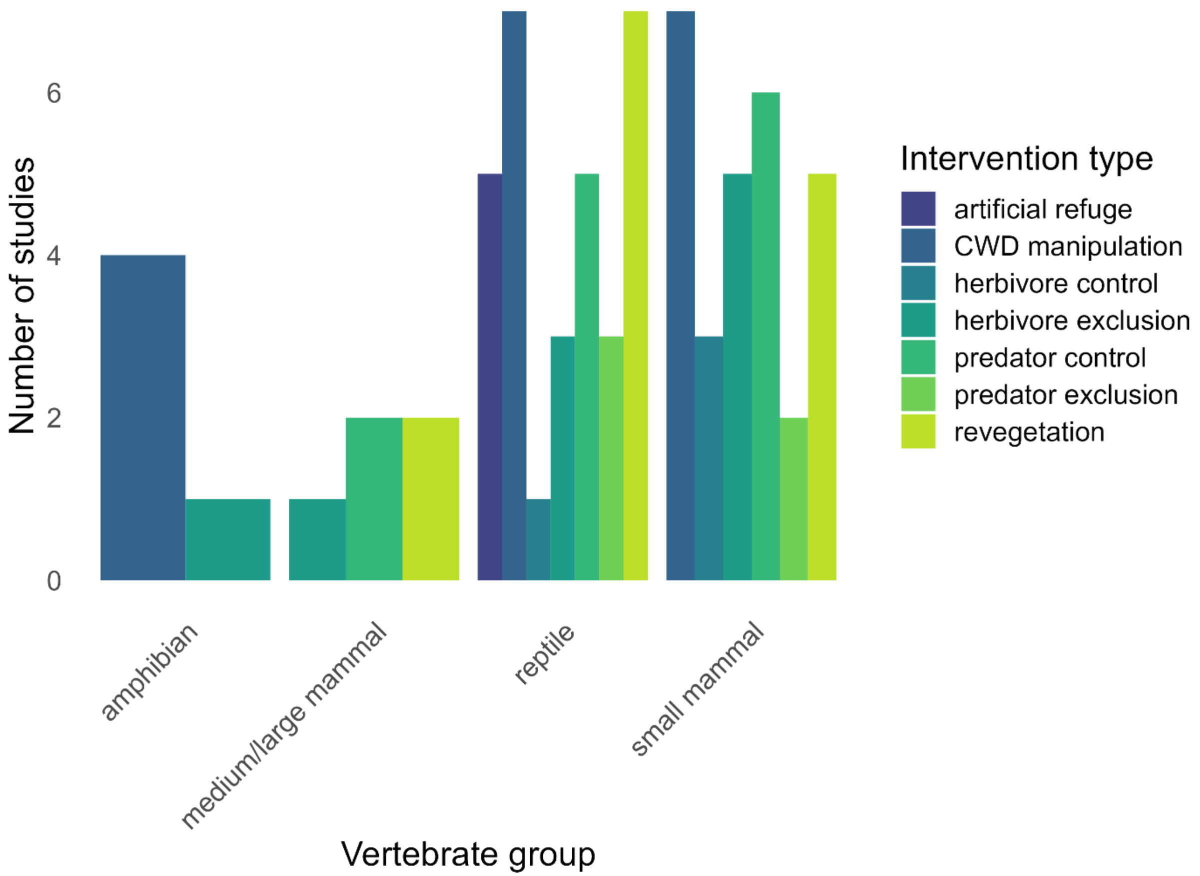


Figure 2-5: The frequency of various interventions used to conserve four vertebrate groups in our review: (1) amphibians; (2) medium/large mammals; (3) reptiles; and (4) small mammals.

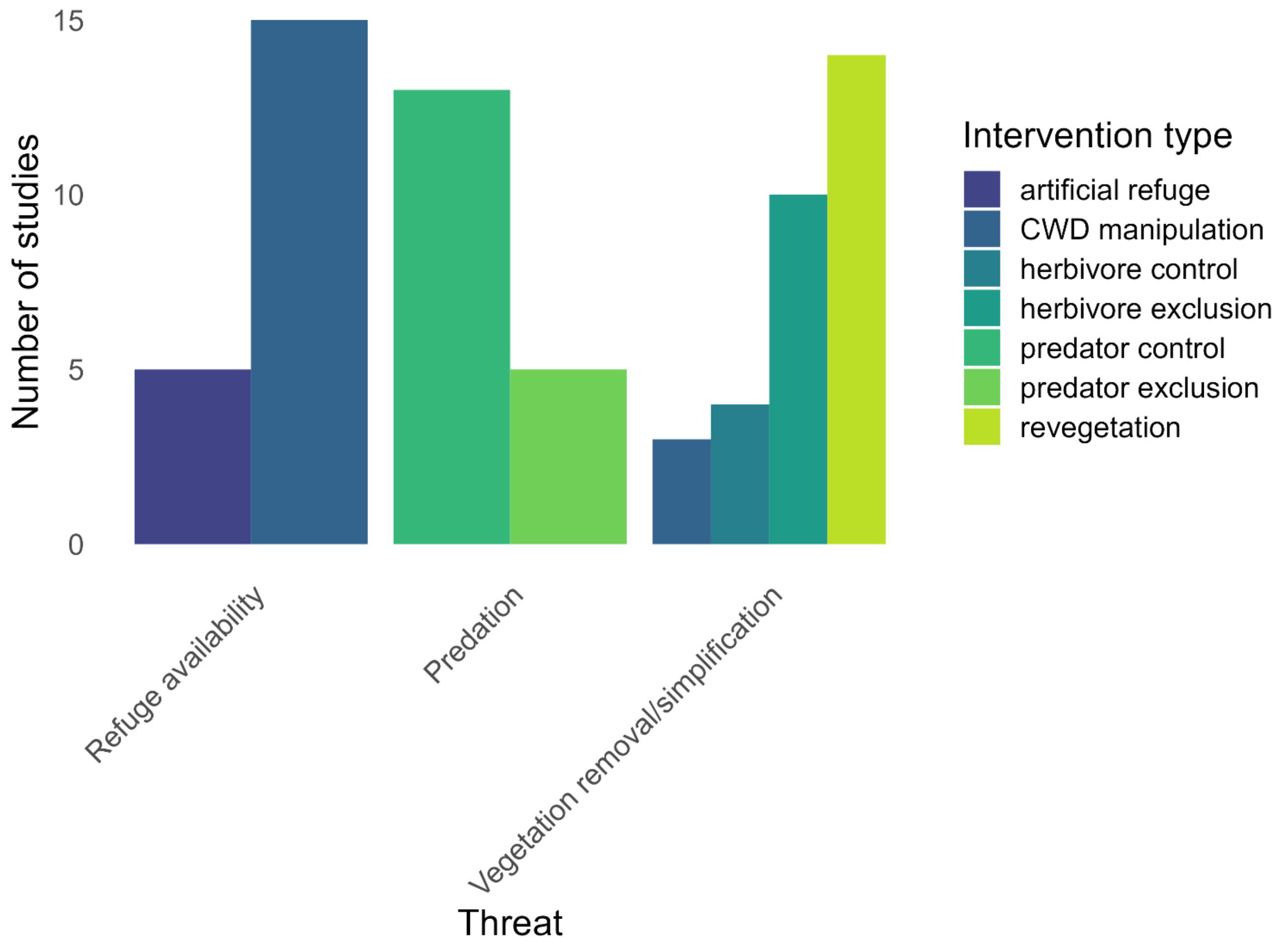


Figure 2-6: The frequency of various interventions used against the three threats identified in our review: (1) predation, (2) refuge availability, and (3) vegetation removal and simplification.

Predator control was the most common response to invasive or overabundant species and the threat of predation (Figure 2-6). Herbivore exclusion was the most commonly tested response to pastoral land use (Figure S2). Coarse woody debris manipulation was the only response tested in production forestry plantations (Figure S3) and the most commonly tested in response to limited refuge availability (Figure 2-6). Revegetation was the most commonly tested response to agriculture and mining (Figures S4, S5). Herbivore exclusion and revegetation were the most commonly tested responses to the threat of vegetation removal and simplification (Figure 2-6). Artificial refuge was the only intervention tested in response to biological resource use and keystone species decline. Predator exclusion and herbivore exclusion were the only interventions tested after prescribed fire.

2.4.6 Management considerations

Of the 69 studies in our review, 29 made some mention of secondary effects on non-target species or ecosystem dynamics. Only four studies mentioned the monetary or labour costs associated with implementing their intervention, three of which also included estimates for ongoing costs. No studies explicitly quantified the public perception of their chosen intervention.

2.5 Discussion

We conducted a systematic review of the literature which aimed to identify (1) which disturbances and threats to ground-dwelling vertebrates have been studied; (2) which management interventions have been used in disturbed landscapes to date; and (3) whether knowledge gaps exist in under-studied disturbances or management interventions. We found that invasive or overabundant species were the most studied disturbance. Coarse woody debris manipulation, predator control, and revegetation were the most studied interventions in our systematic map.

2.5.1 Geographic distribution

The geographic distribution of studies was spread between 14 countries, with the majority of studies conducted in Australia. This may be explained by the structure of our research aims, which focus on solutions to management issues common in Australia (e.g., wildfire, introduced predators, etc.). Our search was also limited to English-only results, likely creating a further bias towards publications from English-speaking countries (Stern and Kleijnen, 2020).

2.5.2 Vertebrate groups

Reptiles and small mammals were the focus of the majority of studies in our review. The abundance of reptiles and small mammals, and limited number of medium or large size mammals, can likely be explained by the types of interventions we included. For example, predator exclusion is typically aimed at protecting small ground-dwelling vertebrates rather than larger mammals which are less vulnerable to predation by invasive mesopredators. In our systematic map Conner *et al.* (2011) and Roshier *et al.* (2020) examined the effects of predator exclusion on small mammals, and Reardon *et al.* (2012) examined its effects on lizards. The smaller home range size of most small mammals and reptiles also present more feasible opportunities for implementing interventions. The limited number of studies on amphibians can be partly explained by our focus on terrestrial ecosystems. For example, there is a larger body of literature that examines protecting amphibians from aquatic predators that threaten juvenile development. However, amphibians also spend key life stages on land where they are vulnerable to some of the same threats as reptiles and other small vertebrates (i.e., land clearing; (Cordier *et al.*, 2021; Todd *et al.*, 2009)). This gap in the literature may be addressed in future by (1) expanding survey effort to include amphibians in areas where they are sympatric with relevant small mammals or reptiles; and (2) specifically studying interventions designed to mitigate barriers to amphibian survival after terrestrial disturbances.

2.5.3 Vegetation types

The majority of studies in our review were conducted in forested landscapes. This may be the result of the broad nature of the category, which encompasses most treed landscapes. The low number of studies in desert landscapes represents a knowledge gap, particularly as feral cats exert strong predatory pressure on reptiles in Australia's arid zone (Woinarski *et al.*, 2018). More studies in desert or arid landscapes would provide land managers with more information to mitigate predation and other emerging threats. Artificial terrestrial environments most commonly refers to agricultural land use in the context of this review.

2.5.4 Disturbances and threats

Only two studies in our review examined landscapes disturbed by fire. This is a relatively small number given the frequency of wildfire in Australia, the United States, and other countries represented in our review (Abram *et al.*, 2021b; Bradstock, 2010; Halofsky *et al.*, 2020; Jones *et al.*, 2020). Both studies tested interventions after prescribed fires rather than wildfires. This is important, as prescribed fires are often lower severity and can affect the landscape in different ways than unplanned wildfires (Price *et al.*, 2022). Wildfires are unplanned while prescribed burns are often planned and easier to fit into an experimental timeline. Of the two post-fire studies in our review, Conner *et al.* (2011) focused on addressing the threat of increased predation with predator exclusion while Foster *et al.* (2016) focused on promoting groundcover vegetation recovery using herbivore exclusion.

Limited refuge availability was not addressed by any post-fire studies in our systematic map, indicating a key gap in the literature. Small mammals and reptiles often use resources such as logs and leaf litter (Dickman, 1991; Flanagan-Moodie *et al.*, 2018; Loeb, 1996; McGregor and Burnett, 2014; Whiles and Grubaugh, 1996) which can be consumed by fire. These resources can take time to regenerate (Haslem *et al.*, 2011), and it is reasonable to expect these vertebrate groups might benefit from temporary alternatives immediately after a disturbance. Although not included in our systematic map, Sutherland (1999) examined the efficacy of artificial nests on semi-arboreal mammals after a prescribed burn. Studies such as this provide an example of the research that is necessary in this area.

In contrast, invasive or overabundant species were the most studied disturbance category in our review. Specifically, the threat from predation was most commonly addressed. This is unsurprising given the ecological damage caused by introduced predators in countries such as Australia and New Zealand. Feral cats, foxes, stoats and rats have rapidly adapted to these landscapes and are threatening a range of vertebrate groups, including reptiles and critical weight range mammals (Cree *et al.*, 1995; Doherty *et al.*, 2017; Stobo-Wilson *et al.*, 2021; Towns and Daugherty, 1994; Woinarski *et al.*, 2020; Woinarski *et al.*, 2018). In our review, responses to introduced or overabundant predators were predominantly predator control interventions, with nearly twice as many studies choosing it over predator exclusion. The other threat studies sought to mitigate in response to invasive or overabundant species was vegetation removal and simplification. This was in response to overabundant or invasive herbivore species (e.g., feral deer, rabbits, kangaroos, etc.; (Burns *et al.*, 2021; Porter *et al.*, 2014; Morgan, 2021; Mills *et al.*, 2020)). The two interventions used to mitigate vegetation removal were herbivore control and exclusion (e.g., (Buesching *et al.*, 2011; Bush *et al.*, 2012; Elsworth *et al.*, 2019)). The threat of limited refuge availability following this disturbance was not addressed by studies in our review, despite the success of coarse woody debris manipulation in mitigating predation pressure from introduced species in more controlled settings (Arthur *et al.*, 2005). This response is worth further study as land managers around the world work to conserve native species at high risk of predation by introduced predators.

Pastoral land use was the second most frequently studied disturbance in our review. In pastoral landscapes, grazing by livestock alters vegetation and other habitat characteristics (Yates *et al.*, 2000). Vegetation removal and simplification was the most common threat addressed in livestock-disturbed areas, which most studies sought to mitigate through herbivore exclusion (e.g., (Frank *et al.*, 2013; Kutt *et al.*, 2012; Leynaud and Bucher, 2005)). These interventions are relatively simple to undertake, as livestock can either be completely excluded or grazing pressure can be reduced by using existing fencing and infrastructure (Frank *et al.*, 2013; Giuliano and Homyack, 2004). Vertebrates in pastoral landscapes are also subject to predation (Graham *et al.*, 2012) and reduced refuge availability (Cunningham *et al.*, 2007). Six studies attempted to mitigate predation through predator control (e.g.,(Glen *et al.*, 2019; Allen *et al.*, 2014; Hoare *et al.*, 2007)), and two studies used coarse woody debris to improve habitat availability (Shoo *et al.*, 2014; Freeman *et al.*, 2021). However, the two studies using coarse woody debris to mitigate limited refuge availability were not conducted in active grazing landscapes, rather they were used to restore historically degraded landscapes.

Landscapes disturbed by agriculture face similar threats to pastoral land use (Gallmetzer and Schulze, 2015). While the reduced groundcover and simplified habitats are the result of monocultures rather than introduced livestock, the lack of shelter and predation protection remain problematic for ground-dwelling vertebrates. Studies in agricultural landscapes sought to mitigate vegetation removal and simplification by adding plantings between cropped areas or reclaiming unused or abandoned fields (Berry *et al.*, 2017; Cunningham *et al.*, 2007; Jellinek *et al.*, 2014; Kittipalawattanapol *et al.*, 2022). Coarse woody debris manipulation was the only intervention in agricultural areas that aimed to mitigate limited refuge availability (Evans *et al.*, 2019b; Manning *et al.*, 2013). With observed declines in mature trees in agricultural landscapes (Gibbons and Boak, 2002), it is reasonable to assume the availability of coarse woody debris and other related habitat elements would also become scarce. Only two studies in our review attempted to mitigate the lack of refuges in agricultural landscapes. These two studies were focused on restoring degraded landscapes, rather than mitigating threats in active agricultural landscapes. The low number of studies mitigating the threat of predation in agricultural settings is surprising, given the high hunting success introduced predators have in simplified landscapes (McGregor *et al.*, 2015) and the frequent use of this intervention in areas disturbed by pastoral land use. The lack of studies may be related to the landscape-scale effort necessary to achieve successful predator control, which requires coordination over multiple properties in agricultural landscapes to be effective. Facilitating or incentivising landowners to cooperatively engage in predator control may provide an opportunity for conservation and threat mitigation in this context.

In our review, limited refuge availability was the only threat addressed in response to disturbance from production forestry. Limited refuge availability was solely mitigated using coarse woody debris manipulation, where offcuts and slash were typically used to construct piles or rows that could be potential habitat for ground-dwelling vertebrates (Sullivan and Sullivan, 2012; Sullivan and Sullivan, 2021; Sullivan *et al.*, 2012). While these techniques are a logical response given the abundance of woody material from offcuts and slash, there is a significant gap in the literature examining alternative or supplementary interventions to mitigate predation, limited refuge availability, and vegetation simplification in landscapes used for production forestry. Artificial refuges may help mitigate habitat loss that cannot be replicated by coarse woody debris piles. While nest boxes have been tested for many arboreal species in forestry plantations (Taulman *et al.*, 1998; Smith and Agnew, 2002), studies of artificial refuges at ground level were not found in our review.

Revegetation was the most common mitigation measure implemented after disturbances from mining (Attuquayefio *et al.*, 2017; Craig *et al.*, 2017; Houston *et al.*, 2018; Nichols and Bamford, 1985; Triska *et al.*, 2016; Twigg and Fox, 1991). As most, if not all, vegetation is removed during mining operations this can be an effective solution to restore landscapes once mining has ceased. Coarse woody debris manipulation was also used in multiple studies to offset limited refuge availability after mining (Craig *et al.*, 2014; Márquez-Ferrando *et al.*, 2009). The threat of predation in mined areas was not addressed in any studies in our review, which may indicate a gap in the literature.

2.5.5 Interventions

Coarse woody debris manipulation was the most frequently used intervention among the studies in our review. It was the most common intervention studied to protect both small mammals and amphibians, likely because they both rely on coarse woody debris as refugia for nesting and protection from predation (Butts and McComb, 2000; Loeb, 1996). Amphibians may also use coarse woody debris to prevent desiccation and dehydration while on land (Whiles and Grubaugh, 1996).

Predator control was used more frequently than predator exclusion. Predator control was also the most commonly studied intervention for reptiles. Introduced predators, such as feral cats, can predate heavily on reptile populations, particularly in arid areas (Stobo-Wilson *et al.*, 2021; Woinarski *et al.*, 2018). Most predator control studies involved baiting at the large patch or landscape scale (e.g., Allen *et al.* (2014)), while most predator exclusion studies used predator-proof fences across smaller patches or sites (e.g., Conner *et al.* (2011)). The costs associated with predator exclusion can be prohibitive (Hayward and Kerley, 2009; Scofield *et al.*, 2011), and it is only feasible over small areas or on islands (Norbury *et al.*, 2014). Constructing and maintaining predator exclusion fences, as well as maintaining healthy populations within exclosures, can be expensive. Lethal control of introduced predators can be done over large areas of land and is less cost prohibitive. However, well-constructed predator exclusion is likely to eliminate predation from introduced species while the efficacy of predator control is variable ((Richards and Short, 2003; Allsop *et al.*, 2017; Fancourt *et al.*, 2021)). Predator control is also not suitable in all contexts because the baits should not be used in sensitive areas such as drinking water catchments.

In contrast to predator management, herbivore exclusion was used in more studies than herbivore control. Most herbivore exclusion utilised existing livestock fencing to exclude domestic livestock from paddocks. This is much less resource-intensive than predator exclusion which involves purpose-built fencing and intensive trapping to remove existing predators. Large herbivores are also much easier to exclude as they cannot climb or dig, making the fencing requirements less stringent than those designed to exclude predators such as foxes or cats. There were only two studies that examined the effects of controlling, rather than excluding, invasive invertebrate herbivores (feral rabbits, Elsworth *et al.* (2019) and North *et al.* (1994)). Control through aerial or ground-based culling is a common practice for large vertebrate herbivores (e.g., feral pigs, deer, horses) in many areas of Australia (e.g., Cox *et al.* (2023)) and New Zealand (Husheer and Robertson, 2005; Latham *et al.*, 2018). Many studies address best practice methodology for herbivore pest control, yet the effects of this control on sympatric native vertebrate populations appears to be underrepresented in the literature.

Revegetation was the subject of 14 studies in our review. It was primarily used as an intervention in response to mining and agriculture (Attuquayefio *et al.*, 2017; Berry *et al.*, 2017; Craig *et al.*, 2017; Cunningham *et al.*, 2007) to address the large vegetation loss that results from these disturbances. Revegetation was also the most common intervention type aimed to protect medium and large mammals (Berry *et al.*, 2017; Houston *et al.*, 2018; Kittipalawattanapol *et al.*, 2022; Smith *et al.*, 2015). Revegetation is typically conducted at larger scales and medium and large mammals, which have large home ranges, benefit from this response.

Artificial refuge was the least commonly studied intervention. The five studies testing artificial refuges examined the benefits of artificial rocks and artificial burrows (Croak *et al.*, 2010; Croak *et al.*, 2013; Grillet *et al.*, 2010; Webb and Shine, 2000; Zappalorti *et al.*, 2014) for a range of reptile species. No studies examined the effects of providing artificial refuges for ground-dwelling small mammals or amphibians during their terrestrial life phase, which face many of the same threats as reptiles (i.e., predation, habitat loss, etc.). Conversely, studies of artificial refuges such as nest boxes are common for arboreal vertebrates (i.e., tree-nesting birds, bats, and arboreal mammals). The lack of studies testing refuges for non-arboreal species represents a significant knowledge gap.

2.5.6 Management considerations

Land managers consider factors such as disturbance type and target species when choosing the most suitable intervention. Other factors that influence their decision include initial and ongoing costs, effectiveness, public perceptions of an intervention, and potential impacts on non-target species. Only four studies in our review reported initial costs (Comer *et al.*, 2020; Elsworth *et al.*, 2019; Engeman *et al.*, 2012; Seip *et al.*, 2018), three of which also included ongoing costs. This is not unique to our study area, as Wortley *et al.* (2013) found similar disparities in cost reporting in landscape restoration literature. No studies quantified the public perceptions of the interventions they studied, and a handful mentioned anecdotal or perceived public perceptions. It is understandable that collecting data on public opinion is not a standard part of land management research, and public opinion varies locally. Public opinion is still important as many land management agencies are publicly funded and responses may be hindered by strong negative opinions for certain management actions. Forty percent of studies referenced the effects of their interventions on non-target species or broader ecosystem dynamics in their study. This may be because it is relatively easy to collect data for multiple species once an experiment or intervention is established. This information is helpful for land managers who manage complex ecosystems and are concerned with the population health of multiple species.

2.5.7 Knowledge gaps and recommendations

- A key knowledge gap exists in studies of management interventions conducted in post-fire landscapes. While localised records of post-fire interventions may exist within land management organisations, there is a lack of data in the peer-reviewed literature. We recommend that more studies experimentally test the effectiveness of various interventions after wildfire. This should include further study of interventions included in this review (herbivore and predator exclusion), as well as interventions that have not yet been tested in a post-fire context (e.g., coarse woody debris manipulation) but have been effective after other disturbances (e.g., Freeman *et al.* (2021); Manning *et al.* (2013)).
- Through discussions with practitioners and our review of the literature, we have identified management actions that are occurring within land management agencies but the outcomes of which are not being reported in the literature (e.g., control of introduced herbivores such as feral deer and horses). We recommend that when such management actions are conducted, the outcomes are measured and subsequently reported beyond the relevant organisation. This would allow for increased knowledge sharing between agencies addressing similar issues. We also recommend increasing the number of studies that report the effects of these interventions on sympatric or prey species, not just the species controlled (e.g., recording prey population statistics after fox baiting rather than simply reporting the number of foxes removed).
- Interventions in response to production forestry and mining were limited to one or two commonly used interventions. We recommend future research broaden interventions to include potential alternatives that address a broader range of threats that benefit target species in these areas. Specifically, we recommend an increased focus on interventions to mitigate predation in areas disturbed by agriculture, mining and production forestry. We also recommend an increase in focus on limited refuge availability in areas disturbed by agriculture, pastoral land use, and invasive or overabundant species.
- We recommend that all future studies report estimates for materials and labour costs associated with each intervention tested. Ongoing costs should also be reported. This information could help land managers make more informed decisions on the relative cost-effectiveness of different interventions.
- We recommend that future studies should continue to report the effects of management interventions on non-target species if they are available.

2.6 Supplementary material

2.6.1 Inclusion criteria

Table S2-1: Inclusion criteria used to determine whether studies were within the scope of the systematic mapping methodology

Element	Criteria
Population	• Management action must be intended to benefit at least one non-volant terrestrial vertebrate (including amphibians)
Intervention	• Management actions must be conservation-focused and mitigate at least one of the following threats (1) predation; (2) loss of habitat complexity; or (3) loss of structural elements/habitat. • Interventions must take place after a natural or human-induced disturbance in a terrestrial environment, all terrestrial vegetation types are suitable for inclusion • Restoration is an acceptable intervention, however, specific restoration methods must be described and meet all other inclusion criteria • Arboreal interventions (i.e., nest boxes) are **not** suitable for inclusion • Studies with models only (and no field-based study) are **not** suitable for inclusion • Interventions that involve reducing biomass (i.e., thinning, grazing, prescribed burning) are **not** suitable for inclusion because these are not viable interventions in a post-fire landscape where biomass is limited • Interventions that cannot be implemented within 1-2 years after disturbance (i.e., thinning 20 years post-clearing) are **not** suitable for inclusion because we aim to assess management interventions that could be quickly implemented after a disturbance
Comparator	• Relevant comparators suitable for inclusion: (1) disturbed areas where management action was not applied (control/impact); (2) disturbed areas before management action was applied (before/after); (3) no comparator—observations of management action implemented after disturbance (after only)
Outcome	• Relevant outcomes include: population demographics, body condition, population size, population density, species richness, species abundance, capture rates, occupancy, and breeding success • For studies with artificial refuge interventions, suitable outcomes also include usage rates and occupancy • Behavioural outcomes and spatial dynamics (i.e., home range size) are **not** suitable for inclusion in relation to any intervention type

2.6.2 Search string

Title-Abstract-Keyword((Fauna* OR mammal OR reptil* OR herp* OR frog* OR amphibian* OR "small mammal*" OR rodent* OR vertebrat*) AND ("habitat complex*" OR "vegetation complex*" OR "ground* vegetation" OR "coarse woody debris" OR "habitat restor*" OR reveg* OR "herbiv* excl*" OR "herbiv* control*" OR "herbiv* remov*" OR "herbiv* reduc*" OR "graz* excl*" OR "graz* control*" OR "graz* remov*" OR "graz* reduc*" OR "brows* excl*" OR "brows* control*" OR "brows* reduc*" OR "debris pile*" OR understor* OR "ground cover") AND ("predat* excl*" OR "predat* control*" OR "predat* remov*" OR "predat* reduc*" OR "predat* bait*" OR "predat* trap*") AND ("artificial habitat" OR "supplement* habitat" OR shelt* OR den OR "artificial nest*" OR hollow* OR "artificial refug*" OR "artificial shelt*" OR "supp* shelt*" OR "tree cavit*" OR burrow* OR hibernic* OR "artificial structur*"))

2.6.3 Metadata terms and definitions

Reference ID: Resource identification number associated with the authors' Covidence database

Title: Title of journal article or resource

Author: Authors listed in order of publication

Year published: Year resource was published

DOI: DOI link to article (if available)

Publication type: type of resource

Continent: Continent where study took place

Study country: Country or countries where study took place

Study region: Region or state within the country where study took place (if available)

Site name: Specific site name where study took place (if available)

Study latitude and longitude: Coordinates of study site (if available)

Vertebrate group: Species in each study were categorised into one of four classes: reptile, amphibian, small mammal (35 g—5.5 kg), and medium/large mammal (>5.5 kg)

IUCN habitat type: The habitat/ecosystem type in each study was categorised using the IUCN Habitats Classification Scheme (Version 3.1)

IUCN disturbance type: The disturbance type in each study was categorised using the IUCN Threats Classification Scheme (Version 3.3). For succinctness, livestock farming or ranching will be referred to as 'pastoral land use', wood and pulp plantations will be referred to as 'production forestry', annual and perennial non-timber crops will be referred to as 'agriculture', and energy production and mining will be referred to as 'mining'.

Intervention type: The management intervention(s) tested in each study were categorised into seven classes: *predator control* (i.e., baiting, shooting, trapping), *predator exclusion* (providing a physical barrier), *herbivore exclusion* (providing a physical barrier, complete destocking), *herbivore control* (partial destocking, reduced grazing intensity, baiting, shooting, trapping), *coarse woody debris manipulation* (adding CWD, creating CWD piles, CWD retention), *revegetation* (intentional replanting or reseeding), *artificial refuge* (burrows, corrugated metal, etc.)

Threat: Given the disturbance described in the study, which of the following barriers to population persistence is the management intervention mitigating: predation, vegetation removal/simplification, reduced refuge availability

Experiment type: The experimental design of the study, categorised as either: control/impact, before/after, before/after/control/impact, or after only

Comparator: A description of what the interventions examined in the study are being compared to. For example, areas undergoing predator control would be compared to areas without predator control. For Before/After/Control/Impact experiments, there may be a comparator listed for the before/after aspect of the experiment as well as the control/impact aspect. Some control/impact studies may also have multiple comparators listed. After only experiments do not have any comparators.

Replication/number of disturbed sites: Information about the replication of each intervention and comparator in the study (i.e., how many times the intervention or comparator was tested/measured). For studies where data was collected for multiple species within the same experimental design, replication information will be listed for each species separately. For studies where multiple interventions or a control were tested, the number of replicates for each intervention/control will be listed.

Outcome metrics measured: Information about how intervention success was measured. For example, exclusion plot success can be measured using abundance or species richness within the herbivore/predator exclusion area.

Time in use: The amount of time that an intervention had been in place before the start of the study.

Time studied: The duration of time that the intervention, and its outcomes, were studied

Time after disturbance: The amount of time (measured in years) between when the disturbance occurred and the start of the study. For ongoing disturbances (i.e., introduced predators) the time after disturbance is zero years.

Secondary impacts: Any recorded/measured impacts on non-target species or ecosystem dynamics.

Scale of implementation: We used two levels of study scale: landscape and site. Landscape scale refers to interventions where each replicate was implemented over entire land management areas or multiple land use types (i.e., some baiting programs). Site scale refers to interventions where each replicate was confined to a single patch or management unit.

Cost of implementation: Must be explicitly linked to intervention tested in study and listed in a currency or labour amount.

Additional ongoing/maintenance costs: Must be explicitly linked to intervention tested in study (not implied) and listed in a currency or labour amount. For example, most herbivore or predator exclusion fences require ongoing maintenance but unless this is specifically quantified it is not listed.

Public perceptions of intervention: Must be explicitly mentioned and linked to intervention tested in study, not inferred. For example, culling of invasive or overabundant species can spark public debate or opinion, but unless it is explicitly stated it is not listed.

2.6.4 Supplementary figures

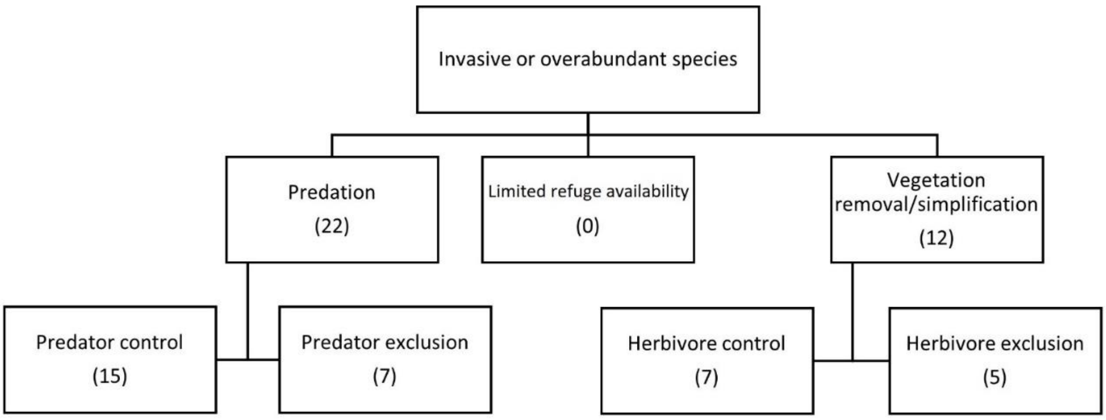


Figure S2.1: This flow chart represents the distribution of (1) studies in our review aimed to mitigate various threats to terrestrial vertebrates after disturbance by invasive or overabundant species; and (2) the subsequent interventions tested in response to those threats. Numbers in parentheses represent the number of studies in our review in the relevant category.

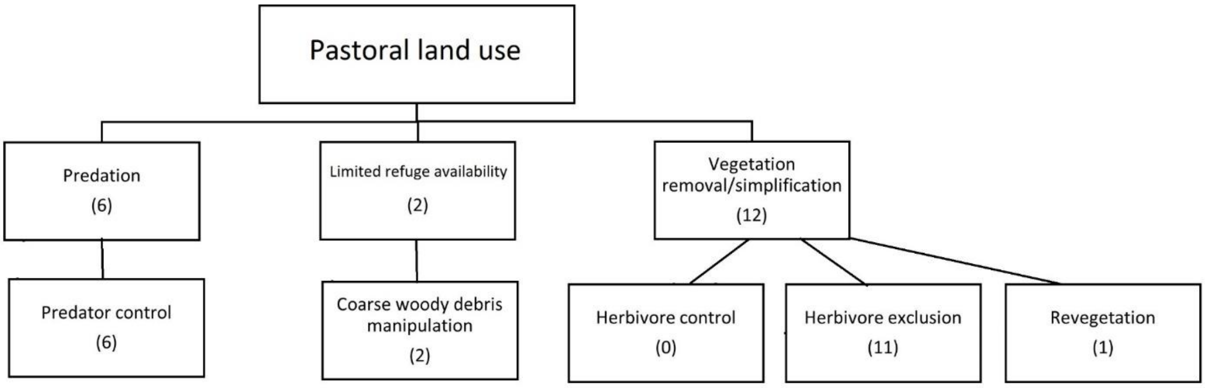


Figure S2.2: This flow chart represents the distribution of (1) studies in our review aimed to mitigate various threats to terrestrial vertebrates after disturbance from livestock farming and ranching; and (2) the subsequent interventions tested in response to those threats. Numbers in parentheses represent the number of studies in our review in the relevant category.

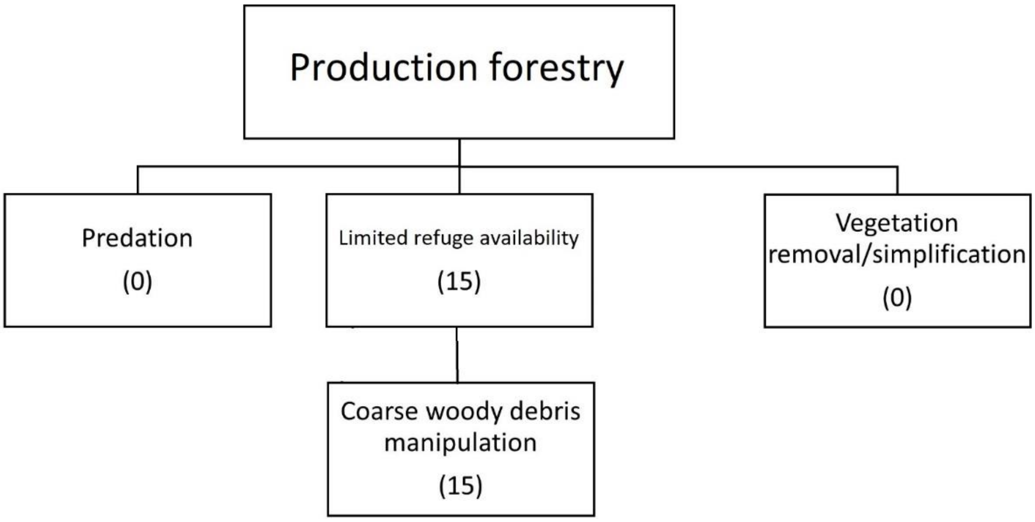


Figure S2.3: This flow chart represents the distribution of (1) studies in our review aimed to mitigate various threats to terrestrial vertebrates after disturbance from wood and pulp plantations; and (2) the subsequent interventions tested in response to those threats. Numbers in parentheses represent the number of studies in our review in the relevant category.

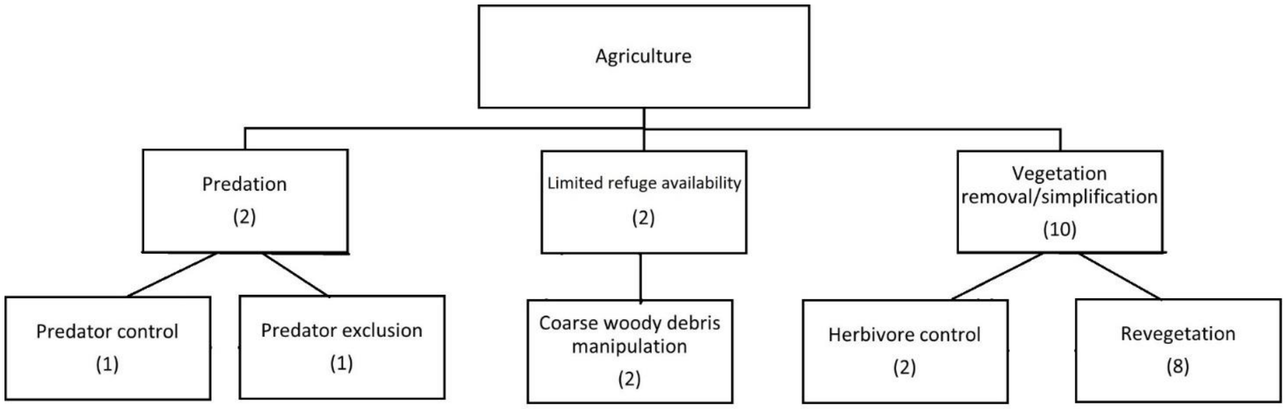


Figure S2.4: This flow chart represents the distribution of (1) studies in our review aimed to mitigate various threats to terrestrial vertebrates after agricultural disturbance; and (2) the subsequent interventions tested in response to those threats. Numbers in parentheses represent the number of studies in our review in the relevant category.

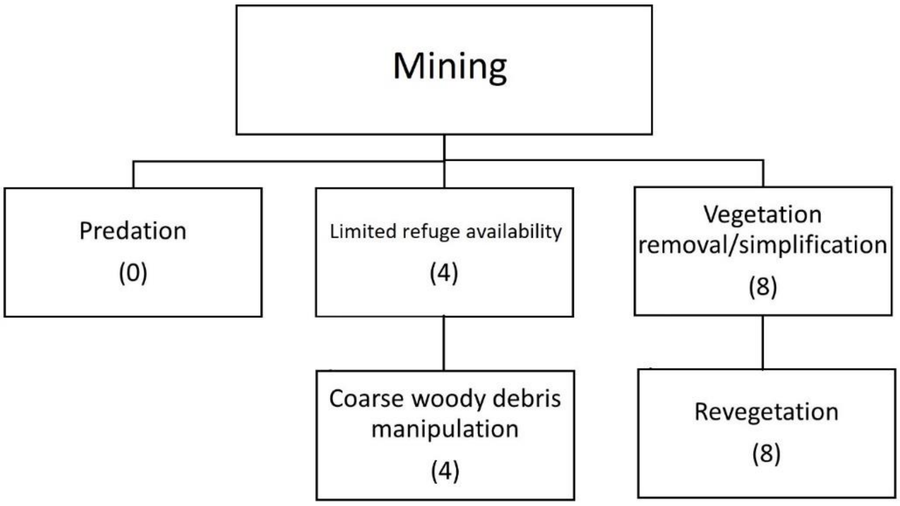


Figure S2.5: This flow chart represents the distribution of (1) studies in our review aimed to mitigate various threats to terrestrial vertebrates after energy production and mining disturbance; and (2) the subsequent interventions tested in response to those threats. Numbers in parentheses represent the number of studies in our review in the relevant category.

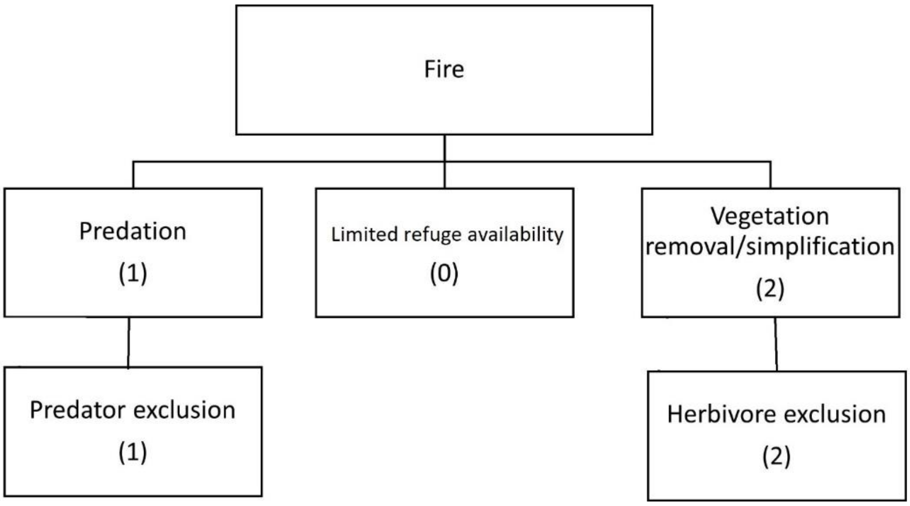


Figure S2.6: This flow chart represents the distribution of (1) studies in our review aimed to mitigate various threats to terrestrial vertebrates after fire; and (2) the subsequent interventions tested in response to those threats. Numbers in parentheses represent the number of studies in our review in the relevant category.

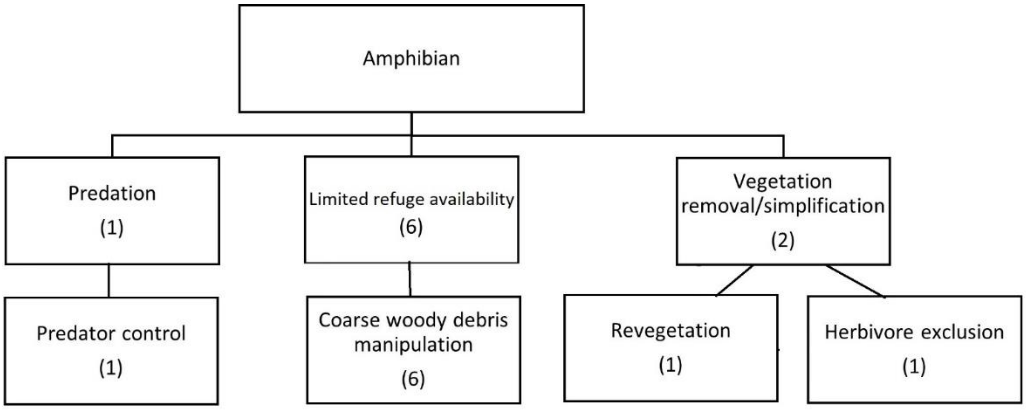


Figure S2.7: This flow chart represents the distribution of (1) studies in our review aimed to mitigate various threats to amphibian populations; and (2) the subsequent interventions tested in response to those threats. Numbers in parentheses represent the number of studies in our review in the relevant category.

Chapter 3: Coarse woody debris supplemental habitat significantly increases reptile abundance in burnt areas

Statement of Contribution

This thesis is submitted as a Thesis by Compilation in accordance with

https://policies.anu.edu.au/ppl/document/ANUP_003405

I declare that the research presented in this Thesis represents original work that I carried out during my candidature at the Australian National University, except for contributions to multi-author papers incorporated in the Thesis where my contributions are specified in this Statement of Contribution.

Title: Coarse woody debris supplemental habitat significantly increases reptile abundance in burnt areas

Authors: Burns, Heather; Cary, Geoffrey; Brawata, Renee; Lindenmayer, David; Connolly O'Donnell, Liam; Rathod, Divyang; Gibbons, Philip

Current status of paper: Not Yet Submitted/Submitted/Under Revision/Accepted/Published

Contribution to paper: HB designed the experiment with feedback from LCO, DR, PG, GC, RB and DL. LCO and DR established the experimental blocks, and HB conducted reptile and habitat surveys with help from LCO and DR. HB analysed the data and wrote the manuscript. PG, GC, RB and DL provided feedback on the manuscript before it was submitted to *Journal of Applied Ecology*.

Senior author or collaborating authors endorsement: Philip Gibbons ______________________________

Heather Burns ________________________ ________________________________ 08/07/2024 _____

Candidate – Print Name Signature Date

Endorsed

Philip Gibbons ________________________ ________________________________ 15/07/2024 _____

Chair of Supervisory Panel – Print Name Signature Date

Robert Dyball ________________________ 17/07/2024

Delegated Authority – Print Name Signature Date

Foreword

Chapter 2 found that there is a considerable lack of examples of post-fire interventions focused on conserving terrestrial vertebrates such as reptiles. When the broader literature was considered, predator control was the most common intervention used to conserve reptiles after disturbance. However, lethal predator control often requires permits and landscape-scale coordination which would have been difficult to achieve within the scope of this project. Coarse woody debris, the second most common intervention for reptiles, provided a more feasible option that had also been frequently used for small mammals (Chapter 2).

In this chapter we experimentally tested how supplementary coarse woody debris piles affected reptile abundance in areas burnt at high and low severity during the 2020 Orroral Valley Fire. The design of our coarse woody debris structures was based on a study of house mice by Arthur *et al.* (2005). I also examined how reptile abundance at each treatment was affected by other habitat features occurring in the same landscape, such as the presence of rocky outcrops, abundance of natural large coarse woody debris, and structural complexity of regenerating vegetation.

This chapter has recently been resubmitted to *Journal of Applied Ecology* with revisions based on feedback from reviewers.

3.1 Abstract

Prey species such as small reptiles are vulnerable to increased predation in burnt landscapes, particularly in the presence of introduced predators. Coarse woody debris (CWD) piles have supported the recovery of reptile populations in a range of other disturbed landscapes but have not yet been tested after fire. We tested three supplementary habitat treatments in an area of south-eastern Australia burnt during the 2019-2020 wildfires: (1) coarse woody debris piles (structure only); (2) segments of chicken wire (predation only); and (3) coarse woody debris piles covered with segments of chicken wire (structure + predation).

We aimed to answer the following questions: (1) What is the response of reptiles to adding supplementary habitat treatments to burnt areas?; (2) Are reptile responses to supplementary habitat affected by variables such as fire severity or vegetation structure?; and, (3) What is the cost-effectiveness of each treatment? To assess the impacts of our treatments we conducted visual and active surveys of reptiles in and around our supplementary habitat over the course of two austral summers. We also surveyed habitat complexity, naturally occurring CWD, and rocky outcrops at each site to determine whether the surrounding habitat affected how reptiles used each treatment. We found that a significantly higher abundance of reptiles and a significantly higher probability of occurrence of the most common taxon (*Lampropholis* spp.) were associated with constructed piles of CWD. Compared to the control, reptile abundance was twice as high at sites with CWD piles and three times as high at sites with the CWD + chicken wire treatment. The structure only treatment was the most cost-effective, with an estimated increase of 0.019 reptiles per $AUD spent on labour and materials. These results demonstrate the benefits of a novel application of coarse woody debris piles as a management tool for reptile conservation in burnt areas.

3.2 Introduction

Predation, particularly in landscapes with introduced predators, can threaten reptile populations. For reptile species that rely on groundcover to avoid predation, recently burnt areas can present increased predation risks (Doherty *et al.*, 2022). Research has shown that introduced predators prefer to hunt in recently burnt areas (Hradsky, 2020; Robley *et al.*, 2016; Hradsky *et al.*, 2017c) and are more effective hunters in simplified landscapes. For example, in north-western Australia feral cats strongly selected recently burnt areas for hunting (McGregor *et al.*, 2017; McGregor *et al.*, 2014). Hradsky *et al.* (2017a) also found that introduced predators occurred five times as often in recently burnt areas compared with unburned controls. As fire becomes more frequent in certain parts of the globe (Abram *et al.*, 2021b; Bradstock, 2010; Halofsky *et al.*, 2020; Jones *et al.*, 2020; Lindenmayer *et al.*, 2023), there is a risk of amplified predation.

Eradicating introduced predators to reduce predation risk is not practical in areas where populations have become established (Victoria Department of Primary Industries, 2010), and predator management is a widely adopted alternative. Traditional predator management centres on predator exclusion and predator control. Predator exclusion, typically through the construction of purpose-built fencing, is resource intensive and best suited to areas over 5,000 hectares (Clapperton and Day, 2001). There are substantial initial and ongoing costs, including fencing materials and labour costs, the initial eradication of predators, and the continued maintenance of the fence (Bode and Wintle, 2010). Nevertheless, exclusion fencing can be successful at protecting reptile populations. For example, Stokeld *et al.* (2018) found that reptile abundance increased twice as quickly when feral cats were excluded compared to a control.

An alternative option for mitigating the impacts of introduced predators is lethal predator control (e.g., shooting, trapping, poisoning, or biological control). Predator control can be more cost-effective than exclusion fencing in smaller reserves (Clapperton and Day, 2001) and has proven to be effective when appropriate levels of effort are maintained (e.g., Engeman *et al.* (2012)), but there are limitations. The effects of predator control is temporary, except on islands, due to migration from surrounding areas (Lieury *et al.*, 2015) and therefore control must be continued indefinitely to be effective. Bait aversion and resistance can also develop in target populations, limiting the success of future control efforts (Allsop *et al.*, 2017).

As neither predator exclusion nor control is feasible in every management scenario, land managers would benefit from additional mitigation strategies. Coexistence conservation (Evans *et al.*, 2022) states that predation management, rather than predator control, can lead to more sustainable solutions to manage dynamics between native prey and introduced predators. Management strategies should be developed, where possible, to allow prey species to successfully coexist with predators outside exclosures or heavily baited landscapes. One potential avenue for promoting successful coexistence is improving or restoring habitat complexity in degraded or disturbed landscapes. Introduced predators prefer to hunt in simplified landscapes (McDonald *et al.*, 2020; McGregor *et al.*, 2014), therefore Doherty *et al.* (2015) argue that the maintenance and creation of more complex landscapes may be a more effective long-term solution than predator control or exclusion. Creating more complex habitats can include more immediate post-disturbance solutions such as the creation of artificial refuges (e.g. (McDougall *et al.*, 2016; Webb and Shine, 2000)), or returning structures that provide refuge from predators such as coarse woody debris (CWD) but that are depleted after disturbance (Christie *et al.*, 2013; Davis *et al.*, 2010).

The addition of CWD of varying ages and decay stages can benefit a variety of reptile species (Michael *et al.*, 2004). This approach has been successfully used to facilitate the recolonisation of skinks in restored mines in Western Australia (Christie *et al.*, 2013) and young rainforest revegetation plantings in Queensland (Freeman *et al.*, 2021; Shoo *et al.*, 2014). In areas degraded by grazing and agriculture, Evans *et al.* (2019b) and Manning *et al.* (2013) found that supplementary CWD is most beneficial for reptiles in areas with reduced canopy or shrub cover. Although shrub cover may increase rapidly after fire, ground-level attributes such as leaf litter can take much longer to recover (Arthur *et al.*, 2012). This may be particularly important for species that rely on complex habitat to provide protection from predators, where post-fire landscapes can present increased risk (Doherty *et al.*, 2022).

In 2019-2020, the eastern coast of Australia experienced the most extensive wildfires since European colonisation of Australia (Boer *et al.*, 2020; Bowman *et al.*, 2020). As a result, south-eastern Australia experienced the largest area of land burnt at high severity in the previous three decades (Collins *et al.*, 2021). Compared to areas burnt at low severity, high severity fires can promote dense understory growth in eucalypt forests (Barker *et al.*, 2022), which can benefit some taxa but not others (Catling *et al.*, 2001). These factors can influence habitat availability and species responses to post-fire landscapes (Haslem *et al.*, 2011; Arthur *et al.*, 2012; Fox, 1982).

Our work aimed to answer the following questions: (1) What is the response of reptiles to adding various supplementary habitat treatments to areas burnt extensively during these fires?; (2) Are reptile responses to supplementary habitat affected by variables such as fire severity or vegetation structure?; And, (3) What is the cost-effectiveness of each treatment?. Specifically, three supplementary habitat treatments were tested: CWD piles, segments of chicken wire, and CWD piles covered in segments of chicken wire. We hypothesise that our CWD structures will increase the amount of protection from predators and result in an increase in reptile abundance. Although this has not been directly tested on reptile populations previously, Arthur *et al.* (2005) found that CWD piles covered in chicken wire provided similar predation protection to a complete predator exclusion treatment for house mice. We also hypothesise that the response of reptiles to fire severity and vegetation may vary by species, depending on species-specific preferences for vegetation structure and availability of direct sunlight for thermoregulation (Hossack *et al.*, 2009; Klingenböck *et al.*, 2000; Pearson *et al.*, 2003; Pringle *et al.*, 2003).

3.3 Methods

Our study area was located in 1061 km² Namadgi National Park, Australian Capital Territory in south-eastern Australia (35.558147 S, 148.940715 E; Land Management and Planning Division (2010)). Namadgi National Park contains a mix of alpine and sub-alpine habitat, dry sclerophyll forest and wet sclerophyll forest (Keith and Simpson, 2012). Elevation in the park ranges from 900 to 1800 metres above sea level, with our study sites situated between 900-1200 metres. Mean annual rainfall in the park is 1091 mm, winter (July) temperature ranges from -2.4°C to 2.6°C and summer (January) temperature ranges from 10.1°C to 20.7°C (Bureau of Meteorology, 2024). The park is home to a range of reptile species, including seven species of snakes and 33 species of lizards (Land Management and Planning Division, 2010). Introduced predators include feral cats (*Felis catus)*, foxes (*Vulpes vulpes)* and feral pigs (*Sus scrofa)*. The national park has experienced multiple wildfires in the past two decades, most recently in January and February of 2020 where an estimated 80% of the park was burnt.

In September 2021, we established a randomised block design experiment consisting of 20 blocks that were equally divided between two strata of fire severity: (1) blocks burnt at high severity in the 2020 fires; and (2) blocks burnt at low severity in the 2020 fires (Figure 3-1). Fire severity was mapped and classified using RdNBR values (Gale *et al.*, 2021) in ArcGIS (Environmental Systems Research Institute, 2022). Once these classes were mapped and tentative study sites were selected, we ground-truthed each location before establishing experimental blocks. For a site to be classified as high severity, the canopy must have been >75% consumed by fire. For a site to be classified as low severity, the canopy must have experienced <25% crown scorch. There were no minimum patch sizes for each burnt area, but each block was located completely within its designated severity class with a buffer of at least 100 metres.

Unburnt sites were not included in this study due to the only unburnt forest being a small patch in the southern portion of the park. All blocks were located within the dominant dry sclerophyll forest vegetation type in the study area. Due to the remote nature of our study area, blocks were necessarily clustered along fire trails and roads. Within these clusters, blocks were a minimum of 150 metres apart and distances between clusters ranged from one to 11 kilometres.

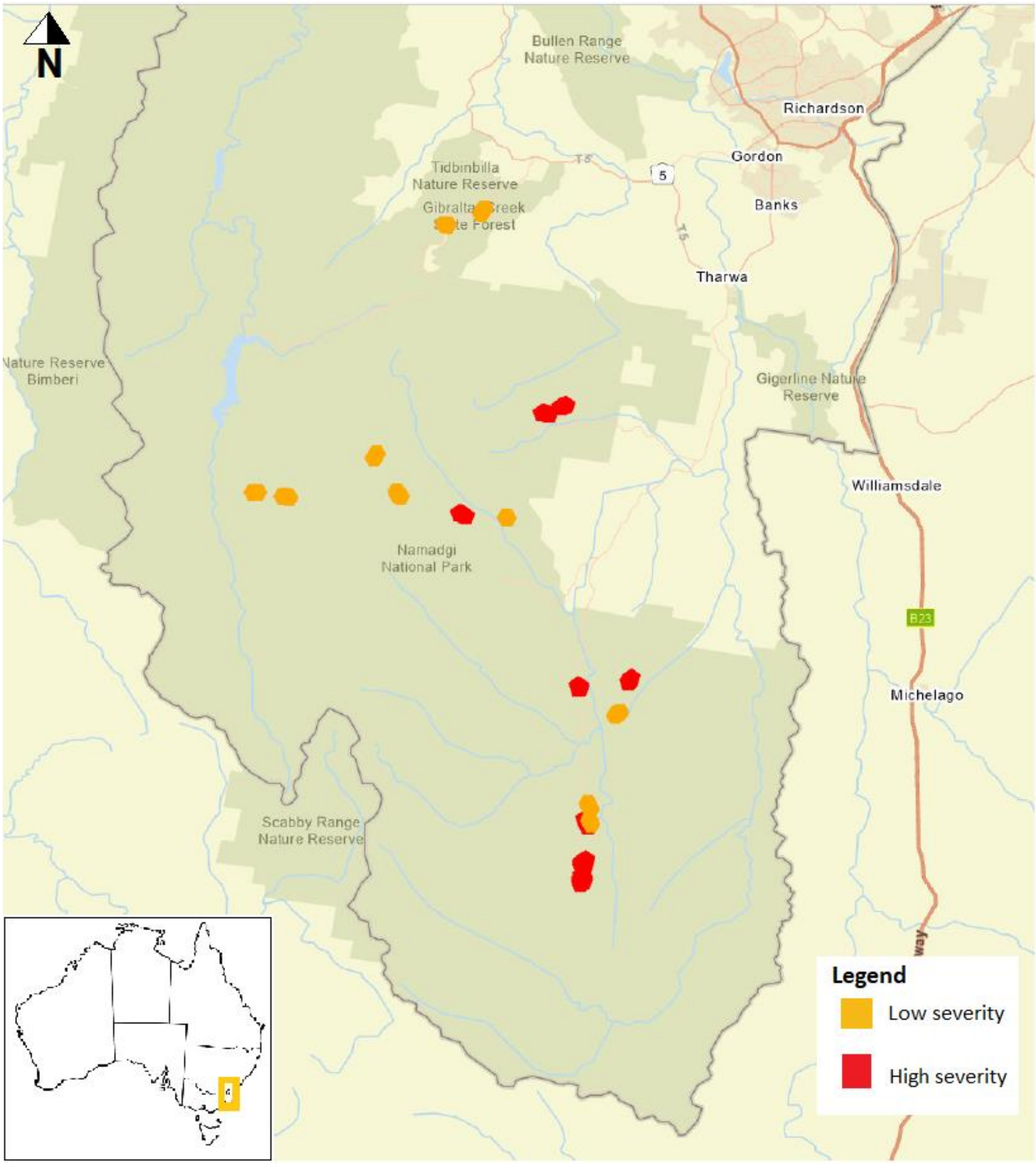


Figure 3-1: A map of 20 experimental blocks across Namadgi National Park, Australian Capital Territory. Blocks were distributed evenly across areas burnt at high severity (red) and low severity (orange) in the 2020 Orroral Valley Fire.

At each block we established four refuge treatments: (1) control; (2) structure only; (3) predation only; and (4) structure + predation. Structure only treatments were constructed using small pieces of CWD (predominantly 5-10 cm. diameter) collected from the surrounding landscape. The CWD was stacked into four piles approximately 3 metres long and 1 metre wide, which were arranged in a square (see Figure 3-2). Predation only treatments were built using 3 metre x 1 metre sections of chicken wire, secured with steel pegs, arranged in the same formation as the structure-only treatment. The chicken wire had a 5 cm aperture that allowed reptiles to move freely but would limit access by predators. Structure + predation treatments were constructed using four CWD piles (similar to the structure only treatment) that were covered with chicken wire (see Arthur et al. (2005) for similar design). The chicken wire had a 5 cm aperture, and was approximately 4 m long and 1.5 m wide to cover the entire CWD pile. Once in place, the chicken wire was secured with steel pegs around the perimeter of the CWD pile. Control sites were selected in the same way as all other treatments but did not contain any CWD piles or chicken wire. All treatments were at least 50 metres from the road and 100 metres from another treatment to minimise the risk of reptiles moving between sites (see Figure S3-1). Although some *Lampropholis* spp. can disperse over a kilometre, most individuals have been found to move less than 30 metres (Pulsford et al., 2018). For the purpose of this paper, we refer to a ‘site’ as a unique treatment x block combination. In total, there were 80 sites in our study (20 blocks x 4 treatments per block).

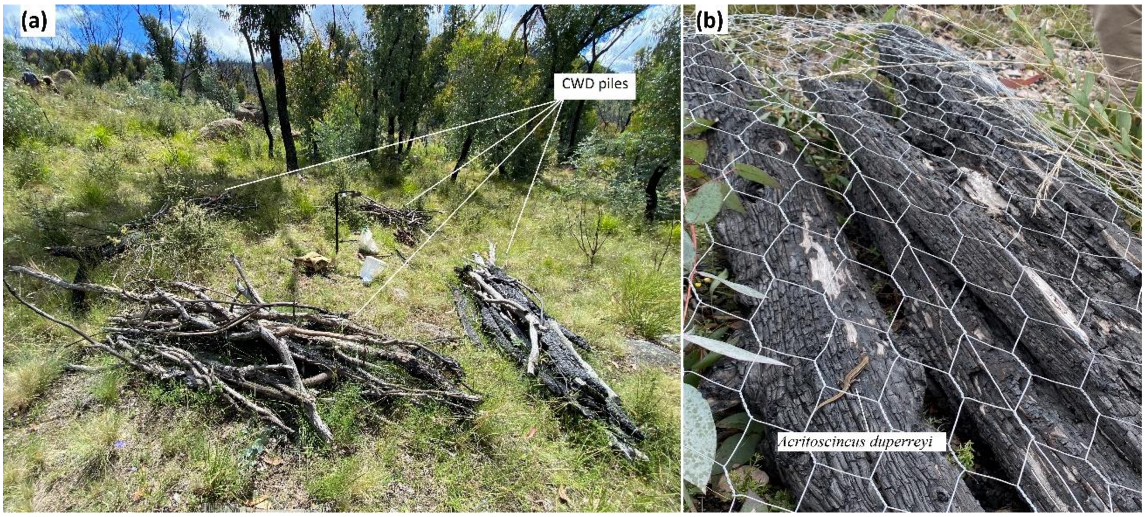


Figure 3-2: Photographs of (a) a structure treatment with four CWD piles established in our study area; and (b) an *Acritoscincus duperreyi* individual seen using the structure + predation treatment during a visual survey. All treatments (structure, predation, and structure + predation) were built in a similar orientation to photo (a) with four replicates of each treatment arranged in a square.

3.3.1 Data collection

We surveyed each site during two Austral summers, once in January-March 2022 and again in December 2022-March 2023. Each survey round included a visual survey and an active survey to maximise the diversity of species recorded (Dixon *et al.*, 2018; Michael *et al.*, 2012). For visual surveys, two observers methodically searched the site area and recorded all reptiles that were visible basking on logs and tree trunks (Dixon *et al.*, 2018). For active surveys, two observers carefully searched potential refugia such as under logs and rocks, within grass tussocks, and behind loose bark and recorded all reptiles (Dixon *et al.*, 2018). Both active and visual surveys were limited to 20 minutes and covered the area within a 10-metre radius of the treatments. For both structure only and structure + predation treatments, active surveys included deconstructing CWD piles, and removing chicken wire where present, to search for reptiles. All surveys were conducted on sunny or mostly sunny days when temperatures were between 16°C and 30°C (Dixon *et al.*, 2018). Although one survey per season may result in some false absences, the occurrence of La Nina events during each survey season made it difficult to access each site more than once (Bureau of Meteorology, 2023). The use of multiple complimentary survey approaches also increases the likelihood of detection across a range of species (Dixon *et al.*, 2018).

At each site we collected data on habitat variables that could affect reptile abundance. This included the presence/absence of rocky outcrops, total length of natural CWD, and habitat complexity (Table 3-1). Habitat complexity surveys allowed us to account for the varying effects of time since fire and fire severity on vegetation structure. This was particularly relevant as our study area experienced above average rainfall as the result of La Nina conditions during our study (Bureau of Meteorology, 2023). Rocky outcrops and large coarse woody debris also represent natural alternative sources of habitat for reptiles, and their availability may have affected reptile abundance at our artificial treatments.

Table 3-1: A description of explanatory variables used in the modelling of *Lampropholis* spp. probability of occurrence and combined reptile abundance. Descriptions include survey methods and survey frequency.

Explanatory variables	Survey method	Data description
Fire severity	Fire severity was determined using spatial relativised delta normalised burn ratio (RdNBR; Miller and Thode (2007)) data for Namadgi National Park (Gale *et al.*, 2021).	Categorical variable with two categories: high fire severity (>75% canopy consumption) and low fire severity (<25% canopy scorch or consumption).
Rocky outcrops	We recorded the presence/absence of rocky outcrops within 25 metres of each treatment. This treatment was conducted once as the presence or absence of rocky outcrops was unlikely to change over time.	Categorical variable with two categories: present and absent
Natural CWD	We recorded the total length (in metres) of large CWD (> 20 cm. diameter) within 10 metres of each treatment. We conducted this survey once during the second survey season.	Continuous variable
Habitat complexity	We established two 25-metre transects that ran parallel to our treatments. At five metre intervals along the transect we used percent cover of five variables to assess habitat complexity; (1) canopy cover (above 2 m); (2) shrub cover (0.5-2 m); (3) ground cover (<0.5 m); (4) bare ground; and (5) leaf litter and fine woody debris. Canopy cover was measured using the %Cover app (Public Interest Enterprises, 2021), while all other variables were estimated using a 50cm x 50 cm quadrat.	Continuous variable

These surveys were conducted annually (2 times in total) due to the rapid regrowth of vegetation in our study area

3.3.2 Data analysis

We combined data from visual and active surveys to produce a single abundance measure for each reptile species. Rather than simply summing the abundances across both surveys, we recorded the maximum value for each species across both surveys as their single abundance value for analysis. Given the visual and active surveys were conducted consecutively in the same site visit, this provided a maximum number of individuals for each species per survey day. This prevented double counting of individuals between surveys, as it is difficult to identify individuals in many skink species and the same individuals may have been observed in both surveys.

Due to low recorded abundances we only modelled presence-absence for one species (*Lampropholis* spp.). We also created a combined reptile abundance metric to assess the effects of our treatments on the general reptile community in our study area. To create this metric we summed the abundance of all species, including *Lampropholis* spp., at the site x survey period level. This resulted in a maximum count of all reptile individuals seen at each treatment x site during each survey season.

Habitat variables were tested for correlation, and bare ground was removed from the analysis because it was highly correlated ($r > 0.7$) with the ground cover variable. The remaining four variables (ground cover, leaf litter, shrub cover, and canopy cover) were averaged at the site level to create a single value per site each year.

To test the effects of our treatments on reptile abundance, we fitted several linear mixed models (LMMs) using two response variables (combined reptile abundance and *Lampropholis* spp. presence/absence) using data collected during visual and active surveys. To account for variation between samples that is not associated with treatments, we also included survey season, fire severity, presence/absence of rocky outcrops, total length of natural CWD, and average percent cover of three habitat variables (ground cover, shrub cover, and leaf litter) as potential explanatory variables in each model (see Table 3-1). For both reptile abundance and *Lampropholis* spp. presence, we ran 10 candidate models designed to test: (1) our null hypothesis that reptile abundance is not affected by habitat variables or supplementary habitat; (2) the effects of the supplementary habitat treatments; (3) the relative importance of fire severity and habitat structure; (4) the effects of natural habitat such as rocky outcrops and large CWD; and (5) potential interactions between the supplementary habitat treatments and fire severity or habitat structure (Table S3-1, Table S3-2). Block and site were treated as random effects in each model.

We fitted the models using the 'glmmTMB' package (Brooks, 2017) in R (R Core Development Team, 2022) and compared model fits using Akaike's Information Criterion (AIC), selecting the simplest model within two delta AIC scores of the top-ranked model for each response variable (Arnold, 2010; Burnham and Anderson, 2002). *Lampropholis* spp. presence/absence models were modelled using a binomial distribution while combined reptile abundance was modelled using a Poisson distribution. All predictions were made while holding all other fixed effects in the model either at their mean (for continuous variables) or mode (categorical variables). We checked models for normality among residuals, overdispersion, and correlation among fixed effects using the Performance package in R. Across all models, variables that produce a p-value <0.05 will be presented in the Results section.

For each best-fit model of combined reptile abundance and *Lampropholis* spp. we conducted post-hoc estimated marginal means tests using the package emmeans in R (Lenth *et al.*, 2018). This allowed us to compare treatments to each other as well as the control.

Our analysis did not include detection probability in the range of candidate models tested. While temperature and cloud cover may have affected activity levels of reptiles in our study area, we attempted to mitigate this by limiting our surveys to periods when temperatures were within the range previously established by Dixon *et al.* (2018) in our study area. The combination of active and visual survey results into a single metric also increases the likelihood that reptiles in our study area are accounted for, either while basking or while seeking refuge under rocks or logs.

We also conducted cost-effectiveness analysis of each treatment at three levels of cost: (1) labour costs; (2) material costs; and (3) total costs (labour + materials). To do this, we used the following modified version of the cost-effectiveness formula found in (Carwardine *et al.*, 2012).

$$Cost\ effectiveness = \frac{Abundance_{treatment} - Abundance_{control}}{Cost}$$

We defined abundance as the abundance at each treatment or control predicted by the best model of combined reptile abundance. We recorded the costs for materials at the time of purchase (2021) and in Australian Dollars. We used $45 AUD/hour as the cost of labour, which was the rate for a casual research assistant at the time fieldwork was conducted (2021-2022). We calculated the number of hours required for each treatment by recording and averaging the total time it took to construct each treatment during our experiment.

3.4 Results

In total, 140 reptiles comprising seven identified species were recorded during the surveys (Table 3-2). Maximum species richness recorded at a site was three species, maximum abundance recorded at a site was eight individuals. The majority of sites had a species richness of one, therefore we did not have sufficient data to fit models to predict species richness.

Table 3-2: The total number of individuals of each reptile species recorded from visual and active searches in each survey season. We used the category 'unidentified skink' to record individuals that were seen during surveys but were unable to be identified to species level.

Species	Abundance season 1	Abundance season 2
Lampropholis spp.	31	35
Acritoscincus duperreyi	10	7
Acritoscincus platynotum	12	3
Pseudemoia entrecasteauxii	13	11
Pseudemoia spenceri	2	0
Rankinia diemensis	0	1
Unidentified skink species	5	10
Total	73	67

3.4.1 The effect of refuge treatments on combined reptile abundance

The best-fitting model for predicting combined reptile abundance included treatment as the only fixed effect (Supplementary material, Table S3-1). While holding other variables constant, reptile abundance at the structure and structure + predation treatments was significantly greater than the control (Figure 3-3, Table 3-3). Analysis of estimated marginal means found that predicted abundance at both structure and structure + predation treatments were significantly greater than abundance at predation treatments ($p=0.001$ and $p<0.0001$, respectively). There was no significant difference between structure and structure + predation treatments ($p=0.53$). Fire severity, habitat structure (i.e., ground cover, leaf litter, etc.) and the presence of rocky outcrops and natural CWD were not included in the best model.

Table 3-3: The best model (lowest AIC) predicting combined reptile abundance contained Treatment as the only fixed effect. Coefficients, confidence intervals, and significance (p-values) for each component of the best model are listed in the table.

Explanatory variable	Estimate	95% C.I.	p-value

Intercept	-1.66	(-2.58, -0.74)	<0.001
$Treatment_{predation}$	-0.15	(-0.91, 0.61)	0.700
$Treatment_{structure}$	1.11	(0.49, 1.72)	<0.001
$Treatment_{structure + predation}$	1.40	(0.81, 1.99)	<0.001

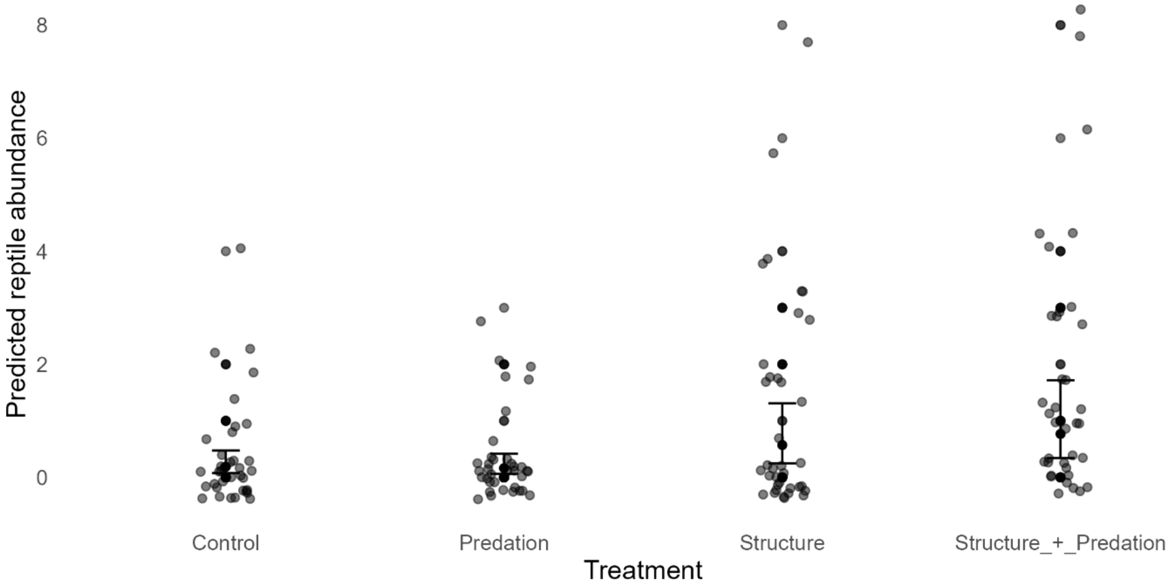


Figure 3-3: Predicted combined reptile abundance (mean ± 95% confidence intervals) for each treatment across our study. All predictions were made using the best-fit model while holding all other fixed effects in the model either at their mean (for continuous variables) or mode (categorical variables).

3.4.2 The effect of refuge treatments on *Lampropholis* spp.

Thirty-one individuals of *Lampropholis* spp. were present at 14 sites (17.5%) during the first season and 35 individuals were present at 18 sites (22.5%) during the second season. The best-fitting model for predicting the presence or absence of *Lampropholis* spp. included survey season, canopy cover, shrub cover, groundcover, leaf litter cover, and treatment as fixed effects (Supplementary material, Table S3-2). While holding other variables constant, the probability of *Lampropholis* spp. being present was significantly greater at structure and structure + predation treatments compared to the control (Figure 3-4, Table 3-4). The probability of occurrence of this taxon was greatest at the structure + predation treatment, although analysis of estimated marginal means indicated that the difference between structure + predation and structure was not significant (p=0.85). The difference in probability of occurrence between structure + predation and predation was not significant (p=0.08). Survey season and shrub cover also had significant effects on the probability of *Lampropholis* spp. presence. Predicted probability increased with increasing shrub cover and time (i.e., probability was lower for season 1 than season 2).

Table 3-4: The best model (lowest AIC) predicting the presence of *Lampropholis* spp. contained treatment, survey season, canopy cover, shrub cover, ground cover and leaf litter. Coefficients, confidence intervals, and significance (p-values) for each component of the best model are listed in the table.

Explanatory variable	Estimate	95% C.I.	p-value
Intercept	-7.45	(-13.28, -1.63)	0.012
Survey season	1.68	(0.03, 3.31)	0.045
Canopy cover	-0.79	(-1.89, 0.30)	0.158
Shrub cover	1.24	(0.15, 2.35)	0.025
Ground cover	-0.83	(-1.89, 0.23)	0.123
Leaflitter	-0.42	(-1.55, 0.72)	0.472
$Treatment_{predation}$	1.04	(-1.21, 3.29)	0.365
$Treatment_{structure}$	2.92	(0.52, 5.33)	0.017
$Treatment_{structure+\ predation}$	3.72	(1.06, 6.37)	0.006

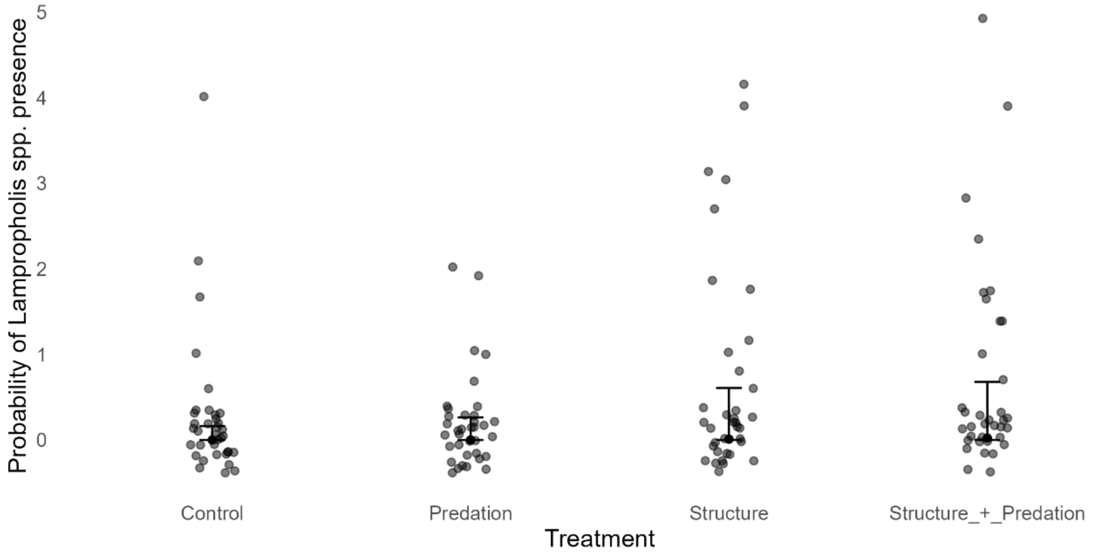


Figure 3-4: Predicted probability of *Lampropholis* spp. presence (mean ± 95% confidence intervals) for each treatment across our study. All predictions were made using the best-fit model while holding all other fixed effects in the model either at their mean (for continuous variables) or mode (categorical variables).

3.4.3 Treatment cost-effectiveness analysis

We conducted a cost-effectiveness analysis for the three treatments examined in our study (predation, structure, and structure + predation). Our analysis included: (1) cost-effectiveness in terms of labour hours; (2) cost-effectiveness in terms of material costs; and (3) cost-effectiveness in terms of total costs (labour + materials). The cost of materials for each treatment was, on average, $38 AUD for predation treatment and $53 AUD for structure + predation treatment. The structure treatment did not require any chicken wire or materials and therefore had $0 materials cost. Labour costs, in total people hours, for each treatment was 0.67 hours for predation treatment, 1.33 hours for structure treatment, and 2 hours for structure + predation treatment. When total cost was calculated (materials cost + labour costs at $45 AUD/hour) the cost for each treatment was $68 for predation treatment, $59 for structure treatment, and $143 for structure + predation treatment. Cost-effectiveness was similar for structure and structure + predation treatments when labour costs were the only costs considered (0.85 and 0.70 reptiles/$AUD respectively, Table 3-5). When materials costs and total costs were considered, the structure treatment had a greater cost-effectiveness than structure + predation (Table 3-5). The predation treatment had the lowest cost-effectiveness across all metrics

Table 3-5: A comparison of the cost-effectiveness of each treatment across three cost categories: (1) materials only; (2) labour only; and (3) labour and materials. All calculations were done using Australian dollars and costs incurred in 2021.

Treatment	Materials-only cost effectiveness (reptiles/$AUD)	Labour-only cost effectiveness (reptiles/hour)	Labour + materials cost effectiveness (reptiles/$AUD)
Predation	0 ± 0.01	0 ± 0.58	0 ± 0.005
Structure	1.1* ± 0.31	0.85 ± 0.24	0.019 ± 0.005
Structure+ Predation	0.03 ± 0.01	0.70 ± 0.15	0.009 ± 0.002

3.5 Discussion

We examined the response of reptiles in burnt areas to three treatments: (1) structure only (coarse woody debris piles); (2) predation only (strips of chicken wire); and (3) structure + predation (coarse woody debris piles covered with chicken wire). We found that the structure and structure + predation treatments significantly increased overall reptile abundance and the probability of occurrence for *Lampropholis* spp.. We found the addition of chicken wire did not lead to an additional increase in reptile abundance. The structure only treatment (coarse woody debris piles) was the most cost-effective treatment tested. There were insufficient data to draw conclusions about the effects of the treatments on reptile richness.

3.5.1 The effect of refuge treatments on combined reptile abundance

We found that total reptile abundance was significantly greater at structure and structure + predation treatments when compared to the control. Our results fit within a wider body of work that has found coarse woody debris supplementation to be beneficial for reptile populations in other disturbed landscapes. Manning *et al.* (2013) and Evans *et al.* (2019b) found that the addition of CWD to degraded agricultural landscapes increased reptile richness and abundance over both the short term (4 year) and the medium term (9+ year). However, their studies involved adding 20 to 40 tonnes of CWD per hectare. Adding large amounts of CWD to remote areas may require specialist equipment (i.e., helicopters or cranes). More cost-effective options are likely to be more attractive, for example, where access is limited or where volunteers or citizen scientists are involved, which can be a valuable resource after wildfires (Kirchhoff *et al.*, 2021; Lee *et al.*, 2023; Rowley *et al.*, 2020). Christie *et al.* (2013) found that creating a larger number of piles from smaller woody debris was more effective at promoting reptile colonisation than a low density of large logs and CWD. Their results support our findings that creating complex structures from smaller woody debris can mitigate the loss of larger CWD after disturbances such as fire.

We also found that predicted abundance was not significantly different between structure and structure + predation treatments and predicted abundance for both treatments were significantly greater than for the predation treatment. This indicates that the CWD, rather than the chicken wire, is providing the greatest benefit to reptile populations in our study. These benefits may be attributed to invertebrate food sources which are attracted to decomposing CWD (Boggs *et al.*, 2020; Vanderwel *et al.*, 2006), or the thermoregulatory benefits refugia such as CWD can provide (Hossack *et al.*, 2009; Klingenböck *et al.*, 2000; Seebacher and Alford, 2002; Webb and Shine, 1998). Although we did not quantify invertebrate abundance, a diverse array of high abundance of invertebrates were observed during active surveys of the CWD piles and may have provided a viable food source (H. Burns, pers. obs.).

Despite the absence of a measurable benefit from the addition of chicken wire, the CWD piles may have still provided protection from some predation. While our conclusions are limited by an inability to quantify predation pressure in our study area, the literature suggests reptiles may be subject to increased predation after fire. This is due to the increased mobility of predators (Geary *et al.*, 2020) and increased abundances of predators such as foxes in burnt areas (Nimmo *et al.*, 2019). Given these background conditions in our study, there is a possibility that the presence of our CWD piles provided sufficient protection from predators that contributed to the increased reptile abundance at those sites. In addition to these points, Arthur *et al.* (2005) provide an example of CWD piles providing clear predation benefits to small terrestrial vertebrates. In an enclosure accessible by foxes and cats, they found that CWD covered in chicken wire (similar to our structure + predation treatment) supported house mouse populations that were not significantly different than a complete predator exclusion treatment. While our study found benefits from both CWD piles and CWD piles + chicken wire, the lack of a significant difference between the two may be because CWD piles alone provide sufficient protection from predators of small reptiles.

Although fire severity was not a significant part of our combined reptile abundance model, we found strong associations between a range of vegetation characteristics and fire severity in our study area. Specifically, areas burnt at high severity had low canopy cover, high shrub cover, and low leaflitter while areas burnt at low severity had high canopy cover, low shrub cover, and high leaf litter (Fig. S3-2). We also observed that in areas burnt at high severity, the mid-storey layer became extremely dense through time. As a result, these regenerating stands often lacked complexity beneath the shrub layer, which may have been supplemented by CWD piles in our study. Data from forests across south-eastern Australia show that CWD consumption increases with fire severity (Hollis *et al.*, 2011), and pulses when branches and stems fall in the post-fire period (Bassett *et al.*, 2015a). Given the fluctuating availability of naturally occurring CWD in severely burnt forests, we conclude that these treatments would be valuable in highly disturbed post-fire landscapes.

In addition to the importance of habitat characteristics, the thermal properties of artificial refuges and structures are critical to their successful and ethical implementation (Cowan *et al.*, 2021; Watchorn *et al.*, 2023). Artificial refuges that do not mimic the insulative properties of natural refugia can result in extreme temperatures well beyond the thermal range of a species preferred microhabitat (e.g., Griffiths *et al.* (2017)). A subsequent study has been conducted to examine the thermal properties of the structure and structure + predation treatments. It determined coarse woody debris treatments provided a significant thermal buffer against ambient temperatures across summer and winter seasons (Chapter 5).

3.5.2 The effect of refuge treatments on *Lampropholis* spp.

We found that probability of *Lampropholis* spp. presence was significantly greater at the structure and structure + predation treatments and increased with shrub cover and time. The preference for CWD is consistent with the work of Evans *et al.* (2019b) discussed above, who found *Lampropholis delicata* was one of the species that benefitted from additional CWD. *Lampropholis* spp. abundance at structure and structure + predation treatments may be due to the availability of food resources in decaying CWD. Lunney *et al.* (1989) found that the diets of multiple *Lampropholis* spp. were dominated by invertebrates, which appeared to be abundant in CWD piles in our study (H. Burns personal obs.; Boggs *et al.* (2020)). Increasing probability of presence over time is likely explained by natural population recovery after disturbance. Multiple studies have also found that *Lampropholis* spp. populations increase in the years after fire (Evans *et al.*, 2019b; Foster *et al.*, 2016; Taylor and Fox, 2001).

3.5.3 Treatment cost-effectiveness analysis

We found that in most circumstances the structure treatment was the most effective for the associated costs of labour and/or materials, with cost-effectiveness values ranging from 0.019 to 1.1 reptiles/$AUD. The effectiveness of the structure and structure + predation treatments were similar only when labour was the only factor considered (0.85 and 0.70 reptiles/$AUD respectively). However, we caution that structures with chicken wire may have associated costs that were not included in this analysis such as maintenance during the use of the treatments and disassembly of structures when the project is completed. The use of chicken wire may also necessitate additional animal ethics considerations and be unsuitable for long-term use in reserve areas or national parks. Therefore, the structure treatment is the most feasible and cost-effective option.

Our analysis relies on the assumption that there is enough CWD remaining in the landscape after fire to support the construction of woody debris piles. In areas with insufficient remnant CWD there may be additional procurement and transportation costs associated with the construction of these CWD piles, and the cost-effectiveness of each treatment may need to be reassessed.

3.6 Conclusion

Our results are consistent with previous studies that found supplemental CWD enhanced reptile populations in disturbed landscapes (Freeman *et al.*, 2021; Manning *et al.*, 2013; Christie *et al.*, 2013). We found that predicted presence of *Lampropholis* spp. and combined reptile abundance were significantly greater for structure and structure + predation treatments, compared to control sites in burnt areas. Whether the treatments tested in this experiment mitigate the threats of predation directly (i.e., providing a physical barrier to predation) or indirectly (providing food sources and quality habitat for native reptiles), they successfully increased predicted reptile abundance in a disturbed landscape. Given the low materials cost and relative simplicity of construction we recommend replication of the structure treatment (i.e., CWD piles) for use in future post-fire landscapes, including sites where volunteers are seeking to engage in activities to help native species recovery.

3.7 Supplementary material

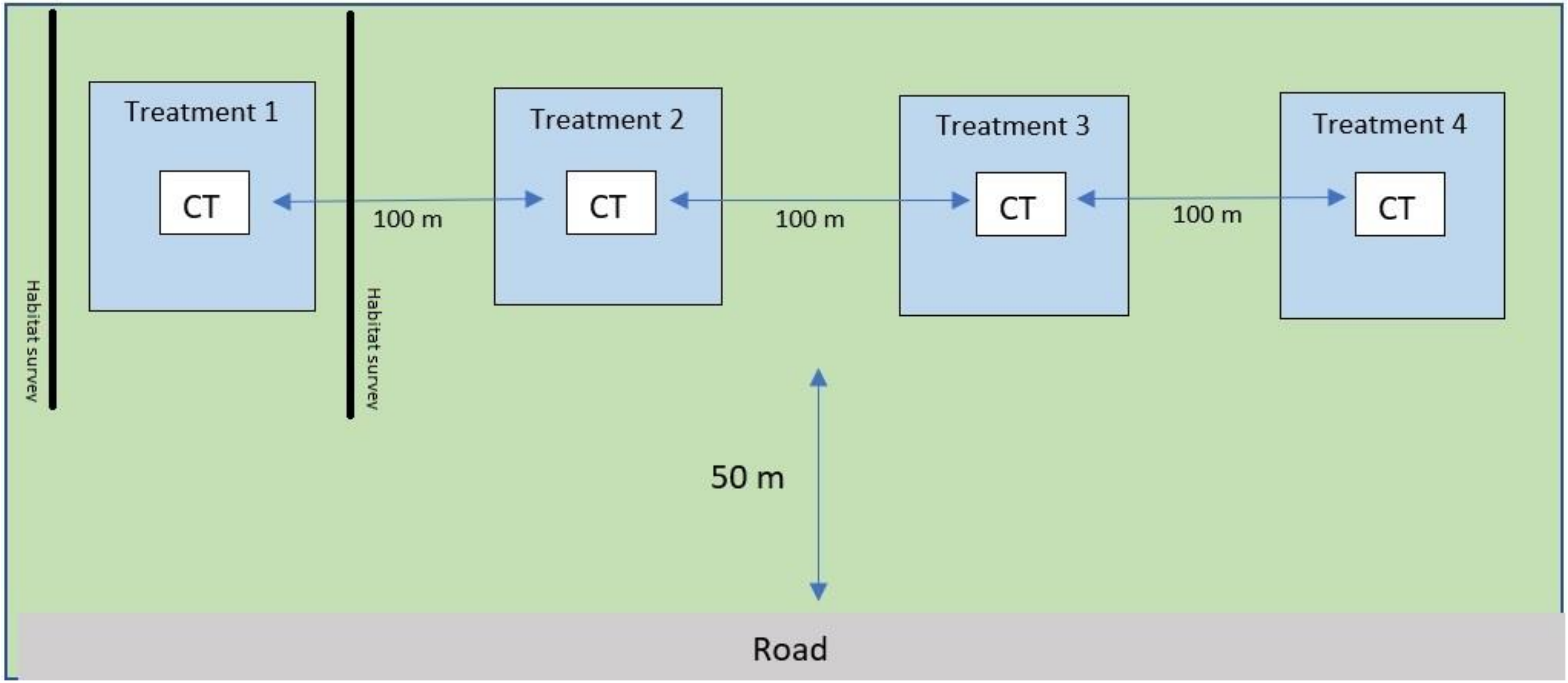


Figure S3-1: A diagram of the block design. For treatments were arranged parallel to the road, offset by 50 m and spaced 100 m apart. CT indicates where the four replicates of each treatment were arranged in a square, with a wildlife camera in the centre (this was part of a separate study). Habitat surveys were conducted along 25 metre transects that ran parallel to each treatment area but perpendicular to the road.

Table S3-1: Candidate linear mixed models tested to predict reptile abundance, ranked by delta AIC. Model components are indicated by an asterisk * in the corresponding column.

Model	Treatment	Survey season	Total CWD	Rocky outcrop P/A	Canopy cover	Canopy cover*treatment	Shrub cover	Shrub cover *treatment	Ground cover	Ground cover *treatment	Leaf litter	Leaf litter *treatment	Fire severity	Fire severity *Treatment	AICc	Delta AICc
1	*														328.41	-
4	*	*			*		*		*		*				331.51	3.10
5	*	*											*		332.78	4.37
6	*	*	*	*											334.44	6.03
2	*	*	*	*	*		*		*		*		*		335.04	6.63
3	*	*	*	*									*		336.73	8.32
8	*	*											*	*	339.93	11.52
0 (null)															347.43	19.02
7	*				*	*	*	*	*	*	*	*			348.22	19.81
9		*	*	*	*		*		*		*				357.92	29.51

Table S3-2: Candidate linear mixed models tested to predict *Lampropholis* spp. presence, ranked by delta AIC. Model components are indicated by an asterisk * in the corresponding column.

Model	Treatment	Survey season	Total CWD	Rocky outcrop P/A	Canopy cover	Canopy cover*treatment	Shrub cover	Shrub cover *treatment	Ground cover	Ground cover *treatment	Leaf litter	Leaf litter *treatment	Fire severity	Fire severity *Treatment	AICc	Delta AICc
5	*	*			*		*		*		*				120.97	-
3	*	*	*	*	*		*		*		*				121.99	1.02
2	*	*													124.65	3.68
6	*	*											*		126.79	5.82
1 (null)															126.89	5.92
7	*	*	*	*											128.13	7.16
4	*	*	*	*									*		130.37	9.40
9	*	*											*	*	133.73	126
10		*	*	*	*		*		*		*				134.09	132
8	*				*	*	*	*	*	*	*	*			137.36	16.39

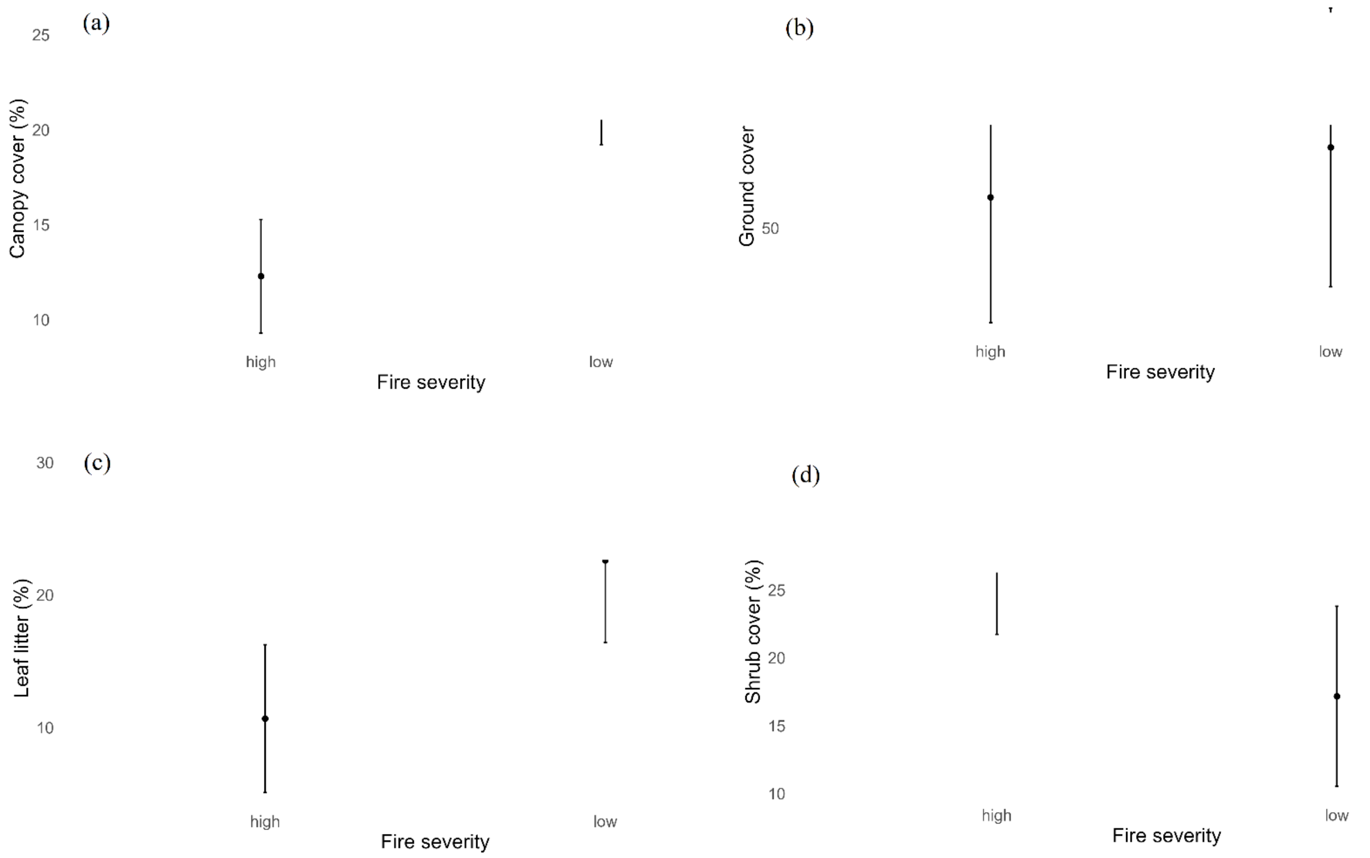


Figure S3-2: Predicted cover for (a) canopy cover, (b) ground cover, (c) leaf litter, and (d) shrub cover at high and low fire severity.

Chapter 4: Vegetation complexity shapes small mammal community responses to fire in a sub-alpine region of south-eastern Australia

Statement of Contribution

This thesis is submitted as a Thesis by Compilation in accordance with

https://policies.anu.edu.au/ppl/document/ANUP_003405

I declare that the research presented in this Thesis represents original work that I carried out during my candidature at the Australian National University, except for contributions to multi-author papers incorporated in the Thesis where my contributions are specified in this Statement of Contribution.

Title: Vegetation complexity shapes small mammal community responses to fire in a sub-alpine region of south-eastern Australia

Authors: Burns, Heather; Cary, Geoffrey; Brawata, Renee; Lindenmayer, David; Connolly O'Donnell, Liam; Rathod, Divyang; Gibbons, Philip

Current status of paper: Not Yet Submitted/Submitted/Under Revision/Accepted/Published

Contribution to paper: HB designed the experiment with feedback from LCO, DR, PG, GC, RB and DL. LCO and DR established the experimental blocks, and HB managed wildlife cameras and conducted habitat surveys with help from LCO and DR. HB analysed the data and wrote the manuscript. PG, GC, RB and DL provided feedback on the manuscript in preparation for submission to a journal.

Senior author or collaborating authors endorsement: Philip Gibbons _

Heather Burns ________________________ ________________________________ 08/07/2024

Candidate – Print Name Signature Date

Endorsed

Philip Gibbons ________________________ 15/07/2024

Chair of Supervisory Panel – Print Name Signature Date

Robert Dyball ________________________ 17/07/2024

Delegated Authority – Print Name Signature Date

Foreword

Chapter 2 found that there is a considerable lack of examples of post-fire interventions focused on conserving terrestrial vertebrates such as small mammals. Coarse woody debris was a common intervention for small mammals, particularly in production forestry landscapes, and had been demonstrated to provide benefits to house mice experimentally exposed to predation pressure (Arthur *et al.*, 2005).

Chapter 4 aims to determine the relative value of naturally occurring features of a post fire-landscape (i.e., vegetation complexity, naturally occurring CWD, and rocky outcrops) as well as supplementary refuge in the form of CWD piles, on small mammal abundance and richness. This chapter uses the experimental treatments outlined in Chapter 3 as the basis of its data collection. Wildlife cameras, baited with tuna and sesame oil, were deployed at each site to monitor small mammal activity across fifteen months.

This chapter is in preparation for submission to a journal.

4.1 Abstract

Wildfire frequency and extent are expected to increase with a changing climate. As megafires become more common, it is important to understand how small mammal populations respond to these events and identify the factors that influence population recovery. We aimed to answer the following research questions: (1) What is the relative value of vegetation complexity, coarse woody debris (CWD) and rocky outcrops for small mammals after fire; (2) Do these variables influence small mammal richness or abundance in a post-fire landscape; and (3) Does adding supplementary structural habitat assist the recovery of small mammals after fire. To answer these questions, we implemented 20 experimental blocks across areas burnt in the 2020 Orroral Valley Fire in the Australian Capital Territory. At each block, we established three supplementary refuge treatments: (1) CWD piles; (2) strips of chicken wire; and (3) CWD piles covered in chicken wire, in addition to a control. We monitored small mammal activity across each site using wildlife cameras centred over a baited cork tile. We found that increasing shrub cover was associated with significant increases in the abundance and occurrence of most species, while species richness also increased significantly in areas burnt at high severity and areas with high levels of natural CWD. Supplementary refuge treatments did not significantly increase the abundance or occurrence of any small mammal species. Our results indicate that access to structural refugia was not a limiting factor for small mammals, and abundance and occurrence was driven primarily by vegetation characteristics, particularly shrub cover.

4.2 Introduction

As climate change drives shifts in fire regimes (Canadell *et al.*, 2021), it is critical to understand species responses and recovery mechanisms in post-fire environments (Driscoll *et al.*, 2010). Small mammals, in particular, have been shown to survive large-scale fires in-situ (Banks *et al.*, 2011), but a range of factors including vegetation complexity, the availability of structural elements and refuge, predation pressure, and climatic conditions affect the trajectory of population recovery (Arthur *et al.*, 2012; Letnic *et al.*, 2005; Doherty *et al.*, 2022). Fire severity can also affect habitat availability by changing understory vegetation structure (Barker *et al.*, 2022).

Vegetation complexity has been linked to post-fire population dynamics for a range of mammal assemblages (Arthur *et al.*, 2012; Di Stefano *et al.*, 2011; Dixon *et al.*, 2019; Fordyce *et al.*, 2016). Fox's habitat accommodation model suggests that changes in small mammal populations are not linear after fire, and instead are driven by changes in vegetation characteristics (Fox, 1982). Specifically, species abundances increase when their preferred conditions develop (e.g., dense shrub layer, open understory, etc.), regardless of how long ago the fire occurred. Vegetation regeneration also interacts with climatic events, such as high rainfall or drought, to affect small mammal populations (Hale *et al.*, 2016). Sufficient rainfall can cause small mammal populations to boom, while drought conditions can restrict small mammals to a narrow niche (Hale *et al.*, 2016).

Small mammals also rely on structural refugia, such as rock crevices, hollow logs and coarse woody debris (CWD), for shelter and nesting opportunities in burnt landscapes (Hale *et al.*, 2022; Matthews *et al.*, 2017). While rocky outcrops are usually not significantly affected by fire, the availability of hollows in logs, snags, and trees may change after fire. In landscapes with hollow-bearing logs and snags, fire can enlarge existing hollows (Collins *et al.*, 2012). However, fire frequency may affect the availability of hollows in mature trees and snags (Gibbons *et al.*, 2024; Lindenmayer *et al.*, 2012). The availability of suitable small mammal habitat can also vary with topographic features and fire severity, with Bassett *et al.* (2015a) finding a reduced levels of CWD in gullies after high severity fire.

Reduced habitat complexity after fire, from loss of vegetation and woody debris, can create open habitats that favour introduced predators such as feral cats (*Felis catus*) (McGregor *et al.*, 2015). This can exert additional pressure on small and medium-sized mammals, frequently referred to as the 'critical weight range' (Johnson and Isaac, 2009; Kutt, 2012; Moseby *et al.*, 2009; Murphy and Davies, 2014). In an Australian example, analysis of fox scats showed that the proportion of their diet formed by native mammals doubled after prescribed fire (Hradsky *et al.*, 2017a). Given the observed increase in fire frequency in some regions of the world (Bradstock, 2010; Jones et al., 2020; Canadell et al., 2021), including in south-eastern Australia (Lindenmayer et al., 2023) , there is the potential for an interaction between more frequent fires and introduced predators to exert greater pressure on small mammal populations.

Given the complex post-fire dynamics affecting small mammal populations after fire, we aimed to examine the relative effects of vegetation complexity, structural features (i.e., CWD and rocky outcrops) and supplementary refugia on the richness and abundance of small mammals in a post-fire landscape. Specifically, we sought to answer the following research questions: (1) What is the relative value of vegetation complexity, CWD and rocky outcrops for small mammals after fire; (2) Do these variables influence small mammal richness or abundance in a post-fire landscape; and (3) Does adding supplementary structural habitat assist the recovery of small mammals after fire. We hypothesise that vegetation complexity will strongly influence the presence or abundance of small mammals in our study area (Fox, 1982). In addition, we hypothesise that the addition of supplemental habitat will increase small mammal abundance given the positive effects found in other disturbed landscapes (Sullivan and Sullivan, 2021; Sullivan *et al.*, 2012).

4.3 Methods

Our study area was located in 1061 km² Namadgi National Park, Australian Capital Territory in south-eastern Australia (35.558147 S, 148.940715 E; Land Management and Planning Division (2010)). Namadgi National Park contains a mix of alpine and sub-alpine habitat, dry sclerophyll forest, and wet sclerophyll forest (Keith and Simpson, 2012). Elevation in the park ranges from 900 to 1800 metres above sea level, with our study sites situated between 900-1200 metres. Mean annual rainfall in the park is 1091 mm, winter (July) temperature ranges from -2.4°C to 2.6°C and summer (January) temperature ranges from 10.1°C to 20.7°C (Bureau of Meteorology, 2024). The study area supports a range of small mammal species, including several species of antechinus (*Antechinus* spp.) and native rodents (*Rattus* spp.) (Land Management and Planning Division, 2010). The national park has experienced multiple wildfires in the past two decades, most recently in January and February of 2020 where an estimated 80% of the park was burnt.

In September 2021, we established a randomised block design experiment consisting of 20 blocks that were divided equally between two levels of fire severity: (1) blocks burnt at high severity in the 2020 fires; and (2) blocks burnt at low severity in the 2020 fires. Fire severity was mapped and classified using RdNBR values (Gale *et al.*, 2021) in ArcGIS (Environmental Systems Research Institute, 2022). Once these classes were mapped and tentative study sites were selected, we ground-truthed each location before establishing experimental blocks. For a site to be classified as high severity, the canopy must have been >75% consumed by fire. For a site to be classified as low severity, the canopy must have experienced <25% crown scorch. There were no minimum patch sizes for each burnt area, but each block was located completely within its designated severity class with a buffer of at least 100 metres.

Unburnt sites were not included in this study due to the only unburnt forest being a small patch of one vegetation type not widespread throughout the park. All blocks were located within the dominant dry sclerophyll forest vegetation type in the study area.

At each block, we measured a range of variables that potentially provide refugia for small mammals in post-fire landscapes: litter, ground cover vegetation, shrub cover, CWD and rocky outcrops. In addition, we established four supplementary refuge treatments: (1) control (no supplementary refuge); (2) structure only; (3) predation only; and (4) structure + predation. Structure only treatments were constructed using small pieces of CWD (predominantly 5-10 cm. diameter) collected from the surrounding landscape (i.e., charred branches from canopy scorched trees). This class of CWD is smaller than what is considered suitable habitat for small mammals, but our aim was to create a complex structure using materials available in a post-fire landscape. The CWD was stacked into four piles approximately 3 metres long and 1 metre wide, which were arranged in a square (see Figure 1). Predation only treatments were built using 3 metre x 1 metre sections of chicken wire, secured with steel pegs, arranged in the same formation as the structure-only treatment. The chicken wire had a 5 cm aperture that allowed small mammals to move freely but would limit access by the most common predators of small mammals in the study area, namely the European red fox, feral cat, and dingo (*Canis familiaris dingo*). Structure + predation treatments were constructed using four CWD piles (similar to the structure only treatment) that were covered with chicken wire (see Arthur *et al.* (2005) for similar design). The chicken wire had a 5 cm aperture, and was approximately 4 m long and 1.5 m wide to cover the entire CWD pile. Once in place, the chicken wire was secured with steel pegs around the perimeter of the CWD pile. Control sites were selected in the same way as all other treatments but did not contain any CWD piles or chicken wire. All treatments were at least 50 metres from the road and 100 metres from another treatment to minimise the risk of small mammals moving frequently between sites. While greater distances between treatments would have further reduced the chances for individuals moving between treatments, we were limited by the patchiness of the fire severity classes and the high topographic variability in our study area. For the purpose of this paper, we refer to a 'site' as a unique treatment x block combination. In total, there were 80 sites in our study (20 blocks x 4 treatments per block).

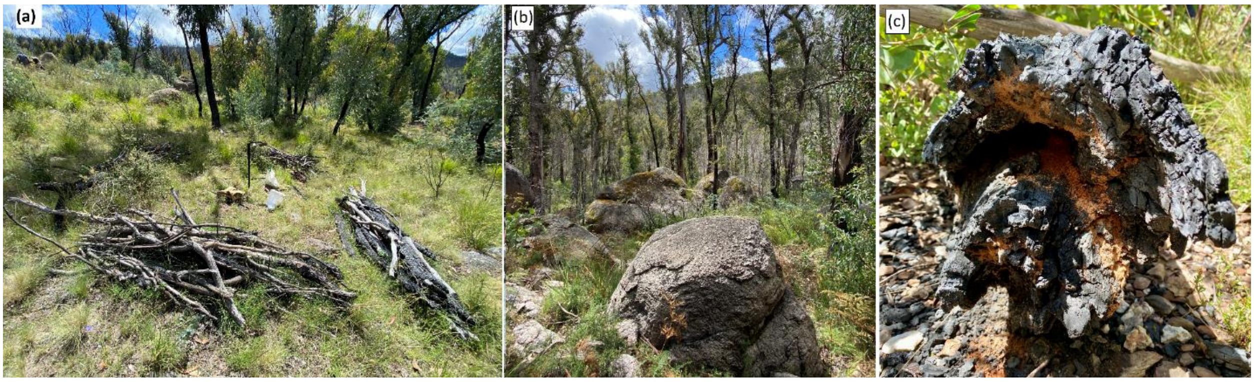


Figure 4-1: From left to right, images of potential small mammal habitat: (a) supplementary refuge treatments; (b) rocky outcrops; and (c) hollow logs.

4.3.1 Data collection

We surveyed all treatments at each block continuously for 15 months across two austral summers, from January 2022 to March 2023. We used 80 Campark T-70 wildlife cameras to detect small mammal activity across our study (one per treatment at each block). We set each camera at a height of 70 cm, facing the ground, and centred over a cork tile baited with both (1) sesame oil and (2) tuna oil and rice bran oil (Dixon *et al.*, 2019). At each treatment, we set cameras at the centre of the four CWD piles and/or segments of chicken wire. At each control, we set cameras at a randomly allocated point using the same methods as the supplementary refuge sites. We visited each camera every two months to replace SD cards and refresh the oil baits.

We set cameras to take one photo per detection, and the resolution was set to the highest setting (12 megapixels). We used the shortest available interval between photos which was five seconds. We set the infrared LED flash was set to high to maximise detail and used the highest shutter speed (1/30) to reduce blur in night-time photos.

At each site we collected data on habitat variables that could affect the richness and abundance of small mammals in burnt landscapes. This included structural features such as the presence/absence of rocky outcrops and total length of natural CWD, as well as vegetation complexity (Table 4-1). Vegetation complexity surveys allowed us to account for the varying effects of time since fire and fire severity on vegetation structure. This was particularly relevant as our study area experienced above average rainfall as the result of La Nina conditions during our study (Bureau of Meteorology, 2023). Rocky outcrops and large coarse woody debris also represent natural alternative sources of habitat for small mammals (Hale *et al.*, 2022; Matthews *et al.*, 2017), and their availability may have affected small mammal abundance at our artificial treatments and across the landscape.

Table 4-1: A description of explanatory variables used in species-specific modelling of *M. musculus, R. rattus, R. fuscipes, A. agilis* and modelling combined small mammal richness and abundance. Descriptions include survey methods and survey frequency.

Explanatory variables	Survey method	Survey frequency
Fire severity	Fire severity was determined using spatial relativised delta normalised burn ratio (RdNBR; Miller and Thode (2007)) data for Namadgi National Park (Gale *et al.*, 2021). Fire severity is a categorical variable with two categories: high fire severity (>75% canopy consumption) and low fire severity (<25% canopy scorch or consumption).	N/A
Rocky outcrops	We recorded the presence/absence of rocky outcrops within 25 metres of each treatment	We conducted this survey once as the presence or absence of rocky outcrops was unlikely to change over time
Natural CWD	We recorded the total length (in metres) of large CWD (> 20 cm. diameter) within 10 metres of each treatment	We conducted this survey once during the second survey season
Habitat complexity	We established two 25-metre transects that ran parallel to our treatments. At five metre intervals along the transect we used percent cover of five variables to assess habitat complexity; (1) canopy cover (above 2 m); (2) shrub cover (0.5-2 m); (3) ground cover (<0.5 m); (4) bare ground; and (5) leaf litter and fine woody debris. Canopy cover was measured using the %Cover app (Public Interest Enterprises, 2021), while all other variables were estimated using a 50cm x 50 cm quadrat.	We conducted habitat complexity surveys annually (summer 2021, summer 2022) due to the rapid regrowth of vegetation in our study area
Survey period	We conducted camera surveys during seven distinct periods. These survey periods are defined in the 'survey frequency' column.	Survey period 1: January 2022 Survey period 2: February and March 2022 Survey period 3: April 2022 Survey period 4: June 2022 Survey period 5: August 2022 Survey period 6: October – December 2022 Survey period 7: January and February 2023
Supplementary refuge treatment	Three supplementary refuge treatments were tested along side a control (no supplementary refuge) at each block. The supplementary treatments were (1) predation only; (2) structure only; (3) structure + predation. More details on each treatment can be found in the methods section of the main text.	N/A

4.3.2 Data analysis

Although our cameras were continuously deployed for fifteen months, we only used a small portion of this period for analysis. We visited each camera every two months and chose to analyse data from the first week of each period to maximise bait potency and small mammal activity. Once photos from the relevant time period were selected, we manually removed blank photos and used TimeLapse software (Greenberg, 2013) to identify species and record metadata from each photo. For each relevant photo in TimeLapse, we collected information on: (1) file name; (2) file path; (3) date and time; (4) survey period; (5) block; (6) treatment; (7) species; and (8) count.

To avoid duplicate counts of the same individual triggering the camera in quick succession, we used distinct time intervals to identify unique events (Meek *et al.*, 2014). Two consecutive photos of the same species were recorded as unique events if there was more than 10 minutes between when the photos were taken. Photos of the same species taken less than 10 minutes apart were considered unique events if they were separated by photos of a different species. Records for all duplicates were removed from the dataset until there was only one record for each unique event. These provide activity counts as a proxy for abundance (hereafter referred to as 'abundance').

Measurements of vegetation complexity from each survey season were associated with the camera survey period that best fit the dates of each survey. This led to camera survey periods 1-5 being associated with vegetation survey 1, and camera survey periods 6 and 7 being associated with vegetation survey 2. Vegetation complexity variables were tested for correlation, and bare ground was removed from the analysis because it was highly correlated ($r > 0.7$) with the groundcover variable. The remaining four variables (ground cover, leaf litter, shrub cover, and canopy cover) were averaged at the site level to create a single value per site each year.

We fit several linear mixed models to test the relative contribution of leaf litter, ground cover, shrub cover, canopy cover, CWD, rocky outcrops and supplementary refuge to small mammal richness and abundance (see Tables 4-1 and 4-2). Block and site were treated as random effects in each model. Small mammal richness and abundance were used as response variables, and CWD, rocky outcrops, supplementary refuge, and four measures of vegetation complexity (canopy cover, leaf litter, shrub cover, ground cover) were included as potential explanatory variables. We fitted the models using the 'glmmTMB' package (Brooks, 2017) in R (R Core Development Team, 2022) and compared model fits using Akaike's Information Criterion (AIC), selecting the simplest model within two delta AIC scores of the top-ranked model for each response variable (Arnold, 2010; Burnham and Anderson, 2002). Relationships between the predictor and response variables were assessed by plotting model predictions and confidence intervals. All predictions were made while holding all other fixed effects in the model either at their mean (for continuous variables) or mode (categorical variables). We calculated effect sizes, with 95% confidence intervals, for each variable in the best model and tested the significance of each effect size in relation to zero. We checked models for normality among residuals, overdispersion, and correlation among fixed effects using the Performance package in R. Collinearity between variables in fitted models was evaluated using the Variance Inflation Factor (VIF) implemented within the Performance package in R (Lüdecke *et al.*, 2021).

Table 4-2: A list of model components for each model used in analysis. Models 1-17 were used to model small mammal abundance, small mammal richness, house mouse abundance, black rat abundance, bush rat presence, and agile antechinus presence. For each component, (+) signifies an additive effect and (*) signifies an interaction between two components. FS = fire severity; CWD = naturally occurring CWD; Vegetation complexity = shrub cover, ground cover, canopy cover and leaf litter.

Model	Random effect	Survey period	FS	Vegetation complexity	CWD	Rocky outcrops	Refuge treatment
1	(1|block) + (1|site)						
2	(1|block) + (1|site)	+	+	+			
3	(1|block) + (1|site)	+	+		+		
4	(1|block) + (1|site)	+	+			+	
5	(1|block) + (1|site)	+	+				+
6	(1|block) + (1|site)	+	+	+	+		
7	(1|block) + (1|site)	+	+		+	+	
8	(1|block) + (1|site)	+	+		+	+	+
9	(1|block) + (1|site)	+	+	+		+	
10	(1|block) + (1|site)	+	+	+			+
11	(1|block) + (1|site)	+	+	+	+	+	+
12	(1|block) + (1|site)	+	+	*	*		
13	(1|block) + (1|site)	+	+	*		*	
14	(1|block) + (1|site)	+	+	*			*
15	(1|block) + (1|site)	+	+		*	*	
16	(1|block) + (1|site)	+	+		*		*
17	(1|block) + (1|site)	+	+			*	*

We used similar methods to run species-specific models to test the relative contribution of leaf litter, ground cover, shrub cover, canopy cover, rocky outcrops and supplementary refuge on four key small mammal species: house mouse (*Mus musculus*), black rat (*Rattus rattus*), bush rat (Rattus fuscipes) and agile antechinus (*Antechinus agilis*). There was insufficient data to model presence or abundance for eastern pygmy possum (*Cercartetus nanus*) and long-nosed bandicoot (*Perameles nasuta*). For species with large datasets (house mice and black rats) we modelled species abundance, for species with smaller datasets (bush rat and agile antechinus) we modelled presence absence data extracted from the original abundance dataset. The abundance of introduced predators (i.e., foxes and feral cats) was not included in any model due to a low number of detections across the study area. Across all models, variables that produce a p-value <0.05 will be presented in the Results section.

4.4 Results

In total we recorded 7,710 unique events across 3,752 trap days. This included 6745 events of six species of small mammal (Figure 4-2, Table 4-3), 9.7 % of which were native species. Maximum species richness recorded at a site in one survey period was six, maximum small mammal abundance in a survey period was 229. Across our study sites mean canopy cover was 16.9% (±0.9), mean shrub cover was 22.0% (± 1.5), mean ground cover was 56.5% (± 2.0), and mean leaf litter was 17.5% (± 1.3).

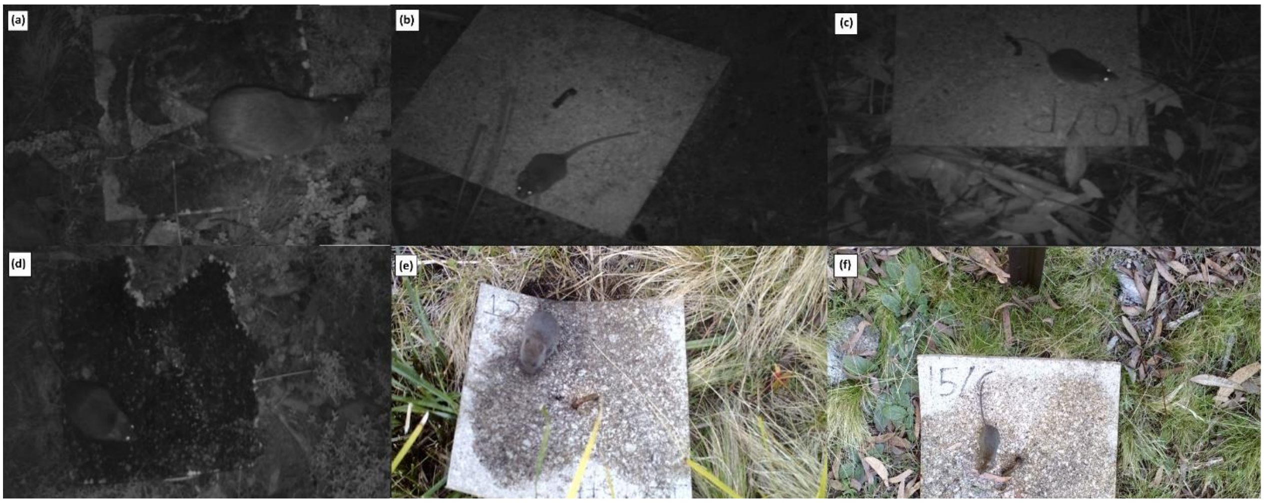


Figure 4-2: Photographs of small mammals from wildlife cameras taken during our study: (a) long nosed bandicoot (*P. nasuta);* (b) eastern pygmy possum (*C. nanus*); (c) agile antechinus (*A. agilis*); (d) black rat (*R. rattus*); (e) juvenile bush rat (*R. fuscipes*); (f) house mouse (*M. musculus*).

Table 4-3: Unique event data for six mammal species (*M. musculus, R. rattus, R. fuscipes, A. agilis, C. nanus, P. nasuta*) recorded in our study, separated by survey period and fire severity.

Species	**Survey period**								**Fire severity**	
	Jan 2022	*Feb-March 2022*	*April 2022*	*June 2022*	*Aug. 2022*	*Oct-Dec. 2022*	*Jan–Feb 2023*	*Total*	*High*	*Low*
House mouse (*Mus musculus*)	123	441	3356	720	49	36	2	4727	3164	1533
Black rat (*Rattus rattus*)	23	78	642	73	407	111	30	1364	855	509
Bush rat (*Rattus fuscipes*)	44	51	251	24	90	8	2	470	355	115
Agile antechinus (*Antechinus agilis*)	42	24	10	0	30	0	13	119	76	43
Eastern pygmy possum (*Cercartetus nanus*)	17	16	6	0	0	1	0	40	32	8
Long nosed bandicoot (*Perameles nasuta*)	7	3	6	2	5	2	0	25	22	3

4.4.1 Factors affecting combined small mammal abundance and species richness

The best-fitting model for predicting combined abundance of native and introduced small mammals included survey period, fire severity, rocky outcrops, canopy cover, shrub cover, ground cover, and leaf litter as fixed effects (Table 4-4, Table S4-1). While holding other variables constant, small mammal abundance increased significantly ($p < 0.001$) with increasing levels of shrub cover (Figure 4-3). Survey period significantly affected small mammal abundance, with survey period 3 (April 2022) recording significantly ($p < 0.001$) greater abundance compared to all other periods (Figure 4-3).

Table 4-4: The best model (lowest AIC) predicting small mammal abundance contained survey period, fire severity, canopy cover, shrub cover, ground cover, leaf litter and rock outcrop presence/absence.

Explanatory variable	Estimate	95% C.I.	p-value
Intercept	1.16	(0.61, 1.71)	<0.001
Survey period 2	0.89	(0.47, 1.32)	<0.001
Survey period 3	3.12	(2.71, 3.54)	<0.001
Survey period 4	0.96	(0.54, 1.38)	<0.001
Survey period 5	0.86	(0.43, 1.28)	<0.001
Survey period 6	-0.78	(-1.30, -0.27)	0.002
Survey period 7	-1.87	(-2.45, -1.30)	<0.001
Fire severity (low)	-0.40	(-1.12, 0.31)	0.26
Canopy cover	-0.11	(-0.34, 0.11)	0.32
Shrub cover	0.36	(0.18, 0.53)	<0.001
Ground cover	0.16	(-0.06, 0.39)	0.15
Leaf litter	0.07	(-0.19, 0.32)	0.61
Rocky outcrop P/A	-0.02	(-0.43, 0.39)	0.92

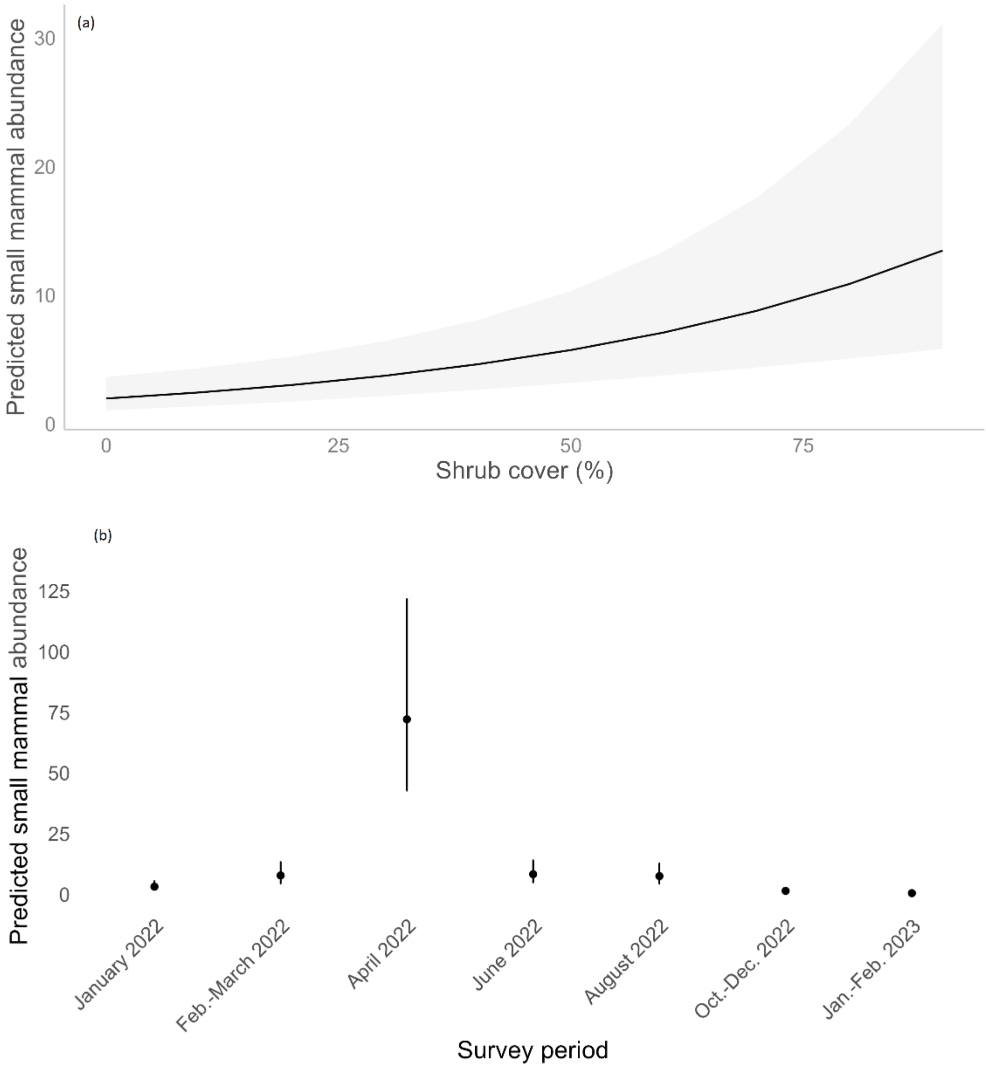


Figure 4-3: The response of small mammal abundance to (a) shrub cover and (b) survey period

The best-fitting model for predicting combined richness of native and introduced small mammals included survey period, fire severity, CWD, canopy cover, shrub cover, ground cover, and leaf litter as fixed effects (Table 4-5, Table S4-1). While holding other variables constant, small mammal richness was significantly greater in areas burnt at high severity compared to areas burnt at low severity (Figure 4-4). Small mammal species richness increased significantly ($p < 0.001$) with increasing levels of shrub cover and natural CWD (Figure 4-4). Species richness varied significantly ($p < 0.001$) with survey period, with the greatest predicted values in periods 2 and 3 (February – April 2022), and the lowest predicted values in periods 6 and 7 (November 2022 – February 2023) (Figure 4-4).

Table 4-5: The best model (lowest AIC) predicting small mammal richness contained survey period, fire severity, canopy cover, shrub cover, ground cover, leaf litter and natural CWD.

Explanatory variable	Estimate	95% C.I.	p-value
Intercept	0.33	(0.126, 0.628)	0.003
Survey period 2	0.22	(-0.04, 0.47)	0.1
Survey period 3	0.53	(0.29, 0.78)	<0.001
Survey period 4	-0.40	(-0.70, -0.09)	0.01
Survey period 5	-0.07	(-0.34, 0.20)	0.62
Survey period 6	-1.03	(-1.43, -0.63)	<0.001
Survey period 7	-1.27	(-1.71, -0.83)	<0.001
Fire severity (low)	-0.31	(-0.61, -0.03)	0.03
Canopy cover	-0.08	(-0.21, 0.05)	0.23
Shrub cover	0.20	(-0.11, 0.29)	<0.001
Ground cover	-0.08	(-0.20, 0.04)	0.21
Leaf litter	-0.03	(-0.18, 0.11)	0.64
CWD	0.14	(0.04, 0.24)	0.004

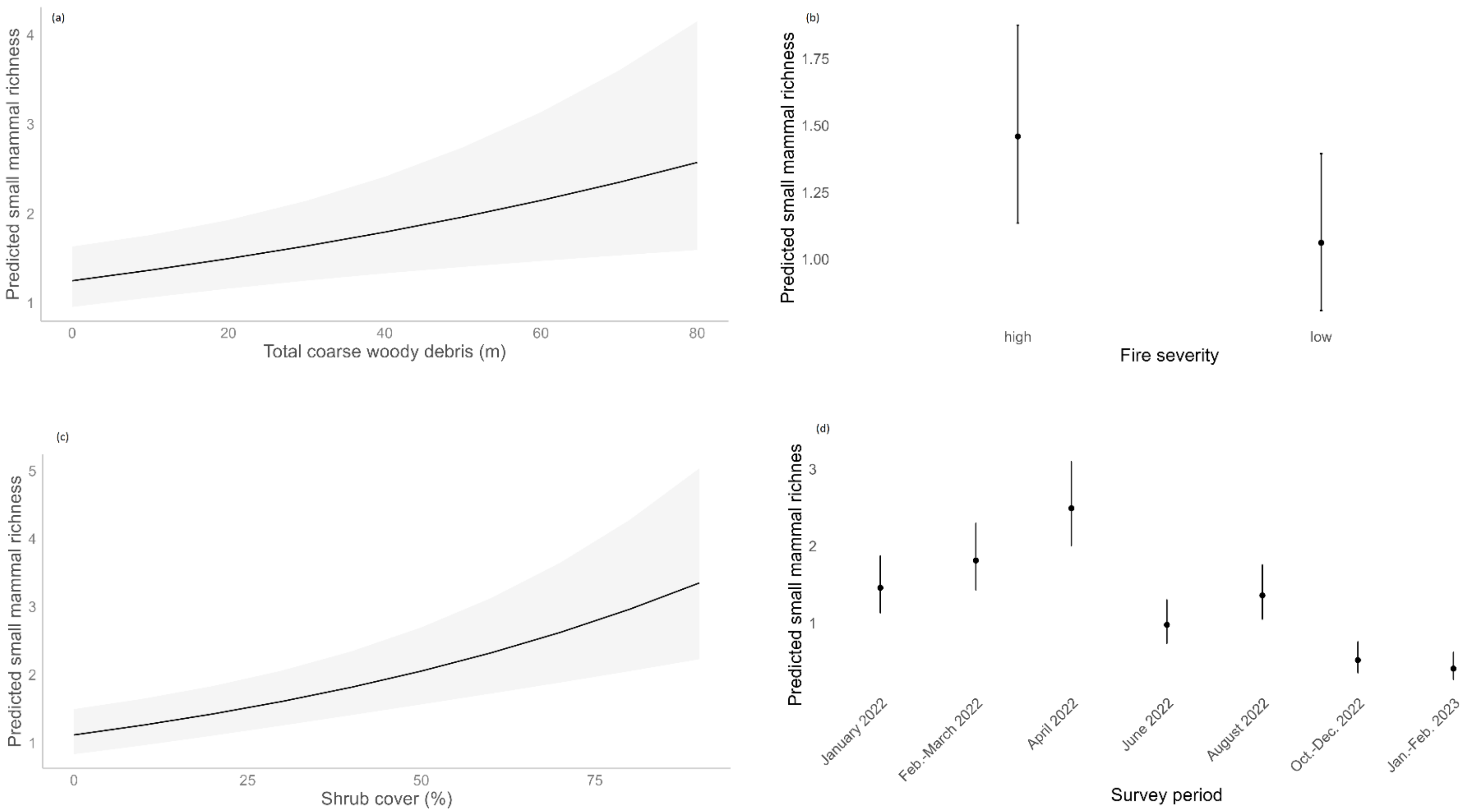


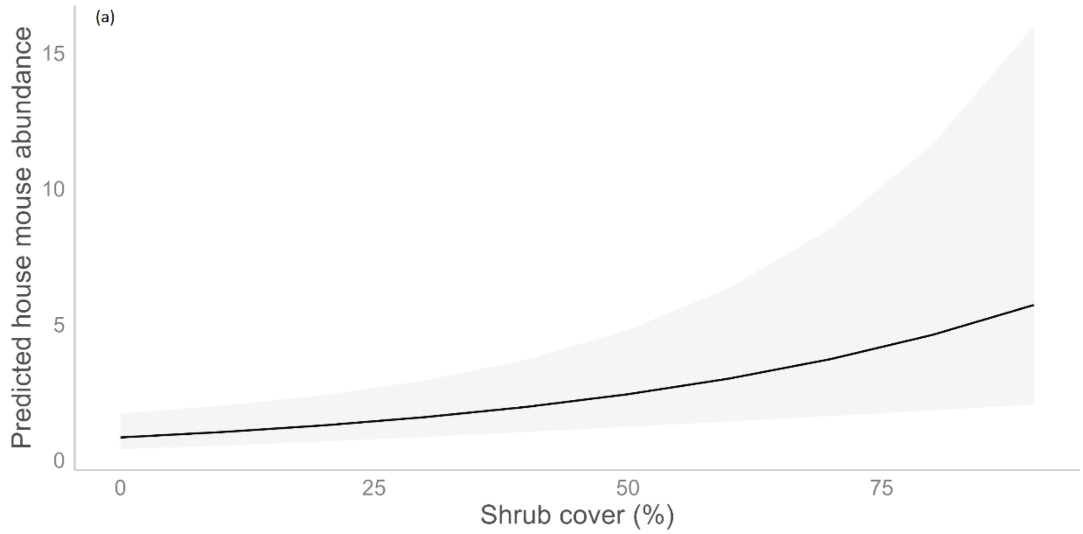
Figure 4-4: The response of small mammal species richness to (a) natural CWD, (b) fire severity, (c) shrub cover, and (d) survey period.

4.4.2 Factors affecting species-specific post-fire landscape use

The best-fitting model for predicting house mouse abundance included survey period, fire severity, canopy cover, shrub cover, ground cover and leaf litter as fixed effects (Table 4-6, Table S1). While holding other variables constant, house mouse abundance increased significantly ($p = 0.001$) with increasing shrub cover (Figure 4-5). Survey period significantly ($p < 0.001$) affected house mouse abundance, with survey period 3 (April 2022) having the greatest predicted abundance (Figure 4-5).

Table 4-6: The best model (lowest AIC) predicting house mouse abundance contained survey period, fire severity, canopy cover, shrub cover, ground cover and leaf litter.

Explanatory variable	Estimate	95% C.I.	p-value
Intercept	0.28	(-0.33, 0.91)	0.37
Survey period 2	1.24	(0.74, 1.74)	<0.001
Survey period 3	3.67	(3.19, 4.16)	<0.001
Survey period 4	1.64	(1.14, 2.14)	<0.001
Survey period 5	-1.61	(-2.28, -0.95)	<0.001
Survey period 6	-2.49	(-3.34, -1.64)	<0.001
Survey period 7	-4.42	(-6.00, -2.84)	<0.001
Fire severity (low)	-0.10	(-0.92, 0.72)	0.809
Canopy cover	-0.18	(-0.50, 0.14)	0.260
Shrub cover	0.36	(0.14, 0.58)	0.001
Ground cover	0.005	(-0.27, 0.28)	0.971
Leaf litter	0.09	(-0.26, 0.45)	0.597



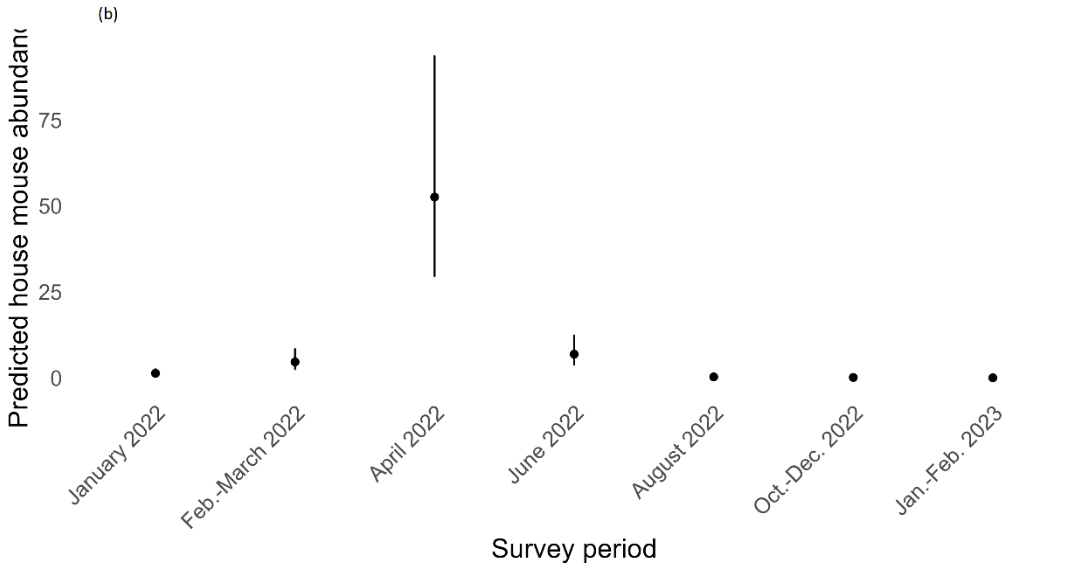


Figure 4-5: The response of house mouse abundance to (a) shrub cover and (b) survey period.

The best-fitting model for predicting black rat abundance included survey period, fire severity, supplementary refuge treatment, canopy cover, shrub cover, ground cover and leaf litter as fixed effects (Table 4-7, Table S4-1). While holding other variables constant, black rat abundance was significantly ($p = 0.004$) lower at the structure + predation treatment compared to the control (Figure 4-6). There were no significant differences between any other treatments. Black rat abundance increased significantly ($p <0.001$) with increasing shrub cover (Figure 4-6). Survey period significantly ($p <0.001$) affected black rat abundance, with periods 3 and 5 (April 2022, August 2022) having the greatest predicted abundance (Figure 4-6).

Table 4-7: The best model (lowest AIC) predicting black rat abundance contained survey period, fire severity, canopy cover, shrub cover, ground cover, leaf litter and treatment.

Explanatory variable	Estimate	95% C.I.	p-value
Intercept	-1.10	(-2.08, -0.11)	0.028
Survey period 2	1.20	(0.40, 2.01)	0.003
Survey period 3	3.73	(2.96, 4.51)	<0.001
Survey period 4	1.02	(0.14, 1.90)	0.02
Survey period 5	3.22	(2.43, 4.01)	<0.001
Survey period 6	1.56	(0.67, 2.45)	<0.001
Survey period 7	0.26	(-0.70, 1.23)	0.59
Fire severity (low)	-0.64	(-1.73, 0.45)	0.25
Canopy cover	0.14	(-0.18, 0.47)	0.39
Shrub cover	0.41	(0.16, 0.65)	<0.001
Ground cover	0.23	(-0.12, 0.58)	0.20
Leaf litter	0.01	(-0.34, 0.38)	0.92
Predation treatment	-0.53	(-1.07, 0.02)	0.06
Structure treatment	-0.30	(-0.87, 0.26)	0.29
Structure + predation treatment	-0.81	(-1.38, -0.24)	0.004

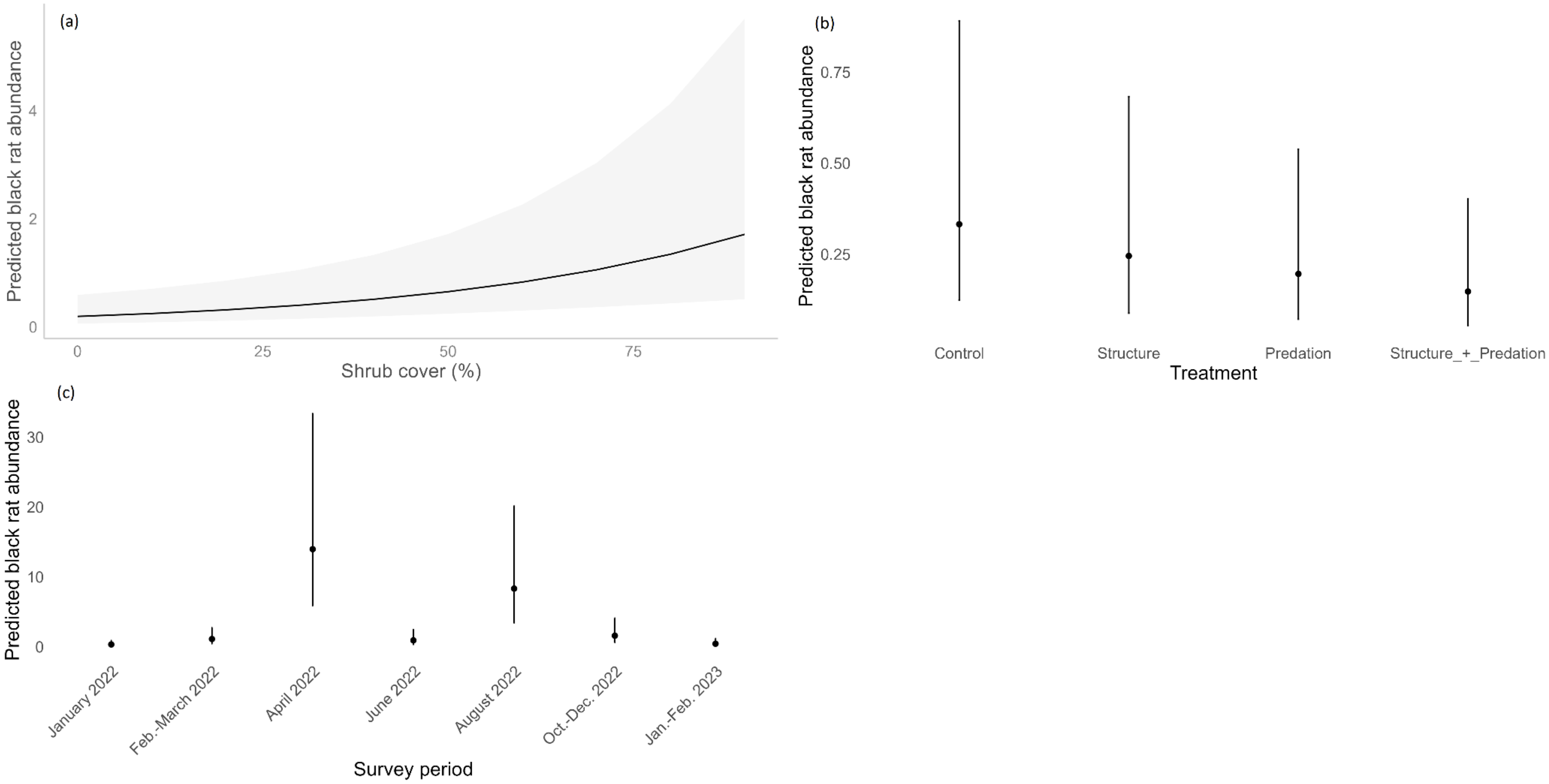


Figure 4-6: The response of black rat abundance to (a) shrub cover, (b) supplementary refuge treatment, and (c) survey period.

The best-fitting model for predicting bush rat presence included survey period, fire severity, canopy cover, shrub cover, ground cover, leaf litter, and supplementary refuge treatment as fixed effects (Table 4-8, Table S4-1). While holding other variables constant, bush rat abundance was significantly lower at the structure ($p < 0.001$) and predation ($p = 0.004$) treatments compared to the control (Figure 4-7). Bush rat abundance decreased significantly ($p = 0.008$) with increasing ground cover (Figure 4-7). Survey period significantly ($p = 0.002$) affected bush rat presence, with the greatest likelihood of bush rat presence occurring in periods 2 and 3 (February – April 2022) (Figure 4-7).

Table 4-8: The best model (lowest AIC) predicting bush rat presence contained survey period, fire severity, canopy cover, shrub cover, ground cover, leaf litter and treatment.

Explanatory variable	Estimate	95% C.I.	p-value
Intercept	-0.18	(-1.26, 0.89)	0.74
Survey period 2	0.32	(-0.59, 1.24)	0.49
Survey period 3	1.09	(0.19, 2.00)	0.02
Survey period 4	-1.89	(-3.11, -0.67)	0.002
Survey period 5	-0.11	(-1.05, 0.82)	0.81
Survey period 6	-2.22	(-3.72, -0.73)	0.003
Survey period 7	-2.71	(-4.42, -0.99)	0.002
Fire severity (low)	-1.49	(-2.67, -0.31)	0.012
Canopy cover	-0.09	(-0.66, 0.46)	0.73
Shrub cover	0.38	(-0.01, 0.76)	0.057
Ground cover	-0.61	(-1.07, -0.15)	0.008
Leaf litter	-0.08	(-0.67, 0.49)	0.76
Predation treatment	-1.44	(-2.44, -0.45)	0.004
Structure treatment	-2.01	(-3.11, -0.92)	<0.001
Structure + predation treatment	-0.79	(-1.74, 0.15)	0.100

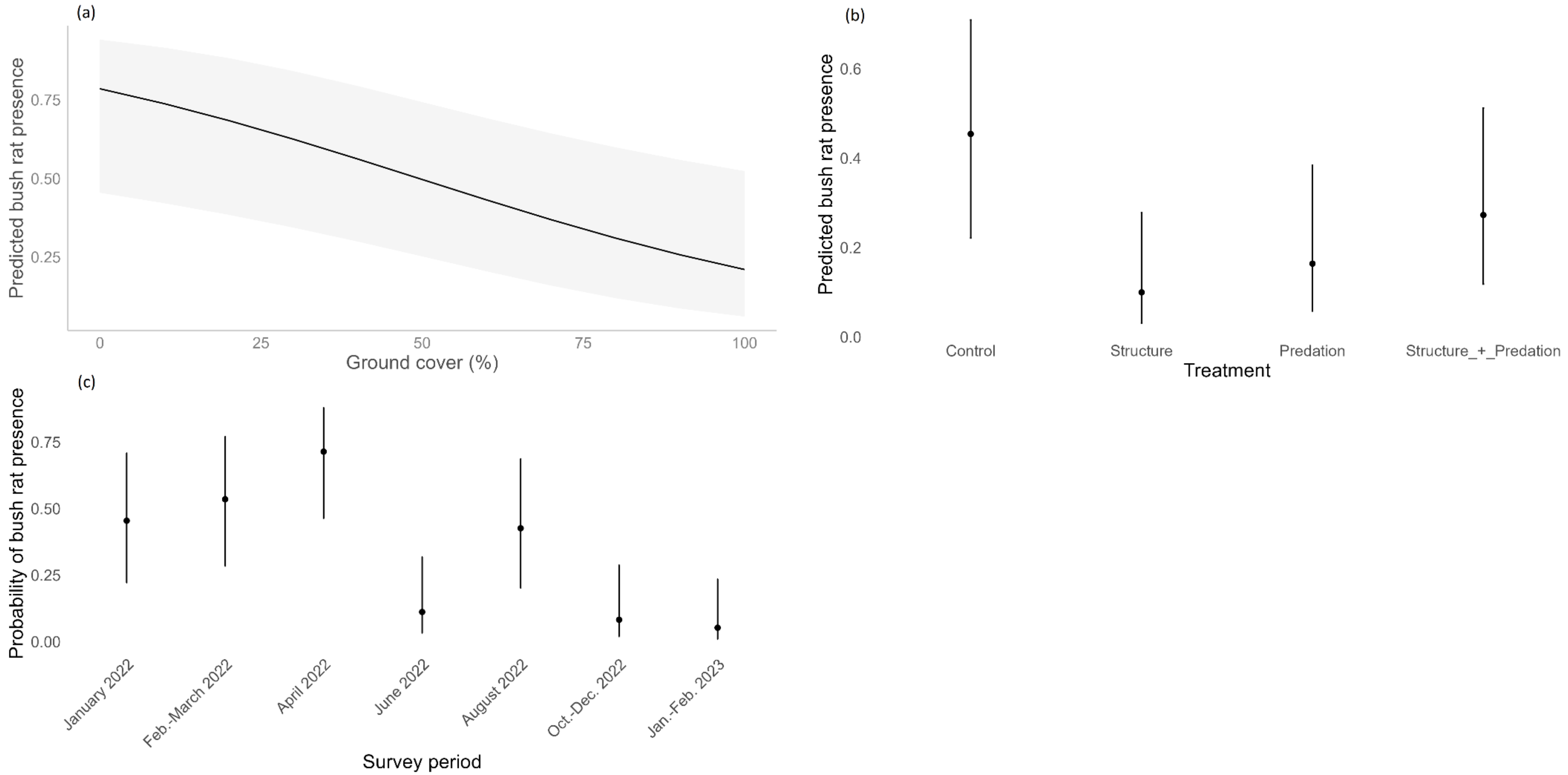


Figure 4-7: The probability of bush rat presence in response to (a) ground cover, (b) supplementary refuge treatment, and (c) survey period.

The best-fitting model for predicting agile antechinus presence included survey period, fire severity, canopy cover, shrub cover, ground cover and leaf litter as fixed effects (Table 4-9, Table S4-1). While holding other variables constant, agile antechinus increased significantly ($p = 0.032$) with increasing shrub cover (Figure 4-8). Survey period significantly ($p = 0.002$) affected agile antechinus presence, with likelihood decreasing from survey periods 1 to 4 (January – June 2024) (Figure 4-8).

Table 4-9: The best model (lowest AIC) predicting agile antechinus presence contained survey period, fire severity, canopy cover, shrub cover, ground cover, leaf litter and treatment.

Explanatory variable	Estimate	95% C.I.	p-value
Intercept	-0.18	(-1.26, 0.89)	0.74
Survey period 2	0.32	(-0.59, 1.24)	0.49
Survey period 3	1.09	(0.19, 2.00)	0.02
Survey period 4	-1.89	(-3.11, -0.67)	0.002
Survey period 5	-0.11	(-1.05, 0.82)	0.81
Survey period 6	-2.22	(-3.72, -0.73)	0.003
Survey period 7	-2.71	(-4.42, -0.99)	0.002
Fire severity (low)	-1.49	(-2.67, -0.31)	0.012
Canopy cover	-0.09	(-0.66, 0.46)	0.73
Shrub cover	0.38	(-0.01, 0.76)	0.057
Ground cover	-0.61	(-1.07, -0.15)	0.008
Leaf litter	-0.08	(-0.67, 0.49)	0.76
Predation treatment	-1.44	(-2.44, -0.45)	0.004
Structure treatment	-2.01	(-3.11, -0.92)	<0.001
Structure + predation treatment	-0.79	(-1.74, 0.15)	0.100

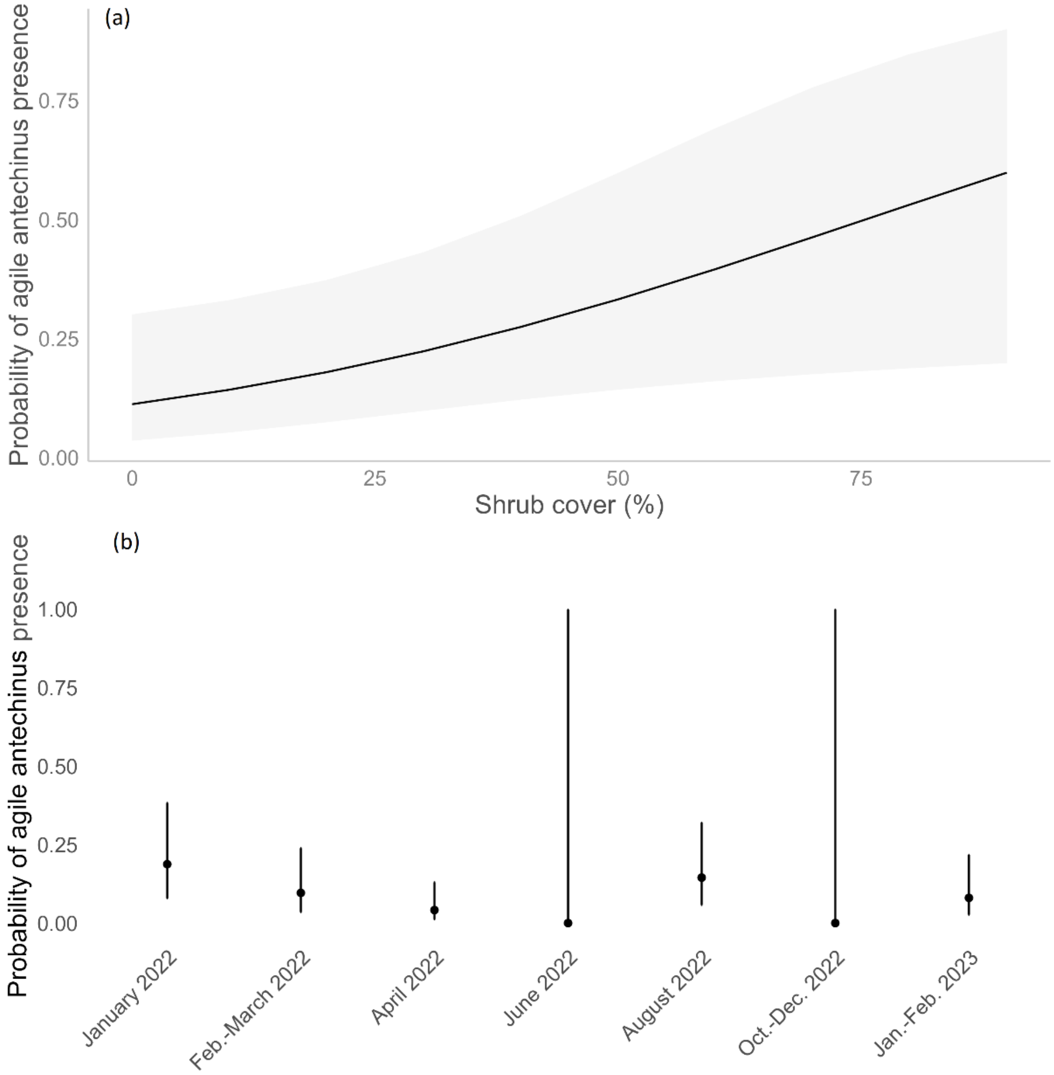


Figure 4-8: The probability of agile antechinus presence in response to (a) shrub cover and (b) survey period.

4.5 Discussion

We aimed to examine the relative effects of vegetation complexity, structural features (i.e., CWD and rocky outcrops) and supplementary refugia on the richness and abundance of small mammals in a post-fire landscape through three key questions: (1) What is the relative value of vegetation complexity, coarse woody debris (CWD) and rocky outcrops for small mammals after fire; (2) Do these variables influence small mammal richness or abundance in a post-fire landscape; and (3) Does adding supplementary structural habitat assist the recovery of small mammals after fire. We found that increasing shrub cover was associated with significant increases in the abundance and probability of occurrence for most species, while species richness also increased significantly in areas burnt at high severity and areas with high levels of natural

CWD. Supplementary refuge treatments did not significantly increase the abundance or occurrence of any small mammal species.

4.5.1 Fire severity and vegetation structural complexity

Habitat structure is an important factor in determining small mammal responses to fire (Arthur *et al.*, 2012). Fox's habitat accommodation model found that small mammals respond to successional changes in vegetation according to their habitat requirements (Fox, 1982) . In our study, we found that shrub cover had a significant positive relationship with small mammal richness and abundance, house mouse abundance, black rat abundance, and the presence of agile antechinus. The association of house mice with high shrub cover in our study is consistent with previous studies, which have found that house mice actively use dense vegetation and complex groundcover to mitigate predation pressure (Dickman, 1992; Jensen *et al.*, 2003). While agile antechinus in our study preferred areas with higher shrub cover, they have been found to use all parts of the burnt landscape equally after prescribed fire (Lees *et al.*, 2022).

In many Australian forests, shrub cover rapidly increases after fire (Arthur *et al.*, 2012; Bowd *et al.*, 2021), and observations from our study area validate these trends with particularly high levels of shrub cover in areas burnt at high severity (unpublished data, see also (Crotteau *et al.*, 2013; Minor *et al.*, 2017; Barker *et al.*, 2022)). The positive relationship between many small mammal species and shrub cover, and increased shrub cover in areas burnt at high severity, illustrate the dynamics behind our finding that species richness was greatest in areas burnt at high severity. The link between small mammals and vegetation may have also been strengthened by above average rainfall in our study area (Bureau of Meteorology, 2023) as a result of the La Niña phase of the El Niño Southern Oscillation. Letnic *et al.* (2005) and Kelly *et al.* (2012) provide examples of Australian small mammal richness and abundance responding positively to increased rainfall.

Availability of natural and supplementary refuge

In addition to vegetation structure, we examined the response of small mammals to natural refuge in burnt areas. We found that no species were affected by the presence or absence of rocky outcrops, but species richness was significantly greater in areas with high levels of natural coarse woody debris. This finding is supported by a previous study of small mammals in Namadgi National Park, which also found small mammal richness increased with CWD volume (Dixon *et al.*, 2019). Individual species, such as the bush rat, have been found to be associated with high levels of CWD in other areas of south-eastern Australia (Claridge *et al.*, 2008). Agile antechinus are also associated with high volumes of CWD (Johnstone *et al.*, 2011), and use hollow logs as refuge alongside arboreal hollows (Cockburn and Lazenby-Cohen, 1992).

We found that adding supplementary refugia did not affect small mammal richness, abundance, or the occurrence of most species. Several factors, including time since fire and above average rainfall, may have reduced the measurable benefit of the piles in our study. Logistical constraints meant we were unable to establish our experiment until nearly two years after the fire, when some of the vegetation had begun to grow back. Structurally complex CWD refuge may have been more beneficial immediately after fire, as species habitat preferences are most pronounced when resources are limited (Hale *et al.*, 2016). The abundance of resources, from food to dense vegetation, that resulted from heavy rainfall in the years following the fire may have broadened the range of viable habitats for small mammals during our study. In addition, the proximity of our sites may have affected movement of individuals between treatments. Although we set the distance between treatments to reduce frequent movement between sites (LAZENBY-COHEN and Cockburn, 1991; Maitz and Dickman, 2001; Marchesan and Carthew, 2008; Wood, 1970), it is possible that individuals occasionally moved between treatments.

Previous studies have provided examples of species found in our study using artificial refuges. Notably, Arthur *et al.* (2005) found that house mice exposed to predation benefited from CWD piles covered in chicken wire, a design which formed the basis of our supplementary refuges. We were unable to quantify predation pressure during our study period, therefore we are unable to determine whether our CWD piles provided similar benefits for small mammals. Agile antechinus have also been recorded using chainsaw hollows cut into trees (Best *et al.*, 2022), although they did not show a positive response to ground-level artificial refuge in burnt areas (Watchorn *et al.*, 2024). When supplementary refugia was included in our best model, we found that black rat abundance was significantly lower for the structure + predation treatment (CWD piles covered with chicken wire) compared to the control. The black rat is a widespread introduced species in Australia, and are not considered to be habitat specialists and have been recorded using similar supplementary log piles in urban areas of Sydney (Price and Banks, 2018). The lack of a strong positive response from our study species indicates that coarse woody debris refuge may not have been a limiting factor for small mammals in our study area. Sutherland (1999) found that *Antechinus stuartii* did not utilise artificial refuges in burnt areas until increasing population size led to reduced natural refuge availability.

4.5.2 Management implications

Although supplementary CWD refugia were not necessary for small mammal population recovery in our study area, they have been beneficial in a range of other disturbed landscapes (Seip *et al.*, 2018; Sullivan and Sullivan, 2012; Sullivan and Sullivan, 2021; Sullivan *et al.*, 2012). We hypothesise that alterations to the design of the CWD piles may improve the rates at which they are used by small mammals. These may include creating more suitable nesting substrates (e.g., PVC tubes for nesting, small nest boxes, etc.) within the CWD piles to increase their occupancy by small mammals. We found evidence of only one nest, likely agile antechinus, within the roughly 160 structures we constructed (pers. obs.).

4.6 Conclusion

Our results indicate that access to structural refugia was not a limiting factor for small mammals after a large-scale fire in south-eastern Australia. In line with the findings of many previous studies, small mammal activity was driven primarily by vegetation characteristics, particularly shrub cover. Species richness increased in areas burnt at high fire severity and was greatest in areas with high volumes of natural CWD. The addition of supplementary refugia, in the form of CWD piles, did not significantly increase the abundance or occurrence of any small mammal species.

4.7 Supplementary material

Table S1: Candidate linear mixed models tested to predict the effects of habitat variables on small mammal richness, small mammal abundance, house mouse abundance, black rat abundance, bush rat presence and agile antechinus presence, ranked by delta AIC. Details for the components of each model can be found in Table 4-2.

Model	df	AIC	ΔAIC
Small mammal richness			
11	19	1300.224	-
6	15	1300.577	0.337
10	17	1303.814	3.574
9	15	1305.102	4.862
2	14	1305.986	5.746
12	19	1307.743	7.503
8	15	1310.488	10.248
13	19	1310.881	10.641
5	13	1311.821	11.581
17	17	1312.554	12.314
14	29	1313.041	12.801
7	12	1313.888	13.648
3	11	1314.377	14.137
4	11	1314.761	14.521
16	17	1314.796	14.556
15	13	1315.617	15.377
1	3	1504.669	204.429
Small mammal abundance			
2	15	2769.764	-
6	16	2771.313	1.549
10	18	2771.383	1.619
9	16	2771.754	1.99
11	20	2775.277	5.513
12	20	2775.636	5.872
5	14	2776.876	7.112
17	18	2777.608	7.844

13	20	2777.776	8.012
3	12	2780.321	10.557
4	12	2780.609	10.845
8	16	2780.715	10.951
7	13	2782.153	12.389
16	18	2782.463	12.699
15	14	2784.14	14.376
14	30	2788.116	18.352
1	4	3122.985	353.221
House mouse abundance			
9	16	1993	-
2	15	1993.084	0.084
10	18	1993.706	0.706
11	20	1994.164	1.164
6	16	1994.45	1.45
12	20	1994.795	1.795
5	14	1995.767	2.767
4	12	1997.099	4.099
8	16	1997.791	4.791
3	12	1997.889	4.889
7	13	1998.218	5.218
13	20	1998.515	5.515
17	18	1998.755	5.755
15	14	1999.515	6.515
16	18	2000	7
14	30	2005.102	12.102
1	4	2391.825	398.825
Black rat abundance			
10	18	1510.735	-
17	18	1512.909	2.174
2	15	1513.281	2.546
5	14	1513.685	2.95

11	20	1514.258	3.523
6	16	1514.797	4.062
8	16	1515.067	4.332
9	16	1515.265	4.53
3	12	1517.21	6.475
4	12	1517.539	6.804
16	18	1518.184	7.449
7	13	1518.412	7.677
15	14	1519.922	9.187
12	20	1520.189	9.454
13	20	1521.362	10.627
14	30	1522.268	11.533
1	4	1656.924	146.189
Bush rat abundance			
11	19	357.6809	-
10	17	359.1416	1.4607
5	13	361.771	4.0901
8	15	361.8341	4.1532
6	15	362.6471	4.9662
14	29	364.3495	6.6686
16	17	364.5732	6.8923
17	17	365.975	8.2941
13	19	366.8887	9.2078
3	11	367.1652	9.4843
9	15	367.235	9.5541
12	19	367.2526	9.5717
7	12	367.3212	9.6403
2	14	368.2307	10.5498
4	11	368.8282	11.1473
15	13	369.1823	11.5014
1	3	441.0568	83.3759

Agile antechinus abundance			
9	15	303.9587	-
6	15	304.6666	0.7079
13	19	305.3033	1.3446
2	14	305.6072	1.6485
11	19	308.6105	4.6518
4	11	308.8173	4.8586
12	19	309.4633	5.5046
10	17	309.8159	5.8572
7	12	310.278	6.3193
3	11	311.1284	7.1697
15	13	312.0958	8.1371
5	13	314.8036	10.8449
17	17	315.2303	11.2716
8	15	315.7271	11.7684
16	17	317.9336	13.9749
14	29	325.0929	21.1342
1	3	371.4852	67.5265

Chapter 5: Thermal properties of natural and constructed refuges for terrestrial vertebrates after fire

Statement of Contribution

This thesis is submitted as a Thesis by Compilation in accordance with

https://policies.anu.edu.au/ppl/document/ANUP_003405

I declare that the research presented in this Thesis represents original work that I carried out during my candidature at the Australian National University, except for contributions to multi-author papers incorporated in the Thesis where my contributions are specified in this Statement of Contribution.

Title: Thermal properties of natural and constructed refugia for terrestrial vertebrates after fire

Authors: Burns, Heather; Cary, Geoffrey; Brawata, Renee; Lindenmayer, David; Evans, Maldwyn; Gibbons, Philip

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Contribution to paper: HB designed the experimental methods with advice from PG, GC, RB and DL. HB conducted fieldwork to deploy thermal sensors, and HB and ME analysed the data. HB wrote the manuscript with feedback from PG, GC, RB and DL.

Senior author or collaborating authors endorsement: Philip Gibbons

Heather Burns ________________________ ____________ 08/07/2024

Candidate Signature Date

Endorsed

Philip Gibbons ________________________ _____________ 15/07/2024 _____

Chair of Supervisory Panel – Print Name Signature Date

Robert Dyball ________________________ 17/07/2024

Delegated Authority – Print Name Signature Date

Foreword

Chapters 3 and 4 examined the effects of supplementary coarse woody debris on reptile and small mammal abundance and richness in areas burnt during the 2020 Orroral Valley Fire. My findings suggest that supplementary CWD benefits reptiles but has no effect on small mammal abundance or richness. In each of the previous chapters, I examined the potential effect of habitat variables such as shrub cover, abundance of natural coarse woody debris, and the presence of rocky outcrops on my results. In Chapter 5, I take a closer look at the thermal properties of supplementary CWD piles.

A review of artificial refuge literature by Cowan *et al.* (2021) found that very few studies reported data on the thermal properties of their refuge, despite the importance of refuge for the thermoregulation to both endothermic and ectothermic vertebrates (Scheffers *et al.*, 2014; Milling *et al.*, 2017; Rezende and Bacigalupe, 2015). Where the thermal properties of artificial refuges have been reported, there are examples of measurable differences between temperatures within natural refuges and corresponding artificial refuges (i.e., nest boxes (Griffiths *et al.*, 2018; Griffiths *et al.*, 2017)). To address this knowledge gap, I recorded temperatures within my CWD piles as well as in natural habitat such as rocky outcrops and hollow logs. I compared how well each refuge type provided a buffer against ambient temperatures, particularly during the warmest and coldest periods of the year.

This manuscript is currently being prepared for submission to a journal.

5.1 Abstract

Terrestrial refuges are essential for many small vertebrates, providing suitable nesting and hibernation habitat as well as a thermal buffer from extreme temperatures. Frequent disturbances, such as fire, alter the availability of natural refuges for a number of ground-dwelling vertebrate species. Artificial refuges can be used to supplement or replace natural refuges in disturbed areas, but their thermal properties are often poorly understood. We established supplementary coarse woody debris piles (a common type of artificial refuge) to promote small mammal and reptile recovery in areas burnt during a wildfire in south-eastern Australia. Our study consisted of 20 blocks spread evenly between areas burnt at high and low fire severity. At each block we used iButtons to measure how the thermal properties of our supplementary habitat treatments compare to: (1) ambient temperature; and (2) naturally occurring refuges such as cavities in natural logs and crevices in rocky outcrops. We also compared the number of days that each refuge type (CWD piles, rock crevices and log cavities) recorded temperatures above maximum (30°C, 35°C) and minimum (5°C, 0°C, -5°C) temperature thresholds. Temperatures within CWD piles were significantly cooler than ambient maximum temperatures and significantly warmer than ambient minimum temperatures. The number of days that CWD temperatures were below minimum temperature thresholds was greatest in areas burnt at high severity. We conclude that while CWD piles provide a thermal buffer against ambient temperatures, a combination of all three refuges in the landscape is required to maximise opportunities for thermal regulation in ground-dwelling vertebrates.

5.2 Introduction

Terrestrial refuges are essential habitat for small vertebrates. Examples include coarse woody debris (CWD), ground-level vegetation, crevices in rocky outcrops, and underground burrows (Huey *et al.*, 1989; Körtner *et al.*, 2008; Matthews *et al.*, 2017; Seebacher and Alford, 2002; Sullivan *et al.*, 2012; Whitaker and Shine, 2002). The availability of such refuge is important because they provide valuable habitat for nesting and hibernation as well as protection from predation (Christie *et al.*, 2012; Flanagan-Moodie *et al.*, 2018; Mac Nally and Horrocks, 2002). Refuges can also provide essential protection from harsh environmental conditions (Cowan *et al.*, 2020; Goldingay, 2019; Goldingay and Thomas, 2021; Scheffers *et al.*, 2014). The addition of refuges, whether natural or artificial, to landscapes has been shown to increase population numbers for a variety of species (Christie *et al.*, 2013; Dracup *et al.*, 2015; Evans *et al.*, 2019a; Manning *et al.*, 2013; Michael *et al.*, 2004; Sullivan and Sullivan, 2012; Arthur *et al.*, 2005).

The thermal conditions of ground-level microhabitats are important because such places are primary refuges for a range of small vertebrates. Endotherms, such as small mammals, expend less energy regulating their temperature when ambient or refuge conditions are close to their preferred body temperature (Cooper and Withers, 2005; Rezende and Bacigalupe, 2015). Behaviour such as basking or sheltering in warm refuges can help reduce these energetic costs (Matthews *et al.*, 2017). When these energetic costs are too high (e.g., winter) or resources are limited (e.g., post-fire), some small mammals use torpor to reduce energetic expenditure (Körtner *et al.*, 2008; Matthews *et al.*, 2017; Stawski *et al.*, 2016). Ectotherms, such as lizards and snakes, are unable to self-regulate body temperatures and often move between basking and shelter to stay within a suitable range (Curry-Lindahl, 1979; Sunday *et al.*, 2014). For example, Whitaker and Shine (2002) found that eastern brown snakes (*Pseudonaja textilis*) selected refuges that required minimal thermal regulation, and their preferred refuge characteristics changed seasonally to reflect changes in thermal requirements.

As a disturbance, fire alters habitat that in turn affects thermal conditions. Ground-storey vegetation and coarse woody debris are important habitat for terrestrial vertebrates, yet are often removed by fire (Tinker and Knight, 2000; Bassett *et al.*, 2015a). Prescribed burns, which can be lower intensity than wildfires, also reduce the availability of trees and logs (Flanagan-Moodie *et al.*, 2018) and understory vegetation (Hradsky *et al.*, 2017a). Fire alters the microclimate of forests through the removal of overstorey, particularly in severe fires where the canopy is often partly or completely consumed (Gale *et al.*, 2023). Canopy cover impacts the amount of solar radiation reaching ground level (Zou *et al.*, 2007) and the resulting thermal conditions for ground-dwelling vertebrates. Pringle *et al.* (2003) found that ground-level microhabitat temperatures increase as canopy cover decreases and solar radiation increases. These conditions can last for years after fire, as Hossack *et al.* (2009) found that microhabitats in areas burnt at high severity were consistently warmer than those in unburnt sites for up to three years.

Frequent disturbances, such as fire, can alter the availability of natural refuges for a number of ground-dwelling vertebrate species. When natural refuges are scarce, installing artificial refuges are a common management intervention for conservation (e.g., nest boxes Goldingay *et al.* (2018) and artificial rock crevices, Cowan *et al.* (2020)). While it may be possible to mimic the structure of natural refuge, creating artificial refuges that mimic the thermal properties of naturally-occurring habitat can be challenging and is often under-reported (Cowan *et al.*, 2021). Multiple studies have also shown that artificial refuges can produce more extreme thermal conditions than natural refuges and temperatures beyond a species preferred range (Griffiths *et al.*, 2018; Griffiths *et al.*, 2017; Rowland *et al.*, 2017).

Australia has experienced severe drought and extensive fire seasons in recent decades (Abram *et al.*, 2021a), with the Black Summer fires of 2019-2020 affecting 5.7 million hectares in the south-eastern region (Mackey *et al.*, 2021). In particular, the total area burnt at high severity across the Black Summer fires was the largest in Australia since 1988 (Collins *et al.*, 2021). Given the impact fire can have on the availability of refuges and their microclimates, it is important to reassess the thermal dynamics of remnant and artificial refuge for ground-dwelling vertebrates after fire. In this study we aimed to test the thermal properties of two natural refuges (natural logs and rock crevices) and one artificial refuge (piles of coarse woody debris) at sites burnt at different fire severities across both summer and winter seasons.

5.3 Methods

5.3.1 Study site

Our study area is located in Namadgi National Park, Australian Capital Territory in south-eastern Australia (35.558147 S, 148.940715 E). Namadgi National Park covers a mix of alpine and sub-alpine habitat, with dry sclerophyll forest and wet sclerophyll forest being the dominant vegetation types (Keith and Simpson, 2012). Elevation in the park ranges from 900 to 1800 metres above sea level, with our study sites situated between 900-1200 metres. Mean annual rainfall in the park is 1091 mm, winter (July) temperature ranges from -2.4°C to 2.6°C and summer (January) temperature ranges from 10.1°C to 20.7°C (Bureau of Meteorology, 2024). Summer (January) mean temperature ranges from 10.7°C to 23.3°C, with 80% of days experiencing daily maximum temperatures between 20.5°C and 33.4°C (Bureau of Meteorology, 1981). The national park has experienced multiple bushfires in the past two decades, most recently in January and February of 2020 where an estimated 80% of the park burnt during the Black Summer fires.

In September 2021, we established an experiment consisting of 20 blocks that were equally divided between two strata of fire severity: (1) blocks burnt at high severity in the 2020 fires; and (2) blocks burnt at low severity in the 2020 fires. Fire severity was mapped and classified using RdNBR values (Gale *et al.*, 2021) in ArcGIS (Environmental Systems Research Institute, 2022). Once these classes were mapped and tentative study sites were selected, we ground-truthed each location before establishing experimental blocks. For a site to be classified as high severity, the canopy must have been >75% consumed by fire. For a site to be classified as low severity, the canopy must have experienced <25% crown scorch. There were no minimum patch sizes for each burnt area, but each block was located completely within its designated severity class with a buffer of at least 100 metres. Unburnt sites were not included in this study due to the only unburnt forest being a small patch in the southern portion of the park. All blocks were located within the dominant vegetation type (dry sclerophyll forest) in the study area.

At each block we measured temperature within four refuge treatments: (1) ambient temperature (i.e., control with no refuge); (2) artificially constructed coarse woody debris (CWD) piles (artificial refuge); (3) cavities in natural logs; and (4) crevices within rocky outcrops. We measured ambient temperature approximately 70 cm. above the ground by hanging temperature loggers from the wildlife camera infrastructure at each block. We used a solar shield made from aluminium foil to reduce exposure to direct sunlight (see Figure 5-1). We included CWD piles as a treatment because of their potential to provide refuge for small vertebrates in landscapes lacking natural refuges (Arthur *et al.*, 2005; Sullivan *et al.*, 2012). For further details on the construction of the CWD piles, see Chapters 3 and 4. We selected natural logs and rock crevices as close to the CWD piles as possible, typically within 100 metres. Due to topographical and geological variability across the study area, rock crevices were not present at all blocks and therefore have a smaller sample size.

Figure 5-1: Examples, from left to right, of refuge treatments in this study: (1) rock crevice; (2) ambient temperature; (3) CWD pile.

5.3.2 Data collection

We used iButton DS1921G-F5 (Maxim Integrated Products, 2022) data loggers to record temperatures across all treatments and blocks. iButtons were inserted into plastic fobs and inserted into CWD piles, log cavities, and rock crevices using a sturdy piece of wire attached to a small stick, which allowed them to be easily retrieved at the end of the study. We selected logs that were > 20 cm diameter and aimed to place the iButtons 30-40 cm into an existing hollow. Where rocky outcrops were present in a block, we aimed to place iButtons in rock crevices as close to the ground as possible and at a depth of 30-40 cm. In CWD piles, we attempted to place iButtons in parts of the structure with minimal exposure to direct sunlight and no direct contact with the ground.

We deployed iButtons between January and March 2022 (summer to early autumn) and collected them in August 2022 (winter). One block did not have samples until February 8, 2022 and a second block did not have samples until March 15, 2022. Samples from these two blocks were included, but with truncated sampling periods. This study period allowed us to sample temperatures during summer, when our sites were likely to experience the greatest maximum temperatures, as well as winter when our sites may experience snow and sub-zero ambient temperatures. We programmed all iButtons to log temperatures every 3 hours. In addition to temperature, we collected data at each treatment on environmental characteristics such as canopy cover that could potentially influence the thermal properties of each treatment (see Table 5-1).

Table 5-1: Potential explanatory and response variables, with definitions and methods of data collection.

Variable	Definition	Data collection
Refuge treatment (explanatory)	Natural or artificial refuge that could be used by terrestrial vertebrates in our study	Four refuge types were examined in our study: CWD piles, log cavities, rock crevices, and ambient temperature (no refuge)
Day (explanatory)	A continuous variable representing day of the year	Calendar dates were transformed into Julian day (i.e., January 1= day 1; December 31= day 365)
Canopy cover (explanatory)	Continuous variable representing the percentage of canopy cover present at a point above each iButton location	Canopy cover (defined as woody biomass/branches and leafy material) was measured using the %Cover app (Mignanelli, 2022).
Fire severity (explanatory)	Categorical variable with two levels: high fire severity and low fire severity	Fire severity was determined using spatial relativised delta normalised burn ratio (RdNBR; Miller and Thode (2007)) data for Namadgi National Park.
Solar radiation (explanatory)	Continuous variable representing maximum potential insolation (in watt hours per km²) at the location of each iButton in our study.	Calculated using the area solar radiation tool in ArcGIS (Environmental Systems Research Institute, 2022), using inputs of latitude and Julian day to calculate the amount of solar radiation at a point. This represents the solar radiation reaching a specific location at a given time of the year, but it does not consider canopy cover or other factors that may influence the amount of solar radiation reaching the ground. Insolation can be calculated over a given period or across multiple time windows. We broke our study period into 16 two-week intervals and calculated insolation values for the location of every iButton during each of these intervals.
Maximum temp measured using iButtons (response)	The highest temperature recorded in a 24-hour period, beginning at midnight, and ending at 11:59 pm the same day	
Minimum temp measured using iButtons (response)	The lowest temperature recorded in a 24-hour period, beginning at midnight, and ending at 11:59 pm the same day	
Temperature range measured using iButtons (response)	Daily temperature variation, measured as daily maximum – daily minimum	

5.3.3 Data analysis

To test the thermal properties of our treatments (ambient, CWD piles, log cavities, and rock crevices), we fitted several linear mixed models (LMMs) using two response variables (daily maximum temperature and daily minimum temperature) (Table 5-1, Table 5-2). To account for variation between samples that is not associated with treatments, we also included day, canopy cover and fire severity as potential explanatory variables in each model (see Table 5-1). Solar radiation was removed from all models as it was highly correlated with the 'day' variable. Block and iButton number were included as a random effect in all models. We also fitted a 'day' variable to both sine and cosine curves to mimic seasonal temperature fluctuations (Lindenmayer *et al.*, 2022). We fitted the models using the 'glmmTMB' package (Brooks, 2017) in R (R Core Development Team, 2022) and compared model fits using Akaike's Information Criterion (AIC), selecting the simplest model within two delta AIC scores of the top-ranked model for each response variable (Arnold, 2010; Burnham and Anderson, 2002). All predictions were made while holding all other fixed effects in the model either at their mean (for continuous variables) or mode (categorical variables). We checked model assumptions by inspecting the histograms of the residuals for normality and checking the variable inflation factor in the Performance package in R (Lüdecke *et al.*, 2021).

We calculated the average number of days each treatment exceeded minimum and maximum temperature thresholds at five distinct levels. The maximum temperature thresholds were set at 30°C and 35°C, given that the preferred body temperature for many Australian reptiles is 30°C (Klingenböck *et al.*, 2000; Pearson *et al.*, 2003; Schwarzkopf and Shine, 1991; Whitaker and Shine, 2002) and our study area experiences temperatures above 35°C infrequently (Commonwealth of Australia, 2012). Minimum temperature thresholds were 5 °C, 0 °C, and -5 °C. These include critical maximum and minimum temperatures, as well as preferred refuge temperatures for a range of small mammals and reptiles (Curry-Lindahl, 1979; Goldingay, 2019; Goldingay and Thomas, 2021; Herrando-Pérez *et al.*, 2019; Senior *et al.*, 2019). To calculate average values, the total number of days above or below each threshold was calculated across all blocks for each treatment and then divided by the number of blocks with data for the given treatment. We used data from the entire study period (i.e., January to August) in these calculations and were not season specific.

Table 5-2 A list of model components for each model used in analysis. Models 1-15 were used to model maximum and minimum temperature. For each component, (+) signifies an additive effect and (*) signifies an interaction between two components. In models 14 and 15, the treatment variable was used in two interactions, treatment*fire severity and treatment*day.

<table>
<tr><th>Model</th><th>Random effect</th><th>Treatment</th><th>Day</th><th>Fire severity</th><th>Point canopy cover</th></tr>
<tr><td>1</td><td>(1|block) + (1|site)</td><td></td><td></td><td></td><td></td></tr>
<tr><td>2</td><td>(1|block) + (1|site)</td><td>+</td><td>+</td><td>+</td><td></td></tr>
<tr><td>3</td><td>(1|block) + (1|site)</td><td>+</td><td></td><td>+</td><td></td></tr>
<tr><td>4</td><td>(1|block) + (1|site)</td><td>+</td><td></td><td>+</td><td>+</td></tr>
<tr><td>5</td><td>(1|block) + (1|site)</td><td>+</td><td>+</td><td>+</td><td>+</td></tr>
<tr><td>6</td><td>(1|block) + (1|site)</td><td>+</td><td>+</td><td>+</td><td>+</td></tr>
<tr><td>7</td><td>(1|block) + (1|site)</td><td>+</td><td>+</td><td></td><td>+</td></tr>
<tr><td>8</td><td>(1|block) + (1|site)</td><td>+</td><td></td><td></td><td>+</td></tr>
<tr><td>9</td><td>(1|block) + (1|site)</td><td>*</td><td>*</td><td></td><td></td></tr>
<tr><td>10</td><td>(1|block) + (1|site)</td><td>*</td><td>*</td><td>+</td><td>+</td></tr>
<tr><td>11</td><td>(1|block) + (1|site)</td><td></td><td>+</td><td>+</td><td>+</td></tr>
<tr><td>12</td><td>(1|block) + (1|site)</td><td>*</td><td>+</td><td>*</td><td>+</td></tr>
<tr><td>13</td><td>(1|block) + (1|site)</td><td>*</td><td>+</td><td>+</td><td>*</td></tr>
<tr><td>14</td><td>(1|block) + (1|site)</td><td>**</td><td>*</td><td>*</td><td></td></tr>
<tr><td>15</td><td>(1|block) + (1|site)</td><td>**</td><td>*</td><td>*</td><td>+</td></tr>
</table>

5.4 Results

We deployed 72 iButtons across four refuge types (ambient n=20; CWD piles n=20; log cavities n=20; rock crevices n=12) for a period of seven months. Of these 72 iButtons, 62 were used to provide temperature data for analysis (ambient n=20; CWD piles n=19; log cavities n=13; rock crevices n=10). The remaining 10 iButtons were unable to be collected at the end of the study or failed to collect data. Each functional iButton recorded between 1092 and 1709 temperature readings, depending on time of deployment.

5.4.1 Change in thermal profile within refuge types across seasons

The best-fitting model (lowest AIC) for predicting maximum temperature included a treatment*day interaction, fire severity, and canopy cover as fixed effects (Table S5-1; Table 5-3). While holding other variables constant, maximum temperatures within CWD piles, rock crevices and log cavities were significantly lower than maximum ambient temperatures across the year (Figure 5-2). Additionally, maximum temperatures decreased significantly with increasing canopy cover.

Table 5-3: The best model (lowest AIC) predicting maximum temperature contained a treatment*day interaction, as well as fire severity and canopy cover as fixed effects.

Explanatory variable	Estimate	95% C.I.	p-value
Intercept	16.91	(16.47, 17.36)	<0.001
Canopy cover	-0.77	(-1.00, -0.54)	<0.001
Low fire severity	0.83	(0.32, 1.34)	0.1
Coarse woody debris	-1.89	(-2.3, -1.45)	<0.001
Log cavities	-1.93	(-2.43, -1.43)	<0.001
Rock crevices	-4.01	(-4.57, -3.45)	<0.001
Sin(day)	1.49	(1.27, 1.71)	<0.001
Cos(day)	8.26	(8.07, 8.45)	<0.001
CWD*sin(day)	0.06	(-0.04, 0.16)	0.058
Log cavities*sin(day)	-0.18	(-0.30, -0.06)	0.13
Rock crevices*sin(day)	0.67	(0.54, 0.81)	<0.001
CWD*cos(day)	-0.35	(-0.44, -0.25)	<0.001
Log cavities*cos(day)	0.76	(0.65, 0.86)	<0.001
Rock crevices*cos(day)	-0.60	(-0.72, -0.48)	<0.001

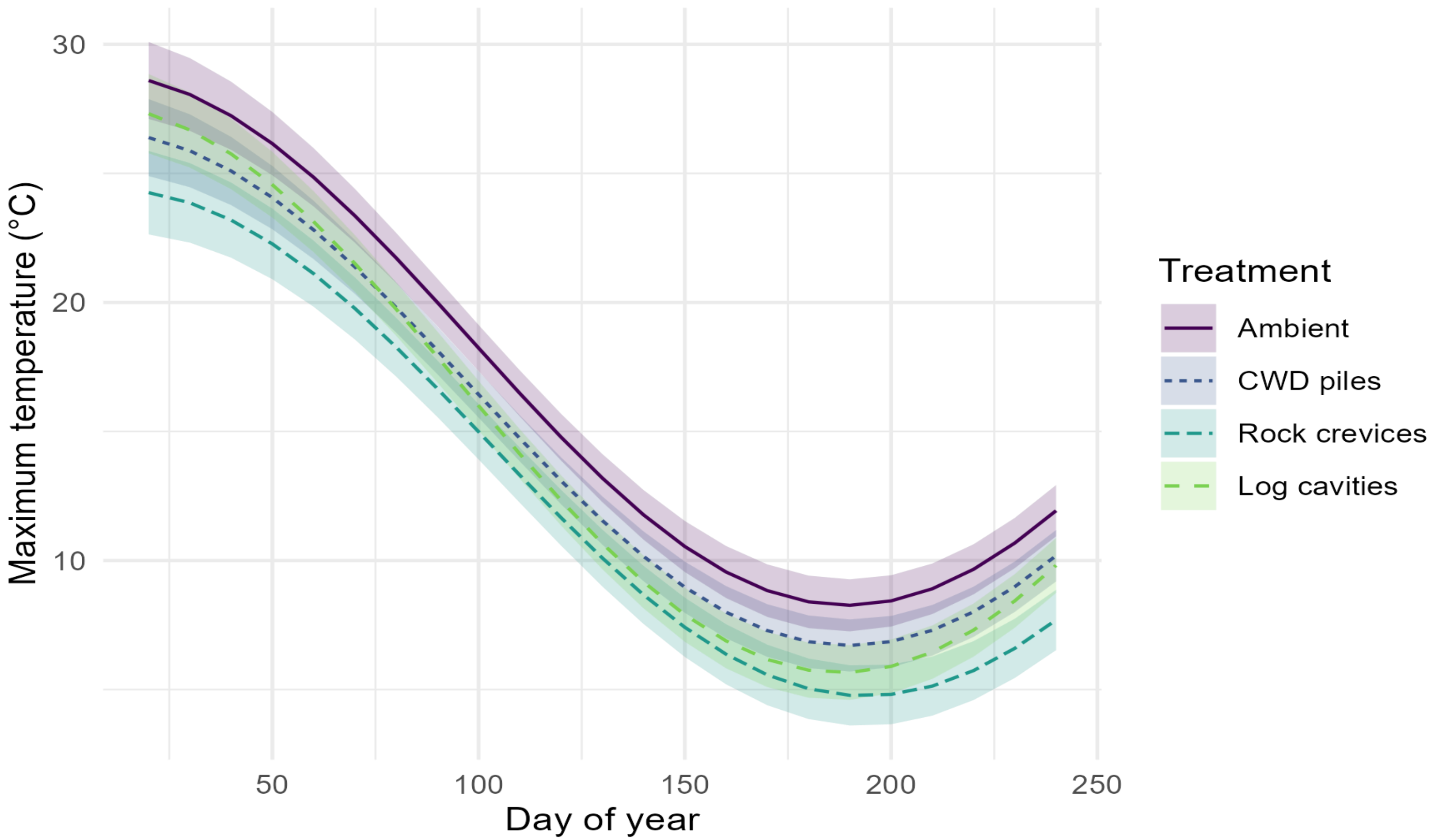

Figure 5-2: Predicted mean (± 95% confidence intervals) maximum temperature values (°C) for each treatment across the study period. All predictions were made using the best-fit model (Table 3) while holding all other fixed effects in the model either at their mean (for continuous variables) or mode (categorical variables). Days 0-60 represent summer, days 61-150 represent autumn, and days 151-240 represent winter.

The best-fitting model for predicting minimum temperature included treatment*day and treatment*fire severity interactions as fixed effects (Table 5-4; Table S5-2). Minimum temperatures were significantly higher than ambient minimum temperatures in CWD piles, rock crevices and log cavities (Figure 5-3). Within log cavities, minimum temperatures were significantly higher in areas burnt at low fire severity compared to areas burnt at high fire severity (Figure 5-4).

Table 5-4: The best model predicting minimum temperature contained treatment*day and treatment*fire severity interactions.

Explanatory variable	Estimate	95% C.I.	p-value
Intercept	6.20	[5.91, 6.48]	<0.001
Coarse woody debris	0.82	[0.53, 1.12]	0.004
Log cavities	1.26	[0.88, 1.63]	<0.001
Rock crevices	3.70	[3.32, 4.08]	<0.001
Low fire severity	0.43	[0.06, 0.81]	0.24
sin(day)	1.65	[1.47, 1.82]	<0.001
cos(day)	6.32	[6.17, 6.48]	<0.001
Coarse woody debris*low fire severity	0.13	[-0.29, 0.55]	0.75
Log cavities*low fire severity	1.17	[0.68, 1.66]	0.016
Rock crevices*low fire severity	0.66	[0.13, 1.19]	0.208
Coarse woody debris*sin(day)	0.22	[0.16, 0.28]	<0.001
Log cavities*sin(day)	0.20	[0.13, 0.26]	0.002
Rock crevices*sin(day)	0.67	[0.60, 0.75]	<0.001
Coarse woody debris*cos(day)	0.34	[0.29, 0.39]	<0.001
Log cavities*cos(day)	0.53	[0.47, 0.59]	<0.001
Rock crevices*cos(day)	0.40	[0.34, 0.47]	<0.001

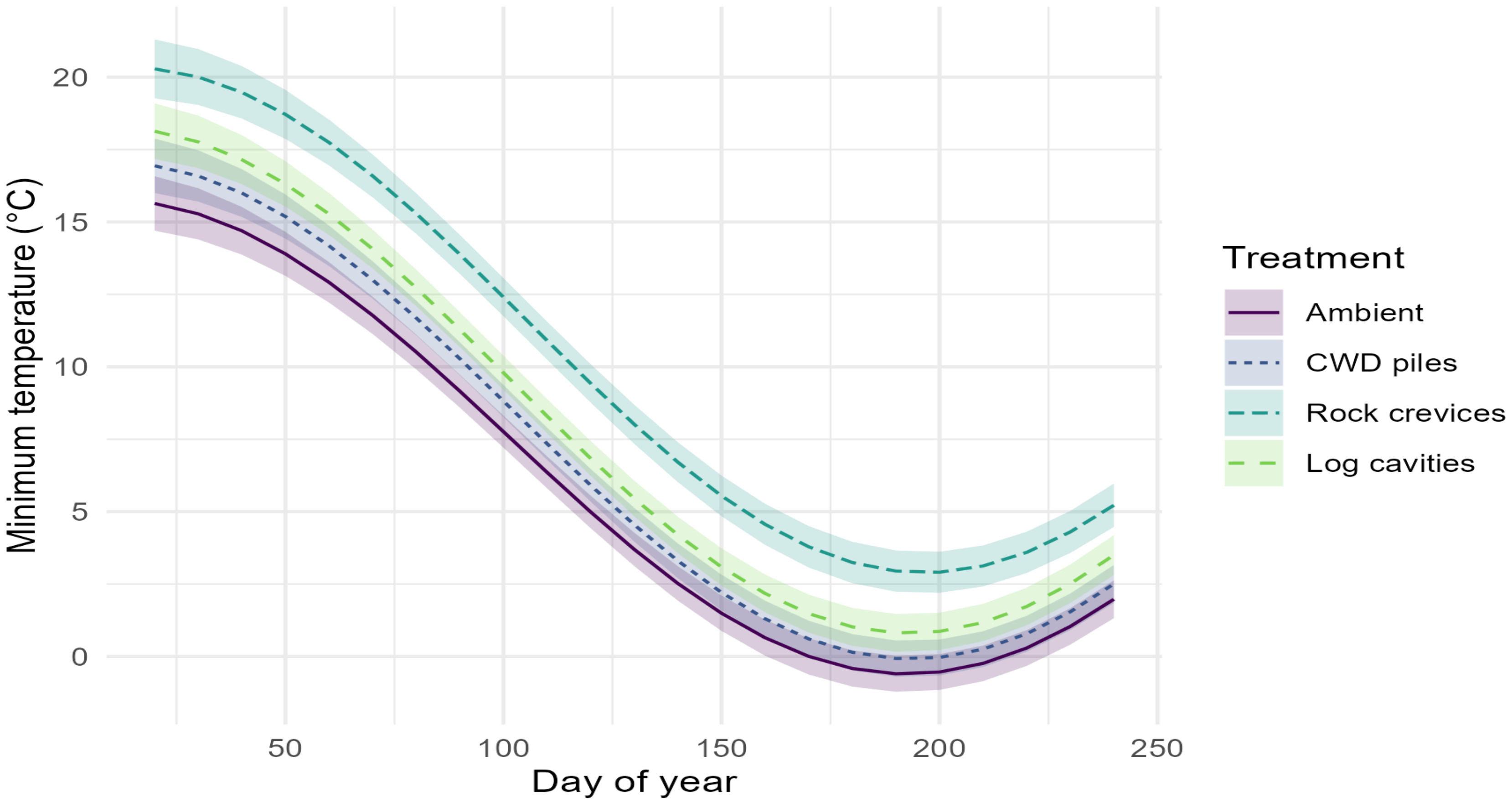

Figure 5-3: Predicted minimum (± 95% confidence intervals) temperature values (°C) for each treatment across the study period. All predictions were made using the best-fit model (Table 5) while holding all other fixed effects in the model either at their mean (for continuous variables) or mode (categorical variables). Days 0-60 represent summer, days 61-150 represent autumn, and days 151-240 represent winter.

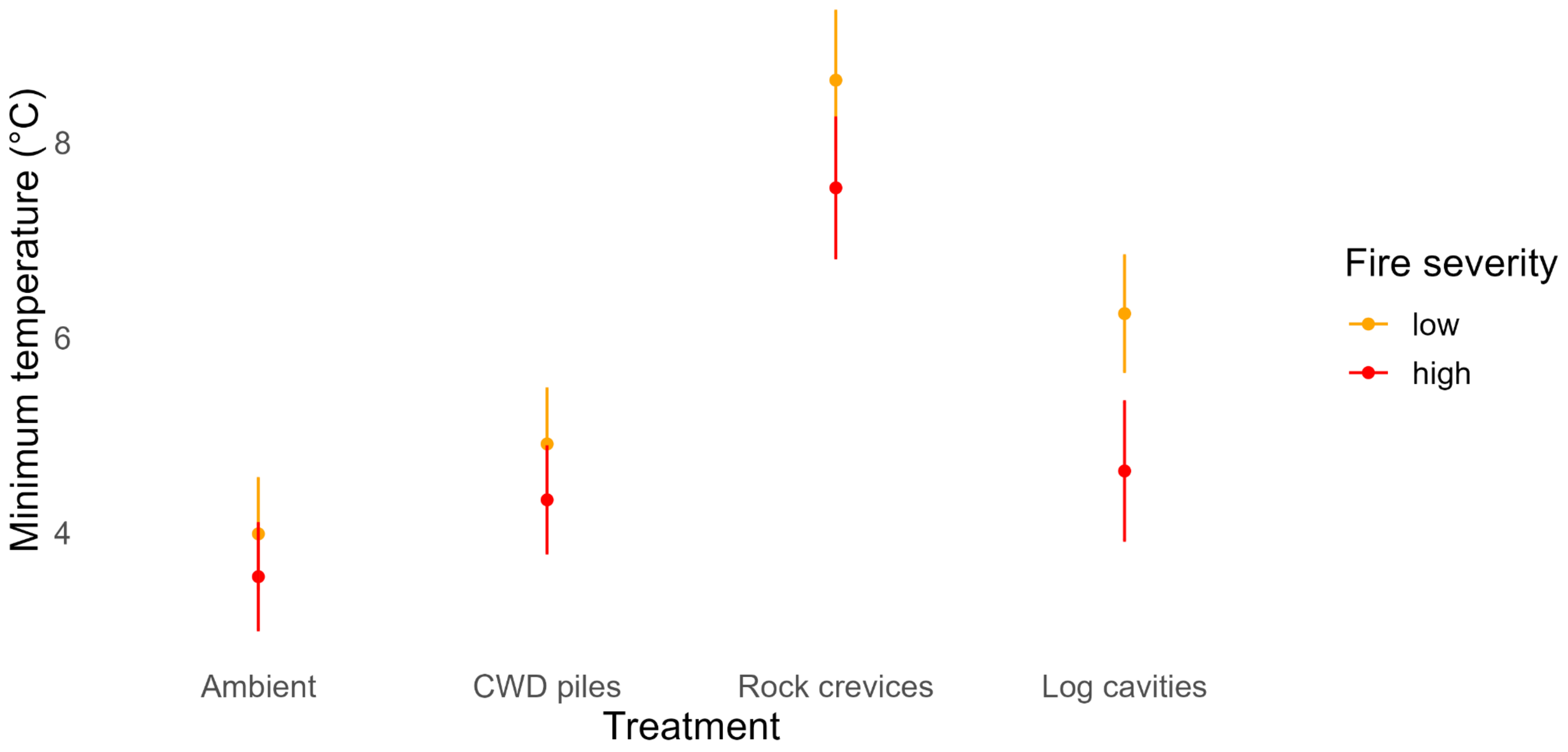

Figure 5-4: Predicted mean (± 95% confidence intervals) minimum temperature values (°C) by fire severity for each treatment. All predictions were made while holding all other fixed effects in the model either at their mean (for continuous variables) or mode (categorical variables).

5.4.2 Occurrence of upper and lower thermal thresholds

Maximum and minimum temperature thresholds data were stratified by fire severity level. We calculated the total number of days each treatment experienced above or below a threshold and divided that number by the number of blocks for each treatment in our study (i.e., ambient n=20, CWD piles n=19). CWD piles averaged a similar number of days above 30°C at high and low fire severity but averaged more days above 35°C and below minimum thresholds in areas burnt at high severity compared to low severity (Figures 5-6 to 5-10). Log cavities averaged more days beyond each thermal threshold, except the 5°C threshold, in areas burnt at high severity compared to areas burnt at low severity (Figures 5-6 to 5-10). Rock crevices experienced comparatively few days beyond thermal thresholds compared to all other refuge types in our study (Figures 5-6 to 5-10). For the extreme minimum thresholds, rock crevices averaged more days below each threshold in areas burnt at high severity (Figures 5-8 to 5-10).

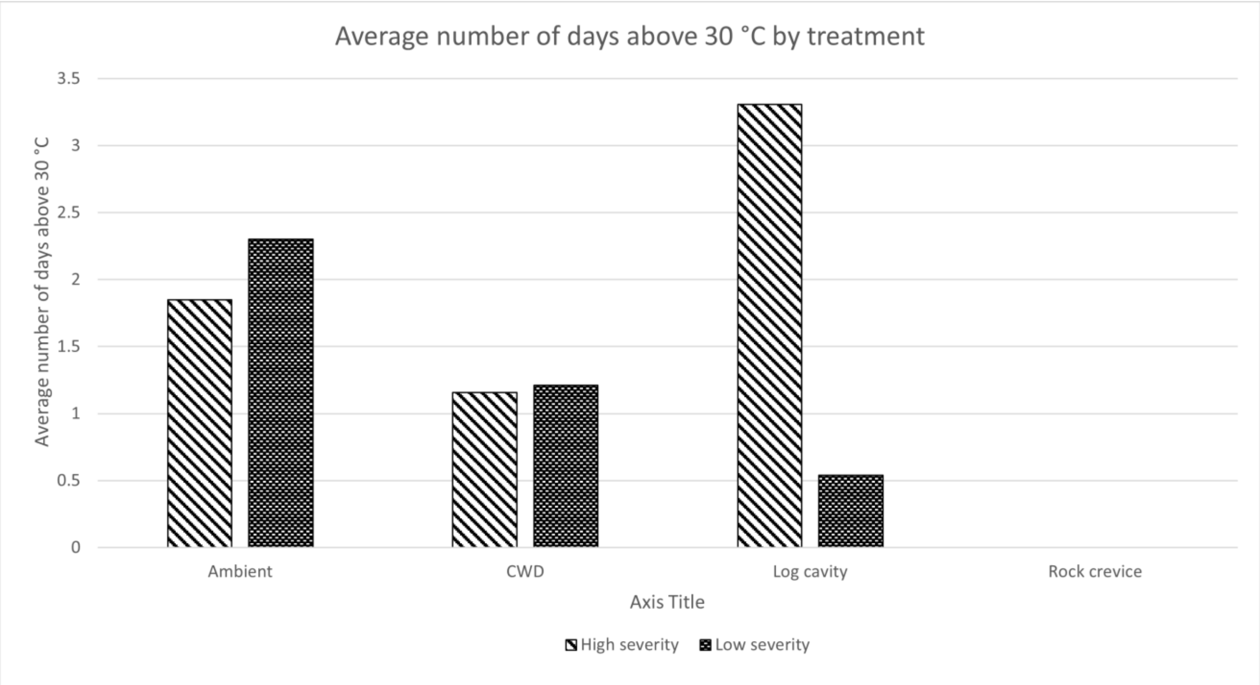

Figure 5-3: Average number of days above 30 °C for each refuge treatment, separated by high and low fire severity

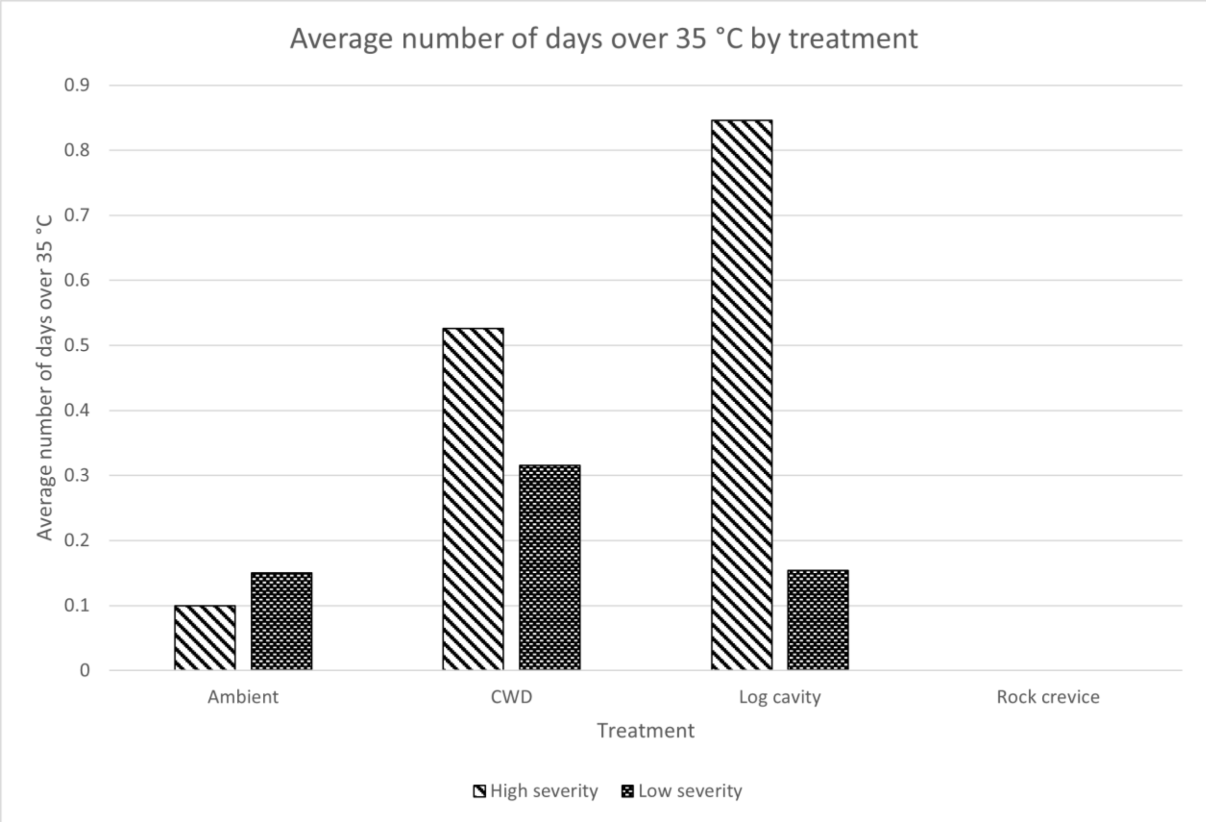

Figure 5-7: Average number of days above 35 °C for each refuge treatment, separated by high and low fire severity.

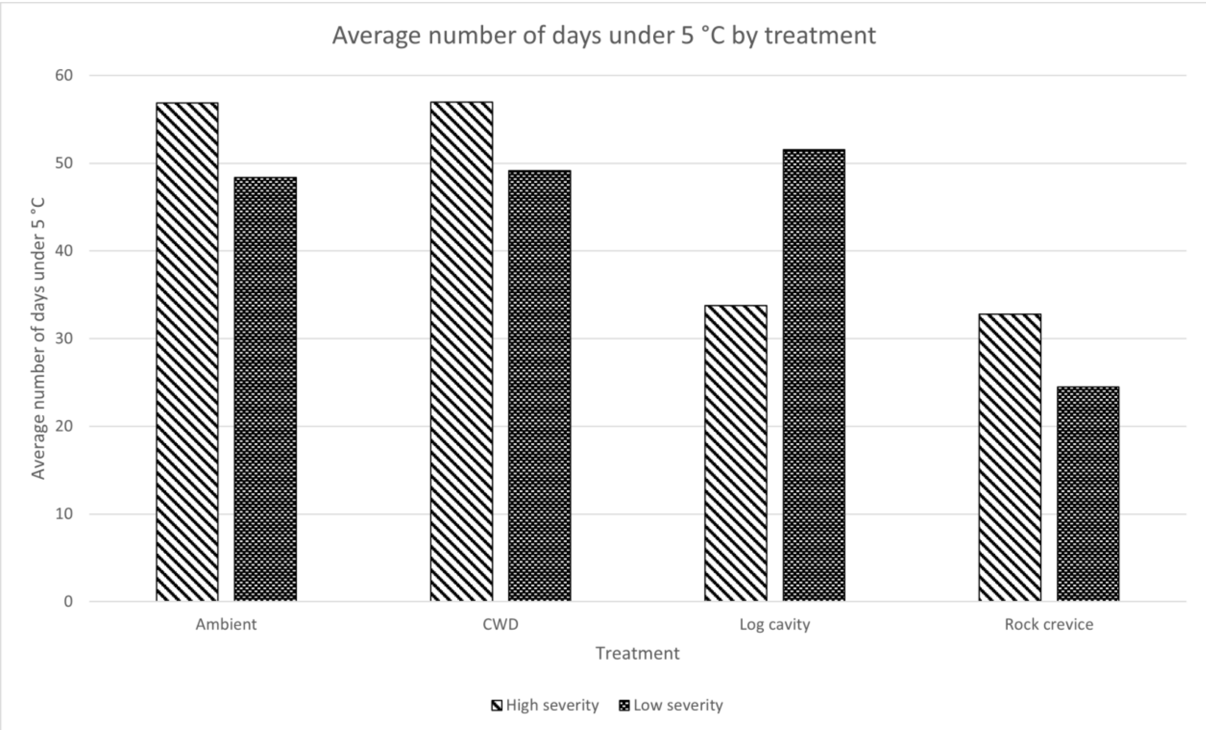

Figure 5-8: Average number of days below 5 °C for each refuge treatment, separated by high and low fire severity.

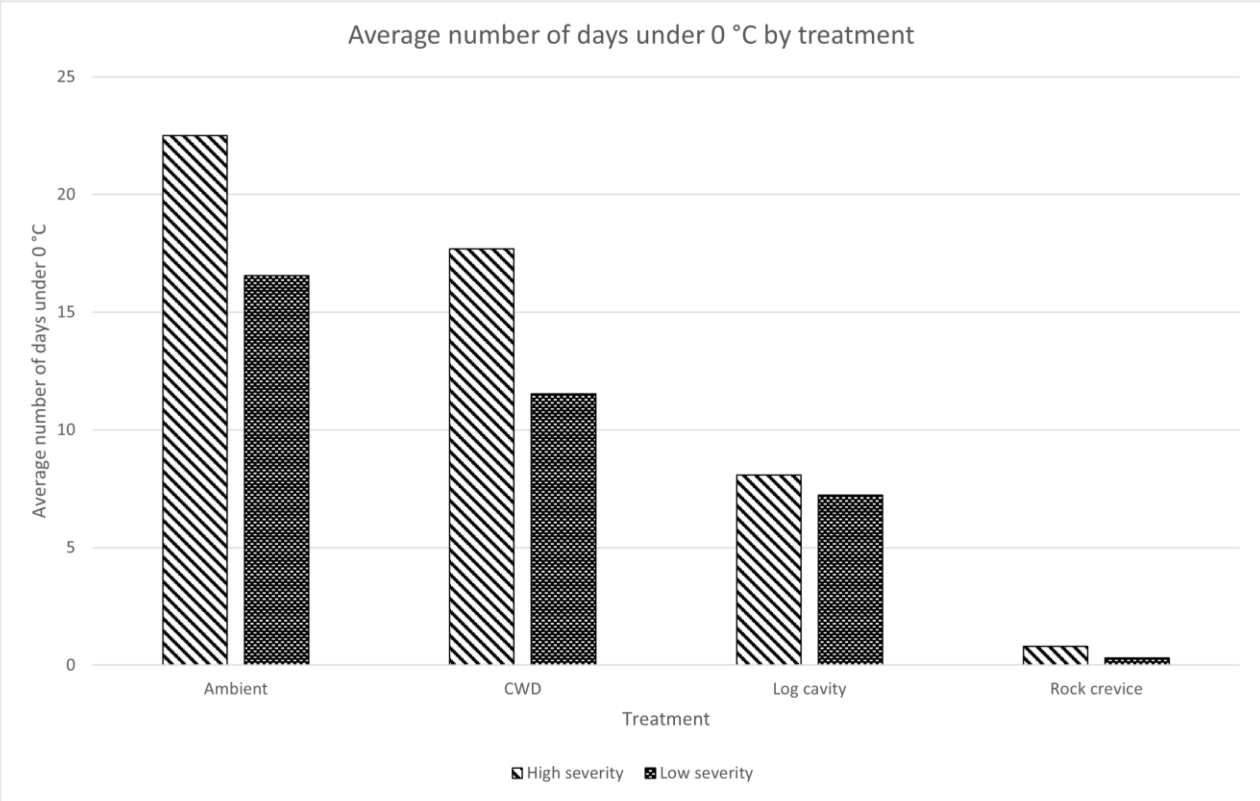

Figure 5-9: Average number of days below 0°C for each refuge treatment, separated by high and low fire severity.

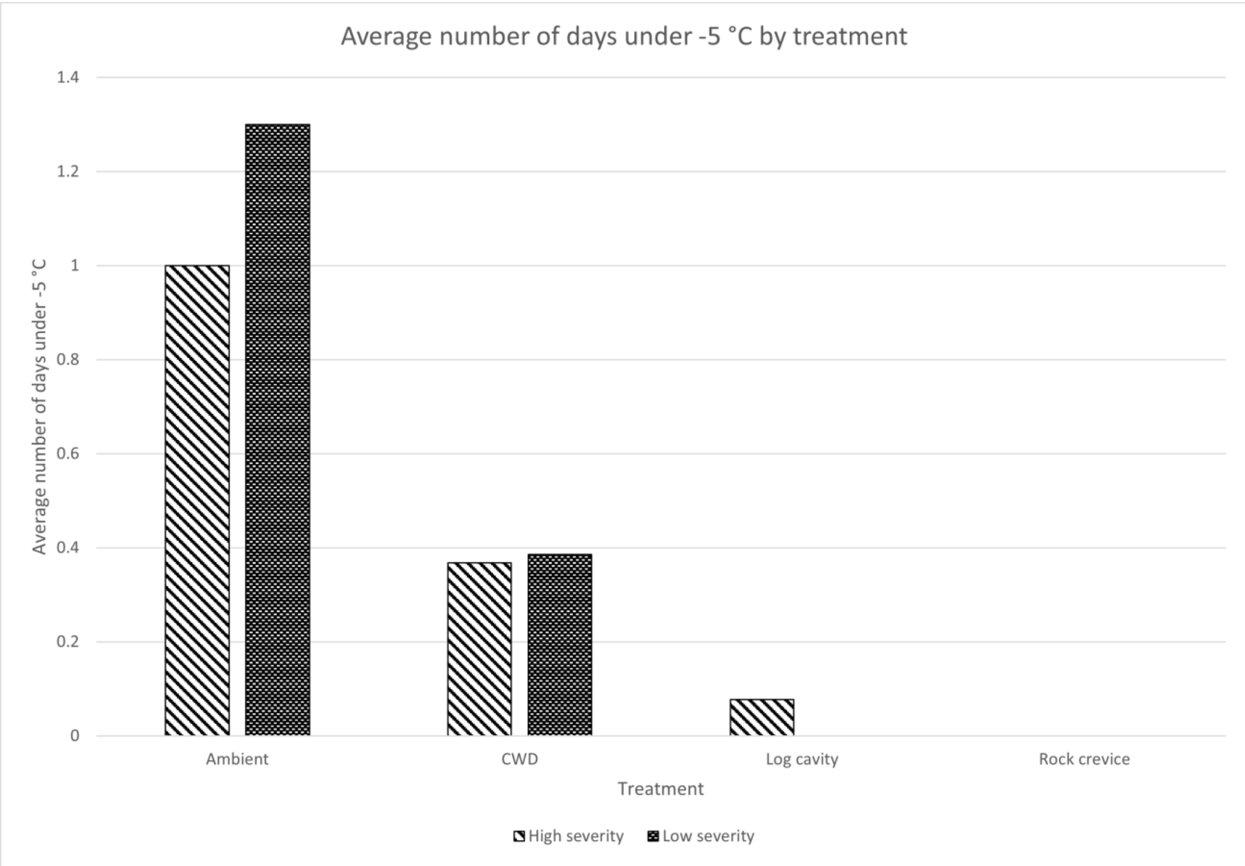

Figure 5-10: Average number of days below -5 °C for each refuge treatment, separated by high and low fire severity.

5.5 Discussion

We aimed to test the thermal properties of two natural refuges (natural logs and rock crevices) and one type of supplementary habitat (CWD piles) in a post-fire landscape. Specifically, we sought to understand how the thermal properties of supplementary habitat treatments (piles of woody debris) compared to ambient temperature and naturally occurring refuge under different fire severities. We found that temperatures within CWD piles were significantly different from ambient temperatures. However, CWD piles did not provide the same degree of buffering from daily minimum temperatures as natural log cavities and rock crevices. All types of refuge were more likely to experience extreme temperatures (i.e., those above our set maximum and minimum thresholds) in areas burnt at high severity compared to areas burnt at low severity. This trend was particularly pronounced for the number of days each refuge experienced temperatures below minimum thresholds.

5.5.1 Thermal properties of habitat refuge in a post-fire landscape

Although endotherms and ectotherms maintain their body temperatures in different ways, both have similar thermal requirements from refuges. In summer months the primary thermal benefits of refuges are to provide shelter from extreme heat and direct sunlight, and small mammals and reptiles often seek out refuge that is cooler than ambient temperatures (Sunday *et al.*, 2014). In winter the opposite is true, and small vertebrates seek refuges that are warmer than ambient temperatures. For example, small mammals in Australia's arid zone prefer refuges with average temperatures of 15 °C, while overnight temperatures in winter can be as low as -3 °C (Körtner *et al.*, 2008).

We found that supplementary CWD piles provided a significant thermal buffer compared to ambient temperatures. Across the survey period, temperatures within CWD piles were cooler than maximum ambient temperatures during the day and warmer than minimum ambient temperatures overnight. Log cavities provided a significantly better buffer against minimum temperatures compared to CWD piles but experienced no significant difference in maximum temperatures. Rock crevices provided a significantly greater buffer against ambient temperatures across all periods compared to CWD piles. Given the preferred habitat characteristics of many small mammals and reptiles, refuges that are cooler than ambient temperatures in summer and warmer than ambient temperatures in winter (Milling *et al.*, 2017; Sunday *et al.*, 2014; Körtner *et al.*, 2008), CWD piles may be a valuable form of thermal refuge in post-fire landscapes.

In addition to this study, we conducted two additional studies to test the effects of the supplementary CWD piles on small mammals (Chapter 4) and reptiles (Chapter 3). We found that reptiles, but not small mammals, were significantly more abundant at sites with CWD piles compared to a control (Chapter 3; Chapter 4). Although thermal conditions within CWD piles may not have been the only factor in these results, the difference in minimum temperatures between CWD piles and natural refuges could have contributed to low small mammal abundance at these sites. In contrast, reptile surveys were conducted over two summer seasons when CWD piles provided more suitable thermal conditions.

Rock crevices consistently provided a thermal buffer from ambient maximum and minimum temperatures across all seasons. Temperatures within rock crevices were significantly cooler than maximum ambient temperatures and warmer than minimum temperatures across the study period. These conditions indicate that where they occur in the landscape, crevices within rocky outcrops provide valuable thermal refuge for small vertebrates in post-fire landscapes. Although we did not record the occupancy of natural refuges in our study, Matthews *et al.* (2017) found that rock crevices were a preferred refuge for *Antechinus flavipes* in a post-fire landscape. Webb and Shine (1998) also found that broad-headed snakes (*Hoplocephalus bungaroides*) used rocky outcrops and crevices to maintain their preferred body temperature throughout the year.

Log cavities were significantly cooler than ambient maximum temperatures and significantly warmer than ambient minimum temperatures across the study period. Our results show log cavities provided a significant thermal buffer against night-time minimum ambient temperatures throughout the winter. The value of hollow logs as overnight refuge has been reported in other studies, with Cooper and Withers (2005) finding that hollow logs and tree hollows were warmer than the ambient temperature by an average of 5°C. Additionally, numerous studies have demonstrated the benefits of tree hollows to provide suitable microhabitats for a variety of species (Goldingay, 2019; Goldingay and Thomas, 2021; Griffiths *et al.*, 2018).

In addition to analysing temperature trends over time, it is important to examine the performance of refuges in comparison to specific minimum and maximum temperature thresholds (e.g., Goldingay and Thomas (2021)). Many species have critical thermal maximums and minimums, and temperatures above or below these thresholds are not conducive to survival. For two skink species in our study area, *Liopholis montana and L. whitii,* the critical maximum temperature is approximately 41 °C (Senior *et al.*, 2019). As this is well above the highest temperatures recorded in our study area (Commonwealth of Australia, 2012), we decided to set an upper thermal threshold closer to the preferred body temperature for many reptile species at 30°C. On average, all refuges recorded fewer than four days above this threshold, with CWD piles exceeding 30°C on fewer days than the ambient temperature. Log cavities recorded temperatures above 30°C more frequently in areas burnt at high severity compared to areas burnt at low severity. All three refuges did not perform as well with respect to critical minimum temperatures. Critical minimum temperatures for *L. montana* and *L. whitii* are 4 °C and 5.74 °C respectively (Senior *et al.*, 2019), which are generally within expected July minimum temperatures in our study area (Commonwealth of Australia, 2012). Rock crevices reported the lowest number of days below each minimum threshold, but the average number of days below 5°C ranged from 25 days (low fire severity) and 32 days (high fire severity). This indicates that across all refuge types in our study, there were periods below the critical minimum temperature for resident species *L. montana* and *L. whitii.* For most minimum temperature thresholds, refuges in areas burnt at high severity reported more days below the threshold than similar refuges in areas burnt a low severity.

5.5.2 Impact of fire severity on the thermal properties of habitat refuge

With an increase in fire extremity predicted in the coming decades (Abram *et al.*, 2021b; Bradstock, 2010; Halofsky *et al.*, 2020), it is important to understand how both high and low severity fires affect the thermal properties of various microhabitats. Our assessment of maximum and minimum temperature thresholds indicated that refuges in areas burnt at high severity experienced a greater average number of days both above maximum and below minimum thresholds for most refuge treatments. Minimum temperature models also indicated that minimum temperatures within log cavities were significantly colder in areas burnt at high severity compared to areas burnt at low severity. Although our study had a limited number of treatment x fire severity replicates (maximum 10), our findings align with existing research. Cummer and Painter (2007) found that during the summer months, higher severity fires can result in warmer post-fire microhabitats than lower severity fires. Hossack *et al.* (2009) found that these differences in refuge temperatures can last for several years. Our results also support this finding, as our study was conducted two years after the Orroral Valley Fire in Namadgi National Park. These findings indicate that species with more moderate thermal requirements may have limited refuge availability in areas burnt at high severity.

5.6 Conclusion

We aimed to compare the thermal properties of supplementary coarse woody debris piles to two natural refuges after a large-scale fire in south-eastern Australia. We found that temperatures within CWD piles were significantly more moderate than ambient temperatures. Compared to other natural refuges, rock crevices consistently provided a better thermal buffer than CWD piles while log cavities only provided additional benefits against minimum temperatures. Given these findings, we conclude that a combination of all three refuges in the landscape would maximise opportunities for thermal regulation in ground-dwelling vertebrates. Refuges in areas burnt at high severity generally experienced more days at extreme minimum or maximum thresholds, compared to refuges in areas burnt at low severity.

5.7 Supplementary material

Table S1: Candidate linear mixed models tested to predict maximum refuge temperature, ranked by delta AIC. Model components are indicated by an asterisk * in the corresponding column.

Model	Treatment	Day	Fire severity	Point canopy cover	AIC	Delta AIC
1					60264.37	702.08
2	+	+	+		59728.98	166.69
3	+		+		60244.46	682.17
4	+		+	+	60241.73	679.44
5	+	+	+	+	59720.63	158.34
6	+	+	+	+	59727.18	164.89
7	+	+		+	59721.10	158.81
8	+			+	60239.91	677.62
9	*	*			59570.60	8.31
10	*	*	+	+	59564.24	1.95
11		+	+	+	59746.43	184.14
12	*	+	*	+	59717.75	155.46
13	*	+	+	*	59726.37	164.08
14	**	*	*		59572.57	10.28
15	**	*	*	+	59562.29	-

Table S2: Candidate linear mixed models tested to predict minimum refuge temperature, ranked by delta AIC. Model components are indicated by an asterisk * in the corresponding column.

Model	Treatment	Day	Fire severity	Point canopy cover	AIC	Delta AIC
1					48986.66	1068.46
2	+	+	+		48269.92	351.72
3	+		+		48914.37	996.17
4	+		+	+	48916.12	997.92
5	+	+	+	+	48271.65	353.45
6	+	+	+	+	48274.31	356.11
7	+	+		+	48276.49	358.09
8	+			+	48914.60	996.40
9	*	*			47921.0	2.80
10	*	*	+	+	47918.20	2.30
11		+	+	+	48354.61	436.41
12	*	+	*	+	48271.91	353.71
13	*	+	+	*	48276.49	358.09
14	**	*	*		47915.90	-
15	**	*	*	+	59562.29	-

Conclusion: What should we do after the next fire?

My thesis aimed to find conservation solutions to a range of challenges faced by small mammals and reptiles in post-fire environments. After mapping the existing literature and experimentally testing supplementary coarse woody debris, it is important to reassess the practical applications of our findings. This chapter will draw on key points from Chapters 2-5 to answer the following question "What should we do after the next fire?".

5.7.1 Interventions identified through systematic mapping

The systematic map of existing management practices in Chapter 2 identified seven broad intervention categories: coarse woody debris manipulation, lethal predator control, predator exclusion, revegetation, herbivore exclusion, herbivore control, and artificial refuge. Of these seven categories, only two had been experimentally tested in post-fire landscapes at the time of the review. Conner *et al.* (2011) found no significant difference in survival rates for a common rodent, *Sigmodon hispidus*, between predator exclusion treatments and a control after prescribed fire. Foster *et al.* (2016) found that groundcover-reliant small mammals such as *Antechinus stuartii* were more abundant in herbivore exclusion plots after prescribed fire in Booderee National Park, New South Wales. In the same study, the introduced European rabbit (*Oryctolagus cuniculus*) was also more abundant where native macropods were excluded, suggesting the need for potential modifications to the exclusion fencing or the implementation of concurrent rabbit control.

Among the remaining untested interventions that were identified in this review, there are a range of potential post-fire strategies that should be formally tested in future. Lethal predator control, via baiting or trapping, is used in a range of other contexts and should be experimentally tested after future fires to quantify its efficacy. It has the potential to be a rapid post-fire response tool to manage predator activity after fire (Doherty *et al.*, 2023), as many land managers already have licensing and protocols for bait and trap deployment. While it is likely that this is already occurring in conservation reserves across Australia, only a handful of studies formally quantify the outcomes (Rees *et al.*, 2024).

Predator and herbivore exclusion also have the potential to benefit small mammals and reptiles after fire, but the resource requirements to deploy large-scale exclosures soon after fires may make it suitable only in targeted areas for threatened and vulnerable species. While Conner *et al.* (2011) did not find significant benefits for hispid cotton rats (*Sigmodon hispidus*) using electrified mesh fencing, this is only one example and predator exclusion should be tested in a variety of scenarios (i.e., threatened species, natural fire, etc.). Herbivore control may be valuable where livestock and conservation interests overlap, as livestock levels can be managed or removed completely while ground cover vegetation regenerates after fire. Where introduced herbivores, such as feral pigs and deer, threaten native vertebrate recovery after fire, post-fire landscapes may provide an opportunity for effective lethal control to reduce population numbers

(Pulsford *et al.*, 2022; Hamnett *et al.*, 2023). The importance of herbivore control may also vary with rainfall and rates of natural vegetation regeneration in the post-fire period.

Revegetation is unlikely to be necessary in fire-adapted landscapes when fire intervals are consistent with historical frequencies (i.e., what the system is adapted to). However, when fire frequency exceeds the ability of a system's dominant species to regenerate, regardless of their regeneration strategy, it may be necessary to implement reseeding or revegetation measures (Duivenvoorden *et al.*, 2024). This also applies to systems that have historically experienced infrequent or no fire but are now being burnt as fires become more widespread and severe (Bassett *et al.*, 2015b).

Although it was published after our systematic mapping was conducted, there has been a new study of post-fire artificial refuges after the Black Summer fires in Australia. Watchorn *et al.* (2024) found that tunnels, constructed of chicken wire and shade cloth, recorded increased activity compared to control (uncovered) areas for some small mammal and reptile species, as well as a few species of ground-dwelling birds.

5.7.2 Lessons from experimental coarse woody debris piles

Chapters 3 and 4 outlined a field experiment designed to test the efficacy of supplementary coarse woody debris piles as a post-fire intervention for small mammals and reptiles. Our results suggest there are four key takeaways that should be considered before this intervention is used after a future fire:

- In general, **supplementary CWD provided significant benefit for reptiles but not for small mammals**. However, CWD has been used successfully for small mammals in other disturbed landscapes (see Chapter 2), and we recommend that this method should be tested again after modifications to the CWD pile design have been considered. These modifications may include specialised design to suit the needs of target species and improved insulation (see further discussion below).
- There was **no significant difference between the benefits of the structure (CWD only) and structure + predation (CWD and chicken wire) for reptiles**, but the added cost of sourcing chicken wire led to the conclusion that **CWD piles without chicken wire were the best option**. In addition, the use of chicken wire may introduce additional animal ethics considerations (i.e., potential for bycatch), and incur additional removal costs at the end of the recovery period.
- **Supplementary CWD efficacy was not affected by fire severity or vegetation complexity**. Supplementary CWD supported significantly more reptiles across all areas burnt during the Orroral Valley Fire, demonstrating its potential as a widespread recovery tool.

- **Supplementary CWD piles are cheap and easy to build**. For the two most effective refuge types, costs for a set of four piles ranged from $0 (CWD only) to $53 (CWD + chicken wire) for materials and between 40 to 60 minutes of labour for a team of two people (CWD only and CWD + chicken wire, respectively). This provides a more accessible solution than other popular interventions, such as the addition of large CWD (e.g., Shorthouse *et al.* (2012)) or the construction of large predator exclusion fences (Clapperton and Day, 2001).

In addition to these points, it is important to note that our study was conducted 2-4 years after the Orroral Valley Fire during a period of above-average rainfall (Bureau of Meteorology, 2023). While we were limited by logistical constraints (e.g., COVID-19, funding timelines, etc.), it is possible the CWD piles would have had additional benefits if they were implemented immediately after the fire. For example, feral cat activity in burnt areas was found to be greatest in the first two months after fire, with a decline in activity after three months (McGregor *et al.*, 2017). The availability of structural habitat elements such as CWD piles may have provided valuable refuge for small mammals and reptiles during this period, even if it is not the preferred refuge of small mammals years after fire when the ground level vegetation has begun to recover.

The simplicity of the design and implementation of these supplementary CWD piles also provides an opportunity for collaboration with citizen science groups. After the Black Summer fires in 2019-2020 many concerned members of the public sought ways to help wildlife in burnt areas, often putting out bowls of food or water. While it is unclear whether these efforts had measurable benefits for species recovery, the supplementary CWD piles described here represent an evidence-based intervention that benefits native reptiles. With limited training, these structures have the potential to be implemented by community groups across burnt landscapes.

In Chapter 5 we assessed the thermal properties of supplementary CWD piles and compared them to the temperatures of natural refuges such as rock crevices and log cavities. The thermal properties of CWD piles were significantly more moderate than ambient temperature but did not buffer against maximum and minimum temperatures as effectively as rock crevices and log cavities. These results may help explain the responses of small mammals and reptiles to our CWD piles and provide further guidelines for future implementation.

Among other factors, the lack of a significant response of small mammals to supplementary CWD in our experiment may be due to cooler than ambient temperatures during the winter months. When the structures were disassembled for reptile surveys each year only one of the 160 piles showed evidence of nesting material by small mammals, suggesting these CWD piles did not generally provide suitable nesting substrate. Despite their poor thermal performance during the winter, the CWD piles succeeded at providing a buffer from ambient daytime maximum temperatures in summer. This may explain the difference between reptile results (focused on summer surveys) and small mammal results (year-round surveys).

This should not discourage the use of these CWD piles for reptiles after fire, but future research should focus on adapting the design to make the piles more suitable for small mammals, particularly in cooler climates. Potential adjustments include increasing the size of the CWD piles to increase structural complexity and insulative benefits. Examples from the literature indicate that CWD piles benefitting small mammals in other disturbed landscapes use larger piles than the ones constructed in our study (e.g., Arthur *et al.* (2005) and offcuts from timber production Sullivan *et al.* (2012)). Where CWD resources are limited and larger piles are not possible, adding specialised nesting substrate such as nest boxes or insulated PVC may improve the suitability of such refuges for small mammals. In addition, we aimed to create suitable refuge for a range of small mammal species in our study area, but a design based on target species ecology may yield better results.

5.7.3 Areas for further research

As our systematic map showed, this work represents a small part of what will hopefully become a well-studied field. There are a range of potentially suitable interventions that should be tested, with particular emphasis placed on designing experiments that collect data on how an intervention affects small mammal and reptile populations. For example, lethal herbivore control has been well-studied with regards to how different baiting or culling methods affect the target herbivore populations, but very few studies focus on how these impacts translate to sympatric small vertebrate species.

Future research should also seek to describe the conditions and fire characteristics that would require the use of supplementary habitat. For example, are there thresholds of fire severity or extent that would prevent small mammal and reptile populations from recovering naturally? Would the persistence of drought conditions after fire result in different recommendations than a year of above-average rainfall? Addressing these questions would help land managers further refine their decision making for post-fire recovery.

In addition to filling broader knowledge gaps, our experimental coarse woody debris results indicate that future applications of our design should be adjusted if they are intended for use by small mammals. Potential modifications include those that provide more suitable thermal conditions for small mammals in winter, including additional nesting opportunities such as PVC tubes and small nest boxes, and larger pile sizes.

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Appendix: Included studies for systematic mapping

Reference ID	Title
28850	Sympatric prey responses to lethal top-predator control: Predator manipulation experiments
34878	Impact of mining and forest regeneration on small mammal biodiversity in the western region of Ghana
36077	Landscape context affects use of restored grasslands by mammals in a dynamic agroecosystem
22222	Testing the effects of deer grazing on two woodland rodents, bankvoles and woodmice
23741	Woodland recovery after suppression of deer: Cascade effects for small mammals, wood mice (Apodemus sylvaticus) and bank voles (Myodes glareolus)
23165	The effects of deer exclosures on voles and shrews in two forest habitats
20816	Trends in the activity of forest-dwelling vertebrate fauna against a background of intensive baiting for foxes
43022	Integrating feral cat (Felis catus) control into landscape-scale introduced predator management to improve conservation prospects for threatened fauna: A case study from the south coast of Western Australia
22198	Effects of mesopredators and prescribed fire on hispid cotton rat survival and cause-specific mortality
28537	Does coarse woody debris density and volume influence the terrestrial vertebrate community in restored bauxite mines?
34867	Can postmining revegetation create habitat for a threatened mammal?
20810	Using artificial rocks to restore nonrenewable shelter sites in human-degraded systems: Colonization by fauna
25703	The benefits of habitat restoration for rock-dwelling velvet geckos Oedura lesueurii
15967	Reptile and arboreal marsupial response to replanted vegetation in agricultural landscapes
20464	Influence of coarse woody debris on herpetofaunal communities in upland pine stands of the southeastern Coastal Plain
20959	Influence of coarse woody debris on the soricid community in southeastern Coastal Plain pine stands
44841	Experimentally testing the response of feral cats and their prey to poison baiting
39327	Changes in small native animal populations following control of European rabbits (Oryctolagus cuniculus) by warren ripping in the Australian arid zone
23918	An ideal combination for marine turtle conservation: Exceptional nesting season, with low nest predation resulting from effective low-cost predator management
40216	A long-term experiment reveals strategies for the ecological restoration of reptiles in scattered tree landscapes
32159	Herbivory and fire interact to affect forest understory habitat, but not its use by small vertebrates
26066	Interactions of grazing history, cattle removal and time since rain drive divergent short-term responses by desert biota
44844	Prickly skink (Gnypetoscincus queenslandiae) use of supplemental coarse woody debris in rainforest restoration sites
12337	Short-term grazing exclusion effects on riparian small mammal communities
39440	Predator control on farmland for biodiversity conservation: A case study from Hawke's Bay, New Zealand
20734	Rabbit burrows or artificial refguges area critical habitat component for the threatened lizard, Timon lepidus (Sauria, Lacertidae)
16337	Attempting to manage complex predator-prey interactions fails to avert imminent extinction of a threatened New Zealand skink population
36565	Recovery of reptile, amphibian and mammal assemblages in Australian post-mining landscapes following open-cut coal mining
39424	Variable reptile responses to introduced predator control in southern Australia
29126	Reptiles in restored agricultural landscapes: The value of linear strips, patches, and habitat condition
40254	Restoring apex predators can reduce mesopredator abundances
44939	Assessing the value of restoration plantings for wildlife in a temperate agricultural landscape
23469	Signals of change in tropical savanna woodland vertebrate fauna 5 years after cessation of livestock grazing
13819	Restoration of degraded Chaco woodlands: Effects on reptile assemblages

25461	Bringing forward hte benefits of coarse woody debris in ecosystem recovery under different levels of grazing and vegetation density
19452	Recovering the reptile community after the mine-tailing accident of aznalcollar (Southwestern Spain)
27217	Determining the effects of cattle grazing treatments on Yosemite toads (Anaxyrus [=Bufo] canorus) in montane meadows
15023	Effects of structural complexity enhancement on eastern red-backed salamander (Plethodon cinereus) populations in northern hardwood forests
31034	Population density of Dall's sheep in Alaska: effects of predator harvest?
17508	Soricid response to coarse woody debris manipulations in Coastal Plain loblolly pine forests
45028	Converting rangelands to reserves: Small mammal and reptile responses 24 years after domestic livestock grazing and removal
2608	Reptile and frog utilisation of rehabilitated bauxite miesites and dieback-affected sites in Western Australia's jarrah eucalyptus marginata forest
18560	Shrew species diversity and abundance in Ziama biosphere reserve, Guinea: Comparison among primary forest, degraded forest, and restoration plots
5164	Changes in the vegetation and reptile populations on Round Island, Mauritius, following eradication of rabbits
18407	Amphibian and reptile community response to coarse woody debris manipulations in upland loblolly pine (Pinus taeda) forests
17378	The importance of understanding spatial population structure when evaluating the effects of silviculture on spotted salamanders (Ambystoma maculatum)
15272	Effects of experimental forestry treatments on a Maine amphibian community
45642	Restoration of gallery forest patches improves recruitment of motacu palms (Attalea princeps) while diversifying and increasing wildlife population
24107	Predator control allows critically endangered lizards to recover on mainland New Zealand
8690	The impact of cats and foxes on the small vertebrate fauna of Heirisson Prong, Western Australia. II. A field experiment
29141	Long-term and large-scale control of the introduced red fox increases native mammal occupancy in Australian forests
43019	Long-term benefits and short-term costs: small vertebrate responses to predator exclusion and native mammal reintroductions in south-western New South Wales, Australia
38039	Use of constructed coarse woody debris corridors in a clearcut by American martens (Martes americana) and their prey
28166	Putting it back: Woody debris in young restoration plantings to stimulate return of reptiles
29556	Fauna community trends during early restoration of alluvial open forest/woodland ecosystems on former agricultural land
26827	Effects of predator exclusion on nest and hatchling survival in the gopher tortoise
41112	If we build it, will they come? Use of restored dunes by beach mice
37907	Rapid increase of Australian tropical savanna reptile abundance following exclusion of feral cats
46443	Responses of mustelids and small mammal prey to combined retention on clearcuts: woody debris, green trees, and riparian structures
23630	Woody debris, voles, and trees: incluence of habitat structures (piles and windrows) on long-tailed vole populations and feeding damage
25068	If we build habitat, will they come? Woody debris structures and conservation of forest mammals
14845	Survival of grand and Otago skinks following predator control
32672	The relative influence of in situ and neighborhood factors on reptile recolonization in post-mining restoration sites
4289	Recolonization of regenerating open forest by terrestrial lizards following sand mining
8736	Paving the way for habitat restoration: Can artificial rocks restore degraded habitats of endangered reptiles?
27563	Use by small mammals of a chronosequence of tropical rainforest revegetation
16903	Sampling skinks and geckos in artificial cover objects in a dry mixed grassland-shrubland with mammalian predator control
32184	Logging residues conserve small mammalian diversity in a Malaysian production forest
29062	Fidelity of northern pine snakes (Pituophis m. Melanoleucus) to natural and artificial hibernation sites in the New Jersey Pine Barrens

Author	Year published	DOI	Publication type	Continent	Study country	Study region	Site name	Study latitude
Allen, B.L.; Allen, L.R.; I	2014	10.1186/s12983-014-0056-y	journal article	Oceania	Australia	Queensland	Strathmore, Mt. Owen, Cordillo Downs, Quinyambie, Todmorden, Lar	unknown
Allen, B.L.; Allen, L.R.; I								
Allen, B.L.; Allen, L.R.; I								
Allen, B.L.; Allen, L.R.; I								
Attuquayefio, D.K.; Ow	2017	10.1007/s10661-017-5960-0	journal article	Africa	Ghana	Wassa West District	Ben West Block Forest Reserve, Bonsa River Forest Reserve	4° N
Berry, B.; Schooley, R.L	2017	10.1674/0003-0031-177.2.165	journal article	North America	United States	Illinois	Decatur	39°50' 25" N
Buesching, C.D.; Newm	2011	10.1016/j.baae.2011.02.007	journal article	Europe	United Kingdom	Oxford	Wytham Woods	51° 46' 26" N
Bush, E.R.; Buesching,	2012	10.1371/journal.pone.0031404	journal article	Europe	United Kingdom	Oxfordshire	Wytham Woods	unknown
Byman, D.	2011	10.1656/045.018.0408	journal article	North America	United States	Pennsylvania	Lacawac Sanctuary	41° 23' N
Claridge, A.W.; Cunnin	2010	10.1016/j.foreco.2010.05.041	journal article	Oceania	Australia	New South Wales	Ben Boyd National Park, Nadgee Nature Reserve, South-East Forest N	unknown
Comer, S.; Clausen, L.;	2020	10.1071/WR19217	journal article	Oceania	Australia	Western Australia	TPB Nature Reserve, Cape Arid Natinal Park, Nuytsland Nature Reserv	unknown
Conner, L.M.; Castlebe	2011	10.1002/jwmg.110	journal article	North America	United States	Georgia	Joseph W. Jones Ecological Research Centre	unknown
Craig, M.D.; Grigg, A.H.	2014	10.1016/j.foreco.2014.01.011	journal article	Oceania	Australia	Western Australia	Alcoa's Huntly mine	32° 36' S
Craig, M.D.; Grigg, A.H.								
Craig, M.D.; Grigg, A.H.								
Craig, M.D.; White, D.A	2017	10.1111/emr.12258	journal article	Oceania	Australia	Western Australia	Huntly minesite	32° 34' S
Croak, B.M.; Pike, D.A.;	2010	10.1111/j.1526-100X.2008.00476.x	journal article	Oceania	Australia	New South Wales	Nowra	35° 0' 47" S
Croak, B.M.; Webb, J.K	2013	10.1111/1365-2664.12040	journal article	Oceania	Australia	New South Wales	Dharawal National Park	unknown
Cunningham, R.B.; Linc	2007	10.1890/05-1892	journal article	Oceania	Australia	New South Wales	Junee, Albury, Gundagai, Howlong	6140128 N
Davis, J.C.; Castleberry	2010	10.1016/j.foreco.2009.12.024	journal article	North America	United States	South Carolina	Savannah River Site	unknown
Davis, J.C.; Castleberry								
Davis, J.C.; Castleberry	2010	10.1644/09-MAMM-A-170.1	journal article	North America	United States	South Carolina	Savannah River Site	unknown
Davis, J.C.; Castleberry								
Doherty, T.S.; Hall, M.L	2021	10.1071/WR21008	journal article	Oceania	Australia	Western Australia	Charles Darwin Nature Reserve	29.65 S
Doherty, T.S.; Hall, M.L								
Elsworth, P.; Berman, I	2019	10.1071/WR18088	journal article	Oceania	Australia	Queensland	Bulloo Downs Station	unknown
Elsworth, P.; Berman, I								
Engeman, R.; Martin, R	2012	10.1017/S0030605311000020	journal article	North America	United States	Florida	Jupiter Island	unknown
Evans, M.J.; Newport, .	2019	10.1007/s10531-019-01798-5	journal article	Oceania	Australia	Australian Capital Territory	Mulligans Flat and Goorooyaroo Nature Reserves	unknown
Evans, M.J.; Newport, .								
Foster, C.N.; Barton, P.	2016	10.1111/acv.12210	journal article	Oceania	Australia	New South Wales	Booderee National Park	35° 10' S
Foster, C.N.; Barton, P.								
Frank, A.S.K.; Dickman,	2013	10.1371/journal.pone.0068466	journal article	Oceania	Australia	Simpson Desert	Carlo Station, Cravens Peak Reserve	23° 29' S
Frank, A.S.K.; Dickman,								
Freeman, A.N.D.; Freer	2021	10.1111/emr.12500	journal article	Oceania	Australia	Queensland	Cloudland Nature Refuge, Ringtail Crossing Nature Refuge, Rock Road	17° 26' S
Giuliano, W.M.; Homya	2004	10.2307/4003857	journal article	North America	United States	Pennsylvania	Pike Run watershed	40° 5' 10" N
Glen, A.S.; Perry, M.; Y	2019	10.20417/nzjecol.43.8	journal article	Oceania	New Zealand	Hawke's Bay	Opouahi, Rangiora, Toronui and Rimu Stations	39° 10' S
Grillet, P.; Cheylan, M.;	2010	10.1007/s10531-010-9824-y	journal article	Europe	France	Atlantic Coast	Oleron Island	45° 59' 35" N
Hoare, J.M.; Adams, L.I	2007	10.2193/2006-488	journal article	Oceania	New Zealand	North Island	Pukerua Bay Scientific Reserve	unknown
Hoare, J.M.; Adams, L.I								
Houston, W.A.; Melzer	2018	unknown	journal article	Oceania	Australia	Queensland	Blair Athol, Newlands, Callide	22° 40.846' S
Houston, W.A.; Melzer								
Houston, W.A.; Melzer								
Hu, Y.; Gillespie, G.; Je	2019	10.1071/WR18047	journal article	Oceania	Australia	Victoria	Cape Conran Coastal Park, Murrungowar State Forest	37° 37' S
Jellinek, S.; Parris, K.M.	2014	10.1111/acv.12121	journal article	Oceania	Australia	Victoria	Benalla, Wimmera	unknown
Jimenez, J.; Nunez-Arjc	2019	10.1016/j.biocon.2019.108234	journal article	Europe	Spain	Badajoz	Matachel river valley	unkonwn
Kittipalawattanapol, K.	2021	10.1111/rec.13470	journal article	Oceania	Australia	Tasmania	Midlands Biodiversity Hotspot	unknown
Kutt, A.S.; Vanderduys,	2012	10.1071/WR11137	journal article	Oceania	Australia	Queensland	Brooklyn Wildlife Sanctuary	16° 36' S
Kutt, A.S.; Vanderduys,								
Leynaud, G.C.; Bucher,	2005	10.1016/j.foreco.2005.04.003	journal article	South America	Argentina	Salta	Los Colorados Biological Station, Campo Grande cattle ranch	24° 43' S

Manning, A.D.; Cunnin	2013	10.1016/j.biocon.2012.06.028	journal article	Oceania	Australia	Australian Capital Territory	Mulligans Flat and Goorooyarroo Nature Reserves	unknown
Manning, A.D.; Cunnin								
Marquez-Ferrando, R.;	2009	10.1111/j.1526-100X.2008.00404	journal article	Europe	Spain	Sierra Morena	Guadiamar Green Corridor	unknown
McIlroy, S.K.; Lind, A.J.	2013	10.1371/journal.pone.0079263	journal article	North America	United States	California	Stanislaus National Forest, Sierra National Forest	38° N
McKenny, H.C.; Keeton	2006	10.1016/j.foreco.2006.04.034	journal article	North America	United States	Vermont	Mount Mansfield State Forest, Jericho Research Forest	unknown
Mitchell, C.D.; Chaney,	2015	10.1007/s13364-014-0199-4	journal article	North America	United States	Alaska	Chisana and Ptarmigan Lake	62° 0' N
Moseley, K.R.; Owens,	2008	10.1016/j.foreco.2007.12.043	journal article	North America	United States	South Carolina	Savannah River Site	33° 0' N
Neilly, H.; Ward, M.; Ca	2021	10.1111/aec.13047	journal article	Oceania	Australia	South Australia	Calperum Station	unknown
Neilly, H.; Ward, M.; Ca								
Nichols, O.G.; Bamford	1985	10.1016/0006-3207(85)90094-1	journal article	Oceania	Australia	Western Australia	Jarrahdale, Del Park, Huntly and WIllowdale mines	unknown
Nicolas, V.; Barriere, P.	2009	10.1007/s10531-008-9572-4	journal article	Africa	Guinea	south-eastern Guinea	Ziama forest	unknown
North, S.G.; Bullock, D.	1994	10.1016/0006-3207(94)90004-3	journal article	Africa	Mauritius	Mascarene Island Group	Round Island	unknown
Owens, A.K.; Moseley,	2008	10.1016/j.foreco.2008.07.030	journal article	North America	United States	South Carolina	Savannah River Site	33° 25' N
Owens, A.K.; Moseley,								
Patrick, D.A.; Calhoun,	2008	10.1016/j.biocon.2007.12.026	journal article	North America	United States	Maine	Dwight B. Demeritt and Penobscot Experimental Forests	44° N
Patrick, D.A.; Hunter Jr	2006	10.1016/j.foreco.2006.07.015	journal article	North America	United States	Maine	Dwight D. Demeritt forest, Penobscot forest	unknown
Peacock, J.; Tonra, C.M	2021	10.1371/journal.pone.0250183	journal article	South America	Bolivia	Beni	Barba Azul Nature Reserve	13° 45' S
Reardon, J.T.; Whitmor	2012		journal article	Oceania	New Zealand	Otago	Macraes Flat	45° 25' S
Reardon, J.T.; Whitmor								
Risbey, D.A.; Calver, M	2000	10.1071/WR98092	journal article	Oceania	Australia	Western Australia	Heirisson Prong	26° 10' S
Risbey, D.A.; Calver, M								
Risbey, D.A.; Calver, M								
Risbey, D.A.; Calver, M								
Robley, A.; Gormley, A	2014	10.1016/j.biocon.2014.10.017	journal article	Oceania	Australia	Victoria	Lower Glenelg National Park, Cobboboonee National Park. Mount Cla	38° 7' 50" S
Roshier, D.A.; Hotellier	2020	10.1071/WR19153	journal article	Oceania	Australia	New South Wales	Scotia Wildlife Sanctuary	33.15° S
Roshier, D.A.; Hotellier								
Roshier, D.A.; Hotellier								
Roshier, D.A.; Hotellier								
Roshier, D.A.; Hotellier								
Roshier, D.A.; Hotellier								
Roshier, D.A.; Hotellier								
Roshier, D.A.; Hotellier								
Seip, C.R.; Hodder, D.P	2018	10.1093/forestry/cpy010	journal article	North America	Canada	British Columbia	John Prince Research Forest	54° 36.417' N
Shoo, L.P.; Wilson, R.; \	2014	10.1111/emr.12079	journal article	Oceania	Australia	Queensland	Cloudland Nature Refuge, Ringtail Crossing Nature Refuge, Rock Road	17° 26' S
Smith, G.C.; Lewis, T.; I	2015	10.1111/rec.12269	journal article	Oceania	Australia	Queensland	Netherleigh, Futter Creek	24.2213 S
Smith, G.C.; Lewis, T.; I								
Smith, G.C.; Lewis, T.; I								
Smith, L.L.; Steen, D.A.	2013	10.1002/jwmg.499	journal article	North America	United States	Georgia	Ichauway	31° 13' 16.88" N
Stoddard, M.A.; Miller,	2019	10.1111/rec.12892	journal article	North America	United States	Florida	Gulf Islands National Seashore, Perdido Key	unknown
Stokeld, D.; Fisher, A.;	2018	10.1016/j.biocon.2018.06.025	journal article	Oceania	Australia	Northern Territory	Kapalga, Kakadu National Park	12° 38.97' S
Sullivan, T.P.; and Sulli	2021	10.1016/j.foreco.2021.119431	journal article	North America	Canada	British Columbia	Summerland	49° 40' N
Sullivan, T.P.; Sullivan,	2012	10.1016/j.foreco.2011.09.001	journal article	North America	Canada	British Columbia	Summerland	49° 40' N
Sullivan, T.P.; Sullivan,	2012	10.1644/11-MAMM-A-250.1	journal article	North America	Canada	British Columbia	China Valley	50° 44' N
Toucher, M.D.	2006	10.2193/0022-541X(2006)70	journal article	Oceania	New Zealand	Otago	Macraes	unknown
Triska, M.D.; Criag, M.I	2016	10.1111/rec.12340	journal article	Oceania	Australia	Western Australia	Alcoa's Huntly Mine	32° 36' S
Twigg, L.E.; Fox, B.J.	1991	10.1111/j.1442-9993.1991.tb01041.x	journal article	Oceania	Australia	New South Wales	Myall Lakes National Park	unknown
Webb, J.K.; Shine, R.	2000	10.1016/S0006-3207(99)00056-7	journal article	Oceania	Australia	New South Wales	Morton National Park	unknown
Whitehead, T.; Goosen	2014	10.1071/WR14082	journal article	Oceania	Australia	Queensland	Atherton Tablelands	unknown
Wilson, D.J.; Mulvey, R	2007		journal article	Oceania	New Zealand	Otago	Macraes Flat	45° 27' S
Wilson, D.J.; Mulvey, R								
Yamada, T.; Yoshida, S.	2016	10.1016/j.biocon.2015.12.004	journal article	Asia	Malaysia	Perak	Temengor Forest Reserve	5° 24' N
Zappalorti, R.T.; Burger	2014	10.1080/15287394.2014.934497	journal article	North America	United States	New Jersey	Stafford Forge Wildlife Management Area	unknown

Study longitude	Vertebrate group	IUCN habitat type	IUCN Disturbance type	Intervention type	Barrier being mitigated	Experiment type	Comparator
unknown	medium/large mammal	artificial (terrestrial)	livestock farming and ranching	predator control	predation	Control/Impact	sites with no predator control
	small mammal	artificial (terrestrial)	livestock farming and ranching	predator control	predation	Control/Impact	sites with no predator control
	reptile	artificial (terrestrial)	livestock farming and ranching	predator control	predation	Control/Impact	sites with no predator control
	amphibian	artificial (terrestrial)	livestock farming and ranching	predator control	predation	Control/Impact	sites with no predator control
1° 45' W	small mammal	forest	energy production and mining	revegetation	structural complexity	Control/Impact	unmined forest, naturally regenerating forest
88° 57' 17" W	medium/large mammal	grassland	annual and perennial non-timber crops	revegetation	structural complexity	After Only	N/A
1° 19' 19" W	small mammal	forest	invasive/overabundant species	herbivore exclusion	structural complexity	Control/Impact	areas outside herbivore exclusion
unknown	small mammal	forest	invasive/overabundant species	herbivore control	structural complexity	Control/Impact	area within herbivore exclusion plot
75° 17" W	small mammal	forest	invasive/overabundant species	herbivore exclusion	structural complexity	Control/Impact	outside herbivore exclosure
unknown	small mammal	forest	invasive/overabundant species	predator control	predation	Control/Impact	area without predator control
unknown	small mammal	shrubland	invasive/overabundant species	predator control	predation	After only	none
unknown	small mammal	forest	fire and fire suppression	predator exclusion	predation	Before/After/Control/Impact	before fire, outside predator exclusion
116° 6' E	amphibian	forest	energy production and mining	CWD manipulation	habitat availability	Control/Impact	unmined forest (no CWD added)
	reptile	forest	energy production and mining	CWD manipulation	habitat availability	Control/Impact	unmined forest (no CWD added)
	small mammal	forest	energy production and mining	CWD manipulation	habitat availability	Control/Impact	unmined forest (no CWD added)
116° 6' E	small mammal	forest	energy production and mining	revegetation	structural complexity	Control/Impact	unmined forest
150° 17' 19" E	reptile	rocky areas	biological resource use	artificial refuge	habitat availability	After only	none
unknown	reptile	rocky areas	biological resource use	artificial refuge	habitat availability	Control/Impact	unrestored sites (no artificial refuges)
0552952 E	reptile	forest	livestock farming and ranching	revegetation	structural complexity	Control/Impact	natural regeneration, remnant patches
unknown	reptile	forest	wood and pulp plantations	CWD manipulation	habitat availability	Control/Impact	sites with no CWD added
	amphibian	forest	wood and pulp plantations	CWD manipulation	habitat availability	Control/Impact	sites with no CWD added
unknown	small mammal	forest	wood and pulp plantations	CWD manipulation	habitat availability	Control/Impact	sites with no CWD added
	reptile	forest	wood and pulp plantations	CWD manipulation	habitat availability	Control/Impact	sites with no CWD added
116.97 E	small mammal	forest	invasive/overabundant species	predator control	predation	Before/After/Control/Impact	before predator control, no predator control
	reptile	forest	invasive/overabundant species	predator control	predation	Before/After/Control/Impact	before predator control, no predator control
unknown	small mammal	desert	livestock farming and ranching	herbivore control	structural complexity	Before/After/Control/Impact	sites without warren ripping vs. sites with warren
	reptile	desert	livestock farming and ranching	herbivore control	structural complexity	Before/After/Control/Impact	sites without warren ripping, sites before treatme
unknown	reptile	marine coastal	invasive/overabundant species	predator control	predation	Before/After	previous predator control methods, no predator c
unknown	reptile	forest	livestock farming and ranching	CWD manipulation	habitat availability	Control/Impact	(1) dispersed CWD, (2) clumped CWD, (3) disperse
	reptile	forest	livestock farming and ranching	herbivore control	structural complexity	Control/Impact	low and high kangaroo density
150° 40' E	small mammal	forest	fire and fire suppression*	herbivore exclusion	structural complexity	Before/After/Control/Impact	sites with no herbivore exclusion, sites with partia
	reptile	forest	fire and fire suppression*	herbivore exclusion	structural complexity	Before/After/Control/Impact	sites with no herbivore exclusion, sites with partia
138° 32' E	small mammal	grassland	livestock farming and ranching	herbivore exclusion	structural complexity	Control/Impact	site with herbivores present
	reptile	grassland	livestock farming and ranching	herbivore exclusion	structural complexity	Control/Impact	site with herbivores present
145° 20' E	reptile	forest	livestock farming and ranching	CWD manipulation	habitat availability	Control/Impact	sites with no CWD added
79° 58' 45" W	small mammal	artificial (terrestrial)	livestock farming and ranching	herbivore exclusion	structural complexity	Control/Impact	sites outside herbivore exclusion
176° 46' E	reptile	artificial (terrestrial)	livestock farming and ranching	predator control	predation	Control/Impact	sites with no predator control
1° 20' 45" W	reptile	marine coastal	other (keystone species decline)	artificial refuge	habitat availability	Before/After	before refuges were installed
unknown	reptile	forest	livestock farming and ranching	herbivore exclusion	structural complexity	Before/After	before herbivore removal
	reptile	forest	invasive/overabundant species	predator control	predation	Before/After	before predator control
147° 31.877' E	reptile	forest	energy production and mining	revegetation	structural complexity	Control/Impact	unmined reference sites
	amphibian	forest	energy production and mining	revegetation	structural complexity	Control/Impact	unmined reference sites
	mammal	forest	energy production and mining	revegetation	structural complexity	Control/Impact	unmined reference sites
148° 44' E	reptile	forest	invasive/overabundant species	predator control	predation	Control/Impact	sites without predator control
unknown	reptile	artificial (terrestrial)	annual and perennial non-timber crops	revegetation	structural complexity	Control/Impact	(1) enlarged patches of remnant vegetation, (2) re
unknown	small mammal	artificial (terrestrial)	invasive/overabundant species	predator control	predation	Before/After/Control/Impact	area unoccupied by lynx, before area occupied by
unknown	mammal	forest	annual and perennial non-timber crops	revegetation	structural complexity	Control/Impact	agricultural sites, remnant patches
145° 15' E	reptile	forest	livestock farming and ranching	herbivore exclusion	structural complexity	After only	none
	mammal	forest	livestock farming and ranching	herbivore exclusion	structural complexity	After only	none
63° 17' W	reptile	forest	livestock farming and ranching	herbivore exclusion	structural complexity	Control/Impact	sites without herbivore removal (continued grazin

unknown	reptile	forest	livestock farming and ranching	CWD manipulation	habitat availability	Control/Impact	(1) dispersed CWD, (2) clumped CWD, (3) disperse
	reptile	forest	livestock farming and ranching	herbivore control	structural complexity	Control/Impact	low and high kangaroo density
unknown	reptile	forest	energy production and mining	CWD manipulation	habitat availability	Control/Impact	sites with no CWD added
120° W	amphibian	grassland	livestock farming and ranching	herbivore exclusion	structural complexity	Control/Impact	(1) grazing, (2) exclusion around breeding sites, (3
unknown	amphibian	forest	wood and pulp plantations	CWD manipulation	habitat availability	Control/Impact	unharvested area with no CWD piles, harvested ar
141° 25' W	medium/large mammal	forest	invasive/overabundant species	predator control	predation	Control/Impact	areas with no predator control
81° 25' W	small mammal	forest	wood and pulp plantations	CWD manipulation	habitat availability	Control/Impact	CWD removal, CWD increased, control
unknown	small mammal	artificial (terrestrial)	livestock farming and ranching	herbivore control	structural complexity	After Only	none
	reptile	artificial (terrestrial)	livestock farming and ranching	herbivore control	structural complexity	After Only	none
unknown	reptile	forest	energy production and mining	revegetation	structural complexity	Control/Impact	unmined forest
unknown	small mammal	forest	annual and perennial non-timber crops	revegetation	structural complexity	Control/Impact	recently abandoned fields, cultivated fields, prima
unknown	reptile	savanna	invasive/overabundant species	herbivore control	structural complexity	Before/After	before rabbit eradication
81° 50' W	reptile	forest	wood and pulp plantations	CWD manipulation	habitat availability	Control/Impact	sites with no CWD added
	amphibian	forest	wood and pulp plantations	CWD manipulation	habitat availability	Control/Impact	sites with no CWD added
68° W	amphibian	forest	wood and pulp plantations	CWD manipulation	structural complexity	Control/Impact	control, partial cut, clearcut with CWD removed, c
unknown	amphibian	forest	wood and pulp plantations	CWD manipulation	structural complexity	Control/Impact	clearcut with no CWD retained, uncut forest
66° 7' W	medium/large mammal	savanna	livestock farming and ranching	herbivore exclusion	structural complexity	Control/Impact	sites outside grazing exclusion plot
170° 28' E	reptile	shrubland	invasive/overabundant species	predator exclusion	predation	Control/Impact	sites outside exclosure
	reptile	shrubland	invasive/overabundant species	predator control	predation	Control/Impact	sites without predator control
113° 23' E	small mammal	unknown	invasive/overabundant species	predator control	predation	Before/After/Control/Impact	(1)foxes controlled, cats uncontrolled; (2) foxes ar
	small mammal	unknown	invasive/overabundant species	predator exclusion	predation	Before/After/Control/Impact	foxes and cats not excluded
	reptile	unknown	invasive/overabundant species	predator control	predation	Before/After/Control/Impact	(1)foxes controlled, cats uncontrolled; (2) foxes ar
	reptile	unknown	invasive/overabundant species	predator exclusion	predation	Before/After/Control/Impact	foxes and cats not excluded
147° 37' 45" E	small mammal	forest	invasive/overabundant species	predator control	predation	Control/Impact	area without predator control
141.06° E	small mammal	grassland	invasive/overabundant species	predator exclusion	predation	Control/Impact	(1) sites with predator and herbivore exclusion, (2
	small mammal	grassland	invasive/overabundant species	predator control	predation	Control/Impact	(1) sites with predator and herbivore exclusion, (2
	small mammal	grassland	invasive/overabundant species	herbivore exclusion	structural complexity	Control/Impact	(1) sites with predator and herbivore exclusion, (2
	small mammal	grassland	invasive/overabundant species	herbivore control	structural complexity	Control/Impact	(1) sites with predator and herbivore exclusion, (2
	reptile	grassland	invasive/overabundant species	predator exclusion	predation	Control/Impact	(1) sites with predator and herbivore exclusion, (2
	reptile	grassland	invasive/overabundant species	predator control	predation	Control/Impact	(1) sites with predator and herbivore exclusion, (2
	reptile	grassland	invasive/overabundant species	herbivore exclusion	structural complexity	Control/Impact	(1) sites with predator and herbivore exclusion, (2
	reptile	grassland	invasive/overabundant species	herbivore control	structural complexity	Control/Impact	(1) sites with predator and herbivore exclusion, (2
124° 21.237' W	small mammal	forest	wood and pulp plantations	CWD manipulation	habitat availability	Control/Impact	clearcut with no CWD added
145° 20' E	reptile	forest	livestock farming and ranching	CWD manipulation	habitat availability	Control/Impact	sites with no CWD added
151.2643 E	reptile	forest	annual and perennial non-timber crops	revegetation	structural complexity	Control/Impact	agricultural sites, remnant patches
	small mammal	forest	annual and perennial non-timber crops	revegetation	structural complexity	Control/Impact	agricultural sites, remnant patches
	medium/large mammal	forest	annual and perennial non-timber crops	revegetation	structural complexity	Control/Impact	agricultural sites, remnant patches
84° 28' 37.81" W	reptile	forest	invasive/overabundant species	predator exclusion	predation	Control/Impact	sites outside exclosure
unknown	small mammal	marine coastal	residential development	revegetation	structural complexity	Control/Impact	natural dunes
132° 22.47' E	reptile	savanna	invasive/overabundant species	predator exclusion	predation	Before/After/Control/Impact	outside predator exclusion
119° 53' W	small mammal	forest	wood and pulp plantations	CWD manipulation	habitat availability	Control/Impact	(1) dispersed CWD only, (2) linear CWD piles, (3) ri
119° 53' W	small mammal	forest	wood and pulp plantations	CWD manipulation	habitat availability	Control/Impact	dispersed CWD (no piles)
119° 28' W	small mammal	forest	wood and pulp plantations	CWD manipulation	habitat availability	Control/Impact	(1) dispersed CWD, (2) CWD piles, (3) CWD rows, (
unknown	reptile	grassland	invasive/overabundant species	predator control	predation	Before/After/Control/Impact	sites with no predator control
116° 6' E	reptile	forest	energy production and mining	revegetation	structural complexity	Control/Impact	unmined forest
unknown	reptile	marine coastal	energy production and mining	revegetation	structural complexity	Control/Impact	unmined dunes
unknown	reptile	rocky areas	biological resource use	artificial refuge	habitat availability	After only	N/A
unknown	small mammal	forest	annual and perennial non-timber crops	revegetation	structural complexity	Control/Impact	abandoned grazing pasture
170° 26' E	reptile	grassland	annual and perennial non-timber crops	predator control	predation	Control/Impact	sites with no predator removal
	reptile	grassland	annual and perennial non-timber crops	predator exclusion	predation	Control/Impact	sites with no exclosure
101° 33' E	small mammal	forest	wood and pulp plantations	CWD manipulation	structural complexity	Control/Impact	sites with CWD removed after harvesting
unknown	reptile	forest	residential development	artificial refuge	habitat availability	Control/Impact	natural refuges

Replication/number of disturbed sites	Outcome metrics measured	Time in use	Time studied	Time after disturbance	Scale of implementation	Secondary impacts
6 x predator control, 3 no predator control	abundance	10	5	0	landscape	mesopredator abundances unaffected by apex predator baiting
7 x predator control, 3 no predator control	abundance	10	5	0	landscape	mesopredator abundances unaffected by apex predator baiting
8 x predator control, 3 no predator control	abundance	10	5	0	landscape	mesopredator abundances unaffected by apex predator baiting
9 x predator control, 3 no predator control	abundance	10	5	0	landscape	mesopredator abundances unaffected by apex predator baiting
1x each treatment	abundance	10	0.5	10	site	unknown
30 grassland sites	occupancy	3	0.75	0	site	unknown
4x each treatment	density	12	2	0	site	significant reduction of bushes and shrubs in control vs. herbivore exclus
3x each treatment	abundance, survival	21	0.5	0	site	unknown
2 exclosures, 2 control sites	abundance	1	9	0	site	unknown
2 predator control areas, one non-baited area	abundance	0	9	0	landscape	unknown
3 sites	abundance	0	8	0	landscape	unknown
4x each treatment	weekly survival rates	0	1	0	site	raptor-caused mortality was greater in predator exclosures than controls
12x treatment, 4x control	abundance	3	0.5	0	site	unknown
12x treatment, 4x control	abundance	3	0.5	0	site	unknown
12x treatment, 4x control	abundance	3	0.5	0	site	unknown
4 revegetated sites, 4 unmined sites	occupancy	16	0.1	21	landscape	unknown
198 artificial rocks	occupancy/colonisation	0	1	0	site	unknown
6x each treatment	abundance, survival rate	0	2	0	site	unknown
unsure	species richness	7	0.1	0	landscape	marsupials also less likely to occur in landscapes with more plantings vs.
3 x each treatment	abundance, species richness, sp	0	3	45	site	unknown
4 x each treatment	abundance, species richness, sp	0	3	45	site	unknown
3 x each treatment	abundance	6	1.5	45	site	unknown
4 x each treatment	abundance	6	1.5	45	site	unknown
1 x each treatment	abundance	0	6	0	landscape	feral cat occupancy only slightly decreased after baiting
1x each treatment	abundance	9	6	0	landscape	feral cat occupancy only slightly decreased after baiting
4x each treatment	abundance	0	2	0	site	unknown
4x each treatment	abundance	0	2	0	site	unknown
5.3 km of beach	number of nests, mean clutch s	0	0.25	0	landscape	first year Kemp's ridley turtles nested in the area
18 x each treatment	species richness, abundance	1	8	0	site	unknown
12x each treatment	species richness, abundance	1	8	0	site	unknown
4 x each treatment	abundance	0	2	0	site	control of macropods may favour rabbits
4x each treatment	abundance	0	2	0	site	control of macropods may favour rabbits
1 station with each treatment	abundance	0	2.5	0	landscape	Cattle removal had positive impact on vegetation composition
1 station with each treatment	abundance	0	2.5	0	landscape	Cattle removal had positive impact on vegetation composition
4 sites with 4 control and 12 CWD manipulation tr	abundance, species richness	6.5	0.1	7	site	unknown
9x each treatment	abundance, species richness	2	0.3	0	site	greater litter cover and vegetation height at exclusion plots
2x each treatment	occupancy	0	5	0	landscape	reduced predator density/occupancy, impact on invertebrate abundance
34 artificial refuges	occupancy	0.5	3	0	site	unknown
one monitoring grid	abundance	0	22	0	landscape	seeding grass cover increased after cattle removal and may have led to ir
one monitoring grid	abundance	0	22	0	landscape	seeding grass cover increased after cattle removal and may have led to ir
10x revegetation, 11 unmined	species richness, community co	7	1	0	landscape	unknown
10x revegetation, 11 unmined	species richness, community co	7	1	0	landscape	unknown
10x revegetation, 11 unmined	species richness, community co	7	1	0	landscape	unknown
2x each treatment	abundance	11	0.3	0	landscape	unknown
5x each treatment	species richness, abundance	8	0.25	0	landscape	ground-layer attributes important for reptile abundance/richness
29 transects in areas occupied by lynx, 29 transect	abundance	0	2	0	landscape	mesopredator abundance decreased after lynx reintroduction
9 restoration sites, 9 paddocks, 4 remnanats	abundance, usage	5	0.2	0	landscape	fallow deer (non-native) occurred frequently in restoration plantings nea
40 sampling plots within study area	abundance	1	5	0	landscape	birds followed similar patterns to reptile abundance
41 sampling plots within study area	abundance	1	5	0	landscape	
1x each treatment with 6 sampling plot each	diversity, abundance	18	4	0	landscape	herb and tree cover higher in herbivore removal area

18 x each CWD treatment	species richness, abundance	0	3	0 site	unknown
6x each treatment	species richness, abundance	0	3	0 site	unknown
1x each treatment	species richness, number of ind	0	4	2 site	unknown
5x each treatment	density	0	4	0 site	unknown
4 x CWD and control treatment, 2x harvested but	abundance	1	0.2	0 site	unknown
1x each treatment	density	0	3	0 landscape	severe winter in second year could have contributed to decrease in shee
3x each treatment	abundance	2	2	50 site	unknown
4x casuarina, 10x mallee	abundance, diversity	6	18	0 landscape	feral goats mitigated benefits of cattle destocking in casuarina areas
4x casuarina, 10x mallee	abundance, diversity	6	18	0 landscape	
3x revegetation, 2x unmined forest	abundance	4	0.25	0 landscape	unknown
4x each treatment	species richness, composition	10	0.3	0 landscape	key forest structure was restored 10-34 years after revegetation
12 study sites	abundance	3	0.1	0 landscape	three endemic tree species more abundant after rabbit eradication
3 x each treatment	species richness, species divers	0	3	50 site	unknown
4 x each treatment	species richness, species divers	0	3	50 site	unknown
4x each treatment	abundance	0	2	0 site	unknown
4x each treatment	abundance	0	2	0 site	unknown
1 exclusion plot	abundance, species richness	8	0.2	0 site	recruitment of certain plant species higher in exclusion plot, more compl
1 exclusion plot	abundance, survival rate	0	2	0 site	unknown
3x predator control plots	abundance, survival rate	0	2	0 site	unknown
1x each treatment	abundance	1	4	0 landscape	cat density increased 3x in fox control area
1x each treatment	abundance	0	4	0 landscape	
1x each treatment	abundance	1	4	0 landscape	
1x each treatment	abundance	0	4	0 landscape	
3 predator control areas, 3 non-baited areas	occupancy	0	8	0 landscape	unknown
1x each treatment with 12 survey sites per treatm	abundance	4	10	0 site	unknown
1x each treatment with 12 survey sites per treatm	abundance	0	10	0 landscape	unknown
1x each treatment with 12 survey sites per treatm	abundance	4	10	0 site	unknown
1x each treatment with 12 survey sites per treatm	abundance	0	10	0 landscape	unknown
1x each treatment with 12 survey sites per treatm	abundance	4	10	0 site	unknown
1x each treatment with 12 survey sites per treatm	abundance	0	10	0 landscape	unknown
1x each treatment with 12 survey sites per treatm	abundance	4	10	0 site	unknown
1x each treatment with 12 survey sites per treatm	abundance	0	10	0 landscape	unknown
9 CWD corridors	abundance, usage	1	2	0 site	unknown
4 sites with 4 control and 12 CWD manipulation tr	abundance, species richness	0.5	1	0 site	unknown
3 pasture, 5 revegetation, 5 remnant	species richness, composition	1	10	0 site	bird species richness also higher than pasture but lower than remnant
4 pasture, 5 revegetation, 5 remnant	species richness, composition	1	10	0 site	bird species richness also higher than pasture but lower than remnant
5 pasture, 5 revegetation, 5 remnant	species richness, composition	1	10	0 site	bird species richness also higher than pasture but lower than remnant
4 exclusion plots	nest survival, hatchling survival	0	6	0 site	unknown
4 restoration planting areas	usage	2	3	0 site	unknown
2 exclosures, 2 control sites	abundance, species richness	0	2	0 site	unknown
4x each treatment	abundance, species richness, sp	0	3	0.5 site	unknown
3x each treatment (dispersed, piles, windrows)	abundance, reproduction, recru	0	2	0.5 site	more weasel activity in pile treatments, herbivory on newly planted seed
3x each treatment	abundance	1	2	0 site	unknown
7 predator control, 3 control	survial rate	0	3	0 site	feral cat and ferret abundance decreased in predator control treatments
35 sites in unmined, 104 sites in revegetated	community assemblage	3	7	0 landscape	unknown
15 revegetated, unknown controls	abundance	4	0.1	0 landscape	unknown
128 pavers across 3 sites	usage	0	1	0 site	temperature and crevice size impact usage
2x each age class and control	diversity, abundance, communi	3	0.5	0 landscape	unknown
2 removal plots, 1 control	abundance	1	0.2	0 site	unknown
1 exclusion plot, 1 control	abundance	1	0.2	0 site	unknown
17x each treatment	genus-level diversity	2	0.1	0 site	unknown
6 artificial refugia	occupancy, site fidelity	0	6	0 site	unknown

Cost of implementation	Additional ongoing/maintenance costs	Public perceptions of intervention
unknown	unknown	unknown
unknown	unknown	unknown
unknown	unknown	unknown
unknown	unknown	unknown
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unknown	unknown	unknown
unknown	unknown	unknown
unknown	unknown	unknown
unknown	unknown	unknown
unknown	unknown	unknown
cost of bait delivery: AUD$37/km2	Avg. annual costs for monitoring and additional feral cat control was AUD$650,000	unknown
unknown	unknown	unknown
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unknown	unknown	unknown
unknown	unknown	unknown
unknown	unknown	unknown
unknown	unknown	unknown
unknown	unknown	unknown
unknown	unknown	unknown
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see Berman et al (2011) https://doi.org	unknown	unknown
see Berman et al (2011) https://doi.org	unknown	unknown
USD$0.09 per additional hatchling com	repeated annually	unknown
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unknown		
unknown	added CWD can degrade quickly in tropical rainforest and will need to be replaced	unknown
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unknown	trap maintenance $5.53-$2.30/ha	unknown
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unknown	unknown	results have potential to improve social acceptance of apex predator reintroduction
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trapping cheaper than predator exclusi	unknown	unknown
trapping cheaper than predator exclusion		
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approx. CAD$200/100 metres of CWD p	unknown	unknown
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