VICTORIAN MURRAY FLOODPLAIN RESTORATION PROJECT

HEALTHY LANDSCAPES, STRONG COMMUNITIES

Replacing natural hollows with hollow alternatives – a review of the available scientific literature

Case study -Vinifera and Nyah projects

Contents

Tables

Figures

Document Control

Document status and revision control

Revision	Date	Author	Reviewer	Approval	Comments
1.0	24/09/2024	Dr M. Birtchnell	VMFRP MCMA	J. David	For publication on VMFRP website

Executive summary

Overview

Altered land use practices, wildfire regimes, timber harvesting, altered hydrology, insect attack, wind and storms, and other changes in environmental conditions across the world have changed the composition of forests and woodlands. As a result, globally, old growth forests are transitioning to shorter, younger forests and woodlands. This has broadscale ramifications for habitat in terms of the quality and quantity of natural tree hollows, which naturally occur in older trees but largely are absent from young trees.

Many species of vertebrates and invertebrates use hollows for shelter, nesting and raising young, finding prey and feeding, protection from predation, and to aid movement and dispersal through the landscape. As many species are obligate hollow users, hollows are critical for their survival and no other habitat resource represents a feasible alternative structure.

In Australia, approximately 100 species that use hollows are listed as rare, threatened or near-threatened under State and/or Commonwealth legislation. Indeed, the loss of hollow-bearing trees from native forests and woodlands, particularly due to firewood harvesting practices, is recognised as a key threatening process declared under the Victorian *Flora and Fauna Guarantee Act 1988* (FFG Act). However, the broader ecosystem implications of hollow loss rarely are considered. Hollow loss often is viewed through a simplified lens, where loss of hollows impact hollow-dependant species. Yet, hollow loss may impact on population size, abundance, and diversity (including in predator or pest species populations), resulting in complex ripple effects that ultimately may destabilise entire ecosystems.

In landscapes where hollow-bearing trees are depleted and where hollow-dependent fauna are a conservation priority, local tree hollow abundance needs to be considered by land managers. Where a tree hollow paucity is identified, there is a need to devise a response to increase hollow numbers. However, in areas where hollows are in local abundance (that is, there are more hollows than hollow-using fauna), it is unclear what impact (if any) the removal of hollow-bearing trees may have on population dynamics.

In recent decades, land managers increasingly have been using nest boxes as a tool where natural hollows are in short supply. Despite the popularity and increasingly widespread use of nest boxes in Australia since the 1970s, in many settings the effectiveness of nest boxes and other forms of artificial/constructed hollows largely is unknown. Nest boxes (and other hollow alternatives) have potential to provide some support for the conservation of endangered hollow-dependent species. However, a range of factors can influence the effectiveness of nest boxes for target species, including lack of knowledge of the target species' habitat requirements and nest box attrition. Alternative approaches to providing cavities in an attempt to redress localised lack of natural hollows include chainsaw-carved hollows, hollow salvage, and erecting dead trees and utility poles

Only in more recent years has there been a growing realisation that nest boxes are deficient in many ways. This realisation of the deficiencies of nest boxes as a suitable alternative to hollows has driven a widespread and growing shift towards the use of alternatives, especially chainsaw hollows, despite there not being a clear understanding of the benefits and shortcomings. Deficiencies are especially obvious when comparing nest boxes to naturally-occurring hollows, yet it is even the case when comparing other hollow alternatives, including chainsaw-carved hollows.

Nonetheless, projects that propose to remove hollow-bearing trees are often required to replace hollows being removed with nest boxes and/or other constructed hollows. Planting trees or supporting new tree growth through ecosystem restoration provides a long-term solution to a paucity of hollows, however, these options have a long lag time to hollow formation, with estimates for hollow production in Australian tree species ranging from 60- 150 years.

The Victorian Murray Floodplain Restoration Project (VMFRP) aims to restore more natural patterns of inundation to the Murray Floodplain to support flood-dependent ecosystems that currently are in decline. In time, the Project will improve the health of the floodplain communities and enable these ecosystems to transition to 'old' systems that bear many naturally- occurring hollows. To achieve this floodplain restoration, the ER Central assessment package: Vinifera, Nyah and Burra Creek Floodplain Restoration Projects ('ER Central') provides a conservative estimate that 117 hollow-bearing trees are proposed to be impacted and/or removed across the ER Central project area (note: the project area no longer includes the Burra Creek site. The loss of these trees is to enable the construction and operation of infrastructure designed to restore health to floodplain and riparian ecosystems. This restoration work is designed to support the survival of thousands of large, hollow-bearing trees across the project areas; these trees are at significant risk of senescence and death without the proposed works. Whilst the loss of any hollow-bearing tree has consequences at some scale, the loss of thousands across the degraded floodplain likely would be catastrophic.

The Minister's Assessment for ER Central recommended a hollow replacement plan - based on available scientific knowledge - be developed to the satisfaction of DEECA, in the hope it will provide for the needs of priority hollow-utilising fauna (see p. 51-52 of the ER Central – Ministerial Assessment). The Minister previously made a similar recommendation for the EES Central package (Hattah Lakes North and Belsar-Yungera), contrary to the recommendations of the SIAC which found that a hollow replacement program was not warranted.

The volume of literature relating to hollows and hollow alternatives is significant. This review aims to distil relevant findings to inform and guide stakeholders as to the most appropriate and effective way to develop and implement an approach to mitigate (replace, offset) the short-term loss of hollow-bearing trees before the acknowledged and longer-term benefits of the project are realised.

Alternatives to naturally-occurring hollows

Prior to exploring alternatives to naturally-occurring hollows, it is important to note there are relatively few studies of naturally-occurring hollows compared to studies of artificial hollow alternatives. Thus, relatively little is known about naturally-occurring hollows, for example in terms of: occupancy rates; dimensions and how cavity dimensions influence occupancy patterns; internal microclimatic conditions and how these are regulated; and more. As a result of the paucity of literature relating to natural hollows, analyses comparing hollow alternatives to naturally-occurring hollows are fraught, although the literature is unequivocal about the superiority of naturally-occurring hollows over any currently available hollow alternative.

Despite natural hollows widely being believed to be superior to artificial hollows, there are several artificial alternatives that have been utilised to varying levels of success. These include:

- Nest boxes
- Spout boxes
- Bat boxes
- Chainsaw-carved hollows
- Chainsaw-carved bat fissures
- Hollow salvage for log hollows
- Erecting dead trees and utility poles
- Stimulation of hollow development in tree; and
- Entrance creation to access naturally-occurring tree cavities

As well as a paucity of information on the success of alternatives to naturally-occurring hollows, there also are a number of issues with these alternatives. Shortfalls reported in the literature include:

- A relatively short lifespan, with attrition typically ranging from approximately <5 to 10 years.
- A short lifespan *relative to the time it takes for natural tree hollows to form*.
- High microclimatic variation: humidity and temperature.
- A mismatch between nest box dimensions and species preferences.
- Often-low rates of uptake, particularly of the target species.
- Can harbour invasive species, including European honey bees, common starlings, common mynas and noisy miners, or support already-abundant common species.
- Tree health implications arising from attaching and installing hollow alternatives.
- The significant expense associated with creating, installing, maintaining, replacing and monitoring nest boxes and hollow alternatives while hollows are developing, including to ensure the nest boxes are safe for wildlife to occupy.

Regarding the last point, in a landscape such as the VMFRP construction footprint where naturally-occurring hollows are abundant, this expense would likely not be justified owing to low occupancy rates of hollow alternatives and the wildlife and ecological implications associated with providing hollow alternatives in these landscapes.

Occupancy patterns

Through review of the literature, it is evident that occupancy rates of natural hollows and different hollow alternatives are not well documented. In addition to this, is it poorly understood whether breeding rates differ between hollow alternatives and naturally-occurring hollows.

Design and installation of hollow alternatives

Design and installation of hollow alternatives varied significantly between studies. In this way, each study introduces new variables (i.e. design and dimensions), which limits the possible comparative analysis that can be undertaken between studies. Thus, it is difficult to determine which design and dimensions are best, by species or by hollow alternative. A number of studies indicated target species were observed using hollow alternatives designed for another target species (and not using the alternative designed for that species), highlighting there is a clear need for further research to be conducted into design of hollow alternatives, ideally prior to implementation of new hollow replacement strategies.

Longevity and attrition of hollow alternatives

Longevity and attrition, particularly of nest boxes, is a common issue in many of the studies reviewed. Studies found increasing numbers of nest boxes fell to the ground over time and that next boxes typically could not be re-erected, as they often were so badly damaged when they fell due to being in a relatively advanced state of decay. Unsurprisingly, significant maintenance was required during the lifetime of the nest boxes.

Storms and the weight of animals occupying the nest box also contributed to decayed boxes falling. As well as maintenance and replacement considerations, falling nest boxes have animal welfare considerations if the box is sheltering animals when it falls.

Ecological considerations

There are a number of ecological considerations that should contribute to the design of hollow replacement programs. These include competition for nest boxes between different species, which could result in some level of influence over the ecosystem dynamics compared to not providing hollows. It is unclear whether nest boxes (or the absence of nest boxes) exert an influence in landscapes where hollows are abundant, but findings indicate older forests (i.e. with some naturally-occurring hollows) have significantly lower rates of nest box occupancy compared to young forest (i.e. a paucity of naturally-occurring hollows). This suggests nest boxes are less likely to be used in a landscape context that includes old, hollow-bearing trees and so, less likely to change any ecosystem dynamics. However, this requires further research and consideration as nest boxes may provide a competitive advantage over naturally-occurring hollows for some occupants (including pest or exotic species) and predators. Studies have found that pest or exotic species including honeybees, common starling, common myna and black rats inhabit and may benefit from nest box programs.

Studies also have suggested that the use of plywood bat boxes are unlikely to lead to increased bat diversity. Further, studies concluded that if bat boxes are being used by a small proportion of the species that make up the local community, this may have unintended consequences for the bat diversity/ population dynamics in the region.

The literature does not consider the ecological implications of translocating hollows from their original location and installing them elsewhere and consequently, it is unknown whether aspects (such a scent-markers on hollows that have been translocated) have an impact on ecosystem dynamics.

Wildlife considerations

Declining hollow availability means artificial hollows, particularly nest boxes, are used widely to offset the removal of hollow-bearing trees or augment a landscape where there is a paucity of hollows, to provide alternative denning or nesting sites. However, as this review demonstrates, rates of occupancy of hollow alternatives can vary substantially among fauna species and habitats. Many studies demonstrate the shortfalls of these artificial hollows, including their limited functional life, potential colonisation by non-target - or even, pest - species and unsuitable internal microclimate.

The literature does not indicate studies have been undertaken to determine the ethics of installing hollow alternatives. However, the wildlife considerations covered in this review demonstrate concerns have been raised (directly and indirectly) about the ethical implications of:

- Creating nest-box reliance;
- Providing nest boxes and hollow alternatives that have potentially dangerous thermal properties;
- Providing nest boxes and hollow alternatives that experience water ingress and may support pathogens that are potentially fatal to wildlife;

- Failing to adequately fund and undertake maintenance of decayed/damaged nest boxes that may fall and kill the occupants;
- Providing hollow alternatives using designs that do not afford adequate protection to occupants from predators;
- Creating a dynamic of abundant hollow resources (by providing hollow alternatives) which encourages population growth of hollow-dependent fauna, but which are then displaced and perish when the only available alternative to their overheating nest box is another overheated nest box; and more.

These ethical considerations should form a central role in developing a hollow replacement strategy, as they have critical implications for the survival of the very species well-intentioned land managers would seek to protect through a hollow replacement strategy.

Host tree health considerations

The suitability of tree species, including River Red Gum, to host chainsaw-carved hollows has been investigated. This tree species has mechanisms to deal with physical damage caused by natural processes (e.g. limb abscission) which result more-readily in formation of natural hollows. This may reduce the risk of host-tree damage arising from installation of chainsaw-carved hollows.

Other studies have tested drilling 10-mm inspection holes drilled into trees to test the level of damaged sustained by the tree. The damage to tree stems caused by the drilling method was minimal compared to creating chainsaw hollows or excavating an entire cavity using a series of drill holes and routing a void.

No research is available in the published or available unpublished literature that addresses what treatment of the hollow section is required when translocating to a host tree, to prevent disease/pathogen transfer, termite infestation, or infection of new wounds. Host tree health and potential damage to old trees must be given careful consideration prior to any large-scale adoption of a hollow replacement approach that requires the attachment of hollow alternatives to old growth, hollow-bearing trees.

Safety, logistics and economic considerations

As well as ecological considerations, there also are safety, logistical and economic considerations associated with using hollow alternatives. Provision, maintenance, replacement and monitoring of effectiveness of hollow alternatives requires a significant investment of time and money. Creating entrance holes to existing internal cavities seems the approach with the least safety and economic implications. Chainsaw carved hollows need to be carved at the height at which the hollow is to be installed and require operating a chainsaw at height. Nest box installation also needs to be done at height, lifting heavy nest boxes into place and attaching them to the host tree whilst navigating a ladder or work platform.

For projects of scale approaching that of VMFRP, it has been estimated the cost of implementing and maintaining a nest box strategy would cost at least several tens of millions of dollars and ultimately may not be effective – or worse, may be deleterious to wildlife and ecosystems – in a landscape where naturally-occurring hollows are diverse and abundant. Nest boxes require significant materials and labour to produce *en masse* and require specialist installation (especially due to working at heights with heavy lifting), maintenance, replacement (more materials and specialist installation), and monitoring (especially, for safety and effectiveness). Given the issues identified with each of these hollow alternatives (including the considerable ethical and ecological implications associated with hollow alternatives), it is extremely unlikely the economic and time costs could be justified for funding the implementation, maintenance, replacement, management and monitoring of a hollow replacement programme using these approaches. This is particularly relevant for large-scale projects such as VMFRP that extend across a wide geographic area where there is an abundance of naturally-occurring hollows. Funds and resources (time, materials, expertise, etc.) required to implement and maintain such a hollow replacement strategy would likely achieve better outcomes for hollow-using species if expended in other more considered and beneficial ways.

Social considerations

Often, nest box (and other hollow alternative) programmes are implemented, monitored and maintained by passionate and dedicated community groups. As these projects are often community-led, nest box monitoring can engage the community in citizen science. In this way the opportunities are broader than ecological, and can result in education and engagement, leveraged community networks, and can be resilient to funding fluctuations owing to volunteer involvement. Certainly, nest box programmes can build community. However, establishing such a group to install and maintain nest boxes (or other hollow alternatives) in remote landscapes such as the VMFRP sites may be challenging.

Performance measures

The literature does not provide guidance as to what level of occupancy should be considered a criterion for success. Likewise, the literature indicates no single approach to measuring 'success' or 'failure' has been accepted as standard. Hence, it is difficult to establish what might define 'success' of hollow alternatives or a hollow replacement strategy.

Ultimately, the efficiency of a hollow replacement strategy must be determined by the extent to which the hollow alternatives provide 'suitable habitat' commensurate to that of a natural hollow. To date, no such alternatives to natural hollows exist. The literature clearly demonstrates issues with the effectiveness of hollow alternatives. For example: occupancy rates commonly are low; it is not always possible to attract target species; non-target species may occupy the provided hollow alternatives; attrition rates are high; maintenance is high; as well as many wildlife (and other) considerations. Much of the literature considers the effectiveness of hollow alternatives at short time scales (e.g. 1.5 years – 25 years) and reveals much disagreement as to whether hollow alternatives really are effective at offsetting the loss of naturally-occurring hollows. Most studies conclude hollow alternatives provide some benefit to hollow-dependent fauna but do not adequately or effectively replace natural hollows. There is a paucity of longer-term studies. Given hollow alternatives must be effective (that is, provide 'suitable habitat') for between 50 and 120 years after installation, to allow for the development of new hollows, the available literature demonstrates it is highly unlikely any hollow replacements will meet this performance target.

Land managers, governments, planners and others must consider whether the implications (wildlife, economic and ecological) of failed hollow replacement strategies create more harm than good. Investing funds in developing, studying and reporting findings of novel hollow replacements may be a much more effective use of funding than another (likely) failed hollow replacement strategy that may be harming the very ecosystems it intended to support. Indeed, the funds for implementing even one hollow replacement strategy could be well spent on developing a more effective hollow replacement model which could advance all future hollow replacement efforts.

Data collection

Although there is a growing body of literature relating to use of nest boxes and other hollow alternatives in Australia, research into using nest boxes as a management tool is still in its infancy. Largely, this is because many nest boxes across the Australian landscape have been installed as part of community-led projects. The lack of monitoring in Australian nest box programmes was linked to low availability of people, or the capacity of community groups to coordinate checking of boxes in different locations. Several methods have been used to monitor the use of tree hollows by mammals and birds, including radio-tracking, trapping on trees with hollows, detection of tree-use scratch marks, direct hollow observations, remote cameras, hollow observations with a thermal camera, and ultrasonic bat detectors.

Of these methods, visual observations to identify wear marks around hollows has potential, as it would enable the evaluation of many hollows over a short period. However, this is unlikely to yield accurate results regarding time since last use, the nature of the visit, whether the animal used the hollow, for what the animal used the hollow, for how long the animal used the hollow and which animal visited the hollow. In contrast, the method likely to produce the most reliable data is the use of remote cameras but this method has the most limitations due to an inability to install cameras wherever monitoring is needed, with installation likely requiring an elevated work platform (which is limiting as it has safety, logistical and cost implications).

Collaboration

This review highlights the imperative for a collaborative process in developing hollow alternatives and hollow replacement strategies. Some papers rightly assert the design, installation and monitoring of hollow alternatives should be a collaborative process involving land managers, ecologists and, depending on the hollow alternative, arborists. It is vital this collaboration includes the collection, collation and publication of empirical data that can be used to inform an adaptive management framework for strategic hollow replacement. Indeed, to some extent, this review highlights how *ad hoc* the research effort has been for investigating the effectiveness of hollow alternatives. Each study (more or less) represents a standalone experiment with limited potential for comparative analyses that are vital for expanding our understanding of the effective use of hollow alternatives. Collaborating (within and between programmes) will help ensure the loss of naturally-occurring hollows can be mitigated in the most effective way, so scarce conservation resources (funding, time, people) are spent as effectively as possible, for *actual and realised* ecological returns.

Known habitat preferences of hollow dependant fauna

Developing effective approaches to mitigating the loss of hollow-bearing trees is fundamentally and critically linked to an understanding of habitat preferences of hollow-dependent fauna. Thus, it is vital that these habitat preferences are used to direct stakeholders to mitigation strategies, rather than applying a standard approach (e.g. 1:1 hollow replacement) that may not, in fact, provide any *actual* mitigation. Indeed, many of the studies reviewed here have demonstrated the failure of hollow alternatives to effectively meet the habitat requirements of general hollow-dependent fauna, let alone those of targeted fauna.

Through the ecological assessment for VMFRP, nominated priority fauna are considered those hollow-dependent fauna listed under the FFG- and/or EPBC Acts which have been identified in VMFRP environmental assessments as 'present' or 'possible' within a project site.

To provide for nominated priority fauna species, it is critical to first consider evidence of their habitat requirements. Secondly, it is critical to establish what implications the proposed tree removal has for these habitat requirements and whether the removal of these specific trees means there is a need to provide any additional provisions for these species.

The nominated priority fauna species at the ER Central sites include:

- Regent Parrot
- South-eastern Long-eared Bat
- Barking Owl
- Major Mitchell's Cockatoo
- Carpet Python

For Regent Parrot, a comprehensive literature search indicates no studies have considered use of hollow alternatives in the wild by Regent Parrots – no studies even mention installation of nest boxes or other hollow alternatives. Indeed, provision of nest boxes or hollow alternatives is not listed as a mitigation approach in Federal (DCCEEW 2024a) or State (Baker-Gabb and Hurley 2011) threatened species policy.

An analysis of the literature for the other four nominated priority fauna species is presented in the body of the current review.

Conclusions

The current review has considered a range of aspects associated with hollow alternatives and factors that might affect the effectiveness of hollow replacement strategies. The volume of literature relating to hollow alternatives (particularly nest boxes) is significant; whilst not all studies could be considered directly in this review, it is hoped the studies included here provide a representative and accurate example of the broader landscape of research findings and perspectives. The content of this review is intended to support (not dictate) stakeholders' consideration of hollow alternatives, and whether hollow replacement strategies are likely to produce the most effective outcomes for hollow-using and hollow-dependent fauna, to mitigate the loss of hollow-bearing trees.

This review found current hollow alternatives do not fully mitigate or offset the loss of naturally-occurring hollows. A range of hollow alternatives and their associated issues have been considered, including implications for ecosystems and wildlife, to help guide stakeholders to develop effective approaches to mitigating the loss of hollow-bearing trees.

Of the hollow alternatives considered in this review, the method used in Ellis *et al.* (2022) holds the most potential and is highly worthy of additional investigations, including as part of the VMFRP hollow mitigation approach. The study provides evidence that providing entrances to otherwise inaccessible cavities in live trees has potential to accelerate development and occupancy of habitat for hollow-utilising fauna. Their findings demonstrate this method provides the benefits of naturally-occurring hollows, including a similarly safe microclimate. The method used by Ellis *et al.* (2022) may well prove to be a very effective tool to mitigate the loss of hollow-bearing trees for some hollow-using species.

Where hollow-bearing trees cannot be retained, the increasingly standard 'mitigation' approach is to 'replace' the hollows using an arbitrary ratio, usually 1:1 replacement. However, the literature reports many reasons why this approach is flawed (most notably in landscapes where availability of hollows is *not* a limiting factor in the environment) and may lead to perverse ecological outcomes, including unacceptable impacts on wildlife.

The review of known habitat requirements of five threatened species present (or potentially present) within the VMFRP ER Central sites study area highlights the critical importance of basing a hollow replacement strategy on the habitat requirements, biology and ecology of a species rather than on an arbitrary replacement ratio. This review raises awareness of the paucity of literature that exists pertaining to the *ecology* and *biology* of these species; regrettably, this also is the case for many other rare, threatened and/or cryptic species. A lack of understanding of the ways in which hollows are utilised by particular species (e.g. Inland Carpet Python) demonstrates how important are empirical data for developing effective, considered mitigation strategies.

In the case of most threatened species considered here (that do (or may) occur within the VMFRP study area), there is no evidence in the literature that hollow alternatives have been used in conservation efforts for these species. Coordinated research *must* be prioritised to determine whether hollow alternatives would provide any benefit for these species, particularly in a hollow-rich environment. For other species considered here, the literature shows there is little-no evidence of hollow alternatives having been used successfully, particularly in landscapes where there is a high rate of hollows.

1 Introduction

Globally, old growth forests and woodlands are transitioning to shorter, younger forests and woodlands. Land use changes, timber harvesting, altered hydrology, changes in wildfire regimes, insect attack, wind and storms, and other changes in environmental conditions, are driving this reduction in height and age. It is estimated 33% of the world's remaining forests have been reduced from mature to young (<140 years of age) (McDowell *et al.* 2020). Older trees often have developed hollows: semi-enclosed cavities that naturally form in many species of trees (Gibbons and Lindenmayer 2002) but which largely are absent in young trees and therefore, young forests. This reduction in extent of mature forests has serious implications internationally for hollow-dependent wildlife (e.g. Lindenmayer *et al.* 1991b; Gibbons *et al.* 2000; Cockle *et al.* 2011; Kikuchi *et al.* 2013). Some remnant woodlands in Victoria, Australia, support an average of 17 hollow-bearing trees per hectare (Bennett *et al.* 1994), however agricultural land – which has replaced around 90% of woodlands in some areas – supports around two hollow-bearing trees per hectare (Soderquist *et al.* 1999).

Many species of vertebrates and invertebrates use hollows during the day (diurnal) or night (nocturnal) for shelter, nesting and raising young, finding prey and feeding, protection from predation, and to aid movement and dispersal through the landscape (e.g. Bennett *et al.*1994; Gibbons and Lindenmayer 2002; van der Ree *et al.* 2006; Goldingay 2009; 2011). Many of these species are hollow-dependent (obligate). Thus, hollows are critical for their survival and no other habitat resource represents a feasible alternative structure (Gibbons and Lindenmayer 2002). Some species, particularly many species of echolocating bats, use tree cavities for thermoregulation (Geiser and Ruf 1995; O'Donnell and Sedgeley 1999; Rueegger 2016). Indeed, the availability of roosts in hollow-bearing trees is vital for reproduction and survival of cavity-roosting bats (Kunz and Lumsden 2005) with 66% of the 67 Australian insectivorous bats relying on tree hollows as day roosts (Churchill 2008). Yet none of these Australian vertebrate species are primary cavity-using species - that is, they are not species that create cavities (cf. woodpeckers), though they may enlarge the cavity and/or entrance. Rather, Australia's cavity-using vertebrate fauna are secondary cavity users (Rueegger 2017) which depend on the availability of pre-existing cavities formed by the activities of fungi and invertebrates, wind and fire events, or other natural processes. Cavity development through these natural processes often is slower than the rate at which hollows are removed from the landscape (for example, through urbanization, deforestation, plantation forestry, changed land use and development), leading to a reduction in available hollows and cavities.

In Australia, around 300 vertebrate taxa (for ease in this review: species) use tree hollows (Gibbons *et al.* 2002). Approximately 100 of these species that potentially use hollows are listed as rare, threatened or near-threatened under State and/or Commonwealth legislation in Australia (Gibbons and Lindenmayer 2002). Among Australian terrestrial vertebrate fauna, Gibbons and Lindenmayer (2002) estimate 10% of reptiles, 13% of all terrestrial amphibians, 15% of birds, and 31% of mammals may use hollows at some time. Some species (occasionally or regularly) will use hollows as a resource; others will be dependent upon them for their survival. Untold numbers of invertebrates also use (or depend on) hollows, though there is a critical dearth of studies on hollow use by invertebrates (Gibbons and Lindenmayer 2002).

The reduction in hollow-bearing trees across Australia is recognised as a serious threat to the survival of many hollow-dependent species, including mammals, birds and bats (e.g. Gibbons and Lindenmayer 2002; Gibbons *et al.* 2002; Lindenmayer *et al.* 2014; Griffiths *et al.* 2023). As discussed, many threatened species are dependent on hollows - and many more species are predicted to become threatened if the trend of removal of hollow-bearing trees is not reversed (Lindenmayer *et al.* 2014). Indeed, the loss of hollow-bearing trees from native forests and woodlands, particularly due to firewood harvesting practices, is a key threatening process declared under the Victorian *Flora and Fauna Guarantee Act 1988* (FFG Act). A broader consideration, however, reveals the loss of hollows and consequent impacts on wildlife species is a simplified view, as each species in turn plays a critical role in maintaining ecosystem function across landscapes, for example, through pollination, seed dispersal, regulating invertebrate populations, creating dynamics favourable for other species, and myriad more roles. In this way, the loss of hollows (especially from a landscape where hollows exert a limiting effect on species diversity and abundance) can have a ripple effect that ultimately may destabilise entire ecosystems.

Localised loss of tree cavities, or otherwise tree cavity scarcity, may impact on population size, abundance, and diversity (e.g. Marsden and Pilgrim 2003; Cockle *et al.* 2011), including in predator populations, further generating ecosystem dysfunction. In landscapes where cavity-bearing trees are depleted and where cavity-dependent fauna are a conservation priority, local tree hollow abundance needs to be considered by land managers. Part of that consideration includes assessing whether present and predicted tree hollow abundance is sufficient to support viable local and vagrant populations of hollow-utilising species. Where a tree hollow paucity is identified, there is a need to devise a response to increase hollow numbers (Rueegger 2017). However, in areas where hollows are in local abundance (that is, there are more hollows than hollow-using fauna), it is unclear what impact (if any) the removal of hollow-bearing trees may have on population dynamics (for example, see Lindenmayer *et al.* 2009). The removal of hollows in a landscape where hollows are abundant may not have significant implications for species conservation though may influence fauna populations at the micro-level, for example, through species- or individual-level hollow-selection preferences and utilisation patterns.

Planting trees provides a long-term solution to a lack of hollows, provided the planting is done in such a way that allows for lateral limb development and trunk growth both of which are important for long-term hollow development. However, planting trees has long lag times to hollow formation – estimates for hollow production in Australian tree species range from 60-150 years, depending on the tree species and environmental conditions (e.g. Gibbons *et al.* 2000a). In recent decades, land managers increasingly have been using nest boxes as a tool where natural hollows are in short supply. Nest boxes (and other hollow alternatives) have potential to provide some support for the conservation of endangered hollow-dependent species (e.g. Harley 2006; Mine *et al*. 2014; Durant *et al.* 2009). However, a range of factors can influence the effectiveness of nest boxes for target species, including lack of knowledge of the target species' habitat requirements and nest box attrition (Lindenmayer *et al.* 2009). Alternative approaches to providing cavities in an attempt to redress localised lack of natural hollows include chainsaw-carved hollows, hollow salvage and erecting dead trees and utility poles (see below).

Despite the popularity and increasingly widespread use of nest boxes in Australia since the 1970s (Menkhorst 1984), in many settings the effectiveness of nest boxes and other forms of artificial/constructed hollows largely is unknown (Goldingay *et al.* 2018; Best *et al.* 2022). Indeed, there long has been debate as to the effectiveness of nest boxes (e.g. Harley 2006; Lindenmayer *et al*. 2009, 2015; Le Roux *et al*. 2015; Rueegger 2016; Lindenmayer *et al*. 2017; Goldingay 2018). Only in more recent years has there been a growing realisation that nest boxes are deficient in many ways (e.g. Griffiths *et al.* 2017; 2018; van der Ree 2019). This is especially obvious when comparing nest boxes to naturally-occurring hollows, but is even the case compared to other hollow alternatives including chainsaw carved hollows. This realisation has driven a widespread and growing shift towards the use of alternatives, especially chainsaw hollows, despite there not being a clear understanding of the benefits and shortcomings (e.g. for ecosystem dynamics, wildlife welfare, maintenance requirements, best construction methods, etc.), of these alternatives. Quite rightly, van der Ree (2019) asks: 'why has it taken at least 40 years to understand that nest boxes are not a panacea to the loss of hollows?...and [even more importantly] what do we need to do over the next few years to ensure that we aren't still debating the merits of carved hollows in 40 years' time?'

Nonetheless, projects that propose to remove hollow-bearing trees often are required to replace hollows being removed with nest boxes and/or other constructed hollows. The Victorian Murray Floodplain Restoration Project (VMFRP) aims to restore more natural patterns of inundation to the Murray Floodplain to support flood-dependent ecosystems that currently are in decline. In time, the Project will improve the health of the floodplain communities and enable these ecosystems to transition to 'old' systems that bear many naturally-occurring hollows. To achieve this floodplain restoration, the ER Central Assessment Package: Proposed Vinifera, Nyah and Burra Creek Floodplain Restoration Projects ('ER Central') provides a conservative estimate that 117 hollow-bearing trees are proposed to be removed across the ER Central project area (note: the project area no longer includes the Burra Creek site). The loss of these trees is to enable the construction and operation of infrastructure designed to restore health to floodplain and riparian ecosystems, including thousands of large, hollow-bearing trees across the project area that are at significant risk of senescence and death. Whilst the loss of any hollow-bearing tree has consequences at some scale, the loss of thousands across the degraded floodplain likely would be catastrophic.

The Minister's Assessment for ER Central recommended a hollow replacement plan be developed to the satisfaction of DEECA, in the hope it will provide for the needs of priority hollow-utilising fauna (see p. 51-52 of the ER Central – Ministerial Assessment).The Minister previously made a similar recommendation for the EES Central package (Hattah Lakes North and Belsar-Yungera) contrary to the recommendations of the SIAC which found that a hollow replacement program was not warranted. For ER Central, the Minister states:

> *I agree with the SIAC's recommendation that, if a hollow replacement plan is mandated, there is a need for careful consideration of designs to appropriately accommodate the range of hollow dependent fauna and ensure appropriate insulation against temperature extremes, and that potential for occupation of nesting boxes by pest and non-target species also need to be considered. These recommendations should be considered in the development of the hollow replacement plan and associated monitoring required under the suggested amendments to EDS E2e.* (p. 52).

The volume of literature relating to hollows and hollow alternatives is significant. This review aims to distil relevant findings to inform and guide stakeholders as to the most appropriate and effective way to develop and implement an approach to mitigate (replace, offset) the short-term loss of hollow-bearing trees before the acknowledged and longer-term benefits of the project are realised. The review focuses on hollows in standing trees (dead or living) (cf. say, hollows in ground logs) and on hollows suitable for vertebrate fauna, and considers a number of aspects relating to nest boxes and constructed hollows.. Necessarily, a review must have a manageable focus. However, the critical role hollows play in the survival of invertebrates (and therefore on the many vertebrates that predate these invertebrates) also is of utmost importance, as is the myriad ecosystem functions large trees provide (e.g. provision of nectar and pollen) beyond 'merely' supporting hollows, all of which are vital contributions to health ecosystem dynamics. Furthermore, a review of indigenous cultures and the ways in which their care of Countries has shaped the occurrence of old hollow-bearing trees, is vital. Given hollow-using fauna (e.g. birds, mammals, lizards) are fundamental for healthy Countries and have been a critical food source for indigenous people for tens of thousands of years, it is undoubtable indigenous practices have long-included cultivating and maintaining hollows, if not actively helping trees to create them. Indigenous understanding of how to manage wound wood (for example) may well provide clues as to how we all can support the production (and acceleration) of hollows or effective hollow alternatives. Despite the significant volume of published research on many aspects covered in this review, there remains a yet more significant volume of research still needing to be undertaken, to address research gaps that hinder our ability to appropriately and effectively provide a healthy future for many Australian ecosystems.

2 Alternatives to naturally-occurring hollows

Prior to exploring alternatives to naturally-occurring hollows, it is important to note there are relatively few studies of naturally-occurring hollows compared to studies of artificial hollow alternatives. Thus, relatively little is known about naturally-occurring hollows, for example in terms of occupancy rates; dimensions and how cavity dimensions influence occupancy patterns; internal microclimatic conditions and how these are regulated; and more. Partly, this is because logic dictates natural hollows are best for hollow-utilising fauna, but is also due to other practical considerations including:

- concerns about causing disturbance to occupants of hollows and hollow-occupant dynamics;
- lack of known natural hollows at study sites (e.g. for comparative analyses against artificial cavities); and/or
- safety risks associated with installing remote monitoring loggers in hollows typically located high up in dead and decaying trees (McComb *et al.* 2022).

As a result of the paucity of literature relating to natural hollows, analyses comparing hollow alternatives to naturally-occurring hollows are fraught although the literature is unequivocal about the superiority of naturally-occurring hollows over any currently available hollow alternative.

Further, it must also be noted there is no scientifically standard approach to hollow alternatives, for example, regarding size, type, host tree species, geographic location, construction, etc. nor any robust comparative analyses, therefore there are only limited and ineffective data available regarding the effectiveness (or otherwise) of the hollow alternatives discussed herein. This lack of data presents a significant issue for developing an *effective* hollow replacement strategy and warrants urgent focus and research effort, particularly in light of the considerable and significant potential ethical and ecological implications of implementing an *ineffective* hollow replacement strategy.

Finally, there also is a critical shortage of literature relating to the *ecology* and *biology* of many hollow-dependent and hollow-utilising species, particularly rare, threatened or cryptic species. The absence of such literature means it is highly complex to determine what are effective, safe and targeted hollow alternatives to mitigate for these species the loss of hollow-bearing trees from their ecosystems. Without empirical evidence, efforts to provide hollow alternatives may well be ineffective – and even dangerous. Thus, hollow alternatives may drive further declines in rare and threatened species populations.

2.1 Nest boxes

It is postulated that some attributes of mature trees can readily be replicated using artificial structures – notably, for decades, there has been a common view that hollows can be replaced by nest boxes. Indeed, the literature indicates nest boxes are used by many species as an alternative to natural hollows in trees (e.g. Beyer and Goldingay 2006; Hamerstrom *et al*. 1973; McComb and Noble 1981) and may have conservation value in some scenarios. For example, a large-scale nest box program has resulted in steady population increases of the bluebird (*Sialia* spp.) in North America (Fiehler *et al*. 2006; Newton 1994). In fact, some nest box programmes (particularly in the northern hemisphere) have been highly successful, including one in Germany, where nest boxes installed throughout forests in Germany resulted in a 5 – 20-fold population increase in some bird species (Bruns 1960; see also: von Hartman 1971; Haramis and Thompson 1985; Taulman *et al.* 1998; Smith and Agnew 2002), though the ecological implications of increasing populations of *some* species are unclear.

Nest boxes are widely considered the best available interim solution to the global reduction in tree hollows, however their effectiveness as a solution to tree hollow availability at larger scales (even in the short-term) is questionable. In the Australian context, despite nest boxes currently being the only tool commonly used to offset or mitigate the loss of hollow-bearing trees, the *actual* effectiveness of nest boxes is little known (e.g. Goldingay *et al.* 2018; Best *et al.* 2022; Lindenmayer *et al.* 2017). The thermal properties of nest boxes *compared* to those of natural hollows have not been comprehensively studied across the diverse range of fauna that use boxes, or across the range of environments where they are deployed (Griffiths *et al.* 2018), however there is much evidence demonstrating the thermal properties of nest boxes pose significant impacts to wildlife welfare and have serious ethical and ecological implications (see below). Indeed, there is strong disagreement in the Australian literature regarding the effectiveness of nest boxes for conserving hollow-utilising fauna and their effectiveness in offsetting habitat loss or the loss of mature hollow-bearing trees (e.g. Harley 2006; Lindenmayer *et al*. 2009, 2015; Le Roux *et al*. 2015; Rueegger 2016; Griffiths *et al.* 2017; Lindenmayer *et al*. 2017; Goldingay 2018; Griffiths *et al.* 2018; van der Ree 2019).

Published literature on nest box utilisation and performance is relatively low, considering their popularity for conservation projects. In 2008, Lindenmayer *et al.* (2009) determined that in the previous five years alone, more than 150 research papers included the term 'nest box' in the title, keywords or abstract. They found most of these papers examined occupancy rates and patterns by different vertebrates (e.g. reviewed by Gibbons and Lindenmayer 2002; Beyer and Goldingay 2006). Few of the studies considered unwanted pest species or the rates of attrition/decay of nest boxes (Lindenmayer *et al.* 2009). Throughout the literature, especially within the Australian context, few papers consider long-term patterns of occupancy or nest box longevity. Even fewer consider the ecological implications (i.e., beyond what species might use the resource) of introducing nest boxes or other hollow alternatives into the landscape.

In their 10-year experimental study in Mountain Ash (*Eucalyptus regnans*) forests, Lindenmayer *et al.* (2008) monitored nest boxes specially designed for the endangered target species, Leadbeater's Possum (*Gymnobelideus leadbeateri*), and found almost 70% of nest boxes were never occupied (including by the target species). Nest boxes were more often utilised in younger forests (~58%) than in older forests (~4%) but these younger forests were where infestations of nest boxes by the European Honeybee *Apis mellifera* were greatest and the probability of nest box collapse was highest. For development of hollow replacement strategies in older forests and woodlands where naturally-occurring hollows are abundant, it is worth noting that nest boxes, when installed in older forests where natural hollows occurred, were occupied by almost no animals including the endangered target species (Leadbeater's Possum) (Lindenmayer *et al.* 2009). To further highlight the role the landscape context plays in the usefulness of hollow alternatives, ABC (2016a) reported the outcomes of a nest box programme specifically designed to supplement breeding habitat for the EPBC-listed, Critically Endangered Swift Parrot *Lathamus discolor,* quoting Dr Dejan Stojanovic from the Australian National University: "*Hollows are so rare that you can actually see swift parrots literally searching tree after tree after tree, inspecting every little black hole, every broken branch stub, just checking, is this a hole, is that a hole, is it suitable? Is it not?*" In that hollow-depleted environment, 11 out of 40 nest boxes were being occupied by Swift Parrots, one of which contained six eggs and another contained four eggs (ABC 2016b). This demonstrates the effectiveness of nest boxes (or other hollow alternatives) can heavily be influenced by the abundance or lack of hollows in the landscape, and whether hollows are a limiting factor for hollow-using species. In landscapes where naturally-occurring hollows are abundant, including landscapes within the construction footprint of the VMFRP, nest boxes or other hollow alternatives are unlikely to be used. The preference for naturally-occurring hollows (evidenced by almost no animals using nest boxes where naturally-occurring hollows were available) demonstrates that retaining hollow-bearing trees - and avoiding removal of trees bearing hollows that show signs of occupancy (where possible to determine) - should be a priority.

Until recently, nest boxes primarily were timber or plywood boxes. The use of plastic nest boxes has been trialled (Elston *et al*. 2007; Rueegger *et al.* 2013; Saunders *et al.* 2020) and recycled plastic nest boxes have been used for Leadbeater's Possum since 2008, with over 600 installed across the species' range (D. Harley and J. Antrobus, unpublished data; see McComb *et al.* 2022). However, a recent novel study explored the potential of 3-D printed plastic nest boxes, comparing three plastic prototypes especially with regard for their potential insulative properties (Callan *et al*. 2023a, see also Parker *et al.* 2022). This study sought to overcome known shortcomings of conventional nest boxes, by using additive manufacturing (in this case, 3D printing) to produce and refine prototypes of optimal thickness and construction for mass production of nest boxes made from longer-lasting materials. The plastic prototypes developed for the study were able to be created in a way that more closely resembled the naturally complex shapes and textures of natural tree hollows – and with fewer mechanical joints that become failure points on traditional nest boxes (Callan *et al.* 2023a). The use of plastic in the construction of nest boxes may influence the humidity within the nesting/roosting chamber, meaning the impervious plastic nest boxes are predicted to maintain high levels of humidity in comparison with ambient conditions, and thus may be more consistent with natural tree hollows (Callan *et al.* 2023a). Whilst these novel plastic nest boxes offer potential, consideration must be given to the ecological implications of plastic nest boxes being exposed to weather, gnawing, excavation and attrition, in terms of plastic/microplastic pollution and the introduction of microplastics into food webs.

2.1.1 Issues

In summary, the issues with nest boxes include:

Despite being used and prescribed for several decades as a suitable and effective hollow replacement option, nest boxes have repeatedly come under scrutiny and criticism. There are several shortfalls to using nest boxes (e.g. van der Ree 2019; Terry *et al.* 2021; Callan *et al.* 2023a), particularly due to aspects of their construction. Shortfalls reported in the literature include:

1. A relatively short lifespan, with attrition typically ranging from <5 to 10 years (although some may still function after 20 years (Goldingay *et al*. 2018)). Attrition of nest boxes often is observed within 2-10 years of nest box installation; they have variable lifespans which may be influenced by forest age and rates of limb shedding (Lindenmayer *et al.* 2009).
2. A short lifespan *relative to the time it takes for natural tree hollows to form*.
3. High microclimatic variation: humidity and temperature. Some studies raised concerns that nest boxes (and other hollow alternatives) are prone to overheating in summer and may not provide protection from temperature extremes compared to natural cavities (Rowland *et al*. 2017; Griffiths *et al*. 2018; van der Ree 2019), though there is little direct evidence that this poses a fitness cost to the species occupying the nest boxes (Goldingay 2017; Saunders *et al*. 2020). However, a nest box programme at Hattah Lakes, in proximity of the VMFRP, showed a direct impact on wildlife health. The discussion in this review demonstrates the ethical implications posed by the poor thermal properties of nest boxes (and other hollow alternatives) (see below).
4. A mismatch between nest box dimensions and species preferences (Goldingay, 2009; Goldingay *et al*. 2020a; Rueegger *et al*. 2019).
5. Often, they have low rates of uptake, particularly of the target species.
6. They can harbour invasive species, including European honey bees, common starlings, common mynas and noisy miners, or support already-abundant common species (van der Ree 2019).
7. Tree health implications arising from attaching and installing nest boxes (and other hollow alternatives).
8. The significant expense associated with creating, installing, maintaining, replacing and monitoring nest boxes and hollow alternatives while hollows are developing (Lindenmayer *et al.* 2017), including to ensure the nest boxes are safe for wildlife to occupy. In a landscape such as the VMFRP construction footprint where naturally-occurring hollows are abundant, this expense would likely not be justified owing to low occupancy rates of hollow alternatives and the wildlife and ecological implications associated with providing hollow alternatives in these landscapes.

Nest box effectiveness can be impacted by local rainfall, the method of attachment to trees and the types of materials selected for construction, though some may still function after 20 years (Goldingay *et al*. 2018).

Rueegger (2017) states:

> 'Given the high use of habitat boxes to mitigate or offset lost cavity- bearing trees in Australia (Lindenmayer *et al.* 2017) and the recurrent negative reports on the effectiveness of habitat boxes (Lindenmayer *et al*. 2009, 2015; Le Roux *et al*. 2015; Rueegger 2016; Lindenmayer *et al*. 2017), there is a need to trial other methods in an attempt to improve the effectiveness of artificial hollows.' (p. 405)

2.2 Spout boxes

Spout hollows are an important resource in Australian landscapes, forming within many *Eucalyptus* species. Spout hollows commonly occur when part of a branch breaks and wound wood does not occlude the broken stub of the limb. These hollows are approximately cylindrical-shaped cavities that form within the centre of a branch or small trunk, running parallel to the stem length (Strain *et al.* 2021). In central Victoria, spout hollows often are occupied by Turquoise Parrots *Neophema pulchella* and Brown Treecreepers *Climacteris picumnus victoriae* (Strain *et al.* 2021), both of which are of conservation concern.

In a study designed to determine whether a nest box could be used as an alternative to a spout hollow, Strain *et al.* (2021) developed and installed a novel nest box design: *spout boxes*, based on the structural characteristics of natural spout hollows. Strain *et al.* (2021) monitored the occupancy (n = 193) and internal microclimate (n = 131) of natural hollows and spout boxes within a woodland where natural tree hollows were once abundant. The target species for provision of hollow alternatives were the Turquoise Parrot and Brown Treecreeper. The study found both natural hollows and spout boxes were occupied and used for breeding by birds and mammals, including ten Turquoise Parrot breeding events in spout boxes (four were detected in tree hollows) and ten Brown Treecreeper nests in spout boxes (none in tree hollows) (Strain *et al.* 2021; see below and below). Many of the findings were consistent with studies comparing regular nest boxes and natural hollows. Natural hollows had consistently higher humidity, and thermal maxima and minima were buffered, when compared with spout boxes (Strain *et al.* 2021). These differences were largely explained by wall thickness (Strain *et al.* 2021). Spout boxes displayed even more extreme temperature variation and lower humidity when not shaded (Strain *et al.* 2021). While more extreme microclimate conditions did not prevent usage, tolerable thresholds for hollow-dependent species may soon be exceeded under current climate change projections (Strain *et al.* 2021).

2.2.1 Issues

Unsurprisingly, the study by Strain *et al.* (2021) discovered many of the same issues as those associated with regular nest boxes, although the rates of occupancy and breeding activity were promising – for both target species, the breeding activity in the spout boxes exceeded that in natural hollows. These results warrant further investigation. However, the vulnerability of spout boxes to unsuitable internal microclimates means use of spout boxes must be carefully considered. It can be inferred that issues with attrition of spout boxes would be approximately consistent with those observed for typical nest boxes. Regular monitoring for the lifetime of the spout box (especially to ensure wildlife welfare) would be vital, meaning long-term use of this hollow alternative would be costly (e.g. time, money, personnel). Use of spout boxes may be beneficial in some circumstances, namely where there is a paucity of natural hollows (Strain *et al.* 2021).

2.3 Bat boxes

As is the case for nest boxes, bat boxes are widely considered a beneficial management tool for the conservation of cavity-roosting bats in landscapes where natural tree roosts and hollows have been depleted (Mering & Chambers 2014). Yet in the Australian context, studies of bat boxes consistently have reported relatively low rates of use, and that those used bat boxes are used by only a small proportion of the species that make up the local community of cavity-roosting bats (e.g. Rhodes and Jones 2011; Godinho *et al*. 2019; Rueegger *et al*. 2019).

2.3.1 Issues

As well as concerns about low (and skewed) rates of use, concerns have been raised around the microclimatic conditions bat boxes may provide and so, the potential for them to become ecological traps (Flaquer *et al*. 2014; Crawford and O'Keefe 2021; Griffiths 2022). Consequently, the actual conservation value of bat boxes for bat communities in areas where there is a paucity of natural hollow roosts is disputable (Griffiths et al. 2017; Griffiths *et al.* 2023). Meanwhile, scarce conservation resources continue to be used by land managers, conservation practitioners, and community groups, to both establish and maintain bat box (and other nest box) projects (Macak 2020). Indeed, if bat boxes are being used (if at all) by only a small proportion of the species that make up the local community of cavity-roosting bats, what implications does providing bat boxes have on the interspecific dynamics of bats (e.g. competition, fitness), and what are the ecological implications of this? Indeed, Griffiths *et al*. (2023) postulate that plywood bat boxes are unlikely to lead to increased bat diversity or support entire bat communities, but likely will be used by at least Gould's Wattled Bat. What are the ecological implications of only one (or just a few) bat species using boxes and how might the *intended* benefits to ecosystems be impacted by *unintended* bat diversity/population dynamics. If only one species of bat is able to persist in an area, what does that mean for the balance of invertebrate prey? Further research urgently is required to determine what types of supplementary roosts will be most suitable for communities of cavity-roosting bats (Griffiths *et al.* 2023) (see below). Importantly, in an old-growth wooded landscape (such as that within the VMFRP construction footprint) where there is an abundance of microhabitat for bats (including decorticating bark, fissures and other micro-hollows), the installation of bat boxes would likely pose unacceptable impacts on wildlife health, populations and ecological dynamics.

2.4 Chainsaw-carved hollows

For at least two decades, chainsaw-carved hollows have been considered as a habitat improvement tool to address shortages of natural hollows (e.g. Carey and Gill 1983; Carrie *et al.* 1998; Wood *et al*. 2000; Saenz *et al*. 2001; Carey 2002), particularly in the Northern hemisphere. Chainsaw-carved hollows involve carving hollows directly into standing trees with a chainsaw or other tool. There is a diverse range of types of carved hollows, including the faceplate method, false door method, simple plunge cuts, and coronet cuts, with very little strong or peer- reviewed evidence to support one method over another (van der Ree 2019).

In North American examples, the literature describes two methods for mechanically creating hollows for the Red-Cockaded Woodpecker (*Leuconotopicus borealis*): one where cavities were created in tree trunks through drilling (Copeyon 1990; Taylor and Hooper 1991) – really, a method of accelerating tree hollow formation by a primary cavity-using species - and the other where artificial wooden boxes were inserted into carved-out cavities (Allen 1991). Both techniques used wood filler and paint in the hollow creation process with pine trees being the hollow host. In an Australian study, Rueegger (2017) showed that excavated tree cavities have the potential to become an additional tool in providing artificial hollows. Hollow uptake was fast, taking only a few days for some species

(Rueegger 2017). Furthermore, hollow use by Sugar Glider *Petaurus notatus* and White-throated Tree-creeper *Cormobates leucophaea* was continuous over prolonged periods and also for the rearing of young (Rueegger 2017). It is likely that Long-eared Bats also used at least one hollow as a maternity roost (Rueegger 2017). The diversity of species using the hollows and frequent use of the hollows indicate a range of hollow-dependent mammal and bird species may take up mechanically created hollows (Rueegger 2017). Rueegger (2017) determined that some elements of the chainsaw hollow design used in that study would likely need to be altered to meet species-specific requirements and may be guided in part by habitat box design elements (e.g. Beyer and Goldingay 2006; Goldingay *et al*. 2015; Rueegger 2016). For example, the hollow inspection by Crimson Rosella *Platycercus elegans* - yet its absence in using hollows - suggests cavity characteristics may not have been suitable for this species, with larger cavities potentially required (Rueegger 2017). The small entrance size used (38 mm) in Rueegger (2017) was not small enough to exclude (the over-represented) Sugar Glider; future studies should consider providing a greater variety of entrance sizes to reduce species dominance and competition for hollows (Ruegger 2016). Best *et al.* (2022) trialed the use of chainsaw hollows for Krefft's Glider *Petaurus notatus* (see below), a recently recognised species formerly classified as the Sugar Glider *Petaurus notatus.*

In Australia, relatively few studies describe field experiments trialling mechanical tree hollow creation or acceleration of tree hollow formation. However, some hypothesised potential advantages of mechanical creation of tree cavities over nest boxes include: greater longevity (e.g. Harley 2008; Beyer *et al*. 2008); greater cost effectiveness mid- to long-term; closer resemblance to natural hollows (including cavity microclimate) (Beyer *et al*. 2008; Griffiths *et al*. 2017); and increased attractiveness to a greater diversity of hollow-dependent species.

These constructed hollows recently have received renewed attention, including in Australia, as an alternative to nest boxes (Rueegger 2017; Griffiths *et al*. 2018). In recent years, the Australian arboriculture industry has actively adopted the use of chainsaw-carved cavities and many practitioners across south-eastern Australia are using cavity designs such as those described by the Victorian Tree Industry Organisation (2010) (Best *et al.* 2022). Indeed, the installation of chainsaw hollows for hollow-dependent wildlife is growing in popularity in habitat restoration and conservation management programs (e.g. Rueegger 2017; Stojanovic *et al.* 2018; Griffiths *et al*. 2020; Terry *et al.* 2021).

To date, however, these chainsaw hollows largely have been installed in an *ad hoc* manner; most installations have little or no regular, ongoing monitoring and/or reporting of wildlife use. Across south-eastern Australia, there currently is limited empirical evidence quantifying features (e.g. cavity design and placement within the environment) that influence the effectiveness of chainsaw hollows, in terms of them being found and used by target fauna (Best *et al.* 2022). Little currently is known about how the physical characteristic of chainsaw hollows, the host-trees in which they are installed, or the features of the surrounding habitat, affects the use of chainsaw hollows by native species (Best *et al.* 2022). Likewise, there is little understanding (or, at least, reporting) of the maintenance required to ensure cavities carved into live trees remain functional (e.g. removing wound wood that can close entrances; Carey and Sanderson 1981). Consequently, quantifying the effectiveness and so, conservation value, of chainsaw hollows (including compared to that of nest boxes) is problematic. Urgent research is needed to empirically assess whether supplementary cavities can achieve effective conservation and applied management outcomes for hollow-dependent fauna; can they play a role in offsetting the loss of natural hollows? Worse: do they play a role in further impacting displaced wildlife?

Van der Ree (2019) posits: 'unless carved hollows are installed according to a scientifically robust experimental study design and carefully monitored and evaluated, it is highly likely that we will still be debating the relative merits of nest boxes, carved hollows, log hollows and natural hollows in 40 years' time.' Van der Ree (2019) suggests hundreds or (possibly even) thousands of carved hollows are being installed around Australia each year; each project is an incredible opportunity to 'learn while doing'. However, van der Ree (2019) notes the common approach to hollow replacement simply is to install hollows according to the land manager's request or arborist's recommendations. This may (or may not!) include recording some details about the hollow or the host tree, and some *ad hoc* monitoring/inspections for a short period of time until 'interest wanes or funding runs out', before moving onto the next project (van der Ree 2019). This dour summary of chainsaw hollow programmes also is clearly demonstrated by Macak (2020) in relation to nest box programmes, with these programme design/management issues replicated many times over across the landscape. Van der Ree (2019) rightly considers the lack of robust understanding around the relative merits of chainsaw (or any artificial) hollow can be solved – the solution is *'for land managers to adopt the principle that every hollow being installed is an opportunity to 'learn about the use and effectiveness of carved hollows', as well as be an opportunity to replace hollows'* (p. 45). In effect, 'every hollow installed should be considered part of an experiment, and by considering and adopting a few guiding principles, the maximum amount of reliable and robust information can be learnt in the shortest amount of time, leading to more rapid adoption of evidence-based best-practice techniques' (van der Ree 2019).

To date, the use of constructed hollows specifically for threatened species largely has been limited to Leadbeater's Possum (Lumsden *et al.* 2016) and Swift Parrot (ABCa,b 2016).

2.4.1 Issues

Consideration of whether created tree hollows weaken the host tree to the extent of tree failure is important. Ratios of hollow width:tree diameter could be guided by Mattheck's t/R tree ratio threshold, which states tree failure is more likely to occur if the residual wall thickness surrounding a decay cavity is < 0.3 of the stem's radius (Mattheck *et al.* 1993; Mattheck and Breloer 1994). However, Mattheck's t/R ratio relates to decay-related tree failure; unlike the immediate impact of mechanically creating hollows, trees have time to respond to trunk decay as decay advances. Rueegger (2017) tested this, as it was unknown whether a t/R ratio of 0.3 would be sufficient to limit or prevent hollow host tree failure following the immediate introduction of a created cavity. During 24 months of monitoring, none of the hollow host trees failed, including during a storm event under which conditions frequently result in trees being uprooted or broken according to the Beaufort wind scale (Rueegger 2017).

When considering the effect of the hollow host trees' wound-wood development on hollow faceplates, Rueegger (2017) observed during 24 months of monitoring that all faceplates remained in place with wound-wood development. Exogenous tree trunk growth did not dislodge faceplates for living host trees (Rueegger 2017). Very minor wound-wood development occurred along the horizontal plane of the faceplate cut, whereas strong wound-wood development occurred along the vertical planes of the cuts emerging from the exposed cambium and callus formation (Rueegger). Terry *et al.* (2021) found the extent of callous regrowth varied between tree species – Red Ironbark (*Eucalyptus sideroxylon*) showed the greatest degree of callousing. Grey Box (*E. macrocarpa*), Narrow-leaf Peppermint (*E. radiata*) and Red Box (*E. polyanthemos*) also displayed callousing within the 2.5-year monitoring period (Terry *et al.* (2021). Yellow Box (*E. melliodora*), Red Stringybark (*E. macrorhyncha*) and Messmate (*E. obliqua*) did not develop callousing during the study (Terry *et al.* 2021). Initial kino flow was abundant, particularly by Spotted Gums (*Corymbia maculata*) (Rueegger 2017). Rueegger (2017) reported wound-wood of chainsaw hollows was found to incorporate the faceplates; within 2.5 years of monitoring in their study, Terry *et al.* (2021) also observed callous regrowth of bark around and over the faceplate.

To some extent, evidence of hollow faceplates being partially occluded by wound wood in living trees was considered encouraging (Rueegger 2017), as it ensured the plates became sturdily locked in. Given the potential advantages of wound-wood growth, keeping the hollow host trees alive should be considered where feasible. In Rueegger (2017), wound-wood growth resulted in sealing gaps around the faceplate, increasing water and air tightness of the hollow meaning the hollow may more closely resemble a natural hollow (Rueegger 2017), however Terry *et al.* (2021) found three of the nine larger chainsaw hollows with natural faceplates had begun to warp open, leaving large gaps of up to 3 cm. Terry *et al.* (2021) also discovered high moisture within the carved chainsaw hollows, including water running down the inside of the faceplate, resulting in wet nesting material that had turned to mud (see below). Rueegger (2017) reported only minor faceplate shrinking, cracking and bark falling off the plate, indicating host-species selection is important when choosing carved chainsaw hollows as a hollow replacement strategy. Furthermore, Rueegger (2017) considered the occlusion of the faceplate likely increases the longevity of the hollow and thus, mechanically-created hollows are expected to have a greater longevity to that of the commonly used plywood nest boxes, which have an estimated longevity of 5–10 years (e.g. Bender and Irvine 2001; Lindenmayer *et al*. 2009). This theory would depend on the rate and extent of wound-wood occlusion, as occlusion that partially- or wholly- covered the hollow entrance would have obvious consequences for access by fauna (and create perhaps unintended competitive advantage/disadvantage between species). Such occlusion also may influence moisture content of the internal cavity and/or the extent to which water could be prevented from entering the hollow – or drain from it. The moisture content within the cavity may have serious implications for hollow-occupants (see below).

Occlusion of entrance holes by wound wood may limit the long-term usefulness of chainsaw hollows, although it is likely research will advance to consider how the process of wound wood production can be harnessed to facilitate the creation of effective chainsaw hollows that do not become occluded. Based on current approaches, Rueegger (2017) expected wound wood growth would eventually overgrow the entire faceplate and sporadic maintenance is likely required to ensure hollow entrances remain open. Rueegger (2017) hypothesised that inserting a length of conduit into the entrance hole (e.g. a piece of PVC pipe with a roughened internal wall to provide a foothold for the animal), may prevent or prolong entrance occlusion. Alternatively, killing hollow host trees once the faceplates are partly occluded may be considered to prevent ongoing maintenance (Rueegger 2017), although this has obvious implications for the tree to provide other resources (e.g. nectar, pollen, leaves for invertebrates, etc.). Also, the longevity of chainsaw hollows only can only be determined through long-term monitoring.

As well as consideration of moisture within the carved hollows, consideration needs to be given to the effect of the hollow host trees' wound wood development on hollow faceplates, among other research questions. Indeed, there are demonstrable problems associated with using carved hollows as substitutes for natural hollows, not least the increased risk of host-tree collapse or failure (e.g. during a strong wind event). One study generated a 37% failure rate of host trees within the first year of hosting a chainsaw cavity (Carey and Gill 1983), although Rueegger (2017) reported no tree failures.

In Rueegger (2017), the creation of one hollow took approximately one hour. The time it takes for hollow creation will depend on access and hollow location on the host tree. However, it is plausible that created hollows have a similar initial cost to that of the combined expense of habitat box purchase and installation, with created tree hollows likely outlasting plywood boxes and so, potentially outperforming boxes economically mid to long-term (Rueegger 2017). However, the need to undertake regular maintenance to ensure entrances remain open (see Griffiths *et al.* 2023) may mean the long-term cost of chainsaw hollows is greater than that of nest boxes.

While results are promising for chainsaw-carved hollows to provide an alternative to nest boxes, more research into many aspects of chainsaw carved hollows is required before incorporating such hollows into any hollow replacement strategy or seeing them routinely adopted in habitat restoration.

2.5 Chainsaw-carved bat fissures

In a study in six reserves across Greater Melbourne, 174 fissures designed to provide bats with day roosts were carved by chainsaw into several *Eucalyptus* and *Corymbia* species (Griffiths *et al*. 2023). The fissures (see below) were based on a commonly installed design of chainsaw carved cavity used to provide supplementary habitat for bats (Griffiths *et al*. 2023). The design involves using a chainsaw to carve a 'fissure-type' entrance directly into a trunk or branch; the fissures are designed to mimic the cracks and crevices that naturally form in trees, and which provide roosts to some Australian bat species (Lumsden *et al.* 2020). The fissures were monitored regularly for six years, yielding valuable results on chainsaw-carved bat fissures – occupancy patterns and ongoing maintenance requirements – to consider how carved bat fissures could be used effectively to support bat populations (Griffiths *et al*. 2023).

2.5.1 Issues

Griffiths *et al*. (2023) determined, at one site, some fissures were completely closed within three months of installation as they had filled with kino flow (when exposed to air and sun, kino rapidly dries to form a hard gum). At that same site, of the 61 carved fissures that still had open entrances, no bat activity (direct or indirect) was recorded – no bats used the fissures, including for roosting during the day (Griffiths *et al*. 2023). Within several years, all 74 of the carved fissures installed at that site had been completely closed over by wound wood (Griffiths *et al*. 2023). At other sites within the study, similar results were observed. Within eight – 41 months after installation, all 100 of the installed bat fissures had been completely closed by kino and wood wound (Griffiths *et al*. 2023). No bat activity (direct or indirect) was observed in any of the created fissures whilst they were open and therefore available (Griffiths *et al*. 2023). In summary, the study determined the carved bat fissures, commonly installed to provide supplementary habitat for crevice-roosting bats, had not provided habitat to any bats during the intensive six-year monitoring period. Many fissures had closed within three-eight months and within five years, despite maintenance attempts, entrances to all 174 installed fissures were completely closed by wound wood growth (Griffiths *et al*. 2023). These findings suggest fissure cavities carved into live trees should be surveyed at least annually for the first three years to monitor kino and sap production and that maintenance would be needed to remove wound wood for some (if not, all) fissures within 3-4 years of installation (Griffiths *et al*. 2023).

Griffiths *et al*. (2023) raises an important point for stakeholders considering installation of carved cavities – whilst the price of installing such cavities is less than installing traditional bat boxes, those reduced costs become irrelevant if the cavities are not used by bats and the fissures become unavailable over time. Such a programme delivers no returns on investment from a bat conservation or habitat restoration perspective. The study demonstrates the importance of post-installation monitoring and of publishing results of these experiments, as further empirical evidence from studies investigating the effectiveness of bat fissures (or any hollow alternatives) is critical to avoiding broad-scale (and expensive) roll-outs of hollow alternatives that may not deliver positive conservation outcomes for target species and may, in turn, waste limited resources (Griffiths *et al*. 2023).

2.6 Hollow salvage for log hollows

Hollow trunks or branches (log hollows) may be salvaged and attached to trees or poles to provide a short-term solution for a range of species (Goldingay and Stevens 2009). The re-use of large hollows has been demonstrated to provide higher potential for uptake success by comparison to artificial nest boxes (Central Coast Council 2016). Council monitoring of a relocated hollow section and large nest boxes designed for owls placed at Wadalba NSW showed the hollow received regular activity and use over the winter period from Sulphur-crested Cockatoo (nesting), Barn Owl, Common Brushtail Possum, Australian King Parrot, Sugar Glider, Feather-tailed Glider, Galah and the Southern Boobook owl (Central Coast Council 2016). By comparison, the nest boxes installed in those same areas received no owl activity and the activity by remaining species was less diverse and inconsistent (Central Coast Council). Powerful Owl, in particular, shows preference for natural hollows, suggesting the use of salvaged hollows may benefit Powerful Owls.

Central Coast Council (2016) provides an excellent, detailed guideline for relocating large tree hollows (specifically for potential use by Powerful and Masked Owls based on Council's limited field trials of Powerful Owl use of salvaged hollow trunks), providing guidance for the relocation of 1) a large hollow section and 2) a complete tree section into a living recipient tree. The Guidelines recommend considering relocating a hollow trunk in the following situations:

- Very large and heavy trunk hollow sections that are too big to be supported in the canopy of a recipient tree.
- To simulate a complete natural looking trunk section surrounded by a living canopy to enhance Powerful Owl habitat.
- If the live large hollow is high and (because height is important for Powerful Owl) the height is to be maintained.
- The section contains several high-quality hollows that can be moved together in the same section.

The Guidelines also suggest: *'If targeting Powerful Owl, the relocation of a complete trunk section would be a more suitable outcome as this method is a more natural looking result. If targeting Masked Owl, the relocation of a complete trunk section may also be a suitable option if the cavity of the hollow extends a long way down the trunk, as this species is known to utilise deep trunk hollows'* (Central Coast Council 2016). The Guidelines are specifically aimed at hollow recovery for Powerful and Masked Owls and do not consider the effectiveness of hollow salvage for other hollow-using species. Indeed, very little consideration in the literature has been given to using this approach for salvaged natural hollows or other hollow-using taxa. This may be because natural hollows have value as hollow logs on the ground.

Consideration should be given to any studies not included in this review that show results of use of salvaged natural hollows, as it is possible (though unlikely) salvaged hollows have thermal qualities that may be closer to the microclimate of natural hollows in live trees (compared to, say, that of plywood nest boxes). However, the humidity of the salvaged hollow attached to a live tree would not be microclimatically regulated (compared to a naturally-occurring hollow in a live tree), as the microclimatic conditions in natural cavities are regulated by biological processes within the live tree.

Some studies consider the use of salvaged logs that have chainsaw hollows created within them (e.g. Griffiths *et al.* 2018) but Griffiths *et al.* 2018 demonstrate the considerations of using chainsaw-carved log hollows are akin to those of other chainsaw-carved hollows. Interestingly, sections of felled trees containing hollows can be retained and used successfully by wildlife shelters to provide hollow resources to displaced, injured or rehabilitating wildlife (pers. obs.).

2.6.1 Issues

The weight of natural hollows (including sections of trunk or branch above and below the hollow) may mean log hollows are difficult to move, lift into place and mount. Indeed, even storing the hollows can be challenging, as wildlife often will swiftly occupy the hollows and be impacted when the hollows are moved (pers. obs.). The weight of the hollow sections, coupled with attachment methods, may mean the log hollows fail within a similar (or sooner) time frame to nest boxes (i.e. less than a decade in some cases, e.g. Lindenmayer *et al.* 2009). This has clear implications for wildlife. This also presents significant safety issues for humans visiting and/or managing the area, posing a potential (and likely: unacceptable) public risk liability for land managers. Installing an adequate diversity (e.g. size, entrance holes, etc.) of log hollows onto standing trees, at an effective height, would also present health and safety issues and require specialised platforms or cherry pickers. As well as the financial and time costs, the infrastructure required to install the log hollows may result in unintended environmental damage. Nonetheless, a research project investigating the use of hollow sections would provide valuable insights into the potential use and effectiveness of this important salvaged-hollow resource as a hollow alternative.

2.7 Erecting dead trees and utility poles

In a unique study, Hannan *et al.* (2019) trialed erecting five dead trees and five utility poles in a highly degraded landscape devoid of natural hollows (Figure One), to consider the ability of these structures to offset the loss of mature trees. It is worth noting the erected dead trees (50–80 cm DBH) (or snags) used in this study were removed from adjacent suburbs for safety reasons, translocated to and installed at the site, and enriched with four log carved nest boxes (100 mm, 80 mm, 50 mm front entry diameters, and a bottom entry bat box). Wooden utility poles also were translocated, installed at the site, and enriched with perches (eight wooden cross beams) and four carved nest boxes with the same dimensions as those boxes installed on dead trees (Hannan *et al.* 2019). These nest boxes were *attached* to the trunk and branches of the dead trees, and to the utility pole (Figure One).

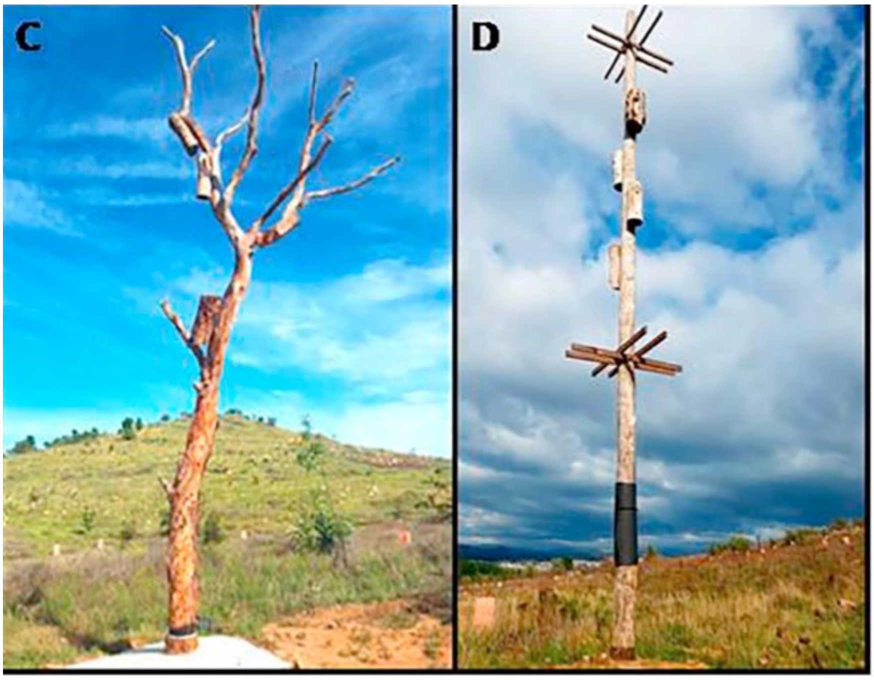


Figure 1 Nest boxes were attached to the truck and branches of a translocated and erected dead tree and to utility poles to increase bird richness in a landscape lacking mature hollow-bearing trees (Hannan *et al.* 2019).

The authors found a significant increase in bird species richness where utility poles or dead trees were erected, with no significant change at control sites or at living mature trees, indicating the erected dead trees and utility poles were providing novel resources in the highly degraded landscape. Whilst the authors reported: 'Nest boxes added to dead trees and utility poles were utilised as nest sites by several bird species', no data were presented to support this finding (Hannan *et al.* 2019). Erecting dead trees provided the greatest gain in bird species richness and was more cost-effective than erecting utility poles. Also, the erected dead trees help address the time lag between losing the mature hollow-bearing tree and the establishment of hollows in newly planted seedlings. However, the study found erected dead trees did not support as high diversity of bird species as living mature trees. Further, 37% of the species observed in the study occurred exclusively at living mature trees, indicating the erection of dead trees and utility poles is (at best) a partial solution only to offsetting the loss of mature, hollow-bearing trees (Hannan *et al.* 2019). That erected dead trees and utility poles do not flower or produce other floral resources further limits the offset potential for erected dead trees and utility poles to offset the loss of living mature trees. It is worth noting this study was conducted in a highly modified landscape where mature trees and hollows largely were absent; evidence of use of hollow alternatives in landscapes where naturally-occurring hollows are present would suggest this approach would have little-no benefit in old-growth, hollow-abundant landscapes.

2.7.1 Issues

The erection process (Hannan *et al.* 2019) involved an excavator digging a footing and inserting a metal cylinder, a truck pouring concrete into the footing, a truck transporting the structure to the nearest road access, and a crane lifting the structure off the truck and placing it in the metal cylinder. Sites where dead trees and utility poles were erected also were subject to disturbance of the ground by machinery. In largely intact landscapes, such as the VMFRP area, the disturbance to vegetation would be highly impactful and unacceptable. Furthermore, the study was restricted to establishing just five replicates of each treatment due to logistical difficulties and the cost of the project, resulting in a total of 20 observational units. Unsurprisingly, the hollow alternatives were similar to nest boxes and carved hollows in terms of issues.

2.8 Stimulation of hollow development in trees

Hollows may be replaced by other means such as inoculation of trees with fungi. Yet more methods include the use of explosives, poisons, girdling, topping by chainsaw and ringbarking (e.g. Horner *et al.* 2010; Gibbons *et al.* 2000; Wainhouse and Boddy 2022). The use of fire to accelerate hollow formation in *Eucalyptus* species also was found to be an option (Adkins 2006).

2.8.1 Issues

Whilst these methods may stimulate hollow development, there may be a lag time of decades for hollow availability which may increase likelihood of (at least, local) extinctions of hollow-dependent fauna.

2.9 Entrance creation to access naturally-occurring tree cavities

Many landscapes across Australia are dominated by *Eucalyptus* species, or species from the closely related *Angophora* or *Corymbia*, both of which genera used to be considered part of *Eucalyptus.* There is much evidence these trees contain internal cavities (voids or areas of internal decay) that lack external access. Interestingly, a study of Bimble Box *Eucalyptus populnea* found all stems greater than 19mm diameter at 30cm above ground were hollow (Harrington 1979) yet Bimble Box need to be 30-40cm Diameter at Breast Height (DBH) to have a 50% probability of having external openings greater than 1cm (Rayner *et al.* 2014). With their average growth rate, this means a tree of around 125-160 years old has a 50% probability of having external access greater than 1cm to these internal hollows. Similar findings demonstrating the discrepancy between the presence of internal hollows and lack of external access exist also for the widespread River Red Gum *Eucalyptus camaldulensis* (Ellis 2018). Ellis (2018) inspected recently felled River Red Gums and found that heartwood decay, a precursor to hollow formation, was common even in relatively small trees but that internal cavities remain inaccessible until stochastic damage exposes them, which may not occur until the tree is old (and therefore likely: large) (Ellis 2018). West (2015) also reports this is common among other species. These studies indicate the prevalence of access to hollows by hollow-dependent fauna may not be limited by hollows in the landscape *per se*, but rather by *external openings* to the otherwise inaccessible hollows.

In a recent 'proof of concept' study in a regenerating Australian landscape, Ellis *et al.* (2022) accelerated availability of natural but inaccessible hollows by mechanically creating entrances in tree stems that have existing voids/cavities or internal decay but have not yet developed entrances. Exploratory drilling (10mm diameter holes) was used to determine the presence of internal cavities or decay; if a cavity or decay was located, an entrance hole was drilled at a size appropriate for the size of the potential cavity. Entrance holes were drilled 2.4-4.8m above ground level. The results were promising – camera traps showed drilled entrances were investigated or used within hours of creation, and all 39 drilled holes were used by animals during the study period, including by several species of marsupial recorded importing nesting material into the newly accessible cavities. The study provides evidence that creating entrances to otherwise inaccessible cavities has potential to accelerate development and occupancy of habitat for hollow-utilising fauna (Ellis *et al.* 2022), immediately addressing the issue of lag time for hollow availability as well as avoiding a number of issues associated with constructed hollow 'replacement' options. Unlike many artificial cavities, the approach used in Ellis *et al.* (2022) does not target particular animal species; instead, the approach taken seeks to emulate in a regenerating landscape an ecological characteristic typical of old-growth vegetation. The internal cavities to which access is created have formed naturally, so are likely to be diverse in character, unlike nest boxes or chainsaw hollows which are constructed to set dimensions and take the form of non-organic shapes. The unique qualities of the naturally-formed internal cavities may explain the wide range of taxa recorded using the entrances (Ellis *et al.* 2022).

Ellis *et al.* (2022) found the following:

- drilled entrance holes are a viable way of providing access to existing internal tree cavities in living and dead trees that have not yet developed external entrances;
- a wide range of fauna will rapidly make use of such access;
- fauna are capable of modifying the cavities through excavation of debris or addition of nesting materials.

Rapid use of drilled entrances (Ellis *et al.* 2022) and artificially excavated cavities (Rueegger 2017) by Feathertail Gliders *Acrobates pygmaeus* clearly shows that activities associated with cutting or drilling into trees do not deter some species from immediate visitation.

2.9.1 Issues

From Ellis *et al.* 2022:

Until late 2019, wound wood growth was restricted to the perimeter of drilled entrance holes and all entrances remained open. In contrast, from 2020 to 2021 rapid growth started to occlude half of the holes. This varied among trees and was not related to time of creation, with two holes created in September 2019 almost totally grown over by May 2021.

It is unsurprising that wound wood growth led to partial or total occlusion of created holes, as this was observed in many studies of chainsaw hollows and indeed, is observed in trees to heal wounds including fallen limbs. The implications of the occlusion of the created hollows are a) that holes might need to be redrilled regularly (which has potential wildlife implications) and/or that holes are left in states of partial/total occlusion (also has wildlife implications) and new holes are periodically drilled into the same cavities. The latter approach also has wildlife implications as many hollow-dependent birds are known to return (seasonally, annually, intergenerationally) to hollows for shelter and breeding. That said, natural hollow entrances can close and new entrances to an old hollow may be created by a stochastic event.

The approach of creating access to internal cavities should be seriously considered as a method of making hollows available in a landscape where hollows have been removed or where there is a paucity, but using methods that harness the trees' natural healing processes to reduce the likelihood (or at least, rate) of occlusion of the entrance hole. For example, if a limb is removed close to the trunk, the wound will be fully occluded (pers. obs.) to prevent disease entering the trunk and heartwood. However, if a limb is cut further from the trunk (~50cm), the tree will heal in such a way that enables the 'stub' of the limb to hollow out into a 'spout', which may eventually provide access to internal cavities (pers. obs.). It is highly recommended consideration be given to how the approach taken in Ellis *et al.* (2022) can be modified to mimic stochastic events which lead to long-term natural access points to internal cavities. Indeed, indigenous knowledge may well provide answers to how to create entrance holes that remain open.

3 Occupancy patterns

3.1 What is known about occupancy patterns in hollow alternatives?

3.1.1 Nest boxes

In a 10-year experimental study of Leadbeater's Possum in the Central Highlands of Victoria, Lindenmayer *et al.* (2009) found that six species of mammals and two species of birds occupied the 96 nest boxes established for this study. The species occupying nest boxes were: Leadbeater's Possum, Mountain Brushtail Possum (*Trichosurus cunninghami*), Common Ringtail Possum (*Pseudocheirus peregrinus*), Sugar Glider (*Petaurus breviceps*), Eastern Pygmy Possum (*Cercartetus nanus*), Crimson Rosella (*Platycercus elegans*), and the Owlet Nightjar (*Aegotheles cristatus*). Only three of these species used nest boxes reasonably frequently in the 10 years of the study. These were the arboreal marsupials Leadbeater's Possum, Mountain Brushtail Possum, and Common Ringtail Possum. This study also confirmed a very strong forest age effect, where almost all incidences of occupancy were recorded in young forest. During the ten-year monitoring, only two of the 48 nest boxes (4.1%) established in 1939 (post fire) regrowth ('old') forest were occupied compared to 28 of the 48 nest boxes (58.3%) occupied in 20-year regrowth ('young') forest. Indeed, in old forest, 46 of 48 nest boxes (=95.8%) remained unoccupied for the 15 checking events over the 10 years of the study. The strong effect of forest age on nest box occupancy indicates that nest boxes may play a role in addressing the paucity of naturally-occurring trees with hollows in young forests but are unlikely to be effective where naturally-occurring hollows are extant, such as in old growth ecosystems like those within the VMFRP area (see below).

Lindenmayer *et al.* (2009) concluded the relative paucity of trees with hollows in young forest meant animals had few (if any) alternative nesting sites to those in nest boxes. The authors found little evidence to suggest factors other than the availability of trees with hollows were limiting populations of arboreal marsupials in young montane ash forests. For example, other studies conducted by the authors have shown these animals can occur in young forests if suitable numbers of large old hollow-bearing trees are retained (Lindenmayer *et al.* 2003). Indeed, Leadbeater's Possum at Yellingbo demonstrated (almost immediately) increased breeding rates when additional nesting boxes were installed (Harley, D. pers. comm.), indicating a lack of nesting hollows (natural or artificial) is the key factor restricting breeding potential. For this species at this site, nest boxes release a population bottleneck that results from lack of adequate cavities.

A similar result was found in a nest box study in degraded Box-Ironbark forest habitat in Central Victoria (Goldingay *et al.* 2020b), where the endangered Brush-tailed Phascogale (*Phascogale tapoatafa*) and more secure Sugar Glider (*Petaurus breviceps*) co-occur. Historic gold mining, forestry and community use throughout the selected study area meant hollows were scarce (Goldingay *et al.* 2020b) – 67% of the random plots across the study landscape supported no hollows that matched the criteria for the focal species (see Goldingay *et al.* 2020b for details on hollow requirements for these species). As anticipated for a landscape of hollow-scarcity, Goldingay *et al.* (2020b) observed repeated use of the 40 installed nest box sites by both species over the three-year study. The Phascogale (at least one individual, but up to three individuals) was observed in nest boxes at 17 sites, with empty nests observed in boxes at another 13 sites (Goldingay *et al.* 2020b). Sugar Gliders, having wider habitat and biological requirements, were observed in nest boxes at 39 sites (plus three nesting sites) across the three years, including 20 sites where they were observed in all four surveys (Goldingay *et al.* 2020b), indicating regular occupation. In the final survey year, the Sugar Glider was observed at 37 sites and nests recorded at an additional three sites. In the same landscape, the Whroo Goldfields Catchment Management Network (WGCMN) have installed at least 600 nest boxes targeting Phascogales and Sugar Gliders; in 2009, the boxes showed nine per cent occupancy (mainly by sugar gliders) and in 2014, the WGCMN recorded 54% active occupancy with one in five (22%) of nest boxes housing Phascogales (Goulburn Broken Catchment Management Authority, unpublished data via www.gbcma.vic.gov.au). Goldingay *et al.* (2018) provides a wonderful review of a range of occupancy rates, including from a study by Harley (2006) that reported 75% of 150 nest boxes targeting Leadbeater's Possum at Yellingbo, Victoria were used by that species.

Nest boxes were designed and installed specifically for large owls and large parrots in New South Wales, but none have been used by these species (Goldingay *et al*. 2020a). Similarly, nest boxes of a generalised design installed for the Brown Treecreeper rarely have been used (Goldingay *et al*. 2020a). However, Brown Treecreepers also were recorded breeding in spout hollows (Strain *et al.* 2021) and one study recorded 48 breeding events by Brown Treecreepers in vertically installed log hollows with entrance sizes of 4–15 cm and depths of 20–57 cm (see Goldingay 2021). The wide range of entrance sizes and depths suitable for Brown Treecreepers indicates they have wide hollow preferences, suggesting providing an adequate abundance of hollows for small and large gliders also should provide for this species.

3.1.2 Spout boxes

Strain *et al.* (2021) observed birds and mammals using both natural and artificial spout hollows during their study in Central Victoria. The target species for the study were Turquoise Parrot and Brown Treecreeper. One hundred and twenty-five (125) spout boxes were attached to trees and pickets; activity in the spout boxes and natural hollows were compared (Strain *et al.* 2021). Ten Turquoise Parrot breeding events were detected in spout boxes (eight on pickets and two on trees), and four were detected in tree hollows. Ten Brown Treecreeper nests also were found in spout boxes (seven on pickets and three on trees), whereas none were detected in tree hollows. This represents an 8% occupancy rate within spout boxes for both focal species in Strain *et al.* (2021). This is a promising result, given Brown Treecreepers occupied only 0-0.6% (two) of 313 surveyed plywood nest boxes as part of a large offsetting programme in New South Wales (Lindenmayer *et al.* 2017). Ironically, neither of the two records of brown treecreeper were from a box designed for the species, despite the offset programme providing 77 nest boxes specifically designed for that species (Lindenmayer *et al.* 2017). This demonstrates the complexity (and ultimately, often, inefficiency) of matching hollow replacements to the (generally, little understood) habitat requirements and preferences of target fauna.

Strain *et al.* (2021) also detected in spout boxes, nests belonging to three other hollow-nesting bird species: White-throated Treecreeper *Cormobates leucophaea* (n=2), Red-rumped Parrot *Psephotus haematonotus* (n=1), and Australian Owlet-nightjar *Aegotheles cristatus*, (n=1). A small marsupial, the Yellow-footed Antechinus *Antechinus flavipes*, also was detected sheltering in a single spout box (Strain *et al.* 2021). It is noteworthy that spout boxes attached either to trees or pickets were used for successful breeding by Turquoise Parrots and Brown Treecreepers – indeed, the spout boxes were used more often for breeding than were natural hollows (Strain *et al.* 2021), though the sample size was small and perhaps not statistically robust. Nonetheless, Strain *et al.* (2021) highlights the need for nest box programs to incorporate empirical data on the physical characteristics of natural hollows used by target species into the design, construction, and installation of artificial hollows.

3.1.3 Chainsaw-carved hollows

In a study located on the southeast coast of NSW, Rueegger (2017) documented five hollow-using vertebrate species using chainsaw hollows: Feathertail Glider (*Acrobates pygmaeus*), Brown Antechinus (*Antechinus stuartii*), White-throated Treecreeper (*Cormobates leucophaea*), Sugar Glider (*Petaurus breviceps*) and Long-eared Bat (*Nyctophilus* sp.). Camera monitoring documented two additional species that inspected the hollows: Crimson Rosella (*Platycercus elegans*) and Sacred Kingfisher (*Todiramphus sanctus*) (Rueegger 2017). Best *et al.* (2022) conducted a study across five Melbourne reserves and recorded 17 hollow-dependent vertebrate taxa (13 native and four introduced) visiting a subset of 40 chainsaw hollows. Over 35,000 visitations were recorded during the study by Best *et al.* (2022); of these, 7,331 were attributed to five native mammals: Krefft's Glider (formerly Sugar Glider) (6,837), Common Ringtail Possum (271), Common Brushtail Possum (171), and Agile Antechinus *Antechinus agilis* (42) [and an unidentified bat]. Visitations by tree-roosting insectivorous bats also were recorded at four chainsaw hollows (Best *et al.* 2022). Best *et al.* (2022) determined it was not possible to identify individual bats to species level, however, based on morphology all bats were classified as Vespertilionidae (Churchill 2008). The highest visitation rates recorded in Best *et al.* (2022) were by native hollow-dependent parrots (Psittacidae): Rainbow Lorikeet *Trichoglossus moluccanus* (10,866), Eastern Rosella *Platycercus eximius* (10,067) and Crimson Rosella *Platycercus elegans* (4523). In addition, one hollow-dependent cockatoo (Cacatuidae), one kingfisher (Halcyonidae) and three passerines also visited chainsaw hollows, but at much lower rates: Galah *Eolophus roseicapilla* (148), Laughing Kookaburra *Dacelo novaeguineae* (4), Spotted Pardalote *Pardalotus punctatus* (12), Striated Pardalote *Pardalotus striatus* (3), and White-throated Treecreeper *Cormobates leucophaea* (11) (Best *et al.* 2022).

Overall, Rueegger (2017) found Feathertail Gliders and Brown Antechinus used 12 different hollows each (75%), Sugar Gliders 10 (63%), Long-eared Bats eight (50%) and White-throated Treecreepers four (25%). The hollows were used for rearing of young by Sugar Gliders (10 hollows) and White-throated Treecreepers (two hollows) (Rueegger 2017). Similarly, Best *et al.* (2022) found that Krefft's Gliders (formerly Sugar Gliders) occupied 60% of the chainsaw hollows in that study at least once. The proportion of available chainsaw hollows occupied by Krefft's Gliders per survey was 9.7% ± 10.1% (range = 0–42%). For the 29 chainsaw hollows used as dens by Krefft's Gliders, the mean number of times they were occupied over the 14 surveys was 2.2 ± 2.0 (range = 1–11) (Best *et al.* 2022).

Camera records in Rueegger (2017) documented first hollow inspections only one day after hollow creation by White-throated Treecreepers and Feathertail Gliders, 14 days by Crimson Rosellas, 19 days by Brown Antechinus, 31 days by Sacred Kingfisher, 39 days by Sugar Gliders and 65 days by Long-eared Bats. A pair of White-throated Treecreepers was recorded to start nest building only three days after hollow creation (Rueegger 2017). During the first physical hollow inspection, 51 days post hollow creation, two White-throated Treecreeper chicks were found in the hollow (Rueegger 2017). A year later, three chicks were found in a different hollow (Rueegger 2017). Prolonged hollow use was documented for Sugar Gliders and for White-throated Treecreepers when rearing chicks. The longest continuous hollow occupation for Sugar Gliders was 11 months (documented through camera monitoring) (Rueegger 2017).

Best *et al.* (2022) considered chainsaw hollows in that study were used mostly as intermittent, secondary dens. Krefft's Gliders did, however, show high fidelity to three chainsaw hollows. Camera trap data confirmed that Krefft's Gliders used two of these chainsaw hollows to raise young during the monitoring period; each comprised a family group with two juveniles that survived to weaning (Best *et al.* 2022). Agile Antechinus were recorded in one chainsaw hollow during two occupancy surveys: three individuals were recorded in May 2017 and four individuals in June 2018 (Best *et al.* 2022). No further records of this species were made on subsequent occupancy surveys, however passive camera trap monitoring revealed Agile Antechinus investigating the same chainsaw hollow, plus one other nearby chainsaw hollow (~30 m away) on multiple occasions (Best *et al.* 2022). Whilst a range of native birds, including parrots, visited the chainsaw hollows, no native birds were recorded as occupying any of the 40 monitored cavities (Best *et al.* 2022 – Table Two).

Sugar Gliders indicated a preference for chainsaw hollows comprising of small entrances (see Rueegger 2017). White-throated treecreepers showed a preference for large entrance hollows, using none of the small entrance hollows (Rueegger 2017). The frequency of use between small versus large entrance hollows was not significant for Feathertail Gliders or Brown Antechinus (Rueegger 2017). Similarly, no hollow entrance preference was shown by Long-eared Bats with bats using five small and three large entry hollows (Rueegger 2017). Bats used only one of the six hollows designed for bats but seven of the 10 designed for marsupials/birds (Rueegger 2017), showing a similar finding of species using boxes designed for other species preferentially compared to boxes designed for that species (Lindenmayer *et al.* 2017).

Chainsaw hollows installed in suburban trees that did not naturally support hollows were shown to increase by 100% visitation and inspection by hollow-dependent fauna compared to visitation to those trees prior to the installation of chainsaw hollows (Griffiths 2019). Over 3,951 camera-trap nights, Griffiths (2019) recorded 439 tree visitation events by hollow-dependent taxa. Of these, 385 visitations were identified to four species of nocturnal mammals: Common Brushtail Possum *Trichosurus vulpecula*, Sugar Glider *Petaurus breviceps*, Common Ringtail Possum *Pseudocheirus peregrinus*, and Agile Antechinus *Antechinus agilis*). One hundred and thirteen (113) visitations were identified to seven species of diurnal cavity-nesting birds: Crimson Rosella *Platycercus elegans*, Eastern Rosella *Platycercus eximius*, Rainbow Lorikeet *Trichoglossus moluccanus*, Galah *Eolophus roseicapilla*, Laughing Kookaburra *Dacelo novaeguineae*, Spotted Pardalote *Pardalotus punctatus*, and Wood Duck *Chenonetta jubata* (Griffiths 2019).

In a Central Victorian study comparing chainsaw carved hollows and nest boxes (Terry *et al*. 2021), Brush-tailed Phascogales and Sugar Gliders were the most frequently encountered species utilising the cavities, accounting for 96% of cavity detections (nests, scats and animals present). A pole mounted camera was used to monitor the cavities. Phascogales and Sugar Gliders used 32% and 84% of the chainsaw hollows respectively, and 21% and 82% of the nest boxes (Terry *et al*. 2021). Phascogale maternal nests were recorded inside four chainsaw hollows during the 2.5-year monitoring period (Terry *et al*. 2021). A Brush-tailed Phascogale was detected by camera trap inspecting the exterior of the chainsaw hollow on the first evening after installation and a Sugar Glider was photographed entering a cavity four days after installation (Terry *et al*. 2021). These results indicate the target species, the Phascogale and Sugar Glider, both will use chainsaw carved cavities. Indeed, detection of both Phascogales and Sugar Gliders was higher in the chainsaw hollows compared to nest boxes, despite the internal dimensions being the same for both types of cavity (Terry *et al*. 2021), indicating some preference for chainsaw carved hollows (cf. nest boxes) was exhibited. The apparent preference for chainsaw hollows over nest boxes may be due to chainsaw hollows providing a more stable and lower thermal environment compared to nest boxes (Griffiths *et al.* 2018), at least in this study (see Thermal properties). Chainsaw hollows also may better mimic the physical appearance of natural hollows and allow easy access and less exposure to predators than nest boxes (Terry *et al*. 2021). The higher detection rate in chainsaw cavities compared to nest boxes by the Brush-tailed Phascogale suggests the installation of chainsaw hollows could be an important method for restoring habitat quality for this threatened species (Terry *et al*. 2021), though the abundance of natural hollows may affect occupancy rates and preferences observed elsewhere. Maternal nests of phascogales were recorded in four chainsaw hollows but none in nest boxes suggesting these cavities satisfied breeding requirements (Terry *et al*. 2021).

Terry *et al*. (2021) recorded two other small mammals showing interest in the hollows: the Yellow-footed Antechinus (*Antechinus flavipes*) and a Feather-tailed Glider. A month after installation, Terry *et al*. (2021) recorded termites present in two chainsaw hollows; one chainsaw hollow was filled entirely by the termites, rendering that hollow unusable by the study’s target species. Fungi (taxa unknown) also was recorded in one chainsaw hollow at two sites one month after installation of the chainsaw hollows (Terry *et al*. 2021).

Terry *et al.* (2021) reported that Phascogales showed a pronounced seasonal variation in detection in cavities, with higher values recorded in summer and autumn. The higher values during these seasons are likely to reflect the dispersal of individuals away from the natal nest (Terry *et al.* 2021). The strong seasonal variation in detection suggests the timing of surveys will be critical to measure the success of artificial cavities in future studies for Phascogales.

Rueegger (2017) concluded that chainsaw hollows should be installed in clusters to best support local populations of any species of arboreal mammal. The spacing of clusters would need to be guided by home range size of each species (Rueegger 2017).

3.1.4 Chainsaw-carved bat fissures

Griffiths *et al.* (2023) assessed the occupancy patterns and maintenance requirements of a chainsaw-carved bat fissures using a design commonly installed to provide supplementary habitat for bats. During occupancy surveys conducted prior to wound wood closing over the entrances, Griffiths *et al.* (2023) recorded no direct evidence of bats (or any other vertebrate species) roosting inside any of the carved fissures.

3.2 Optimal density of carved/provided hollows in a landscape

The literature clearly demonstrates how landscape (including available natural hollows) changes species’ responses to hollow alternatives. There is unlikely to be a generalized optimal density that can be used without consideration of the landscape context in which the hollow alternatives are installed. However, Lindenmayer *et al*. (2009) found the average number of hollow-bearing trees within 40m of each nest box in young forest was 0.33 – in old forest, the average number of hollow-bearing trees within 40m of each nest box was 1.25. It would be interesting to investigate the robustness of this dynamic, as the implications may influence the development of hollow replacement strategies in areas where naturally-occurring hollows are abundant and occupancy rates of these hollows are low.

3.3 Hollow-opportunistic, hollow-using and hollow-dependent taxa

The literature shows a wide range of species have been recorded visiting hollow alternatives, although many studies recorded a similar subset of species. The extent to which a species is dependent on hollows (cf. say, an occasional user of hollows) may affect that species' representation in the literature, although other factors such as breadth and nuance of habitat preferences; hollow fidelity; relative population abundance; patterns of food availability; mate availability; fitness; competitiveness, are likely to exert a significant influence over occupancy patterns reported in the literature.

3.4 Preference for different hollow alternatives by different species

Any preferences for different hollow alternatives exhibited by different species are not well documented in the literature, although Terry *et al*. (2021) reported detection of both Phascogales and Sugar Gliders was higher in chainsaw hollows compared to nest boxes, despite the internal dimensions being the same for both types of cavity, indicating some preference for chainsaw carved hollows (cf. nest boxes) was exhibited. The apparent preference for chainsaw hollows over nest boxes may be due to chainsaw hollows providing a more stable and lower thermal environment compared to nest boxes (Griffiths *et al.* 2018).

3.5 What are the repetitive breeding rates and breeding success rates (by type) and how do these compare to those of natural hollows?

The literature indicates very little is known about breeding rates and the extent to which they vary between hollow alternatives or compare to breeding rates in naturally-occurring hollows.

4 Design and installation of hollow alternatives

Design and installation of hollow alternatives varied significantly between studies (excepting high consistency for the design of nest boxes for Leadbeater's Possum). In this way, each study introduces new variables (i.e. design and dimensions) which limits the possible comparative analysis that can be undertaken between studies to determine which design and dimensions are best by species or by hollow alternative. Given the number of studies that indicated target species were observed using hollow alternatives designed for another target species (and not using the alternative designed for that species), there is a clear need for further research to be conducted into design of hollow alternatives, ideally prior to implementation of new hollow replacement strategies. Nonetheless, some designs and dimensions yielded from the literature are presented here.

4.1 Nest boxes

Leadbeater's Possum (Harley 2004; Lindenmayer *et al*. 2009):

Nest boxes targeting Leadbeater's Possum at Yellingbo and the Central Highlands were constructed boxes from marine plywood. Their dimensions were: (1) small boxes – 400 mm (height), 237 mm (width), 271 mm (depth), 51 mm (entrance hole diameter), internal volume (0.019 m^3); and (2) large boxes – 490 mm (height), 292 mm (width), 330 mm (depth), 103 mm (entrance hole diameter), internal volume (0.038 m^3). All boxes were attached with self-tapping wood-screws. The nest boxes were hinged at the top and had a drain hole in the bottom. The entrance was high on the front of each nest box. (Lindenmayer et al. 2009)

Nest box dimensions only weakly affected occupancy patterns in the Central Highlands of Victoria (Lindenmayer *et al.* 2009). In young forest, Lindenmayer *et al.* (2009) found a weak effect of box height; high nest boxes had a higher probability of being occupied than those closer to the ground. It was unclear what factors influenced occupancy for any individual species of arboreal marsupial (Lindenmayer *et al.* 2009). The study also identified: in young forests, a significant box height effect in which higher boxes were occupied earlier in the study than lower boxes; and a significant effect of site slope, in which boxes on steeper sites were occupied earlier than those on flatter sites (Lindenmayer *et al.* 2009). Data showed that nest boxes on sites with a northern aspect were occupied earlier in the study than boxes on west or south facing sites (Lindenmayer *et al.* 2009). In summary, the time to occupancy was significantly faster in high boxes and for boxes on sites with a northern aspect (Lindenmayer *et al.* 2009). This finding (i.e. that within young forests – where nest box occupancy was predominantly recorded in this study – the dimensions of nest boxes affected occupancy) is not consistent with an earlier finding by Beyer and Goldingay (2006) that fauna generally do *not* show a preference for the height at which a nest box is erected (within the range typically tested).

McComb *et al.* (2022):

Plastic nest boxes were used by McComb *et al.* (2022) for Leadbeater's Possum. The boxes (mean width, depth and height 23, 24 and 40 cm respectively) had 25 mm thick walls and a small entrance hole (51 mm diameter, Figure Two), and were attached to trees 3–4 m above the ground the mid-storey or canopy connectivity is adequate to facilitate possum movements. The nest boxes chosen for this study were predominantly light grey in colour, however a small subset were dark grey. All nest boxes have a south-east orientation to reduce exposure to direct sunlight in the afternoon.

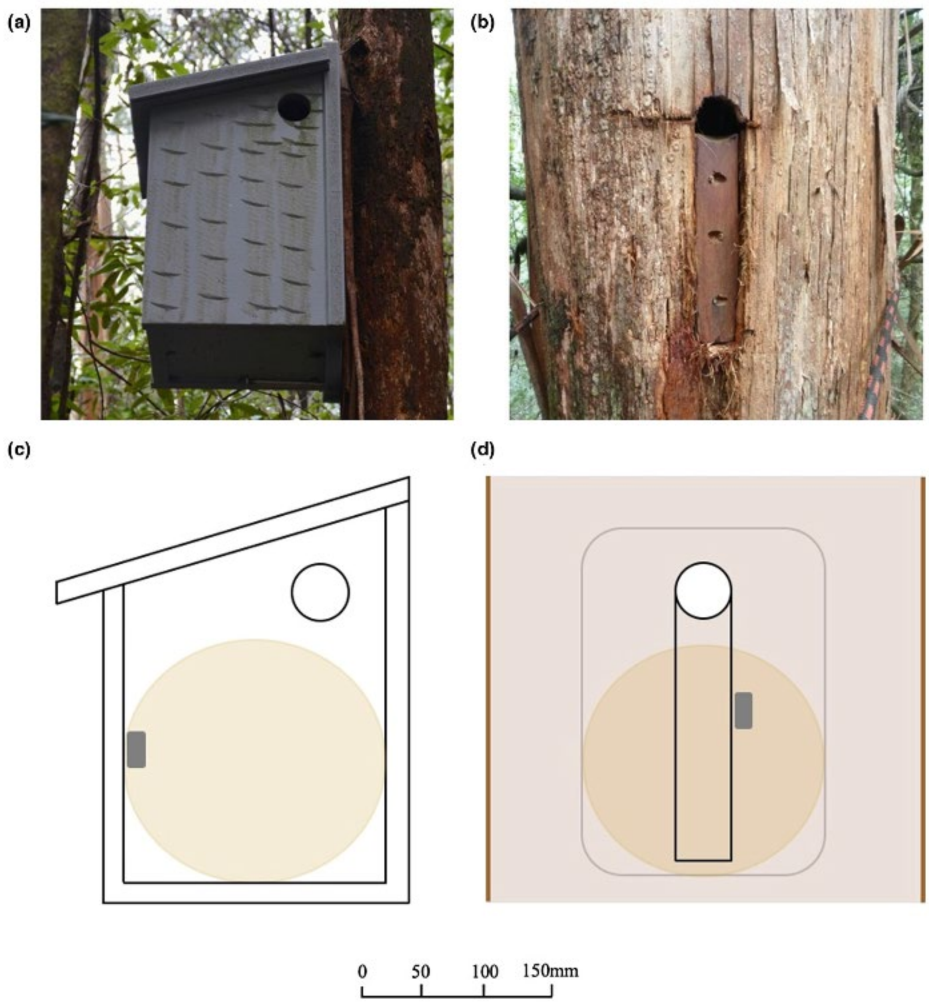


Figure 1 Photos and diagrams of the two artificial den types for Leadbeater's possums: (a, c) recycled plastic nest box (dimensions 40 cm height, 23 cm width, 24 cm depth); (b, d) excavated chainsaw hollow (av. height 30 cm, av. width 23 cm, av. depth 26 cm). Artificial den diagrams show data logger (dark grey rectangle) position on the wall of the nest box (c) and through the gap along the side of the chainsaw hollow door plate (d). The shaded circle represents the position of the large shredded bark Leadbeater's possum nest. The scale bar is indicative for diagrams (c) and (d).

Figure 2 Photos and diagrams of two artificial cavities for Leadbeater's Possum (from McComb *et al.* 2022).

Other taxa:

Rueegger (2017) found Sugar Gliders indicated a preference for hollows comprising of small entrances. Whereas White-throated Treecreepers showed a preference for large entrance hollows, using none of the small entrance hollows. The frequency of use between small versus large entrance hollows was not significant for Feathertail Gliders or Brown Antechinus. Similarly, no hollow entrance preference was shown by Long-eared Bats with bats using five small and three large entry hollows. Bats used only one of the six hollows designed for bats but seven of the 10 designed for marsupials/birds.

A nest box programme for Forty-spotted Pardalote uses plywood nest boxes installed ~4.7m above the ground. The boxes provide a 30mm diameter entrance hole and the internal dimensions of the box are120mm×120mm×220mm (Hingston 2024).

Goldingay *et al.* (2020) installed nest boxes thought suitable for a range of species (Table One).

Table 1 Eight nest box designs and their dimensions (cm) used by Goldingay *et al.* (2020).

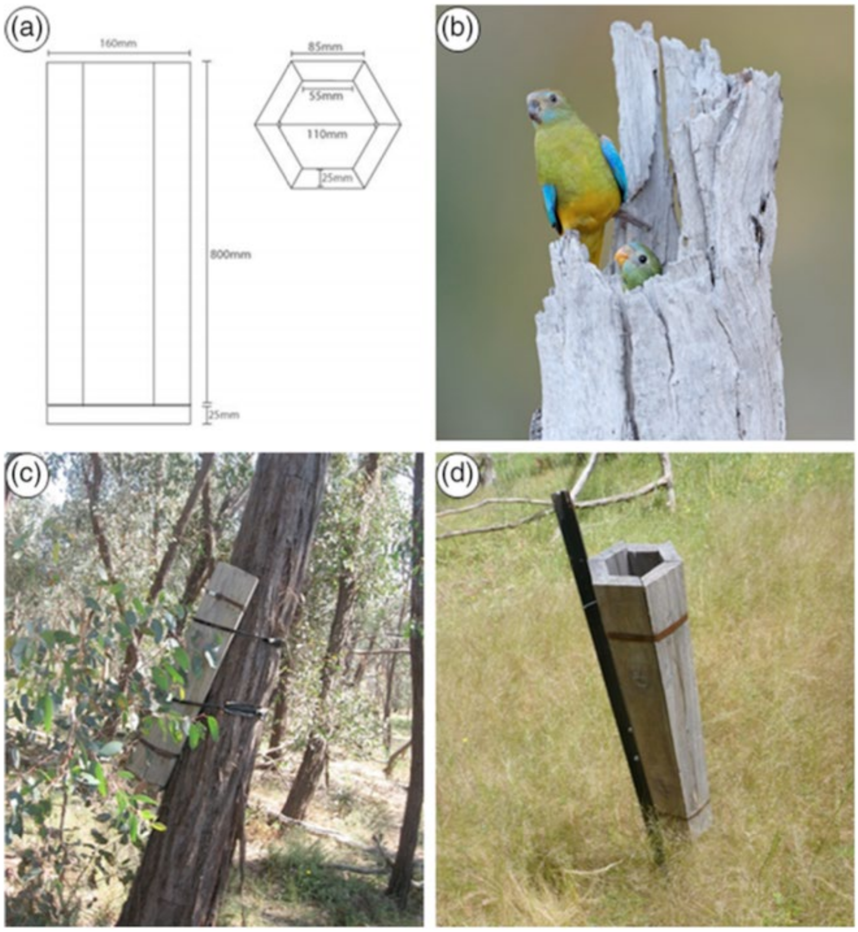
Nest box type	Entrance diameter	Width	Vertical height
Scansorial mammal	3–4	18 × 18	30
Small glider	4–5	20 × 20	30
Small parrot	6.5	20 × 20	40
Large glider	7–9	25 × 30	40
Possum	8.5–10	25 × 30	40
Small owl	10	25 × 30	50
Large owl	20	55 × 55	80
Large parrot	20	30 × 40	120

4.2 Spout boxes

Turquoise Parrot and Brown Treecreepers (Strain *et al.* 2021):

Spout boxes developed by Strain *et al.* (2021) were hexagonal wooden structures, 800 mm long by 160 mm wide (Figure Three). The spout boxes had a consistent wall thickness of 25 mm compared to the natural hollow thickness, which varied within and between each hollow from an average minimum thickness of 55 mm (range: 3–220 mm) to an average maximum thickness of 125 mm (range: 19–600 mm) (Figure Three).

The design was based on hollow preferences displayed by Turquoise Parrots and Brown Treecreepers (Quin and Baker-Gabb 1993; Strain *et al.* 2021).



Spout box measurements (a) side view and top view; a female Turquoise Parrot perched on the edge of a natural spout hollow with chick looking out of the nest (b—image credit Chris Tzaros [Birds Bush and Beyond]); a spout box attached to a tree (c); and a spout box attached to a picket (d)

Figure 3 Spout box measurements and methods of installation, from Strain *et al.* (2021).

4.3 Chainsaw-carved hollows

From Best *et al.* (2022):

Best *et al.* (2022) carved chainsaw hollows directly into tree trunks to replicate 'knot hole' cavities that form naturally at locations where branches break off and are used by a wide range of hollow- dependent fauna (Gibbons and Lindenmayer, 2002). A trapezoid prism-shaped cavity was carved into the heartwood through a rectangular opening (8 × 20 cm, width × height) cut into the outer bark and vascular (cambium and sapwood) tissue (Figure Four). Internal cavity volume across the 48 chainsaw hollows ranged from 4200 to 7644 cm^3 (mean ± SD = 5508 ± 734 cm^3) (see Best *et al.* 2022). Best *et al.* (2022) could not find any published data quantifying the internal dimensions of tree hollows used by Krefft's Gliders, however the volume of the chainsaw hollows created in that study were comparable to those of nest boxes commonly used by this species (Harper *et al.* 2005; Macak 2020). The cavities were sealed with a pre-made hardwood faceplate (8 × 20 × 3 cm, width × height × depth) and an entrance hole (~3.5 cm) was carved into the trunk above each faceplate (Figure Four) (Griffiths *et al.* 2018, 2020). Entrance holes were installed facing a range of orientations: 11 faced north, 9 east, 13 south, and 15 west. There was also some variation in the shape and size of the cavity entrance holes: 14 chainsaw hollows had rectangular entrances and 34 had semicircular entrances (Figure Four and see Best *et al.* (2022)). The entrance dimensions allowed access by Krefft's Gliders while restricting larger, hollow-using competitors such as the Common Ringtail Possum *Pseudocheirus peregrinus* (body mass 700–1100 g) and Common Brushtail Possum *Trichosurus vulpecula* (2600–4200 g). The size of the entrance holes (area, mm^2) was recorded at the time of installation, and again 32 months post-installation to quantify changes caused by wound wood growth (Figure Four). In this study, chainsaw hollows with entrances facing north or west were occupied more frequently by Krefft's Gliders than those facing south or east (Best *et al*. 2022).

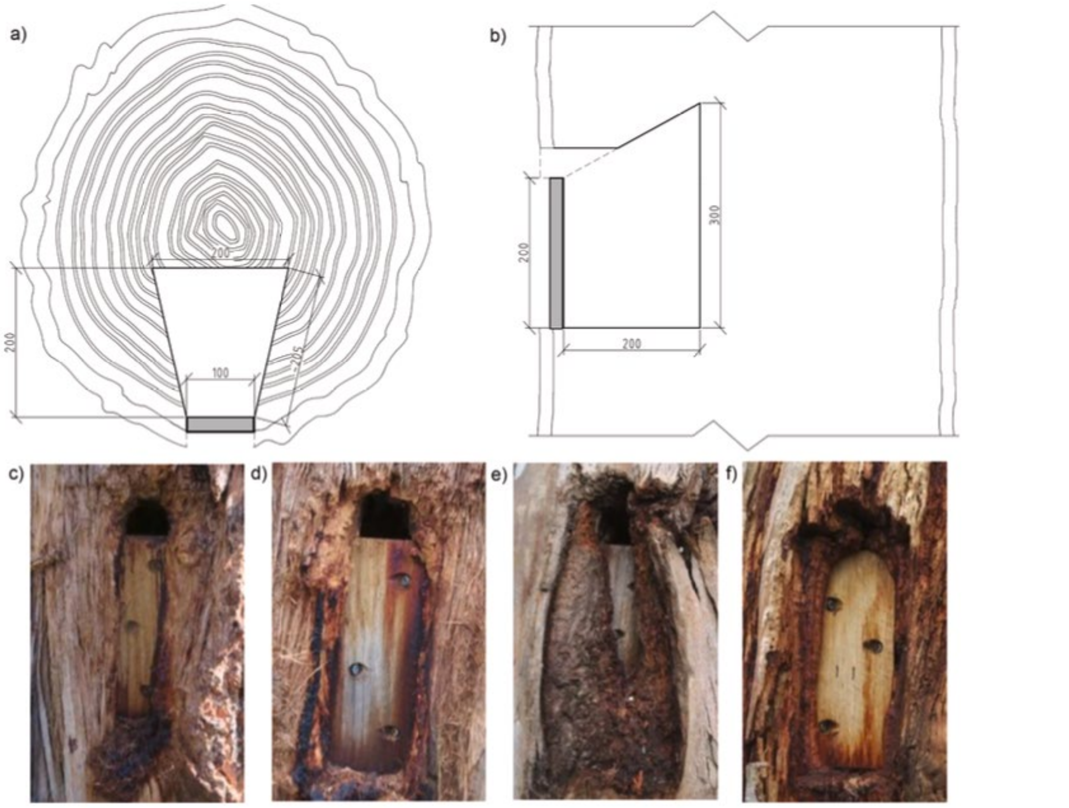


Fig. 2. Diagram of a trapezoid prism-shaped chainsaw hollow cavity showing (a) cross-section and (b) side views of the trunk (cavity dimensions shown are in mm); grey shaded rectangles represent the pre-made hardwood faceplate. Examples of chainsaw hollows with (c) semicircular and (d) rectangular-shaped entrance holes; (e) a rectangular entrance that has begun to be closed over by woundwood; (f) a semicircular entrance completely closed by woundwood. Photographs of the four different chainsaw hollows shown in (c–f) were taken in January 2019, 32 months post installation.

Figure 4 Diagram of chainsaw hollow design used by Best *et al.* (2022).

From McComb *et al.* (2022):

Chainsaw hollows had been installed to trial the suitability of these structures. They were created by arborists who cut out a large cavity (mean width, depth and height 23 cm, 26 cm, and 30 cm respectively) in large-diameter (mean = 83 cm, range = 44–170 cm) living Mountain Ash, Mountain Grey Gum (*Eucalyptus cypellocarpa*) and Shining Gum (*Eucalyptus nitens*) trees. These were fitted with a 5 cm wide wooden door plate, leaving a roughly circular 5 cm diameter entrance above the plate (see Figure Two). The hollows were installed at heights between 3 and 16 m above the ground, where there was dense mid-storey connectivity nearby. Seventy-two chainsaw hollows were installed at sites occupied by Leadbeater's Possum in 2015 and were subsequently monitored at approximately four-month intervals.

Rueegger (2017) used the method shown in Figure Five to install chainsaw hollows. Six of the 16 hollows in that study targeted tree cavity-roosting bats and ten targeted hollow-breeding birds and arboreal marsupials (Figure Five). The differences between the bat and marsupial/bird hollows were the location and size of the entrance (Rueegger 2017) (Figure Five). The entrance for the bat hollow was located near the base of the cavity to provide roost space above the entrance, whereas the entrance for the marsupial/bird hollow was located near the top of the cavity (Rueegger 2017) (Figure Five). The entrance hole for the bat hollow was 38 mm in diameter, whereas the two entrance sizes used for the marsupial/bird cavity were 38 mm and 76 mm in diameter (Rueegger 2017). The two different entrance sizes were used by Rueegger (2017) to provide habitat opportunities for a variety of different sized marsupial/bird species. Overall, the marsupial/bird hollows comprised of five small and five large entrances. All hollow entrances faced north-west. The Projected Foliage Cover (PFC) for the live trees was recorded at the time of hollow creation.

Fig. 1. (a) Faceplates for large entrance marsupial/bird hollow, bat hollow and small entrance marsupial/bird hollow (from left to right); (b) transitional stage of creating the cavity, showing the plunge cuts that allowed the hammering out of individual wood pieces; (c) finished cavity; (d) finished large entry marsupial/bird hollow with heat/motion activated camera installed on bracket above cavity, (e) completed marsupial/bird hollow with small entrance, and (f) completed bat hollow.

Figure 5 Example of method of constructing chainsaw-carved hollows (Rueegger 2017).

Van der Ree (2019) recommended the following attributes of chainsaw hollows be collected and considered, to help build data sets that provide an understanding of the suitability and efficiency of chainsaw hollows as a hollow alternative:

- the type, size and construction technique of hollows installed;
- the species, health and size of the host trees;
- the prevailing weather and other environmental conditions among sites;
- different management regimes (e.g. risk assessments, pruning techniques and intensity); and
- the type and quality of data being recorded.

4.4 Chainsaw-carved bat fissures

From Griffiths *et al.* (2023):

The arborists created the carved fissures by making three plunge cuts upward at an angle of approximately 30° into the stem through a vertical slit-entrance 2 x 15 cm (width x height) (Figure Six). This produced a wedge-shaped internal cavity (maximum depth = 20–25 cm), that was located above the entrance slit, with a volume of approximately 500cm^3 (Griffiths *et al.* 2023). Finally, a small amount of bark and cambium around the entrance slit was scoured to make a rough surface for alighting bats to land on and grip. The design (Figure Six) adopted was not intended to target a particular species, sex, or type of activity (i.e. nonbreeding vs. maternity roost), and attempting to do so would have been difficult given the lack of published information on the internal dimensions of natural roosts. Instead, it represented the authors' best guess at what a generic roost might look like based on past research and observations, and was intended to act as a starting point from which the authors could learn and make improvements following monitoring.

The arborists cut all the carved fissures as systematically as possible, to ensure relative consistency in the dimensions of the entrance slits and internal cavities in trees across the six reserves. Carved fissures were installed at heights ranging from 1.5 to 6.6 m (mean = 4.4 m) above the ground to facilitate passive monitoring with a pole camera and access with a 6-m extension ladder during manual inspections.

Figure 2. An arborist making a plunge cut upward at an angle of approximately 30° into (A) a branch and (B) the trunk of two different live sugar gum (*Eucalyptus cladocalyx*) trees at Nangak Tamboree Wildlife Sanctuary. (C) External view of a completed carved fissure with a vertical slit entrance (2 × 15 cm, width × height) carved into the trunk of a sugar gum. (D) An example of a fissure carved into a felled sugar gum log to show the wedge-shaped internal cavity (adapted from Griffiths et al. 2018).

Figure 6 Method of creating carved bat 'fissure roosts' (from Griffiths *et al*. 2023).

4.5 Hollow salvage for log hollows

The literature does not provide guidance on the relocation or installation of large hollow sections or complete trunk sections for use as natural log hollows attached to standing trees. Hannan *et al.* (2019) conducted the most similar process, translocating and re-erecting entire dead, salvaged trees (see above).

4.5.1 Entrance creation

From Ellis *et al.* (2022):

Ellis *et al.* (2022) selected trees in areas away from roads and walking tracks. From the ground, the trees were visually inspected for existing hollow entrances. In trees lacking entrances, trees were tested for internal decay/voids, by drilling inspection holes of 10-mm diameter and up to 200 mm depth into trunks. Resistance to drilling was used to assess whether a void or decay had been encountered. Where a void/decay was detected, the tree was ascended using a ladder and the internal void tracked upwards by drilling further 10-mm inspection holes until a suitable point for creating an entrance was found (void/decay through <200 mm of wood, > 40 mm diameter, accessible via ladder).

Three entrance diameters (40, 65 and 90 mm) were trialed to facilitate access by vertebrates with a range of body sizes. For each stem, Ellis *et al.* (2022) chose an entrance size no greater than the estimated diameter of the internal cavity. To define the entrance size, a hole saw was used to cut the first few centimetres into the stem and the timber was removed using a hammer and chisel. An auger, twist, or spade bit was then used to make multiple holes to extend the entrance through to the void with the remains of the timber removed with a chisel. The initial trial created entrances in November 2017 in three stems on which monitoring techniques were tested. Once installation and monitoring techniques were determined to be feasible, the trial was extended. Between April 2018 and September 2019, Ellis *et al.* (2022) added entrances into an additional 21 stems in one study site and 15 stems in a second study site. Entrance holes were made through 60–200 mm of wood, at 2.4–4.8 m above ground level, and accessed voids or decay of 40–240 mm diameter. Stem diameter ranged from 28 to 80 cm. Holes were created in: *Eucalyptus blakelyi* (n = 6 stems), *E. albens* (n = 8), and *E. melliodora* (n = 10) (study site one); and *E. tereticornis* (n = 1), *E. sieberi* (n = 4), *E. longifolia* (n = 2), *E. muelleriana* (n = 6), and *Angophora floribunda* (n = 2) (study site two). At study site one, four of the entrances were in trees planted in the 1980s to revegetate cleared areas; other drilled entrances at study site one and all of those at study site two were in naturally regenerated trees.

4.5.2 Implications of design and implementation for VMFRP

The literature indicates a wide range of designs have been used for hollow-alternatives for various species. However, these designs have been shown to attract only a small number of species, none of which are target species for the VMFRP project area. Further, as the literature considered herein demonstrates, the effectiveness and appropriateness of each of these designs is severely compromised and so these designs do not provide a suitable hollow replacement option.

5 Longevity and attrition of hollow alternatives

In a ten-year study of nest box occupancy and attrition in the Central Highlands of Victoria, Lindenmayer *et al.* (2009) found increasing numbers of nest boxes fell to the ground over time. In their initial surveys, fallen boxes were re-attached, so the probability was modelled of a box falling at least once at any time in the study. In the later phases of the ten-year study, nest boxes could not be re-erected as they often were badly damaged when they fell, owing to being in a relatively advanced stage of decay. The modelled probability of a nest box falling at any time in the ten-year study identified a strong forest age effect, where nest boxes had a higher probability of falling in young forest (there were 18 instances of fallen nest boxes in old forest compared with 31 in young forest), mostly due to self-thinning of young trees and the boxes being pushed off their attachment nails as a result of rapid growth rates of the regenerating trees. The authors found no significant or near significant relationship between the probability of falling and whether a nest box had been occupied. Finally, the study found nearly all nest boxes fell during years 7-10 – the nest boxes became decayed and so more subject to being damaged by falling limbs (Lindenmayer *et al.* 2009). Storms and the weight of animal occupancy may also contribute to decayed boxes falling. As well as replacement considerations, falling nest boxes have animal welfare considerations too, if the box is sheltering animals when it falls. Goldingay *et al.* 2018, however, found in their study (in a different environment and forest type) that 84% of nest boxes were functional for the target species (Brush-tailed Phascogales and Sugar Gliders) after 10-11 years and 60% after almost 20 years (although only 28% were still functional at 25 years). The microclimatic conditions (e.g. thermal properties, humidity, water ingress and moisture content) of the boxes was not considered but ideally would form part of the consideration of the effectiveness of the nest boxes over time, given the implications microclimatic conditions may have on wildlife fitness and ecosystem dynamics (see below). Needless to say, some maintenance was required during the lifetime of the nest boxes.

6 Ecological considerations

Potential impacts of hollow alternatives on ecosystem dynamics

Camera data from Rueegger (2017) documented hollow competition between species. Hollow eviction was determined where the evicted animal used a hollow over several consecutive days or weeks before the competitor animal took over the hollow on the same night the evicted animal was last seen exiting/entering the hollow (Rueegger 2017). Sugar Gliders were recorded to have taken over hollows used by Feathertail Gliders and White-throated Treecreepers; Brown Antechinuses were recorded to evict Feathertail Gliders and a White-throated Treecreeper pair (Rueegger 2017). Cameras also provided observations of two occasions of Diamond Python (*Morelia spilota spilota*) predation on Sugar Gliders (Rueegger 2017). Whilst it is not possible to distinguish whether this competition and/or predation would have occurred between naturally-occurring hollows, these observations indicate the nest boxes provided do not preclude competition nor predation so exert some influence over ecosystem dynamics compared to the nest boxes not being provided. Of course, the absence of nest boxes (especially in a landscape where there is a paucity of natural hollows and/or hollow-bearing trees have been removed so hollow resources are constrained) exerts its own influence over ecosystem dynamics.

Whether nest boxes (or the absence of nest boxes) exert an influence to the extent observed by Rueegger (2017) is unclear in landscapes where hollows are abundant, but findings by Lindenmayer *et al.* (2009) indicate older forests (i.e. with some naturally-occurring hollows) have significantly lower rates of nest box occupancy compared to young forest (i.e. a paucity of naturally-occurring hollows). This suggests nest boxes in a landscape context that includes old, hollow-bearing trees are less likely to be used and so less likely to change any ecosystem dynamics, however this requires further research and consideration as nest boxes may provide a competitive advantage over naturally-occurring hollows for some occupants and predators.

The literature does not consider the influence of the microclimate of hollow alternatives may affect predator-prey dynamics, when animals using a hollow alternative with an unsuitable (intolerable) microclimate are forced to vacate their hollow and have no available cavity in which to shelter, nest or roost, so are subject to predation. More research is urgently required, including consideration of whether the installation of poorly insulated, poorly regulated hollow alternatives introduce artificial inter- and intra-specific competition for hollows by creating an unnaturally volatile scenario of hollows being available (suitable) then unavailable (unsuitable).

Griffiths *et al*. (2023) postulate that plywood bat boxes are unlikely to lead to increased bat diversity or support entire bat communities, but likely will be used by at least Gould's Wattled Bat. Indeed, if bat boxes are being used (if at all) by only a small proportion of the species that make up the local community of cavity-roosting bats, what implications does providing bat boxes have on the interspecific dynamics of bats (e.g. competition, fitness), and what are the ecological implications of this? What are the ecological implications of only one (or just a few) bat species using boxes and how might the intended benefits to ecosystems be impacted by unintended bat diversity/population dynamics. If only one species of bat is able to persist in an area (owing to artificial resources that suit only that species), what does that mean for the balance of invertebrate prey? These questions apply across all hollow-utilising species, across all hollow alternatives.

6.1 Ecological implications of translocating hollows any distance from their original location

The literature does not consider the ecological implications of translocating hollows from their original location and installing them elsewhere. Depending on the distance between origin and installation, is possible scent markers may be used by some species showing high hollow-fidelity, to locate the translocated hollow though research would need to be undertaken to determine whether this would occur. It is possible the scent of a translocated hollow would influence the ecosystem dynamics at the receiving site, either through increased territorial behaviour (toward the unfamiliar smell) or by rejection of the translocated hollow. In addition, translocation of a used (scented) hollow may result in increased energy expenditure by predators at the receiving site as they inspect newly installed (but unoccupied) hollows with no energetic return.

6.2 Pest infestations

Lindenmayer *et al.* (2009) found many of their results for pest infestations and nest box attrition were broadly consistent with what the authors had predicted to occur when they commenced their investigation. They found nest boxes were more likely to be occupied by honeybees in young forests than in older stands (Lindenmayer *et al.* 2009). Other studies also found that honeybees can occupy nest boxes (e.g. Coelho and Sullivan 1994; Emison 1996; Soderquist *et al*. 1996; Gibbons and Lindenmayer 2002). Moreover, nest boxes in disturbed areas may be susceptible to occupancy by pest species (e.g. Munro and Rounds 1985; Harper *et al*. 2005), which may explain why nest boxes in more recently disturbed areas of logged and regenerated young forest in Lindenmayer *et al.* (2009) had higher levels of infestation than older stands. However, other (unidentified) factors appear to be important as results in Lindenmayer *et al.* (2009) revealed strong regional effects. It is possible that factors such as topography, aspect and landscape context have an important influence on regional susceptibility to infestations of honeybees (Lindenmayer *et al.* 2009).

In southern New South Wales, several studies found nest boxes occupied by honey bees – 33% of boxes (Lindenmayer *et al.* 2016), 13% of boxes (Le Roux *et al.* 2016) and 9% of boxes (Lindenmayer *et al.* 2017). Between 39-57% of 14-18 boxes were occupied at two locations in Victoria (Soderquist *et al.* 1996). Goldingay *et al.* 2015 found honeybees occupied around 10% of nest boxes in a study near Sydney and in north-east New South Wales, but concluded bee occupation seemed not to be a serious issue associated with nest boxes as the bees were observed in that study to vacate boxes after ~ one year. Of course, honeybees also occupy naturally-occurring hollows and have been known to displace or exclude hollow-dependent native fauna (e.g. Oldroyd *et al.* 1994; Gibbons and Lindenmayer 2002).

Rueegger (2017) found no exotic cavity-using pest species were recorded using the chainsaw hollows in that study. In contrast, Best *et al.* (2022) recorded 2254 records of four introduced taxa visiting chainsaw hollows: Common Starling *Sturnus vulgaris* (1089), Common Myna *Acridotheres tristis* (547), Black Rat *Rattus rattus* (525), and European Honeybee (93). In one chainsaw hollow, Common Starlings produced and raised successfully two clutches (each of three chicks) over consecutive years (2018 and 2019). While camera trap surveys revealed Common Mynas visiting and entering several chainsaw hollows, Best *et al.* (2022) recorded no evidence of current or past breeding during occupancy surveys. European Honeybees were present at all five study sites (Best *et al.* 2022). Overall, Best *et al.* (2022) considered introduced pest species used chainsaw hollows infrequently.

Introduced birds, particularly the Common Starling and the Common Myna, and the Black Rat also have been recorded occupying nest boxes in other studies (e.g. Lindenmayer *et al.* 2016, 2017; and Le Roux *et al.* 2016). However, in each of these studies, the percentage of boxes occupied by these species was low compared to those nest boxes that were unoccupied, indicating the use of nest boxes by pest species may not greatly influence resource availability for target species. Indeed, pest animals may be exploiting nest boxes that are unoccupied because they are unsuitable for native species.

Buff-tailed Bumblebee *Bombus terrestris* recently was recorded as a potential nest site competitor in a nest box programme designed to support the recovery of the endangered Forty-spotted Pardalote *Pardalotus quadragintus* (Hingston 2024). Buff-tailed Bumblebee workers were observed entering and exiting one of the boxes; this activity was observed repeatedly for the next three weeks (Hingston 2024). The nest box was accessed by single rope climbing to inspect the contents and to remove the bumble nest, which had been constructed (at least partially) using an existing pardalote nest (Hingston 2024). Alongside these remains of the bumblebee nest, further from the entrance hole, was a new forty-spotted pardalote nest that did not contain eggs or nestlings (Hingston 2024). This was the first record of a potential non-avian competitor with Forty-spotted Pardalotes for nest boxes and the first evidence of bumblebees using above-ground nesting boxes in Tasmania (Hingston 2024). Although the Forty-spotted Pardalote readily nests in nest boxes, the capacity of nest boxes to provide nesting opportunities is prevented by nest box competitors - when their nest sites are taken over by other birds (e.g. the Striated Pardalote *Pardalotus striatus* and the Tree Martin *Petrochelidon nigricans*), some Forty-spotted Pardalote pairs have been recorded successfully renesting elsewhere, however the majority of pairs were not detected nesting again after being evicted (Edworthy 2016). It is unclear whether the nest taken over by bumblebees belonged to a Striated Pardalote or a Forty-spotted Pardalote; if the nest was that of the latter, it is possible the nesting pair may not establish another nest. Further research is required to determine the frequency with which buff-tailed bumblebees nest in pardalote boxes, and whether this causes nest abandonment by Forty-spotted Pardalotes or renders boxes unsuitable for concurrent or subsequent use by this endangered species of bird (Hingston 2024).

The Tree Martin and Common Starling were identified as significant pests in a study of nest boxes intended for the Critically Endangered Swift Parrot *Lathamus discolor* (Stojanovic *et al.* 2021). These two introduced birds were recorded using nest boxes and, although the target species (Swift Parrot) exploited similar numbers of nest boxes during the study, the pest competitors were the main beneficiaries of established boxes (Stojanovic *et al.* 2021). Stojanovic *et al.* (2021) considers permanent box availability for Swift Parrots may produce perverse outcomes by increasing nesting habitat for Common Starlings and suggests that, for species that only use cavities for part of their life cycle, nest box programme managers should limit access to nest boxes outside of critical times to reduce the likelihood that pest populations can exploit restoration efforts and create new problems. This has interesting implications for the management of hollow alternative programmes for other hollow-using species that display strongly seasonal hollow usage patterns.

7 Wildlife considerations

7.1 Thermal properties

Declining hollow availability means artificial hollows, particularly nest boxes, are used widely to offset the removal of hollow-bearing trees or augment a landscape where there is a paucity of hollows, to provide alternative denning or nesting sites. However, as this review demonstrates, rates of nest box and chainsaw hollow occupancy can vary substantially among species and habitats (see also Lindenmayer *et al.* 2009). Many studies demonstrate the shortfalls of these artificial hollows, including their limited functional life, potential colonisation by non-target (worse: pest) species and unsuitable internal microclimate (e.g. Harper *et al.* 2005; Griffiths *et al.* 2017; Rowland *et al.* 2017).

7.1.1 Nest boxes

Thin-walled, plywood nest boxes are used frequently across Australia, but a key concern is the temperature and humidity extremes and variability within nest boxes which do not replicate the thermal properties of natural hollows (Larson *et al*. 2018; Rowland *et al.* 2017). The results of a survey designed to quantify the number and location of nest boxes across Victoria, Australia, conducted by the Department of Environment, Land, Water and Planning (DELWP, now the Department of Energy, Environment, and Climate Action (DEECA)) demonstrated why thermal properties of nest boxes should be a central consideration (Griffiths 2019); Macak 2020). Respondents from 94 different nest box programs provided data showing approximately 10,000 nest boxes were at that time installed across Victoria (Macak 2020). Many of these nest boxes are installed in Mediterranean or semi-arid environments, meaning they are subject to relatively cold winter and hot summer ambient conditions, and high levels of solar radiation (Griffiths 2019). These conditions, like those experienced in northern Victoria along the Murray River, challenge the suitability and effectiveness of nest boxes, as these conditions (especially during summer) can drive internal nest box temperatures well above critical thermal tolerance limits for endothermic bird and mammal occupants (Rowland *et al.* 2017). Yet, the respondents predominantly represented community-run nest box programmes which do not have resources to monitor the internal microclimates of nest boxes or the suitability of those internal microclimates for the species observed using the boxes (Griffiths 2019). This has obvious issues and likely consequences for the effectiveness of the nest box programmes, as well as for wildlife using the nest boxes.

Nest boxes designed for arboreal mammals were recorded as having a mean maximum temperature 8^{o}C warmer than trees hollows, with peaks of up to 52°C (Rowland *et al*. 2017). Griffiths (2019) found mean hourly temperatures within nest boxes varied considerably more than natural hollows, chainsaw hollows or log hollows (in this study: chainsaw hollows cut into felled logs that were subsequently attached to tree trunks), with nest boxes experiencing both hottest and coldest extremes. Griffiths (2019) also found mean hourly temperatures in chainsaw hollows were comparable to naturally-occurring hollows – both were cooler than ambient temperatures during the day and warmer than ambient temperatures at night. Log hollows followed the same pattern but fluctuated more in temperature than either natural hollows or chainsaw hollows (Griffiths 2019). In contrast, nest boxes in that study (Griffiths 2019) exhibited the opposite pattern, remaining warmer than ambient temperatures during the day and close to or colder than ambient temperatures during the night. This raises questions as to whether nest boxes affect ecological dynamics, as those species that can tolerate a wide variety of climatic conditions (including exotic species) may have some competitive advantage in semi-arid and Mediterranean landscapes where nest boxes are installed and/or replace naturally-occurring tree hollows. During two summers, Goldingay (2015) recorded temperatures in nest boxes over a one-month period. The temperature within the nest box varied by up to 20°C within a 24 hour period, with some nest boxes experiencing temperatures above 40°C. Whilst data from some studies (e.g. Griffiths *et al.* 2018) indicate the microclimate of unoccupied chainsaw hollows is comparable to that of naturally-occurring hollows, and chainsaw hollows may be used by some species in preference to nest boxes (Terry *et al.* 2021), other studies indicate there are microclimatic and other issues with chainsaw hollows, too (see below). Nest box and chainsaw hollow installations (especially for non-flying mammals; Goldingay 2015) should microsite nest boxes carefully (e.g. having regard for aspect, surrounding vegetation, shading, etc.) to minimise extreme temperatures.

Common nest box designs clearly may be of limited value to wildlife during heat events due to extreme temperatures, however even every-day fluctuating temperatures and exposure to temperature extremes may impact the survival and reproductive fitness (and success) of nest box occupants. Some studies raised concerns that nest boxes may not provide protection from temperature extremes compared to natural cavities (Rowland *et al*. 2017; Griffiths *et al*. 2018), though there is little *direct* evidence that this poses a fitness cost to the species occupying the nest boxes (Goldingay 2017; Saunders *et al*. 2020). Although one study showed there was no mortality or decline in abundance of Brush-tailed Phascogales (*Phascogale tapoatafa*) or Sugar Gliders (*Petaurus breviceps*) using nest boxes during a prolonged extreme heat event (Goldingay and Thomas 2021), many studies have found the thermal dynamics of nest boxes can impact individuals using the nest boxes (Catry *et al.* 2011). At least one study has found commonly used bat box designs exceeding 44°C, the lethal limit for bats (see Nagorsen 2009). Birds using nest boxes may be impacted by reduced reproductive success or disrupted offspring development when exposed to extreme heat (Ardia *et al*. 2006; Larson *et al*. 2015).

Though some studies suggest there is little *direct* evidence, logical consideration suggests the microclimate of a den – be it a naturally-occurring hollow or a constructed hollow – may have significant fitness consequences. This is particularly likely for those species that spend many hours a day in the den (e.g. Leadbeater's Possum spends up to 20 hours a day in the den; Smith 1980). Poorly insulated dens or those with poor moisture regulation (see below) may shift individuals out of their safe thermoregulation mode, requiring them to use energy to stay warm or lose water content (and body mass) through evaporative heat loss (see McComb *et al.* 2022). Thermal fluctuations experienced in nest boxes and chainsaw cavities would be exacerbated by issues relating to humidity and moisture dysregulation associated by these artificial dens. Indeed, Leadbeater's Possums have been observed sleeping on top of their nests (i.e. not being insulated by their nest) during extreme summer temperatures in both nest boxes and chainsaw hollows (D. Harley and L. Lumsden unpublished data; see McComb *et al.* 2022). Further studies are required to determine what implications thermal fluctuations play on animal welfare, including individual fitness (and therefore the population's- and species'- fitness), and whether temperature limits the effectiveness of nest boxes and chainsaw hollows at some locations.

7.1.2 Plastic nest boxes

A recent study (Callan *et al.* 2023a) explored the potential of plastic nest boxes to provide improved thermal regulation (and longer field life) than traditional timber or plywood nest boxes. The use of plastic nest boxes has been trialed (Elston *et al*. 2007; Rueegger *et al.* 2013; Saunders *et al.* 2020) and recycled plastic nest boxes have been used for Leadbeater's Possum since 2008, with over 600 installed across the species' range (D. Harley and J. Antrobus, unpublished data; see McComb *et al.* 2022). However, Callan *et al.* (2023a) conducted the first comparative study of various plastic nest box designs using prototypes constructed with 3D printing technology. Trials conducted by Callan *et al.* (2023a) in a temperature-controlled laboratory demonstrated that a double-walled plastic nest box with timber inserts provided a stable internal microclimate, approaching the thermal profile of natural hollows.

7.1.3 Spout boxes

Strain *et al.* (2021) compared animal activity and thermal properties of natural and artificial spout hollows during their study in Central Victoria. One hundred and twenty-five (125) spout boxes were attached to trees and pickets; these were found to be less thermally buffered (i.e. displayed both higher maximum and lower minimum daily temps) than natural hollows (Strain *et al.* 2021). Spout boxes on pickets had (on average) cooler minimum temperatures and higher maximum temperatures in summer than spout boxes on trees (Strain *et al.* 2021). Nesting in both target species was largely confined to October and November, which avoided the extreme lows and highs in ambient temperature that occurred in August (Strain *et al.* 2021). Humidity in spout boxes declined more rapidly than in natural hollows; boxes on pickets had lower maximum humidity than boxes positioned on trees and experienced a greater reduction in humidity into summer (Strain *et al.* 2021). Daily variation in temperature and humidity (difference between minimum and maximum) was greater in spout boxes than natural hollows, and those spout boxes attached to pickets had greater variation than those affixed to trees (Strain *et al.* 2021). The key predictor variables influencing variation between types of hollows included minimum wall thickness of the cavity (variable in natural hollows, but constant and therefore excluded in spout boxes), canopy cover, openness, altitude, slope aspect, and location (Strain *et al.* 2021). These models indicated that for natural hollows, maximum daily temperatures were lower as wall thickness and canopy cover increased (Strain *et al.* 2021). For spout boxes, maximum temperatures were lower with greater canopy cover, minimum temperatures were associated with site location, and minimum humidity was higher with greater canopy cover (Strain *et al.* 2021).

Despite differences in cavity temperature and humidity, spout boxes attached either to trees or pickets successfully were used for breeding by Turquoise Parrots and Brown Treecreepers – indeed, the spout hollows were more often used for breeding than were natural hollows (though the sample size was small and perhaps not statistically robust). During the study, microclimatic conditions within spout boxes clearly were tolerable for the target species, despite large differences in temperature and humidity between artificial and natural hollows. These results suggest spout boxes may provide effective supplementary nesting sites for target woodland birds, however more research is urgently needed into the thermal properties of these and other nest boxes particularly in the face of a changing climate. A changing climate (predicted to be hotter and drier, but with erratic rainfall events) means differences in microclimate between spout boxes and natural hollows likely will be further amplified. Thus, internal cavity microclimates that buffer against climate variation (i.e. higher relative humidity and a more stable thermal profile) are likely to influence the ongoing viability of populations of hollow-dependent fauna. There is a critical need for nest box programs to incorporate empirical data on the physical characteristics of natural hollows used by target species into the design, construction, and installation of artificial hollows, especially to inform development of artificial hollows that provide target species with microclimate conditions that resemble natural hollows (Strain *et al.* 2021). Finding hollow alternatives that provide safe thermal conditions will be critical for the diversity of hollow-using fauna endemic to the extensively cleared and degraded woodland ecosystems across south-eastern Australia (Bennett & Ford, 1997; Gibbons & Lindenmayer, 2002), where nest boxes are commonly installed to provide supplementary habitats (Lindenmayer *et al*., 2017; Macak, 2020). This part of Australia is particularly vulnerable to changes to climatic conditions that might exacerbate the existing Mediterranean climate (Rowland *et al*. 2017).

7.1.4 Chainsaw-carved hollows

While several studies have shown a diversity of fauna will use chainsaw hollows (e.g. Best *et al.* 2022; Griffiths *et al.* 2020; Honey *et al.* 2021; Terry *et al.* 2021), less is known about their thermal properties. Several recent studies carried out in southern Australia found temperatures inside chainsaw hollows and natural hollows in live trees were very similar; both provided temperatures that were warmer than ambient at night, and cooler than ambient during the day (Griffiths *et al.* 2018; 2022). Notably, a recent study by Griffiths *et al*. (2022) found that even when ambient temperatures reached 46.5ºC, average temperatures inside chainsaw-carved hollows in live trees were 30.2ºC, demonstrating that such structures in live trees may provide thermal refuges to wildlife during extreme heat events.

Although the results of Griffiths *et al*. (2022) are encouraging, they were based on temperatures recorded inside chainsaw hollows created in eight mature live trees at heights of 1.5 to 2.5 m above ground. However, in many rural and agricultural landscapes, large live trees suitable for creation of hollows are scarce and so chainsaw hollows often must be created in standing dead trees. Because dead trees lack the shade provided by a canopy and the thermal insulation provided by the flow of water within the sapwood of live trees, it is likely that chainsaw hollows carved into dead trees are hotter than has been reported for chainsaw hollows carved into live trees (Callan *et al.* 2023). Callan *et al.* (2023) conceded there are no published studies on the thermal properties of chainsaw hollows carved into dead trees, so it is not known if such structures provide thermally-suitable diurnal refuges for nocturnal marsupials during summer, particularly during heatwaves.

The interaction between orientation and season, along with canopy cover, have been shown to strongly influence thermal profiles of unoccupied hollow alternatives (Best *et al.* 2022). Best *et al.* (2022) considered the chainsaw hollows that faced north or west in their study may have remained warmer throughout the cooler months than those facing south or east. This would be due to increased conductive heating of the cavity through the walls as they were warmed by direct solar radiation, thereby providing a more favourable den microclimate for Krefft's Gliders (Best *et al.* 2022). Empirical data investigating the factors that drive variation in temperature and humidity within chainsaw hollows would be of great interest when designing shelters for species that require specific microclimate conditions during different seasons (Best *et al.* 2022).

McComb *et al.* (2022) compared nest box and chainsaw hollow thermal profiles in high elevation montane ash and found winter daily minimum temperatures were lower, and summer daily maximum temperatures higher, in nest boxes compared to chainsaw hollows, with the chainsaw hollows providing a more stable environment. Daily temperature fluctuations (calculated as the daily maximum minus the daily minimum) in occupied nest boxes were up to four times greater than in occupied chainsaw hollows (McComb *et al.* 2022). Winter minimum den temperatures were on average 3°C warmer in chainsaw hollows than nest boxes for both occupied and unoccupied dens (McComb *et al*. 2022). This pattern was also apparent over summer, but temperature differences were more pronounced: summer maximum den temperatures were 9°C cooler in occupied chainsaw hollows than nest boxes (22.3°C vs 31.4°C) (McComb et al. 2022). Additionally, a significant interaction between den type and occupancy indicated that occupancy had a greater effect on mean and maximum summer temperatures within chainsaw hollows (McComb *et al.* 2022).

Biophysical modelling undertaken by McComb *et al.* (2022) suggests Leadbeater's Possums can partially mitigate the energetic impacts arising from the cool nest box temperatures recorded in winter via behavioural strategies such as building large, insulative nests and huddling. In contrast, Leadbeater's possum has limited behavioural scope to mitigate high summer nest box temperatures (recorded primarily in lowland swamp forest), other than denning elsewhere (McComb *et al.* 2022). This is likely to be the case for other hollow-dependent nocturnal mammals. Denning elsewhere may mean animals are forced to leave their hollow during the day, facing both increased predation and increased exposure to heat. In a landscape where natural hollows are scarce and hollow alternatives are abundant, this could mean a functional lack of denning opportunities (despite the intended increase provided by provision of hollow alternatives) and indeed a functional limit to population growth as carrying capacity (in this case, suitable habitat) is exceeded.

High temperatures were predicted to require substantial heat loss for Leadbeater's possums, such that during the day, while in their den, they would be outside their thermoneutral zone 9% of the time (McComb *et al.* 2022). During extreme heat events, this could lead to heat stress and dehydration, particularly given the high humidity levels also recorded in nest boxes (McComb *et al.* 2022). The low summer occupancy of nest boxes recorded in lowland swamp forest indicates that possums may shift dens to natural hollows to avoid extremely high temperatures – an issue exacerbated by the loss of dense vegetation structure at this site and associated higher solar exposure (Greet *et al.* 2020). Summer temperature extremes are likely to be more challenging [than low winter temperature extremes] for this species and other similar species, as Leadbeater's possum colonies have limited capacity to avoid high temperatures in nest boxes other than by choosing to den elsewhere (McComb *et al.* 2022). Summer nest box temperatures recorded in McComb *et al.* (2022) add to the growing body of evidence there may be eco-physiological costs for species reliant on nest boxes year-round (Goldingay 2015; Rowland *et al.* 2017). However, this evidence is not universal as Goldingay (2017) recorded no fitness costs to squirrel gliders (*Petaurus norfolcensis*) reliant on nest boxes over a 10-year period, and no observations of mortality of Sugar Gliders following and 11-day heatwave (Goldingay and Thomas 2021). In the case of Leadbeater's possum and other similar species, eco-physiological costs may be greater in more open vegetation types where maximum temperatures are exacerbated by high solar exposure. Nests are likely to provide some insulation from hot ambient temperatures, however they may also trap heat produced by occupants (McComb *et al.* 2022). With reduced air flow inside dens, nest microclimates are likely to lead to elevated physiological stress during the hotter afternoons (McComb *et al.* 2022).

7.1.5 Entrance creation

In a study which involved drilling entrance holes into standing trees to provide access to internal cavities existing within trees, Ellis *et al.* (2022) demonstrated all 10 cavities were similar in temperature at any point in time and fluctuated by up to 4^{o}C per day despite local ambient temperature fluctuating by up to 21^{o}C. Inside cavities, the mean daily maximum was up to 14^{o}C lower than maximum ambient and mean daily minimum up to 10^{o}C above minimum ambient. Like natural cavities with natural openings (Griffiths *et al.* 2018), the cavities accessed by drilled entrances had relatively stable temperature (Ellis *et al.* 2022). In stark contrast, as this review has discussed, wooden nest boxes can become far hotter than ambient air temperature, even in temperate environments, and fluctuate widely. Thus, denning or nesting in artificially opened but otherwise natural cavities such as those in Ellis *et al.* (2022) is likely to be less physiologically stressful than using nest boxes or other hollow alternatives, and similar to modeling by Rowland *et al.* (2017) for natural hollows compared to nest boxes.

7.2 Humidity and water ingress

Carey (2002) constructed 128 chainsaw cavities with faceplates in trees. After two years, faceplates were replaced on 75% of cavities due to water ingress. Similarly, in an Australian study, Terry *et al*. (2021) discovered chainsaw hollows were subject to a build-up of moisture or vulnerability to water ingress. At the final inspection, 70.5% of chainsaw hollows compared to only 13.6% of nest boxes had a build-up of moisture (>1 moisture index); of these chainsaw hollows, 29.5% had moderate to high (>2 moisture index) rates of moisture (Terry *et al.* 2021). One check of cavities was undertaken during rainfall and water was observed trickling down the insides of faceplates on the pole mounted inspection camera (Terry *et al.* 2021). The contents of three chainsaw hollows were so wet that old nesting material had turned into mud-like material (Terry *et al.* 2021). Despite the accumulation of moisture by some cavities, Terry *et al.* (2021) observed high rates of use by the target species across all chainsaw hollows – this perhaps should increase concerns for wildlife welfare posed by damp cavities. A pre-made timber cavity insert inserted into chainsaw hollows (e.g. Saenz *et al*. 2001) may be an effective way to manage water ingress but would increase complexity, time and economic cost of installation. It is worth noting, some traditional nest boxes also recorded a build-up of cavity moisture (Terry *et al.* 2021). This was particularly the case where removal of the outer painted surface (which may have been caused by chewing by parrots) allowed water to penetrate the timber and cause mild warping of lids (Terry *et al.* 2021). Somewhat paradoxically, tree hollows have been found to maintain relatively high humidity (mean ~90%) (Maziarz *et al*. 2017; Rueegger 2019) and several species of bat have been found to actively select roosts with higher humidity (e.g. Sedgeley 2001).

This difference between chainsaw carved hollows and nest boxes in terms of internal rates of moisture, particularly where rates of moisture are so high they result in wet nesting material, may have serious consequences for occupants. As well as contributing to accelerated attrition, wet nesting materials may result in high mould levels, which could prove fatal for species susceptible to Aspergillus mould species. Aspergillosis is a fungal respiratory infection especially harmful (fatal, even) to wild birds (e.g. Helmeted Honeyeater *Lichenostomus melanops cassidix*), including parrots which may use provided hollows. Yet, low humidity in nest boxes also has implications for fauna, resulting in evaporative water loss (and therefore loss of body mass) in bats (Webb *et al.* 1995) and birds and their eggs (Deeming *et al.* 2011; Vleck *et al.* 1983). In this way, artificial cavities cannot regulate humidity and retained moisture in the same way as natural hollows, which have complex regulatory processes that enable the maintenance of humidity without fungal growth. This demonstrates that natural hollows are superior, including in terms of wildlife welfare, to constructed or artificial cavities.

Although humidity, water ingress and moisture levels within the cavity were not tested by Ellis *et al.* (2022), the approach of creating small entrances to internal, existing cavities within the trees is the closest to a natural hollow of any approach considered in the literature. Because this approach just provides access to existing but inaccessible hollows, the processes that regulate temperature, humidity and moisture content in naturally-occurring hollows must apply to these hollows, too; these processes do not apply to carved or attached hollow replacements as they are not tapping into the trees' regulatory processes.

7.3 Implications for wildlife of attrition of nest boxes

A 10-year experimental study investigating nest box occupancy rates and attrition (Lindenmayer *et al.* 2009) found nearly all the 96 nest boxes installed in young- and old- Mountain Ash forests fell during years 7-10. In a Central Victorian study, nest boxes showed some signs of deterioration towards the end of the 2.5-year monitoring period (Terry *et al.* 2021), though another study in the same landscape (a decade or so earlier; note: during a drier – El Nino - period) has documented a large percentage of nest boxes remaining functional for at least 20 years (Goldingay et al., 2018). Lindenmayer *et al.* (2009) found nest boxes became decayed and so more subject to being damaged by falling limbs. Storms and the weight of animal occupancy may also contribute to decayed boxes falling. Indeed, the increased weight and movement of growing juveniles in decayed nest boxes could make them vulnerable to injury if the nest box fell. As well as replacement considerations, falling nest boxes have animal welfare considerations if the box is sheltering animals when it falls.

7.4 Ethics

The literature does not indicate studies have been undertaken to determine the ethics of installing hollow alternatives. However, the wildlife considerations covered in this review demonstrate concerns have been raised (directly and indirectly) about the ethical implications of: creating nest-box reliance; providing nest boxes and hollow alternatives that have potentially dangerous thermal properties; providing nest boxes and hollow alternatives that experience water ingress and may support pathogens that are potentially fatal to wildlife; failing to adequately fund and undertake maintenance of decayed/damaged nest boxes that may fall and kill the occupants; providing hollow alternatives using designs that do not afford adequate protection to occupants from predators; creating a dynamic of abundant hollow resources (by providing hollow alternatives) which encourages population growth of hollow-dependent fauna, but which are then displaced and perish when the only available alternative to their overheating nest box is another overheated nest box; and more. These ethical considerations should form a central role in developing a hollow replacement strategy as they have critical implications for the survival of the very species well-intentioned land managers would seek to protect through a hollow replacement strategy.

7.5 Optimal timing for hollow removal and translocation of displaced fauna

In a dynamic and healthy environment, determining the optimal timing for hollow removal would be challenging, as nocturnal vertebrates find diurnal roosts in hollow- and crevice-bearing trees. Seasonally, removing hollows during the hotter months clearly would impact displaced wildlife and limit the effectiveness of translocation. Few studies provide guidance, beyond considering the biological and ecological characteristics of present species.

7.6 Nest boxes as a research tool

According to Goldingay *et al.* (2020), nest boxes can, through careful study design, enable investigation of factors that ordinarily may be highly variable or complex (e.g. Aitken and Martin 2012; Le Roux *et al*. 2016; Lindenmayer *et al*. 2016). As discussed, van der Ree (2019) recommends each hollow alternative be considered as an experiment that can yield vital information to help stakeholders develop effective, beneficial strategies for compensating for removal of hollow-bearing trees.

8 Host tree health considerations

The suitability of River Red Gum (*Eucalyptus camaldulensis*) (as one of several eucalypts) to host chainsaw-carved hollows was considered by Best *et al.* (2022). Aside from River Red Gum occurring widely across Australia (and so, relevant to many landscapes), this species has mechanisms to deal with physical damage caused by natural processes (e.g. limb abscission) which more readily result in the formation of natural hollows (Best *et al.* 2022). As is the case with many eucalypts, this compartmentalising of physical damage is done through the production of wound-wood – as well as closing over wounds to reduce decay, the production of wound-wood also acts to strengthen areas weakened when wood tissue is removed (Kane and Ryan 2003). This may reduce the risk of host-tree damage arising from installation of chainsaw-carved hollows. At approximately five years post-installation, Best *et al.* 2022 reported no failures had occurred in any of the tree trunks in which chainsaw hollows were carved and none of the trees showed any signs of reduced growth or vitality.

Ellis *et al.* (2022) found the 10-mm inspection holes drilled into trees to determine which trees supported internal cavities, rapidly sealed over in live trees. The damage to tree stems caused by the drilling method used by Ellis *et al.* (2022) was minimal compared to creating chainsaw hollows (Rueegger 2017) or excavating an entire cavity using a series of drill holes and routing a void (e.g. Carey & Sanderson 1981). The amount of stem circumference cut using the Ellis *et al.* (2022) technique was only the diameter of the entrance hole; the internal void/decay being tapped into had developed naturally. Thus, the method used in Ellis *et al.* (2022) should result in the stem undergoing additional resistant growth, thus strengthening the stem, unlike the sudden imposition of structural alteration using cavity excavation methods (Ellis *et al.* 2022). Only long-term monitoring of trees treated by the various methods will resolve how important this difference is to the survival of the treated stems (Ellis *et al.* 2022).

There is no research available in the published or available unpublished literature that addresses what treatment of the hollow section is required to prevent disease/pathogen transfer, termite infestation, or infection of new wounds when translocating hollows (although see Central Coast Council 2016). Host tree health and potential damage to old trees must be given careful consideration prior to any large-scale adoption of a hollow replacement approach that requires the attachment of hollow alternatives to old growth, hollow-bearing trees.

9 Safety, logistics and economic considerations

9.1 Time/economic requirements

Provision, maintenance, replacement and monitoring of effectiveness of hollow alternatives requires significant investment of time and money. For projects of scale approaching that of VMFRP, it has been estimated the cost of implementing and maintaining a nest box strategy would cost at least several tens of millions of dollars (e.g.

Lindenmayer *et al.* 2017) and ultimately may not be effective – or worse, may be deleterious to ecosystems – in a landscape where naturally-occurring hollows are diverse and abundant. Nest boxes require significant materials and labour to produce *en masse* and require specialist installation (especially due to working at heights with heavy lifting), maintenance, replacement (more materials and specialist installation), and monitoring (especially, for safety and effectiveness). Plastic and 3-D printed nest boxes present similar time and economic requirements and are not available for such large-scale projects as VMFRP. Likewise, chainsaw-carved hollows require specialist installation and regular management to ensure they remain open and accessible. Erecting log hollows, dead trees or nest-box bearing utility poles would require specialist installation and considerable use of heavy machinery in sensitive environments, as well as ongoing maintenance, replacement and monitoring. Given the issues identified with each of these hollow alternatives (including the considerable ethical and ecological implications associated with hollow alternatives), it is extremely unlikely the economic and time costs could be justified of funding the implementation, maintenance, replacement, management and monitoring of a hollow replacement programme using these approaches. This is particularly relevant for large-scale projects such as VMFRP that extend across a wide geographic area where there is an abundance of naturally-occurring hollows. Funds and resources (time, materials, expertise, etc.) required to implement and maintain such a hollow replacement strategy would likely achieve better outcomes for hollow-using species if expended in other more considered and beneficial ways.

Of particular note: the entrance-drilling method used by Ellis *et al.* (2022) may be more cost effective than chainsaw excavation, attaching constructed nest boxes, or other hollow alternatives, because far less wood working is involved. Also, there is no attrition associated with creating an entrance into an existing internal cavity, meaning time and funding savings. The drilling method studied in Ellis *et al.* (2022) has the additional element of requiring stem testing to identify the presence of internal voids/decay, but both the exploratory stage and subsequent entrance addition can be done using a drill rather than a chainsaw (Ellis *et al.* 2022). So, this type of cavity access can potentially be safely created without needing arborists. However, this method is reliant on the presence of stems with existing decay or voids. The slow process of identifying suitable trees may be a disadvantage of this method, but the process was accelerated in Ellis *et al.* (2022) by use of local expertise about stem condition. There also are less invasive, but more expensive and cumbersome, techniques that can be effective at detecting internal decay or voids in trees; future development of such techniques may make them feasible as landscape-scale conservation tools (Ellis *et al.* 2022).

9.2 Maintenance

9.2.1 Nest box inspection

Lindenmayer *et al*. (2017) based cost-estimates for maintaining a nest box programme on a 90-year period – this period of maintenance is realistic if a nest box (or any artificial hollow replacement) is to address the lag period for hollow development by providing any kind of denning or nesting resource while natural hollows are developing in the landscape. Also, data sets of this duration would help address a critical knowledge gap in how best to develop *effective* hollow replacement strategies, as *per* international literature on long-term effectiveness of hollow replacements. The estimates by Lindenmayer *et al*. (2017) were premised on two maintenance checks per year or 180 checks in total; this schedule was for a high rainfall area in tall forests. Likewise, Lindenmayer *et al.* (2017) present compelling arguments for establishing adequate monitoring, maintenance and replacement regimes that would be most able to generate an effective hollow replacement/offset strategy – the funding required to implement and maintain an effective hollow offset programme was calculated at around $27 million (AUD in 2010), an amount far greater than would typically be allocated to implementing and maintaining a nest box (or other hollow alternative) programme. However, Goldingay *et al.* 2018 suggests one check every 5 years (i.e. 18 checks over 90 years) may be adequate in their study area (drier woodlands and forests), which would reduce the cost of such a programme by as much as 90%. Further research is needed into the frequency of maintenance and its influence on nest box functionality, particularly in relation to the climatic conditions local to where the nest boxes are installed. As a management tool, there is a need to recognise where local circumstances may not be suitable (Goldingay *et al.* 2018).

9.2.2 Nest box replacement

Lindenmayer *et al.* (2009) determined that infestations of pest invertebrates coupled with high attrition rates of nest boxes would necessitate a regular fumigation and replacement schedule in forests where natural cavities are rare (nest box invasion by European Honeybees was higher in young forests, as was the rate of nest box attrition). Moreover, the relatively limited effective occupancy time recorded in that study suggests many nest box replacements might be required if a management aim is to provide a perpetual supply of potential cavities over a period of ~120–150+ years until sufficient naturally-occurring hollows develop for use by hollow- using arboreal marsupials (Ambrose 1982; Lindenmayer *et al*. 1993). Such a fumigation and replacement schedule clearly would have substantial logistical and financial implications (McKenney and Lindenmayer 1994).

9.2.3 Wound wood occlusion

Ellis *et al*. (2022) drilled inspection holes in approximately 125 tree stems in one study site and 30 stems in a second study site to discover stems containing decay or voids. The 10-mm inspection holes which were not enlarged into entrance holes rapidly sealed over in live trees (Ellis *et al.* 2022). During early stages of the study, wound wood growth was restricted to the perimeter of drilled entrance holes and all entrances remained open. However, rapid growth later in the study (potentially rainfall-related) started to occlude half of the holes (Ellis *et al.* 2022). This varied among trees and was not related to time of entrance creation, with two holes created in September 2019 almost totally grown over by May 2021 (Ellis *et al.* 2022).

Similarly, wood wound occlusion impacted the effectiveness of chainsaw hollows in several studies. For example, Best *et al.* (2022) discovered wound wood growth had closed one chainsaw hollow and reduced the entrances of a further five so that they were too small for Krefft's Gliders to access (i.e. <2.0 cm). Wound wood growth reduced the entrances of another 22 chainsaw hollows, but not enough to restrict access by Krefft's Gliders (Best *et al.* 2022). Entrances to the remaining 20 chainsaw hollows were not reduced by wound wood (Best *et al.* 2022). Data from Best *et al.* (2022) and other studies confirmed that removing wound wood (produced as live, healthy trees close over chainsaw hollows) is likely to be required as an ongoing maintenance action (Rueegger 2017; Griffiths *et al*. 2018; Terry *et al*. 2021). Stakeholders considering carving chainsaw hollows into live, healthy trees should be aware that ongoing monitoring will be required to monitor wound wood growth and identify maintenance actions required to ensure the chainsaw hollows remain functionally available to wildlife (Best *et al.* 2022).

The need for ongoing monitoring and maintenance of traditional hollow alternatives such as timber nest boxes has been well-documented (e.g. Lindenmayer *et al.* 2009; 2017), however, the frequency and timescale over which similar monitoring and maintenance actions will be required for chainsaw hollows carved into live trees is largely unknown, despite the rapid uptake of this (largely untested) approach across Australian landscapes. Carving chainsaw hollows into dead trees would eliminate the problem of wound wood growth closing over entrances but introduces other uncertainties including cavity microclimate and longevity.

9.3 Safety considerations of each hollow type

Creating entrance holes to existing internal cavities seems the approach with the least safety implications. Chainsaw carved hollows need to be carved at the height at which the hollow is to be installed and require operating a chainsaw at height. Nest box installation also needs to be done at height, lifting heavy nest boxes into place and attaching them to the host tree whilst navigating a ladder or work platform. By comparison, the entrance hole creation requires only a ladder and drill, providing access to internal cavities from relatively safe heights (e.g. 2.4 – 4.8m; Ellis *et al.* 2022).

10 Social considerations

Often, nest box (and other hollow alternative) programmes are implemented, monitored and maintained by passionate and dedicated community groups, for example, the impactful Whroo Goldfields Conservation Management Network. Lawton *et al.* (2022) considered the role community played in the installation and monitoring of a nest box project designed to enhance the conservation of the threatened Brush-tailed Phascogale. They found important benefits arose because this project was community-led; it provided significant opportunities for education and engagement, leveraged community networks to extend the project into sites on private land, and has been maintained over the long-term and remained resilient to funding fluctuations owing to volunteer involvement (monitoring continued even after funding ceased for the project (Lawton *et al.* 2022). The study demonstrated that nest box monitoring can engage the community in citizen science (Lawson *et al.* 2022). Elements that enhance community-led monitoring include scientific input to project design, collecting data in a consistent manner, allocating sufficient time for data curation, engaging people invested in project outcomes, maintaining good relationships with stakeholders, and sharing data for analysis (Lawson *et al.* 2022). Macak (2020) provides an excellent summary of the important role community plays in the establishment of nest box programmes, whilst also highlighting the implications when community resources (time, people, funding) become stretched. Certainly, nest box programmes can build community. However, establishing such a group to install and maintain nest boxes (or other hollow alternatives) in remote landscapes such as the VMFRP sites may be challenging.

Further, some aspects of monitoring and maintaining hollow replacement programmes can be disheartening for community. For example, not considering the microclimatic properties of the installed nest boxes may have implications for the morale of the community groups investing time and other resources into nest box programmes that may fail due to nest boxes not providing effective or safe habitat.

11 Performance measures

11.1 What level of occupancy should be considered a criterion for success?

The literature does not provide guidance as to what level of occupancy should be considered a criterion for success. Likewise, the literature indicates no single approach to measuring 'success' or 'failure' has been accepted as standard. Hence, it is difficult to establish what might define 'success' of hollow alternatives or a hollow replacement strategy (Goldingay *et al.* 2018). Indeed, 'success' or 'failure' typically are determined at the individual study level, against criteria or hypotheses established by the authors. Lindenmayer *et al*. (2017) concluded from their study that 10% occupancy (i.e. occupied during inspection) was a plausible expectation for the Squirrel Glider on the south-west slopes of New South Wales – it is worth noting, this expectation is based on occupancy rates of a species (the Squirrel Glider) that is known to have wide hollow-requirements, is common and widespread in the subject landscape, and is known from the literature to readily use a nest box. This is not the case for most target species. Therefore, the expectation of occupancy rates would be considerably lower for threatened, uncommon, migratory, or rare species, or those species with narrow hollow-requirements. The expected occupancy rates would further be reduced to approaching zero *per cent* in landscapes where naturally-occurring hollows are available. However, in the absence of better estimates of predicted (and so, 'successful') occupancy - and factoring in the potential influence of local availability of hollow-bearing trees, a species' home range size, and abundance of other hollow-users – some other studies (e.g. Goldingay *et al*. 2018) also have adopted the 10% value to test against their data, despite the fact in many landscapes this is arbitrary and influenced (as discussed) by a complex intersection of variables. Thus, any measure of 'success' (or other performance target) of hollow replacements should be developed with caution –

setting an arbitrary expectation that is not supported by adequate empirical data in the same (or analogous) landscape using the same hollow replacement approach and the same target species (or faunal communities) may well set up a hollow replacement strategy for failure, potentially leading to millions of dollars of wasted funds, disheartened stakeholders, negative public sentiment, and unintended (and likely, unacceptable) wildlife and ecological impacts. Indeed, if diverse and abundant naturally-occurring hollows are available in the landscape and are suitable for hollow-using species, an adaptive management approach will unlikely ever successfully attract hollow-using species (particularly those species that will not use hollow alternatives!), leading to further failure of the hollow replacement strategy.

In terms of duration, Lindenmayer *et al.* (2009) recorded reasonable levels of occupancy in young forest ~2–3 years after box establishment and then high levels of nest box attrition after 8–10 years. Nest boxes in older forests were occupied by almost no animals and also demonstrated high levels of nest box attrition after 8-10 years. The study found the effective occupancy time was approximately five years for the arboreal marsupials considered in that research. The literature demonstrates occlusion of carved and drilled hollow replacements also limited occupancy; in some studies, the occlusion had completely limited access to cavities within several months of installation.

11.2 Establishing the effectiveness of a hollow replacement strategy

Ultimately, the efficiency of a hollow replacement strategy must be determined by the extent to which the hollow alternatives provide 'suitable habitat' commensurate to that of a natural hollow. To date, no such alternatives to natural hollows exist. The literature clearly demonstrates issues with the effectiveness of hollow alternatives. For example, occupancy rates commonly are low (see above); it is not always possible to attract target species; non-target species may occupy the provided hollow alternatives; attrition rates are high (see above); maintenance is high (above); as well as many wildlife (and other) considerations (see above). Much of the literature considers the effectiveness of hollow alternatives at short time scales (e.g. 1.5 years – 25 years) and reveals much disagreement as to whether hollow alternatives really are effective at offsetting the loss of naturally-occurring hollows. Most studies conclude hollow alternatives provide some benefit to hollow-dependent fauna but do not adequately or effectively replace natural hollows. There is a paucity of longer-term studies – given hollow alternatives must be effective (that is, provide 'suitable habitat') for between 50 and 120 years after installation, to allow for the development of new hollows (Lindenmayer *et al.* 2009), the available literature demonstrates it is highly unlikely any hollow replacements will meet this performance target.

Indeed, Lindenmayer *et al.* (2017) considered 'the anatomy of a failed offset' and discovered the loss of naturally-occurring hollows was not offset, despite the case study being a sizeable nest box programme. Importantly, one of the greatest failings of the case study (and most other published and unpublished hollow replacement programmes) was the lack of monitoring and maintenance (Lindenmayer *et al.* 2017). Lindenmayer *et al.* (2017) noted the expenditure for the case study nest box programme would likely have been ~$200,000 (AUD in 2010), considered by the authors to be a gross overspend given low levels of nest box use by target species and high rates of attrition of nest boxes. Ultimately, it was considered the programme had failed to be effective at offsetting the loss of hollows, despite that being the fundamental aim of the nest box programme (Lindenmayer *et al.* 2017). For this case study hollow offset programme to have been effective, Lindenmayer *et al.* (2017) calculated the appropriate cost would have been approximately $26.9 million (AUD in 2010). This figure includes funding effective levels of monitoring, installation of a more appropriate offset of nest boxes (5:1 rather than 1:1) and replacement of each nest box over a 90-year hollow replacement period (Lindenmayer *et al.* 2017). Yet, no matter the amount of funding directed at an offset plan or hollow replacement strategy, the performance measures of the current hollow alternatives do not indicate there is an alternative that performs anywhere near as well as a naturally-occurring hollow.

Land managers, governments, planners and others must consider whether the wildlife, economic and ecological implications of failed hollow replacement strategies create more harm than good. Investing funds in developing, studying and reporting findings of novel hollow replacements may be a much more effective use of funding than another (likely) failed hollow replacement strategy that may be harming the very ecosystems it intended to support. Indeed, the funds for implementing even one hollow replacement strategy could be well spent on developing a more effective hollow replacement model which could advance all future hollow replacement efforts.

12 Data collection

Although there is a growing body of literature relating to use of nest boxes and other hollow alternatives in Australia, research into using nest boxes as a management tool in Australia is still in its infancy (Goldingay *et al.* 2018). Largely, this is because many nest boxes across the Australian landscape have been installed as part of community-led projects. Macak (2020) found a wide range in the frequency and regularity of monitoring across community-led nest box projects, with some nest boxes checked only once since being installed. Many projects monitored nest boxes every 2-3 years or less, and many initially conducted regular nest box checks but checking decreased over time and sometimes ceased altogether (Macak 2020). The lack of monitoring in Australian nest box programmes was linked to the availability of people, or the capacity of community groups to coordinate checking of boxes in different locations. Much of this monitoring is unfunded and conducted manually by volunteers. Improving funding of monitoring associated with projects (including, but not limited to, community-led projects) seeking to install hollow alternatives, should include adequate funding to support monitoring. Some effort should be made to regularly synthesise and publish these data so the effectiveness of nest boxes and other hollow alternatives as a management tool can be established. In Sweden, North America and England, some nest box programmes (of ~80-150 nest boxes) have been monitored for 41-64 years (Schölin and Källander 2011; Shutler *et al*. 2012; Burgess 2014). Other programmes, involving >200 nest boxes have been monitored for 15-30 years in Canada, England, Sweden and France (Robertson & Rendell 2001; Goodenough *et al*. 2008; Corrigan *et al*. 2011; Hipkiss *et al*. 2013; Lambrechts *et al*. 2016). Such long-term data sets would provide more equivocal evidence about the comparative effectiveness of hollow alternatives, occupancy patterns, use of hollow alternatives by threatened species, landscape dynamics, attachment learnings, management costs associated with installation and management of hollow alternatives, novel construction materials and methods, and more. This investment would help ensure hollow replacement strategies were effective offsets for hollow removal and provided effective alternatives for displaced hollow-dependent wildlife.

Van der Ree (2019) agues the *ad hoc* approach to constructing and monitoring chainsaw hollows (and other hollow alternatives), host trees and occupants, is an approach that '*will at best take a long time and much effort to generate sufficient reliable data to be able to evaluate the effectiveness of carved hollows because of the enormous variation in:*

- *the type, size and construction technique of hollows installed;*
- *the species, health and size of the host trees;*
- *the prevailing weather and other environmental conditions among sites;*
- *different management regimes (e.g. risk assessments, pruning techniques and intensity); and*
- *the type and quality of data being recorded.'* (p. 45)

In effect, *'every hollow installed should be considered part of an experiment, and by considering and adopting a few guiding principles, the maximum amount of reliable and robust information can be learnt in the shortest amount of time, leading to more rapid adoption of evidence-based best-practice techniques'* (van der Ree 2019, p. 45). To this end, van der Ree (2019) recommends land managers recognise the critical role they play in ensuring the hollows they install are part of a commitment to learning whilst installing the hollows and establish an experimental design approach to hollow installation. Once the commitment to learning is established, the land manager should seek to formulate a question or hypothesis to test, for example: does species X prefer a nest box, carved hollow, log hollow or natural hollow; or does species X prefer entrance sizes that are 50mm, 60mm or 100mm diameter? (see van der Ree 2019). Some questions may be able to be answered using existing published and unpublished data sets; though abandoned monitoring protocols may impact the quality and robustness of available data sets, there may still be some useful and interesting data which further research can extend. In this way, data collected can be used help guide an adaptive approach to rolling provision of hollows.

As well as providing guidance on appropriate experimental design to help evaluate the effectiveness of chainsaw hollows, which has relevance to any hollow alternative programme, van der Ree (2019) stresses the critical role collaboration plays in generating successful and high quality hollow installation programmes, and highlights that collaboration can bring together people with different skill sets, resources and approaches leading to diverse benefits including cost reductions and sample size increases.

12.1 Methods to monitor use of tree hollows

Several methods have been used to monitor the use of tree hollows by mammals and birds: i) radio-tracking, ii) trapping on trees with hollows, iii) detection of tree-use scratch marks, iv) direct hollow observations, v) remote cameras, vi) hollow observations with a thermal camera, and vii) ultrasonic bat detectors. Goldingay (2021) (Table Two) provides a helpful review of these methods (pp. 12-15).

Table 2 A brief evaluation of different methods used to monitor the use of tree hollows (from Goldingay 2021).

Methods	Advantages	Disadvantages
Radio-tracking	No ambiguity about current use. Various species can be sampled.	Labour-intensive. Sample size may be low due to the need to trap animals and attach transmitters.
Tree-trapping	Provides age-sex data for captured animals. May provide data for many individuals using hollows.	Labour-intensive. May provide few data relative to effort. Captured animals might not be using hollows.
Tree-use scratch tracks	Very low cost. Only binoculars required. Large sample size of trees surveyed.	Some tree species unsuitable. Animal species making the tracks unknown. Small species may produce less wear and be overlooked. Birds may be under-sampled.
Direct observations	Very low cost. Only binoculars required. Large sample size of trees surveyed. Can be applied day or night to sample birds and mammals.	May require a large amount of effort to obtain any data.
Remote cameras	Long periods of monitoring. Can operate day and night.	Requires climbing trees or using elevated work platforms to install cameras. Sample sizes limited by equipment and access to hollows. Safety issues will preclude monitoring of some/many hollows.
Thermal cameras	May provide more reliable nocturnal sampling.	Unproven reliability. Equipment is very expensive. Limited equipment availability will limit sample sizes. Small species unreliably sampled.
Ultrasonic detectors	May provide abundant data on hollow-using bats.	Data may not accurately reflect hollow use. May be some limitation to the distance of detection.

Goldingay (2021) considers visual observation to identify wear marks around hollows has much potential, also, as it would enable the evaluation of many hollows over a short period. However, this is unlikely to yield accurate results regarding time since last use, the nature of the visit, whether the animal used the hollow, for what the animal used the hollow, for how long the animal used the hollow and which animal visited the hollow. The method likely to produce the most reliable data is the use of remote cameras but this method has the most limitations due to an inability to install cameras wherever monitoring is needed, with installation likely requiring an elevated work platform (working from this has occupational health and safety considerations as well as being time consuming to erect) (Goldingay 2021).

12.2 Direct and remote methods to monitor use of hollow alternatives

The literature demonstrates a range of direct and remote methods are being used to monitor use of hollow alternatives. These methods are consistent with those identified by Goldingay (2021) for monitoring natural hollows, excepting several studies of hollow alternatives that also remotely tracked cavity microclimate. A subset of methods is presented here to assist stakeholders to determine the relative intrusion, efficiency, accuracy and reliability of a range of methods.

In Goldingay *et al.* (2020) nest boxes were checked over 2– 4 days during April–June in each of three consecutive years. (It is worth noting that such short periods of data collection likely will yield different data to long-term data sets (for example as population dynamics and food availability change over time) and will fail to identify the influence of weather cycles (ten years' data for temperature, 30 years' data for rainfall) on long-term occupancy patterns and effectiveness of the programme). A camera attached to a pole with the lens directed downwards was used to lift the nest box lid, and a photograph was taken of the inside. Mammal species or their distinctive nests were identified. The Phascogale typically deposits scats in the top corner of its nest, allowing use to be determined unambiguously. Whether they were fresh scats was determined by how black and shiny they were. Occasionally, Goldingay *et al.* (2020) used a ladder to inspect all the nest boxes to determine recent use but specifically to ascertain how many contained Phascogale maternity nests.

To determine occupancy of nest hollows and chainsaw hollows, McComb *et al.* (2022) used motion-heat-activated camera traps (Reconyx HC600 and PC900 Hyperfire: Reconyx, Holmen, WI, USA) positioned on an adjacent tree 0.7–3 m distant and directed at the den entrance. Camera traps were programmed to take five pictures per trigger at high sensitivity, with no delay between triggers (McComb *et al.* 2022). Camera trap photos were reviewed to assign occupancy status (occupied, unoccupied, uncertain) for each den at 30-minute intervals (McComb *et al.* 2022). As Leadbeater's possums are nocturnal, a den was designated as 'occupied' during the day if photographs showed individuals emerging at dusk and returning at dawn; McComb *et al.* (2022) found emergence and return times to be highly consistent between nights. The batteries of some cameras did not last the full sampling period (eleven boxes in winter and nine in summer), and nest boxes in lowland swamp forest were not monitored with cameras during the winter period owing to a shortage of cameras (McComb *et al.* 2022). Temperature differences between ambient and nest boxes with known occupancy were used to assign occupancy to boxes for which camera images were not available (McComb *et al.* 2022). Nest boxes were treated as occupied if recorded nest box temperatures were consistently >2°C higher than ambient during daytime hours (McComb *et al.* 2022). This method was only applied over winter when heat signals from animals in occupied nest boxes were stronger (McComb *et al.* 2022). In summer, nest box occupancy was classed as uncertain when camera data were not available. For all nest boxes monitored with cameras, McComb *et al.* (2022) also determined colony size and breeding activity (based on the presence of juveniles).

In Rueegger (2017), carved chainsaw hollows were inspected nine times over a 15-month period. Two additional inspections were carried out 21 and 24 months post hollow creation to document wound wood development, tree stability and tree vitality (Rueegger 2017). Hollow inspection started at the end of October 2015, followed by end of November 2015 and mid-February 2016 inspections. The subsequent inspection interval was approximately two months (Rueegger 2017). A snake eye camera (unbranded) with a 1 m long flexible tube was used to inspect the hollows through the entrance hole (Rueegger 2017). As well as identifying the presence of an animal in the hollow, evidence of past hollow use, such as leaf/twig nests were recorded (Rueegger 2017). Passive monitoring took place using heat/motion activated cameras (Scoutguard SG550 & KG680V and one Reconyx HC500) (Rueegger 2017). Cameras were installed at nine of the hollows at the time of hollow creation; after 2.5 months, additional cameras were installed to monitor the remaining seven tree hollows, as well as four 'control' trees (trees that did not contain hollows) to monitor the frequency of animal detection on trees with and without created hollows (Rueegger 2017). Control trees were randomly chosen within the range of the hollow host tree species, DBH and height. Cameras were installed ~1 m above the hollow entrance attached to a bracket and ~5 m above ground for the control trees (Rueegger 2017). The cameras were set up to record 10-second videos (or five pictures in quick succession for the Reconyx HC500 camera), with a minimum interval of five minutes between triggering (Rueegger 2017). Video records of animals were sorted into three classifications: animal entered/exited hollow, animal inspected hollow, or animal walked past hollow (Rueegger 2017). Two-tailed Fisher's exact tests were used to analyse the frequency of hollow type use based on entrance size, though Rueegger (2017) acknowledged the sample size was low as species' hollow type preference was not a focus of that study.

Best *et al.* (2022) also used motion-activated infrared-sensitive cameras (HyperFire HC600, Reconyx, USA) to continuously monitor fauna visiting and inspecting a subset of 40 chainsaw hollows from September 2018 to January 2019 (total camera trap days = 5128, mean days per CH = 102 ± 10.9); equipment failure at eight chainsaw hollows meant not all 48 were surveyed (Best *et al.* 2022). Cameras were attached to trees approximately 1 m above the chainsaw hollow entrance, facing down the trunk toward the ground (see Griffiths *et al*. 2020). A series of five photos were taken per trigger, with no delay between trigger events (Best *et al.* 2022). 'Visitation' was defined as a photograph of an animal interacting with the tree trunk at the location of the chainsaw hollow, and excluded photos of animals present on other parts of the tree, or on the ground (Best *et al.* 2022). Multiple photographs of the same animal taken during a single trigger event were scored as one visitation (Best *et al.* 2022). To record use of the chainsaw hollows by Krefft's Gliders and other wildlife, Best *et al.* (2022) conducted occupancy surveys on 14 separate occasions 4–32 months post installation. Each occupancy survey was conducted over 2–3 days due to distances between reserves; all chainsaw hollows at each reserve were checked on the same day. During the first four occupancy surveys, Best *et al.* (2022) used a wireless digital camera (QC-8712, Signet, Australia) attached to a 4-m telescopic pole, which enabled passive, non-invasive observations inside the CHs. For the remaining 10 surveys, we employed the same procedure, but used a custom-made digital camera (Reach HD250, H. Wegner), which wirelessly streamed video and photo data to a smart- phone (Best *et al.* 2022). All observations of chainsaw hollow occupancy were recorded, regardless of species (Best *et al.* 2022). Best *et al.* (2022) defined 'occupancy' as: (1) the presence of an animal or eggs inside the chainsaw hollow at the time of inspection, or (2) evidence of a distinctive cup-shaped leaf nest constructed by a Krefft's Glider (Harper *et al.* 2005). Empty nests were only counted as a positive record of Krefft's Glider occupancy during the first survey when a newly built nest of fresh, green Eucalyptus leaves was detected (Best *et al.* 2022). During the first four occupancy surveys, all 48 chainsaw hollows were checked. For surveys 5–14, a total of 41 chainsaw hollows were checked; the six chainsaw hollows with wound wood growth that restricted access by Krefft's Gliders were not checked because the pole camera could longer fit through the entrance hole (Best *et al.* 2022). One further chainsaw hollow was not available to Krefft's Gliders during surveys 5–14 because it was occupied by European Honeybees *Apis mellifera* (Best *et al.* 2022).

Terry *et al.* (2021) conducted monthly monitoring of cavities for the first year, bimonthly in the second year and then twice in the final six months. A ground operated inspection camera (Bright- Star, Melbourne) was used to check inside each cavity without disturbing any wildlife (Terry *et al.* 2021). The species, total number of animals, or presence of nesting material or scats were recorded (Terry *et al.* 2021). Infrared flash camera traps (Reconyx Hyperfire 2 HF2X) were installed on three chainsaw hollows at two different locations immediately following construction to determine the interval before they were discovered by hollow-using species (Terry *et al.* 2021). Each camera was set to take five still images with no interval between each trigger over a combined total of 289 camera trap days (Terry *et al.* 2021). In a study that considered seasonal variation in target species movement, Terry *et al.* (2021) reported that Phascogales showed a pronounced seasonal variation in detection in cavities, with higher values recorded in summer and autumn. The strong seasonal variation in detection suggests the timing of surveys will be critical to measure the success of artificial cavities in future studies for Phascogales (Terry *et al.* 2021) and other species that show strong seasonal variation in movement and activity.

In their novel study of providing entrance holes to otherwise inaccessible, existing internal hollows, Ellis *et al.* (2022) installed a camera trap (initially PixController, replaced by Reconyx Hyper- fire 2 IR in 2018), mounted on a bracket above each entrance, with the camera suspended 75 cm from the stem facing the entrance (Figure Seven). This arrangement prevented animals using the bracket for easy access to the entrance (Ellis *et al.* 2022). Cameras were triggered by inbuilt motion sensors aimed near the entrance and continued taking still images until movement ceased (Ellis *et al.* 2022). Images were downloaded regularly until summer 2019–2020 when bushfires forced removal of equipment from the south coast and the subsequent COVID-19 travel restrictions limited further visits (Ellis *et al.* 2022). Images were viewed to determine if an animal had triggered the camera and the animal's behavior recorded (Ellis *et al.* 2022). Photographed events were classified as: 'incidental' for no apparent interaction with the entrance and were not further considered; 'inspection' where the animal approached to the edge of the entrance and inserted its head into the hole; or 'use' where the animal was recorded entering or exiting the entrance or was active inside the hole (Ellis *et al.* 2022). Frequently, individuals were recorded entering but not exiting, or occasionally vice versa. Due to the speed of animals compared to the camera frame rate and the nature of the sensor pattern, some events may have resulted in animals entering holes but the actual moment of entry not being recorded (Ellis *et al.* 2022).

Figure 7 Camera mounted on a bracket attached above a drilled hole giving access to an internal void, allowing the camera to photograph inside the entrance and adjacent area of the trunk. String to the side is attached to a temperature logger (Ellis *et al.* 2022).

Temperature loggers (Thermochron iButtons, Maxim Integrated Products) with a 0.5°C resolution were used in Ellis *et al.* (2022) to record temperature every 30 minutes in 10 cavities for two weeks in summer 2020 in one study area. Loggers were anchored outside entrances and suspended into the cavity on a string (see Figure Seven). Ambient temperature was recorded in a Stevenson's screen nearby (Ellis *et al.* 2022). Attempts to record temperatures in other cavities failed due to animals pushing the loggers out (Ellis *et al.* 2022). It is worth noting that reptiles, being heterotherms, are less likely to be detected by camera as their temperature is indistinguishable from their background much of the time. Since the cameras used in Ellis *et al.* (2022) were designed for detecting large endothermic animals, there was a likely bias in the species photographed. Recording equipment with a faster reaction time than the cameras used in that study would be needed to better understand the interactions between animals and the created entrances (Ellis *et al.* 2022). Additionally, activation of cameras required thermally detectable animals to cross multiple points within the infrared sensor field, meaning not all visits would be recorded (Ellis *et al.* 2022). A trigger mechanism located within the entrance may overcome this but may also deter animals from entering (Ellis *et al.* 2022).

To measure the temperature and humidity of artificial dens and ambient conditions, McComb *et al.* (2022) used iButtons Thermochron DS1922L (temperature) and Hygrochron DS1923 (temperature and humidity) (Maxim Integrated, San Jose, CA, USA), which have an operating range -20°C to 85°C (precision ±0.5°C) and 0% to 100% relative humidity (precision ±5%). The study had a limited number of Hygrochrons, so these were used to measure ambient conditions and placed in a subset of nest boxes (n = 40/57 in winter, n = 34/61 in summer) (McComb *et al.* 2022). All other nest boxes had temperature-only loggers (McComb *et al.* 2022). Data loggers mounted on plastic fobs were positioned in nest boxes on the south-east facing wall (the wall away from the tree; McComb *et al.* 2022). Leadbeater's Possum nests vary in size and can fill an entire nest box. McComb *et al.* (2022) installed data loggers on the nest box wall outside of the nest, and adjacent to the midpoint of the nest height. In chainsaw hollows, data loggers were installed between the side of the cavity and the nest by means of a metal bracket positioned approximately 4–12 cm below the entrance hole (McComb *et al.* 2022). Data loggers were orientated to ensure the sensor surface was facing towards the nest, which has been shown to be the correct orientation to accurately measure relative humidity (McComb *et al.* 2022). Ambient temperature and humidity were recorded for a cluster of four chainsaw hollows within a 200 m radius, or for a cluster of two to four nest boxes within a 500 m radius (McComb *et al.* 2022). Ambient data loggers were attached underneath the nest box or to the trunk of the tree at chainsaw hollow sites 1–2 m off the ground on the south side, to ensure they were not exposed to any direct sunlight (McComb *et al.* 2022). Temperature and humidity were recorded every 30 minutes for 7–12 weeks over winter (June–September) and 10–11 weeks over summer (December–March) (McComb *et al.* 2022).

In an important pilot study, Moore *et al.* (2010) investigated and validated the innovative use of temperature dataloggers (iButtons®, Maxim Integrated Products) for remotely monitoring nest box use without needing to disturb or touch the occupants. The study design used iButton recordings in both a captive and field setting. iButton recordings tracked the duration and time of day when each nest box was occupied (Moore *et al.* 2010). In captivity, where only one species (Western Ringtail Possum *Pseudocheirus occidentalis*) was present, the accuracy of occupancy data was validated by unobtrusive infrared video recording. However, hair sampling at the nest-box entrances was required in the field to identify which of two mammal species (Western Ringtail Possum or Common Brushtail Possum) were present in the nest box (Moore *et al.* 2010). Although there was limited use of nest boxes at the field site, results from Moore *et al.* (2010) confirmed iButtons are useful for remote-monitoring of nest-box use in the field. Nest-box use by captive Western Ringtail Possums also validated iButtons as a useful remote-monitoring tool, with <5–6% error. Continuous (24-h) monitoring with minimal disturbance was found to be convenient (for the researcher and animal), however Moore *et al.* (2010) identified a major advantage from using iButtons was that occupancy could be matched with environmental temperature or rainfall records, as well as other events (e.g. storms or frost). This has significant benefits for establishing the effectiveness of hollow alternatives and their ability to provide suitable habitat for hollow-using fauna. Moore *et al.* (2010) determined iButtons are a useful remote-monitoring tool of nest boxes (also see McComb *et al.* 2022) and considered the results of their study demonstrate iButtons also may be a helpful technology for studies of naturally-occurring tree-hollow occupation. Indeed, Ellis *et al.* (2022) demonstrated successful use of iButtons to record temperature every 30 minutes in ten cavities to which they had provided entrance holes for wildlife to access the internal natural hollows. However, Moore *et al.* (2010) considered it important to note that the criteria they used for determining the presence or absence in the nest box (i.e. temperature difference, $T_{in} – T_{out}$, of 2°C) will not be relevant for all nest-box designs; they recommended investigating the thermal properties of the hollow alternative (e.g. nest box) or tree hollow before using the methods used in their study. Whether use of iButtons or similar technology could support pre-clearance surveys is worthy of consideration, as such surveys require only presence/absence data rather than necessarily a knowledge of which species is present in the hollow soon to be removed. Most importantly, this approach used by Moore *et al.* (2010) can help inform stakeholders as to the conditions under which hollow alternatives are used by fauna, as well as preferences for different nest-box designs. Payne *et al.* (2022) also found thermal imagers provide an effective, non-invasive and efficient method for monitoring nest box occupancy.

Experimental research on use of remote technologies such as camera and temperature loggers, including comparative analyses and validations of the technology, is vital for improving our understanding of the effectiveness of hollow alternatives and, so, the potential effectiveness of proposed hollow replacement strategies. Traditional methods of physical checking of hollow alternatives (mostly, nest boxes) can disrupt usage and occupancy patterns and impact wildlife; as well, physical inspections are time consuming and pose safety risks (e.g. working at height, manual handling, fatigue, exposure, remoteness) to personnel checking the hollows. In particular, undertaking physical checks is not practical or feasible for large-scale hollow replacement programmes (e.g. Lindenmayer *et al.* 2017), though it should be considered that the use, accuracy and reliability of remote technologies must be supported and validated by (comparatively, more occasional) physical checks; thus, even programmes using remote technologies must budget resources to conducting physical inspections.

Establishing reliable protocols for remote monitoring will mean less disturbance to wildlife and ecosystems as well as reducing the financial and time costs associated with monitoring and data collection. In this way, a better understanding of how remote technologies can be used for monitoring the effectiveness of hollow alternatives will mean funding can be extended to achieve longer-term studies and ultimately advance our knowledge of how best to *effectively* mitigate the loss of hollow-bearing trees.

13 Collaboration

Griffiths *et al*. (2023) and van der Ree (2019) highlight the imperative for a collaborative process for developing hollow alternatives and hollow replacement strategies. They rightly assert the design, installation and monitoring of hollow alternatives should be a collaborative process involving land managers, ecologists and, depending on the hollow alternative, arborists. It is vital this collaboration includes the collection, collation and publication of empirical data that can be used to inform an adaptive management framework for strategic hollow replacement (Griffiths *et al*. 2023; van der Ree 2019). Indeed, to some extent, this review highlights how *ad hoc* the research effort has been for investigating the effectiveness of hollow alternatives, with each study (more or less) being a standalone experiment with limited potential for comparative analyses that are vital for expanding our understanding of the effective use of hollow alternatives. Collaborating (within and between programmes) will help ensure the loss of naturally-occurring hollows can be mitigated in the most effective way, so scarce conservation resources (funding, time, people) are spent as effectively as possible, for *actual and realised* ecological returns.

14 Known habitat preferences of hollow-dependent fauna

Developing effective approaches to mitigating the loss of hollow-bearing trees is fundamentally and critically linked to an understanding of habitat preferences of hollow-dependent fauna. Thus, it is vital that these habitat preferences are used to direct stakeholders to mitigation strategies, rather than applying a standard approach (e.g. 1:1 hollow replacement) that may not, in fact, provide any *actual* mitigation. Indeed, many of the studies reviewed here have demonstrated the failure of hollow alternatives to effectively meet the habitat requirements of general hollow-dependent fauna, let alone those of targeted fauna. This section considers a summary of known habitat preferences of a range of hollow-dependent fauna. More research into habitat preferences is critical, however these summarised habitat preferences may provide insight into the possible habitat preferences of similar hollow-dependent fauna, for which studies are lacking.

A number of studies have been published relating specifically to hollow requirements of Leadbeater's Possum and other possums and gliders in Central Highlands of Victoria (e.g. Smith and Lindenmayer 1988, 1992). The forests of the Central Highlands of Victoria provide habitat for over 40 species of cavity-using vertebrates (Lumsden *et al.* 1991), including species of conservation significance such as Leadbeater's Possum (*Gymnobelideus leadbeateri*), Yellow-bellied Glider (*Petaurus australis*), and Sooty Owl (*Tyto tenebricosa*) (Milledge *et al*. 1991; Lindenmayer 2007). Some of these species' populations are known to be limited by a lack of hollow-bearing trees (Lindenmayer *et al.* 1991a).

Gibbons and Lindenmayer (2002) and Goldingay (2021) provide a summary of known habitat preferences for a range of hollow-using (really, hollow-dependent) mammals and birds.

In summary (Goldingay 2021), greater gliders prefer:

- large live DBH trees (≥100 cm)
- large volume hollows (i.e. large branches, opening into wide section of trunk)
- entrances <20 cm in diameter
- a relatively high density of these trees within retained clumps

In summary (Goldingay 2021), yellow-bellied gliders prefer:

- large live DBH trees (≥100 cm)
- large volume hollows (i.e. large branches, openings into wide section of trunk)
- relatively narrow (~10 cm) entrances
- den trees that are widely scattered across their large home ranges

In summary (Goldingay 2021), squirrel gliders prefer:

- live or dead hollow-bearing trees with larger DBH trees (≥50 cm)
- branch and trunk hollows equally
- hollow entrances of <5 cm diameter
- multiple potential den trees at approximately 0.5 per ha

In summary (Goldingay 2021), sugar gliders prefer:

- live or dead hollow-bearing trees
- branch and trunk hollows equally
- hollow entrances of <5 cm diameter
- multiple potential den trees at approximately 0.5 per ha

In summary (Goldingay 2021), feathertail gliders prefer:

- live or dead hollow-bearing trees
- branch and trunk hollows equally
- hollow entrances of <3 cm diameter
- larger cavities with narrow entrances when they den communally
- multiple potential den trees, at a density of 1 per ha, given their home ranges may be equivalent in area to those of the greater glider

In summary (Goldingay 2021), brushtail possums prefer:

- relatively large hollows of at least 10 cm diameter and 40 cm depth
- trunk or branch hollows equally
- multiple potential den trees at approximately 1 per ha

In summary (Goldingay 2021), eastern pygmy-possums prefer:

- live or dead hollow-bearing trees
- branch and trunk hollows equally
- hollow entrances of <3 cm diameter
- a cavity of at least 10 cm diameter for breeding
- multiple potential den trees at approximately 0.3 per ha

In summary (Goldingay 2021), brush-tailed phascogale prefer:

- live or dead hollow-bearing trees
- branch and trunk hollows equally
- hollow entrances of <5 cm diameter
- a cavity of at least 20 cm diameter for breeding
- multiple potential den trees at approximately 0.1 per ha

In summary (Goldingay 2021), hollow-using microbats prefer:

- live or dead hollow-bearing trees
- branch and trunk hollows equally
- hollow entrances of <5 cm diameter
- large DBH trees as maternity sites.

A landscape approach to roost and maternity tree retention should be applied given the long commuting distances of which most bats are capable (see also Gibbons and Lindenmayer 2002).

Goldingay (2021) also provides an excellent summary of hollow requirements for hollow-using birds. In particular, the species of interest considered by Goldingay (2021) were the large forest owls, glossy-black cockatoo and the brown treecreeper.

> *'When the hollow requirements of the Australian hollow-using birds were reviewed in 2009 there had been two studies published describing the hollows of the glossy black cockatoo, two for the powerful owl, one for the sooty owl, three for the masked owl, one for the barking owl and none for the brown treecreeper (Goldingay 2009). The average DBH of the nest trees across studies were: 70–99 cm for the glossy black-cockatoo, 130–163 cm for the powerful owl, 157 for the sooty owl, 137 cm for the masked owl and 120 cm for the barking owl. The only studies that described the entrance size of the nest hollows were one for the glossy black cockatoo (25 cm diameter) and the one study of the barking owl (28 cm).* ***The salient point here is the massive size of the trees selected as nest trees.*** *This may partly reflect the abundance of old- growth forest where these species occurred but also may reflect that the large type of hollows favoured by these species may only be provided by very large trees."* (Goldingay 2021, emphasis added)

15 Known habitat requirements of 'nominated priority fauna' within the VMFRP study area

The Victorian Murray Floodplain Restoration Project aims to restore the health of the Murray floodplain by reintroducing a more natural frequency, seasonality and extent of floodplain inundation. Historic and current river regulation and water extraction mean the Murray floodplain is receiving less water, less often, for shorter durations, and unseasonally, than the floodplain ecosystems require for health and functionality. Despite efforts to reduce the number of hollow-bearing trees being removed as part of the Project, hundreds of hollow-bearing trees are proposed to be removed across the VMFRP sites to facilitate works and operation. During construction, it is expected micro siting will enable the retention of some hollow-bearing trees that had been approved for removal. During operation, it is expected the more natural inundation patterns will restore the floodplain ecosystems and support the longevity of floodplain vegetation, including significant, hollow-bearing trees, which currently are in decline due to insufficient and inappropriate watering.

River Red Gum-dominated woodlands such as those of the Murray floodplain are well-known for supporting hollow-dependent wildlife. Indeed, River Red Gum *Eucalyptus camaldulensis* is considered one of the most important hollow-bearing species in floodplain ecosystems, not least because they are widespread. A study of semi-arid River Red Gum woodlands found 35% (n=113) of sampled River Red Gums were hollow-bearing and the density of hollow-bearing River Red Gums in that landscape was 16-48 (mean = 31) trees per hectare (Westerhuis *et al.* 2019). Westerhuis *et al.* (2019) found the best model for predicting the number of hollows in River Red Gums used a combination of equivalent DBH, stem number and canopy form. Whilst this study (and others) confirms River Red Gums provide important habitat for hollow-dependent fauna across the landscape, it is likely only a fraction of available hollows are suitable for hollow-using animals (Westerhuis *et al.* 2019), with the maximum estimate of hollow usage being around 43% of available hollows (Gibbons *et al.* 2000). This has critical implications for VMFRP, as it indicates (at most) less than half the available hollows in the VMFRP landscape may be being used; indeed, many of the hollows being removed may not provide suitable habitat (or, if suitable, be used) at all.

Hollows provide shelter, roosting and breeding opportunities for a range of fauna, including parrots, woodland birds, reptiles and mammals. Many of these taxa are of (state, national and/or international) conservation significance, largely as a result of widespread removal of hollow-bearing trees on which they are dependent.

Environmental assessments undertaken within the VMFRP ER Central project areas identified that hollow-bearing trees (including likely some of those trees proposed/approved for removal) provide potential or actual nesting and roosting habitat for the threatened:

- Regent Parrot (eastern subspecies) *Polytelis anthopeplus monarchoides*
- South-eastern Long-eared Bat *Nyctophilus corbeni*
- Barking Owl *Ninox connivens*
- Major Mitchell's Cockatoo *Lophochroa leadbeateri*
- Carpet Python *Morelia spilota metcalfei*

For each of several sites, the VMFRP environmental assessments have identified the likelihood of these taxa being present ('present') or possibly present ('possible') (Table Three).

Table 3 Likelihood of occurrence for five hollow-dependent fauna of conservation significance ('nominated priority fauna') across ER Central sites of the Victorian Murray Floodplain Restoration Project area.

	Conservation significance		Likelihood of occurrence by site	
Taxon	**EPBC***	**FFG^**	**Nyah**	**Vinifera**
Regent Parrot	Vulnerable	Vulnerable	Present	Possible
South-eastern Long-eared Bat	Vulnerable	Endangered	Possible	Possible
Barking Owl		Critically Endangered	n/a	Possible
Major Mitchell's Cockatoo	Endangered	Critically Endangered	Present	Present
Carpet Python		Endangered	Possible	Possible

* EPBC: Commonwealth *Environment Protection and Biodiversity Conservation Act 1999.*

^ FFG: Victoria *Flora and Fauna Guarantee Act 1988.*

As part of the Ministerial assessment of the VMFRP central packages, the Minister has recommended a hollow replacement strategy that is to 'provide for nominated priority fauna species on the basis of suitable evidence of their habitat requirements'. For some sites, the Ministerial Assessment did not stipulate the hollow replacement strategy should be developed for 'nominated priority species' however it seems logical to use this approach as the basis of developing a hollow replacement strategy. 'Nominated priority fauna' are considered those hollow-dependent fauna listed under the FFG- and/or EPBC Acts which have been identified in VMFRP environmental assessments as 'present' or 'possible' within a project site.

In order to provide for nominated priority fauna species, it is critical to first consider evidence of their habitat requirements. Secondly, it is critical to establish what implications the proposed tree removal has for these habitat requirements and whether the removal of these specific trees means there is a need to provide any additional provisions for these fauna.

15.1 Regent Parrot Polytelis anthopeplus monarchoides

	Conservation significance		Likelihood of occurrence by site	
Taxon	EPBC*	FFG^	Nyah	Vinifera
Regent Parrot	Vulnerable	Vulnerable	Present	Possible

* EPBC: Commonwealth *Environment Protection and Biodiversity Conservation Act 1999.*

^ FFG: Victoria *Flora and Fauna Guarantee Act 1988.*

Regent Parrot (eastern subspecies) is restricted to a single population occurring in inland south-eastern Australia, in the lower Murray-Darling basin region of South Australia, New South Wales and Victoria (see Baker-Gabb and Hurley 2011) (note: the reporting of 'present' for Regent Parrot at Nyah and Burra is outside the critical habitat of Regent Parrot reported by Baker-Gabb and Hurley (2011); the Regent Parrot is widespread but occasional and more scattered outside the three key breeding areas within the Mallee district (DCCEEW 2024a) indicating the central VMFRP sites are not core breeding or foraging habitat for Regent Parrot). Within this range, the eastern Regent Parrot habitat comprises River Red Gum and sometimes Black Box *Eucalyptus largiflorens* ecological communities for nesting and large areas of mallee woodland (usually Christmas Mallee *E. socialis* and Yellow Mallee *E. incrassata*) for feeding (SWIFFT 2024a). Nest trees usually are close to water and within 20 km (usually, 5-10kms) of mallee foraging habitat (DCCEEW 2024a). Some literature indicates the availability of, and proximity to, large areas of mallee woodland is more of a limiting factor than access to breeding habitat in River Red Gum/Black Box woodlands (Baker-Gabb and Hurley 2011).

It is worth noting, most movements by Regent Parrot across the landscape occur early in the morning, with few movements occurring after 11:15am (Sluiter 2007 in SWIFFT 2024a). Hurley (see SWIFFT 2024a) reported that 80% of the movement associated with nesting occurs between 7am and noon.

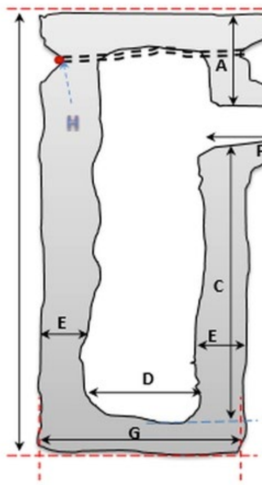
Typical nest trees are mature, senescent or dead with a height of ~30m, 1.6m Diameter at Breast Height (DBH) and a crown diameter of 17m (SWIFFT 2024a), usually close to water (Baker-Gabb and Hurley 2011). River Red Gums with a DBH of 1.6m are likely to be a minimum of 160 years old, most probably significantly older (Baker-Gabb and Hurley 2011). Hollows used for nesting are an average of 18-21m (range: 6-36m) above the ground (Burbidge 1985; Webster 1991; Baker-Gabb and Hurley 2011). Characteristic nest hollow measurements are small diameter openings (averaging 10cm high x 9cm wide) and deep hollows to nest chamber averaging 1 m deep (SWIFFT 2024a). Recent evidence from Regent Parrot (eastern) nest census work indicates that this species is extremely faithful to a particular nesting colony within the three key breeding areas, with hollows often re-used from year to year (DCCEEW 2024a). Occasional predation of nestlings from nest hollows can occur; Australian Ravens and various goannas including the Lace Monitor (*Varanus varius*) are likely to take Regent Parrot eggs or nestlings from their nest hollows (DCCEEW 2024a).

Clearing and degradation of mallee (foraging) habitat and River Red Gum (nesting) habitat are considered among several serious threats to Regent Parrot, especially within 20km of major rivers such as the Murray River (SWIFFT 2024a). In particular, commercial logging of Red Gum forests and woodlands for firewood has impacted Regent Parrot nesting habitat (DCCEEW 2024a). The Victorian Murray Floodplain Restoration Project (through the Mallee Catchment Management Authority (CMA) and North Central CMA) is recognised as a key land manager project in the conservation of the Regent Parrot in Victoria (SWIFFT 2024a); provision of more natural watering regimes across the Murray floodplain is recognised as a key solution to supporting Regent Parrot, as the health of River Red Gum-dominated floodplains has been affected significantly by modified flood regimes and infrequent flooding.

15.1.1 Considerations for hollow replacement strategy

A comprehensive literature search indicates no studies have considered use of hollow alternatives in the wild by Regent Parrots– no studies even mention installation of nest boxes or other hollow alternatives. Indeed, provision of nest boxes or hollow alternatives is not listed as a mitigation approach in Federal (DCCEEW 2024a) or State (Baker-Gabb and Hurley 2011) threatened species policy.

Despite there being no evidence Regent Parrots will use hollow alternatives in the wild, SWIFFT (2024a) provides a recommended Regent Parrot nest box plan (Figure Eight) based on measurements taken from 35 known Regent Parrot nests in naturally-occurring hollows in Victorian River Red Gums. According to SWIFFT (2024a), based on these hollow-dimension preferences, nest boxes must be constructed from hardwood and preferably from River Red Gum logs. The base of the nest chamber must be lined with 50 mm thick layer of activated charcoal covered with a further 50 mm thick layer of frass (termite dung and other detritus) from existing decaying River Red Gum logs on site. The lid must be hinged to allow inspection and be secured via a rare earth magnet and metal attachment plate. The lid must be able to open to at least 90 degrees (Hurley 2014 in SWIFFT 2024a) (Figure Eight). The literature does not demonstrate Regent Parrots will use these nest boxes, despite them being designed to tree hollow preferences displayed by Regent Parrots.



Measurement Code	Description	Measurement (mm)
A	Top of nest to top of entrance	200
B	Entrance Diameter	80
C	Depth from entrance to nest chamber floor	1200
D	Diameter of nest chamber floor	200
E	Nest wall thickness	110
F	Length of entrance tunnel	150
G	External diameter of nest box	420
H	Hinge join of lid	-

Figure 8 SWIFFT (2024a) recommended nest box plan based on measurements taken from 35 known Regent Parrot nests in Victorian River Red Gums.

Knowledge of Regent Parrot habitat requirements indicates nest boxes would need to be installed in mature, senescent or dead trees with a height of ~30m, 1.6m DBH and a crown diameter of 17m (SWIFFT 2024a). The tree should be close to water (Baker-Gabb and Hurley 2011) and in suitable proximity to mallee woodland. To meet habitat preferences, nest boxes would need to be installed at an average of 18-21m (range: 6-36m) above the ground (Burbidge 1985; Webster 1991; Baker-Gabb and Hurley 2011) (SWIFFT 2024a).

Plywood nest boxes do not provide adequate thermal insulation; internal temperatures often exceed external temperatures. Regent Parrots nesting in these boxes would need to cope with daily fluctuations in temperature of up to 25^{o}C, whereas for those in natural cavities the maximum daily variation is 12^{o} C (SWIFFT 2024a). Natural cavities are more thermally stable and likely to remain more attractive to Regent Parrots in the extremes of Victoria's Mallee region (Robertson & Hurley 2015).

The apparent absence of literature relating to use of custom (as above) or standard plywood nest boxes (or any other hollow alternative) by Regent Parrots suggests nest boxes and other hollow alternatives may not be effective hollow replacements for Regent Parrot. It is possible some unpublished data exists on Regent Parrot use of nest boxes (or other hollow alternatives); if made available, such data should be considered in the development of any hollow replacement strategy. Further investigation of the appropriateness and effectiveness of providing Regent Parrot-specific nest boxes, particularly outside recognised core breeding areas, is warranted. Indeed, installation of hollow alternatives that may not be used by Regent Parrots could encourage an increase of pests in the landscape (e.g. Oldroyd *et al*. 1994), increasing the vulnerability of the Regent Parrot. Removal of trees supporting Regent Parrot breeding hollows (most likely, within the three core breeding areas in the Mallee district), should be avoided, as the literature does not provide evidence that mitigation of hollow losses can be achieved through a hollow replacement strategy.

15.2 South-eastern Long-eared Bat Nyctophilus corbeni

	Conservation significance		Likelihood of occurrence by site	
Taxon	EPBC*	FFG^	Nyah	Vinifera
South-eastern Long-eared Bat	Vulnerable	Endangered	Possible	Possible

* EPBC: Commonwealth *Environment Protection and Biodiversity Conservation Act 1999.*

^ FFG: Victoria *Flora and Fauna Guarantee Act 1988.*

The South-eastern Long-eared Bat *Nyctophilus corbeni* (previously *Nyctophilus timoriensis* (south-eastern form)) is a rare species of microbat, which has a scattered distribution mostly within the Murray-Darling Basin. It is more common in box, ironbark and cypress pine woodland on the western slopes and plains of the Great Dividing Range (DCCEEW 2024b). The Piliga Scrub in north-western New South Wales seems to support the great numbers of this bat (DCCEEW 2024b). Threats to the species include habitat loss and fragmentation, fire, and reduction of hollow availability (DCCEEW 2024b).

The species roosts in tree hollows, crevices and under loose bark (DCCEEW 2024b). Law *et al.* (2016) considered habitat requirements of the South-eastern Long-eared Bat in north-western New South Wales. The study reported small maternity colonies (<10 bats) were found in hollows and fissures (often in exposed locations) of trees with a small diameter (means range 23–39 cm) that were usually dead (82.5% of roosts) (Law *et al.* 2016). Buloke *Allocasuarina luehmannii* was most used for roosting (49%), yet prior to Law *et al.* (2016) had been overlooked as a source of hollows for fauna. Landscape-scale habitat use was found to be subtle (see Law *et al.* 2016). In studies of roosting behaviour in Victoria, most bats were found roosting individually in mallee eucalypts in areas of long-unburnt (70+ years) mallee (Bennett 2016), with some under bark or in fissures of dead Buloke (*Allocasuarina luehmannii*) or Belah (*Casuarina cristata*) trees (TSSC 2015). The South-eastern Long-eared Bat is considered a 'narrow-space' bat, which avoids roosting in relatively open, thinned areas (Law *et al*. 2018). Not surprisingly then, areas of high stem density, especially those containing dead trees, were found to provide key roosting habitat for South-eastern Long-eared Bat (Law *et al.* 2016).

Although these studies were conducted in a different ecological community (indicating the River Red Gum floodplain ecosystems are not critical habitat), the findings prompt consideration of whether regenerating River Red Gum floodplains (i.e. those produced by episodic flooding events, which would increase during operation of the VMFRP) would provide greater habitat (i.e. higher stem density) than mature River Red Gum woodland. Further, hollow-bearing River Red Gums may not provide habitat for the South-eastern Long-eared Bat. Environmental assessments of the ER Central VMFRP sites suggest only a possible likelihood of the presence of South-eastern Long-eared Bat within the study area. Although South-eastern Long-eared Bats have been recorded in River Red Gum forests (TSSC 2015), the literature indicates the location and structure of vegetation within the study site is not likely to support this species of bat. The availability of suitable roosting habitats, particularly (at least, in Victoria) in Buloke, Belah and long-unburnt mallee eucalypts, is essential for the conservation of the South-eastern Long-eared Bat (TSSC 2015). This species mainly roosts in tree hollows and so a reduction in hollow availability in their preferred habitat would likely put pressure on the species. However, the literature suggests the removal of hollow-bearing trees within the VMFRP study area may not impact this species.

15.2.1 Considerations for hollow replacement strategy

The literature does not address the use of hollow alternatives for the conservation of South-eastern Long-eared Bat, presumably as there is a paucity of research relating to so many aspects of this species' ecology and biology. However, given the South-eastern Long-eared Bat is a microbat, the use of hollow alternatives recorded in other microbats may be helpful in developing a hollow replacement strategy if one was required. Nonetheless, it seems unlikely the hollow-bearing trees proposed for removal within the VMFRP central sites would provide habitat to South-eastern Long-eared Bat, though pre-removal checks should be undertaken of crevices suitable for microbat roosting. A hollow replacement strategy to mitigate impacts from the proposed loss of hollow-bearing trees seems unlikely to benefit South-eastern Long-eared Bat. Even if the species did occur in the subject vegetation structure and occurred within the study area, research into the use of artificial hollow alternatives indicates very few species of crevice-roosting bats actually use these alternatives (e.g. Griffiths *et al*. 2023).

15.3 Barking Owl Ninox connivens

	Conservation significance		Likelihood of occurrence by site	
Taxon	**EPBC***	**FFG^**	**Nyah**	**Vinifera**
Barking Owl	-	Critically Endangered	n/a	Possible

* EPBC: Commonwealth *Environment Protection and Biodiversity Conservation Act 1999.*

^ FFG: Victoria *Flora and Fauna Guarantee Act 1988.*

The Barking Owl is the most threatened owl in Victoria (Clemann and Loyn 2003). Relatively little is known of the ecology and habitat requirements of the Barking Owl *Ninox connivens* (Kavanagh *et al.* 1995; Taylor *et al.* 2002). The species, one of the Australian large owls, is distributed sparsely through temperate and semi-arid areas of mainland Australia, becoming more abundant in the tropical north (Kavanagh *et al.* 1995). In New South Wales studies, Kavanagh *et al.* (1995) reported the habitat of the Barking Owl in inland and coastal areas of New South Wales appears to be associated with open woodland or semi-cleared land near creeks and rivers, particularly with one or more species of red gums *Eucalyptus* spp., although not recently detected in the Red Gum forests of the Murray River. This is despite records indicating Barking Owls were found in the NSW Riverina and south-western NSW in grassy box woodland, riparian forests and floodplain forests fringing wetlands (McGregor 2011), much of which has been cleared since European settlement. In particular, the removal of large hollow-bearing trees would have reduced suitable trees in which Barking Owls could nest or roost (McGregor 2011). Between 1950 and 2000, Barking Owls were considered 'not uncommon' in the area's River Red Gum forests, but this is no longer their present status (McGregor 2011). Threats to this species include habitat removal, inappropriate fire management, competition with introduced predators, and secondary poisoning by rodenticides (Beranek and Hayward 2021).

Twenty pairs of Barking Owls were located and studied in northeast Victoria in a mosaic of Box-Ironbark forest and pastoral farmland; the study found the most frequently used nest and roost trees were Apple Box *Eucalyptus bridgesiana* and Red Box *E. polyanthemos* associated with Box-Ironbark forests and forests on drier slopes (Taylor *et al.* 2002). Earlier records show four nest hollows were discovered in River Red Gum (eight in Apple Box, six in Red Box, three in Blakely's Red Gum *E. blakelyi* and three in dead trees of unidentified species) (Taylor *et al.* 2002). The mean DBH of nest trees was 120cm (range: 60-260cm) and the estimated height of nest trees was 20m, with nest hollows typically around 10m (range: 4.8-19m) from ground level (Taylor *et al.* 2002). The average size of the entrance to hollows was 31.4 x 24.1cm with the smallest entrance recorded at 16 x 14cm (Taylor *et al.* 2002). Most hollows were slightly larger than the entrance; the minimum cavity dimensions were 23 x 15cm and the average hollow depth was 103.6cm (range: 30 – 200cm) (Taylor *et al.* 2002). Two thirds of nest hollow entrances had a southerly aspect compared to only one third of entrances with a northerly aspect, although this difference may have been due to a southerly bias in the availability of suitable nest hollows (Taylor *et al.* 2002). However, south-facing hollows may be preferred as they would be protected from the sun, especially during early summer when large chicks were in the nest hollows (Taylor *et al.* 2002). A single female Barking Owl was determined to have a territory size of 226 ha (Taylor *et al.* 2002), although the average home range in south-east Australia is reportedly 1400 ha (range: 800 – 8000 ha) (McGregor 2011). Almost 70% of nighttime locations were recorded at the forest edges (i.e. within 100m of forest patches); only 10% were recorded within the forest interior and around 12% were using habitat along wooded creeklines (Taylor *et al.* 2002). The preference for forest edges may be due to prey densities and/or the Barking Owl's need for large hollows for nesting; in the studied landscape, it is possible trees of a suitable size were more abundant or available closer to the forest edge (see Taylor *et al.* 2002 for further analysis).

The Victorian Biodiversity Atlas shows records (between 2000 and 2020) of Barking Owl close to the Murray River, especially west of Wodonga (SWIFFT 2024b). In 2006, a short eight-night survey was undertaken in suitable habitat along the Murray River at Walpolla Island in Victoria's north-west (SWIFFT 2024b). The survey was focused in the vicinity of where Barking Owls were recorded in 1995, but no Barking Owls were recorded (SWIFFT 2024b). A comprehensive study by McGregor (2011) of Barking Owls along the Murrumbidgee River and Murray River (between the Goulburn River west of Moira State Forest and Cottadidda State Forest east of Cobram) reported only one pair of Barking Owls, compared to eight pairs in a previous study. Interestingly, the only pair of Barking Owls was recorded at Narrandera Town Common, where there was a watered wetland (McGregor 2011). None of the forests surveyed had a filled wetland and none supported Barking Owls (McGregor 2011); indeed, few of the wetlands in south-west NSW State Forests had received flood water during the four years prior to McGregor (2011). In this way, the VMFRP may help restore Barking Owl populations along the Murray floodplains by returning water to these wetland systems. The research demonstrates a need to improve understanding of this species' abundance, distribution and ecological requirements (SWIFFT 2024b). A reduction in prey species (e.g. through degraded habitat or poisoning by rodenticides) may be more of a limiting factor in the distribution of Barking Owls than insufficient suitable hollows. Restoring the ecological dynamics of the floodplains through more regular watering may boost numbers of Barking Owls along the Murray.

15.3.1 Considerations for hollow replacement strategy

The literature provides no evidence that nest boxes or hollow alternatives have been used for Barking Owls. However, Clemann and Loyn (2003) identify trialling the use of Barking Owl nest boxes as temporary substitutes for tree hollows at selected sites, particularly in Box-Ironbark forests and on private land. They state trials should commence in areas currently occupied by owls to determine their efficacy with this species (Clemann and Loyn 2003). Given the lack of recent records of Barking Owls where hollow-bearing trees are proposed for removal as part of the VMFRP, it seems unlikely a trial for such nest boxes (or other hollow alternatives, untested for Barking Owl) should take place in this area. Indeed, funding that might be spent on untested nest boxes for Barking Owl may be more effectively directed to undertaking research into what is limiting the use of Murray River ecosystems by Barking Owl: food or hollows/habitat. The introduction of untested hollow alternatives into a landscape where a shortage of prey is the limiting factor is unlikely to lead to increased populations of Barking Owl. Further research is required before any effective hollow replacement strategy could be developed for Barking Owl.

The literature indicates the ER Central VMFRP sites are unlikely to support Barking Owls; at best, the riparian and floodplain ecosystems may be marginal habitat. Whilst the proposed removal of hollow-bearing trees in the VMFRP study sites may not directly impact Barking Owls through the removal of nest or roost trees, it is likely the hollows being removed support small mammals such as Sugar Gliders and possums. Thus, the removal of hollow-bearing trees may reduce the abundance of Barking Owl's prey, though the number of hollows being removed from the VMFRP study area may be relatively few compared to the number of hollows available for prey species in that landscape, especially for a species such as the Barking Owl which has a large territory. It is likely there will remain in the VMFRP study area, a number of large trees with suitable hollows to support Barking Owls. Critically, the return of water to the floodplains and wetlands – and the consequent improvement to the ecological productivity of these landscapes, including to prey populations - may well remove a barrier to Barking Owls returning to this area.

15.4 Major Mitchell's Cockatoo Lophochroa leadbeateri leadbeateri

	Conservation significance		Likelihood of occurrence by site	
Taxon	**EPBC***	**FFG^**	**Nyah**	**Vinifera**
Major Mitchell's Cockatoo	Endangered	Critically Endangered	Present	Present

* EPBC: Commonwealth *Environment Protection and Biodiversity Conservation Act 1999.*

^ FFG: Victoria *Flora and Fauna Guarantee Act 1988.*

The Major Mitchell's Cockatoo (or Pink Cockatoo) *Lophochroa leadbeateri leadbeateri* is an iconic bird species endemic to Australia. It is a hardy species that occurs in low densities throughout arid and semi-arid regions across Australia (Ewart *et al.* 2021). Despite the species being listed as Endangered under the Federal EPBC Act, there is no adopted or drafted Recovery Plan for this species though a decision to develop a Recovery Plan was made on 23/03/2023 (DCCEEW 2024c).

The eastern subspecies occurs in the Murray-Darling, Eyre and Bulloo River basins, from Isisford and Roma in the north, through western NSW to north-west Victoria and west, to eastern South Australia (DCCEEW 2023). Environmental assessments undertaken for ER Central VMFRP sites determined the Major Mitchell's Cockatoo is present at some sites.

Eastern Major Mitchell's Cockatoos generally prefer to nest in hollows with the following dimensions (from Hurley and Harris 2014 in DCCEEW 2023):

- average hollow entrance diameter of 13.3 x 27.7 cm (range 8-30 x 9-80 cm, horizontal x vertical diameter);
- average hollow depth of 53.9 cm (range 19 – 180 cm);
- average nest chamber floor diameter of 18 cm (range 9-34 cm);
- average nest tree diameter at breast height of 72.5 cm (range 34-149 cm).

Nesting trees usually are old; in north-western Victoria, Major Mitchell's Cockatoos often nest in hollows in Cypress Pines that are over 80 years old, preferring trees around 130-140 years old (DCCEEW 2023). Their preferred habitat is always 'within reach' of surface water (DCCEEW 2023), indicating the VMFRP may benefit this species by restoring more natural inundation patterns to the Murray floodplain, thus extending potential habitat (where any exists within the VMFRP area) within easy reach of surface water.

One of the largest known breeding populations of Major Mitchell's Cockatoo in Victoria is at Pine Plains, Wyperfeld National Park, in a Slender Cypress Pine *Callitris gracilis murrayensis* woodland (Hurley and Stark 2015). This location has been the site of long-term monitoring of Major Mitchell's Cockatoo (see Hurley and Stark 2015). Records indicated a continual decline in the breeding population at Pine Plains over the 15 years prior to Hurley and Stark (2015); fires in February 2014 removed 93% (77 of 83) known cavity-bearing trees within the fire-affected area (Hurley and Stark 2015), further exacerbating the decline of Major Mitchell's Cockatoo at this site.

15.4.1 Considerations for hollow replacement strategy

In what seems to be the only published research on hollow alternatives for Major Mitchell's Cockatoo, Simulated Natural Cavities (SNCs) were constructed within suitable *Callitris* trees to replace cavities lost due to fire, storms and decay and were expected to require less maintenance and offer higher thermal stability than would nest boxes (see Hurley and Stark 2015, p. 6 for details of cavity dimensions). The study (data collected during one season) considered whether (24) constructed cavities actually simulated (71) natural cavities and the efficacy of the SNCs as a hollow alternative for nest sites (Hurley and Stark 2015). The study found Major Mitchell's Cockatoo occupied 17% of the SNCs and 42% of the natural cavities. Despite this seemingly large difference, Hurley and Stark (2015) reported the data set for occupancy probability was not statistically significant. Breeding was successful (defined as one or more nestlings fledging) for ~50% the occupied SNCs. One of the four occupying Major Mitchell's Cockatoo pairs successfully bred in a SNC compared to 71% of pairs in the natural cavities; again, a small dataset meant it could not be determined whether this represented a genuine difference (Hurley and Stark 2015). They determined competition for hollows was not a primary factor impacting nest occupancy at the study site; rather, nest preferences may have been driven by biological factors/preferences of each species or simply by population dynamics (e.g. population size, number of breeding pairs). This finding underscores the importance of determining what is limiting a population – if the limiting factor is not a paucity of suitable hollows (for roosting, breeding, etc.), installing (largely untested) nest boxes for target species may not be an efficient use of limited resources (e.g. money, time, personnel) and may, in fact, have unintended ecological consequences. Furthermore, the implementation of an ill-devised hollow replacement strategy may mean the actual reason for a species' decline (e.g. a decline in food source, or dysfunctional secondary or tertiary relationships) is overlooked. In some cases, the funds that might be allocated to providing untested hollow alternatives in an unfounded scenario, may be better spent conducting meaningful research into a species' habitat requirements, ecology and/or biology so more is known about that species, driving better conservation outcomes and better-informed planning decisions.

Studies from nest box programs for other threatened cockatoo species such as the South Australian Glossy Black-cockatoo (*Calyptorhynchus lathami halmaturinus*), Carnaby's Cockatoo (*Zanda latirostris*), Baudin's Cockatoo (*Zanda baudinii*) and Forest Red-tailed Black-cockatoo (*Calyptorhynchus banksii naso*), may also assist in the development of a nest box program for the eastern Major Mitchell's Cockatoo (DCCEEW 2023). Hollow-bearing trees proposed for removal that meet the habitat requirements, including hollow dimensions consistent with hollow preferences of Major Mitchell's Cockatoo, should be identified and prioritised for retention if they might possibly support this species.

An emerging alternative to nest boxes is mechanically created artificial hollows. The simulated cavity approach studied by Hurley and Stark (2015) was innovative and not studied to the extent that nest boxes and other augmentations have been. They argue simulated cavities are a logical solution for secondary cavity nesting birds, as they represent a closer analogue to natural cavities. The literature on hollow alternatives demonstrates the critical need for an analogue of a natural hollow.

Again, it is important for hollow replacement programs to be tailored to the habitat requirements of the targeted threatened species as studies have shown that simply setting up nest boxes or other hollow replacements can attract competitor species rather than intended threatened species (DCCEEW 2023). In general, hollow replacements should only be considered where there is evidence that a shortage of natural hollows exists or is suspected (DCCEEW 2023).

15.5 Carpet Python Morelia spilota metcalfei

	Conservation significance		Likelihood of occurrence by site	
Taxon	**EPBC***	**FFG^**	**Nyah**	**Vinifera**
Carpet Python		Endangered	Possible	Possible

* EPBC: Commonwealth *Environment Protection and Biodiversity Conservation Act 1999.*

^ FFG: Victoria *Flora and Fauna Guarantee Act 1988.*

The Inland Carpet Python *Morelia spilota metcalfei* is a slow-moving, nocturnal snake. It belongs to the python family so is non-venomous, overcoming prey by constricting it in coils of its body (Allen *et al.* 2003).

In Victoria, the Inland Carpet Python inhabits two very different environments in the north of the State; River Red Gum forests and associated Black Box woodlands along the major watercourses (including sections of the Murray River); and rocky hills, often within woodlands of Blakely's Red Gum (Allen *et al.* 2003). There also are some records from other vegetation types, such as mallee shrublands, *Callitris* woodlands and freshwater swamps. Environmental Assessments for ER Central sites of VMFRP indicate this species is likely present in at least one site within the study area.

Hollow-bearing trees and logs, or large rock outcrops, plus thick litter or shrub cover, are essential to the existence of Inland Carpet Pythons. Not only do these features support the Inland Carpet Python as shelter sites, to avoid predators, to ambush prey, and to assist in thermoregulation, they also provide vital habitat for prey (Allen *et al.* 2003). Inland Carpet Pythons also may use rabbit burrows as shelter, with rabbits being a major food source (Allen *et al.* 2003). In some areas, they make use of houses and other structures, where introduced rodents form part of the diet (Allen *et al.* 2003).

Activities which remove large hollow-bearing trees, logs, shrubs, coarse woody debris, and litter may threaten the survival of the Inland Carpet Python (Allen *et al.* 2003). Adults prey on small to medium-sized mammals, as well as birds, particularly those roosting in tree hollows (Allen *et al.* 2003). The radical alterations to the abundance and distribution of mammals which have occurred since European settlement in northern Victoria would likely have profoundly affected the feeding habits of adult Inland Carpet Pythons, the resultant changes in prey availability potentially limiting both the frequency of breeding and the number of young produced (Allen *et al.* 2003). The number of young surviving to reproductive age may also be altered (Allen *et al.* 2003). In recent decades, with the extinction of many species of small to medium-sized mammals, the Inland Carpet Python has switched prey with most of the adult diet being obtained from the introduced (ground-dwelling) European Rabbit *Oryctolagus cuniculus.* Concerted efforts to control rabbit populations (e.g. through poisoning, warren fumigating and ripping) have reduced rabbit numbers and potentially reduced native prey species through off-target poisoning (and so, reduced food availability for the Inland Carpet Python), and destroyed Inland Carpet Python habitat via warren ripping (Allen *et al.* 2003). Changed flooding regimes are reported to impact Inland Carpet Python, as unnatural flood patterns impact the health of floodplain ecosystems including hollow-bearing trees. In this way, the VMFRP may support the long-term health of Inland Carpet Pythons by supporting the survival of hollow-bearing trees. In addition, the more frequent inundation of the floodplain will increase the ecological productivity of the area, increasing abundance of prey for the Inland Carpet Python. Indeed, the Action Statement for the Inland Carpet Python sets an intended management action:

'In riverine areas of Inland Carpet Python habitat, ensure that flooding frequency is appropriate to maintain those vegetation elements which comprise important elements pythons. When determining areas in which to promote flooding for ecological purposes, the presence of Inland Carpet Python habitat should be an important consideration' (Allen *et al.* 2023).

A range of other threats face the Inland Carpet Python (see Allen *et al.* 2003).

The literature does not detail the extent to which Inland Carpet Pythons directly rely on hollows in hollow-bearing trees (e.g. for shelter or protection from predation), as opposed to relying on prey that use hollows in hollow-bearing trees. It is unclear whether the species is an obligate hollow-user (i.e. hollow-dependent) or an opportunistic hollow-user, though this is difficult to determine (Gibbons and Lindenmayer 2002), so the implications of proposed removal of hollow-bearing trees on Inland Diamond Python also are difficult to determine. Gibbons and Lindenmayer (2002) confirm Carpet/Diamond Pythons used hollows, but it is not stated whether this is use of hollows as habitat or for predating hollow occupants, as has been observed in the closely related Carpet Python *Morelia spilota imbricata* (Saunders *et al.* 2020b). A study by Webb and Shine (1997) on the threatened Broad-headed Snake *Hoplocephalus bungaroides* revealed the snake showed strong association with rocks and crevices of rocky outcrops during Spring. However, the study showed >80% of tracked snakes in the study moved away from rocky outcrops during the summer, moving to open woodland where they spent long periods (up to 48 days) sequestered inside tree hollows (Webb and Shine 1997). The snakes selected for dead rather than live trees, large rather than small trees, and trees that supported many hollows, indicating they may select for appropriate thermoregulatory conditions and/or proximity to an abundance of hollow-using prey species (Webb and Smith 1997). It is unknown whether the Inland Carpet Python exhibits similar habitat requirements and hollow use. Indeed, a study by Heard *et al.* (2004) reported nine Inland Carpet Pythons in north-eastern Victoria (within the Murray-Darling Basin) extensively used rocky outcrops and primarily predated rabbits during the summer months, indicating a low reliance on tree hollows or predation of hollow-utilising fauna. Another study found introduced canids (that is, foxes and/or wild dogs) are predators of the Inland Carpet Python, with 23% of tracked pythons killed by predators during the study (Heard *et al.* 2006). Being non-venomous and slow-moving means the python is more susceptible to fatal attacks by foxes and dogs (Heard *et al.* 2006), which is exacerbated by pythons having switched to predating ground-dwelling rabbits over arboreal prey. Further research is required into the use of hollows by Inland Carpet Pythons. Restoration of the Murray floodplain ecosystems through more natural inundation patterns would likely increase the longevity of hollow-bearing trees across the Python's range, as well as increase the breeding and survival rates of the Inland Carpet Python's arboreal prey.

15.5.1 Considerations for hollow replacement strategy

Prior to considering installation of hollow alternatives as part of a hollow replacement strategy for the Inland Carpet Python, two questions should be considered: 1) do Inland Carpet Pythons face a shortage of hollows for their primary use and would this shortage be exacerbated by the removal of those hollow-bearing trees proposed for removal for the VMFRP (or is there an abundance of suitable hollows, regardless)? and 2) do Inland Carpet Pythons **hollow-dependent** *prey species* face a shortage of hollows and would this shortage be exacerbated by the removal of those hollow-bearing trees proposed for removal for the VMFRP (or is there an abundance of suitable hollows, regardless)? The second question deliberately stipulates **hollow-dependent** prey species, as there are clear predation implications for Inland Carpet Pythons relying on ground-dwelling species such as rabbits – ideally, the VMFRP project would seek to create conditions that enable arboreal vertebrate species to flourish so populations of Inland Carpet Pythons also could flourish. Of significant concern is the threat posed to Inland Carpet Pythons by introduced canids (Heard *et al*. 2006) – rather than funds be expended on a (potentially) high-maintenance: low-ecological return hollow replacement strategy that intends to (but may not) support Inland Carpet Python recovery, funds could instead be directed to conducting further research into the biology and ecology (including habitat requirements) of the Inland Carpet Python and reducing predation by introduced canids. Likely, a shortage of hollows across the VMFRP sites is not a limiting factor for populations of Inland Carpet Pythons, though may be a limiting factor for their arboreal prey species. Given Inland Carpet Pythons are known to frequently use crevices in rocks, hollows in logs, gaps in leaf litter, and has been recorded using roof cavities (Heard *et al.* 2006), the literature indicates the Inland Carpet Python has very broad preferences for cavities/hollows. Providing hollow replacements without understanding the broader ecological and biological context for why hollow replacements are being installed may lead to unintended negative impacts on ecosystems and threatened species.

16 Conclusion

The current review has considered a range of aspects associated with hollow alternatives and factors that might affect the effectiveness of hollow replacement strategies. The volume of literature relating to hollow alternatives (particularly nest boxes) is significant; whilst not all studies could be considered directly in this review, it is hoped the studies included here provide a representative and accurate example of the broader landscape of research findings and perspectives. Also, it is important to note this review provides a synthesis of what is known *at this point in time*; new studies relating to hollow alternatives regularly are being published, revealing new findings, approaches, applications and perspectives, so it is important for stakeholders to undertake their own reviews of the contemporary (and broader) relevant literature. The content of this review is intended to support (not dictate) stakeholders' consideration of hollow alternatives, and whether hollow replacement strategies are likely to produce the most effective outcomes for hollow-using and hollow-dependent fauna, to mitigate the loss of hollow-bearing trees.

This review found current hollow alternatives do not fully mitigate or offset the loss of naturally-occurring hollows. A range of hollow alternatives and their associated issues have been considered, including implications for ecosystems and wildlife, to help guide stakeholders to develop effective approaches to mitigating the loss of hollow-bearing trees. In particular, finding hollow alternatives that provide safe thermal conditions will be critical for the diversity of hollow-using fauna endemic to the extensively cleared and degraded woodland ecosystems across south-eastern Australia (Gibbons and Lindenmayer 2002; Strain *et al.* 2021). This area of Australia is particularly vulnerable to changes to climatic conditions that might exacerbate the existing Mediterranean climate (Rowland *et al*. 2017). Clearly, retaining hollow-bearing trees and avoiding removal of trees bearing hollows that show signs of occupancy (where possible to determine) should be a priority.

Of the hollow alternatives considered in this review, the method used in Ellis *et al.* (2022) holds the most potential and is highly worthy of additional investigations, including as part of the VMFRP hollow mitigation approach. The study provides evidence that providing entrances to otherwise inaccessible cavities in live trees has potential to accelerate development and occupancy of habitat for hollow-utilising fauna (Ellis *et al.* 2022). Their findings demonstrate this method provides the benefits of naturally-occurring hollows, including a similarly safe microclimate. The method used by Ellis *et al.* (2022) may well prove to be a very effective tool to mitigate the loss of hollow-bearing trees for some hollow-using species. Like every hollow alternative considered in this review, the approach trialled in Ellis *et al.* (2022) requires regular monitoring and maintenance, especially with regard to entrance hole occlusion by wound wood. Further research into the use of this method could lead to significant advances in provision of effective habitat for hollow-dependent wildlife and thus, effective mitigation for the loss of hollow-bearing trees. Working with indigenous knowledge holders may well reveal practices traditionally used to care for hollows, practices that limited and managed wound wood growth, ensuring hollows were able to support populations of hollow-using wildlife, at least some of which would be a food source for indigenous people. Such practices could be critical for refining the method of creating the entrances to existing internal cavities to ensure the entrances remain open and cavities remain accessible.

Where hollow-bearing trees cannot be retained, the increasingly standard 'mitigation' approach is to 'replace' the hollows using an arbitrary ratio, usually 1:1 replacement. However, the literature reports many reasons why this approach is flawed (most notably in a landscape where availability of hollows is *not* a limiting factor in the environment) and may lead to perverse ecological outcomes, including unacceptable impacts on wildlife. In particular, this approach assumes every hollow in the landscape is suitable for at least one hollow-using taxa, and that every available hollow is used. In fact, it is likely only a fraction of available hollows are suitable for hollow-using animals (Westerhuis *et al.* 2019), with the maximum estimate of hollow usage being around 43% of available hollows (Gibbons *et al.* 2000). This means not every hollow removed will be suitable for hollow-using fauna and fewer than half of those that are suitable are likely occupied, or show evidence of current or historic use. This is consistent with the finding in Lindenmayer *et al.* (2009) that nest boxes erected in older forests (where hollows occurred) were occupied by almost no animals (including the endangered target species, Leadbeater's Possum), suggesting provision of hollow alternatives may be redundant in landscapes that support old hollow-bearing trees. This has important implications for the VMFRP, given the landscape in which sits the construction footprint, supports an abundance of hollow-bearing trees; thus, providing hollow replacements may introduce to these ecosystems more harm than benefit. However, this review demonstrates that hollow replacement strategies may play an important role in landscapes elsewhere where there is a paucity of naturally-occurring hollows.

Undoubtedly, there is an imperative for a collaborative approach. Griffiths *et al*. (2023) and van der Ree (2019) rightly assert the design, installation and monitoring of hollow alternatives should be a collaborative process involving stakeholders, including land managers, ecologists and, depending on the hollow alternative, arborists. It is vital this collaboration includes the collection, collation and publication of empirical data that can be used to inform an adaptive management framework for strategic hollow replacement (Griffiths *et al*. 2023; van der Ree 2019). This will help ensure the loss of naturally-occurring hollows can be mitigated in the most effective way, so scarce conservation resources (funding, time, people) are spent as effectively as possible, for *actual and realised* ecological returns.

The review of known habitat requirements of five threatened species present (or potentially present) within the VMFRP ER Central sites study area highlights the critical importance of basing a hollow replacement strategy on the habitat requirements, biology and ecology of a species rather than on an arbitrary replacement ratio. This review raises awareness of the paucity of literature that exists pertaining to the *ecology* and *biology* of these species; regrettably, this also is the case for many other rare, threatened and/or cryptic species. A lack of understanding of the ways in which hollows are utilised by particular species (e.g. Inland Carpet Python) demonstrates how important are empirical data for developing effective considered mitigation strategies. In the case of most threatened species considered here (that do (or may) occur within the VMFRP study area), there is no evidence in the literature that hollow alternatives have been used in conservation efforts for these species – coordinated research *must* be prioritised to determine whether hollow alternatives would provide any benefit for these species, particularly in a hollow-rich environment. For other species considered here, the literature shows there is little-no evidence of hollow alternatives having been used successfully, particularly in landscapes where there is a high rate of hollows. Indeed, for at least some species, the availability of hollows may not be a limiting factor influencing their abundance or distribution. Instead, primary, secondary or tertiary ecological relationships (e.g. predator and prey dynamics) may exert a greater influence over those threatened species' abundance and distribution. Providing hollow alternatives in an *ad hoc* way in such cases may not provide any benefit to these threatened species and may, in fact, introduce novel complications which further imperil the species. For example, the current review of Major Mitchell's Cockatoo's habitat requirements underscores the importance of determining what is limiting a population – if the limiting factor is not a paucity of suitable hollows (for roosting, breeding, etc.), installing (largely untested) nest boxes for target species may not be an efficient use of limited resources (e.g. money, time, personnel) and may, in fact, have unintended ecological consequences. Furthermore, the implementation of an ill-devised hollow replacement strategy may mean the actual reason for a species' decline (e.g. a decline in food source, or dysfunctional primary, secondary or tertiary relationships) is overlooked.

In some cases, the funds that might be allocated to providing untested hollow alternatives, against a backdrop of lack of understanding of ecosystem dynamics and of target species' ecology and biology, may be better spent conducting meaningful research into a species' habitat requirements, ecology and/or biology so more is known about target species. This research would drive better future conservation outcomes and more informed planning decisions. The review of known habitat requirements of five threatened species present (or potentially present) within the VMFRP ER Central sites study area highlights the complexity of ecosystems and the ways in which a well-meaning but poorly-considered hollow replacement strategy could negatively impact dynamics within these complex ecological systems.

If not a hollow replacement strategy, then what? How can the loss of hollow-bearing trees from a landscape, including within the VMFRP study area, be mitigated for those species listed under Victoria's FFG Act and as *per* the Minister's Assessment in a way that provides effective, positive outcomes for ecosystems and hollow-using fauna, without the potential for unintended, negative consequences? Land managers, governments, planners and other stakeholders must consider whether the implications for ecosystems, wildlife and funding costs of failed hollow replacement strategies create more harm than good. This review provides evidence that, across the VMFRP study area, significant funds would be required to develop, implement, monitor and maintain a 1:1 hollow replacement strategy which the literature shows may well fail to provide effective habitat for hollow-using species, let alone for those known threatened species.

Perhaps a more creative, pragmatic response to mitigating the loss of hollow-bearing trees resulting from VMFRP could be used in this case. Vegetation removal permitted under Clause 52.17 Native Vegetation in the State Planning Provisions of Victoria's *Planning and Environment Act 1987* is ordinarily mitigated through a DEECA-managed offsets programme, which typically requires permit applicants to demonstrate they have secured (sourced and paid for) off-site offsets via a third-party offset delivery arrangement. The offsetting system enables a transaction whereby proponent, whose core business is with matters of the planning application, secure offsets which provide commensurate conservation benefit that will be realised by a contracted third-party. Whilst not without its issues, this offset system is accepted practice for mitigating short-, medium- and long-term conservation impacts resulting from permitted vegetation removal. Perhaps a similar approach could be taken to mitigate – offset – the loss of hollow-bearing trees resulting from VMFRP in the short- to medium-term, between the time of removal of hollow-bearing trees and the longer-term realisation of improved health of hollow-bearing trees across the restored Murray floodplains. For example, the funds required to implement a potentially ineffective and dangerous hollow-replacement strategy could instead be paid into an offset research fund – a hollow-bearing tree offset programme.

This offset programme could finance an agency and/or universities to conduct vital coordinated, scientific third-party research, which develops, studies and publishes findings on, for example: hollow alternatives; threatened species habitat requirements; ecosystem dynamics in relation to the provision of hollow alternatives; ecophysiology and cavity microclimates of hollow alternatives, etc.). Of particular funding merit for such an offset fund would be further research into the viability of creating entrances to access naturally-occurring internal cavities and voids (see Ellis *et al.* 2022). Establishing a hollow-bearing tree offset programme and funding such vital research projects likely would be a much more effective use of funding – with far-reaching benefits for future projects that remove hollow-bearing trees - than another (likely-) failed hollow replacement strategy that may ultimately harm the very ecosystems and species it intended to support.

17 References

ABC (2016a). *Swift parrot egg safe havens a 'bonanza' for endangered species.* Australian Broadcasting Corporation. https://www.abc.net.au/news/2016-10-07/swift-parrot-eggs-found-in-experimental-nesting-boxes/7911138 (accessed 15 Jan 2024).

ABC (2016b). *Swift Parrot: Arborists Hope New Tree Hollows Will Help Save Endangered Bird*. Australian Broadcasting Corporation. http://www.abc.net.au/news/2016-10-13/hope-new-hollows-will-save-endangered-swift-parrot/7929536 (accessed 15 January 2024).

Adkins, M.F. (2006). A burning issue: using fire to accelerate tree hollow formation in *Eucalyptus* species. *Australian Forestry.* **69**(2), 107-113.

Aitken K.E.H. and Martin K. (2012). Experimental test of nest-site limitation in mature mixed forests of central British Columbia, Canada. *Journal of Wildlife Management.* **76**, 557– 565.

Allen, G.G., Barrow, G., Hurley, V. and Robertson, P. (2003). *Flora and Fauna Guarantee Act 1988 Action Statement No. 175: Inland Carpet Python* Morelia spilota metcalfei. Department of Sustainability and Environment, Melbourne.

Ambrose, G.J. (1982). *An Ecological and Behavioural Study of Vertebrates Using Hollows in Eucalypt Branches.* Ph.D. Thesis. La Trobe University, Melbourne.

Ardia, D.R., Pérez, J.H. and Clotfelter, E.D. (2006). Nest box orientation affects internal temperature and nest site selection by tree swallows. *Journal of Field Ornithology.* **77**(3), 339–344.

Baker-Gabb, D. and Hurley, V.G. (2011). *National Recovery Plan for the Regent Parrot (eastern subspecies)* Polytelis anthopeplus monarchoides. Department of Sustainability and Environment, Melbourne.

Bender, R. and Irvine, R. (2001). Bat boxes at organ pipes national park. *Australian Bat Society Newsletter*. **16**, 19–23.

Bennett, A.F. (2016). Eucalypts, wildlife and nature conservation: from individual trees to landscape patterns. *Proceedings of the Royal Society of Victoria*. **128**(1), 71–86.

Bennett, A., Lumsden, L. and Nicholls, A. (1994). Tree hollows as a resource for wildlife in remnant woodlands: Spatial and temporal patterns across the northern plains of Victoria, Australia. *Pacific Conservation Biology*. **1**(3), 222–235.

Beranek, C. and Hayward, M. (2021). *Review of Reintroductions to Ngambaa Nature Reserve – Report Prepared for the NSW National Parks and Wildlife Service.*

Best, K., Haslem, A., Maisey, A.C., Semmens, K. and Griffiths, S.R. (2022). Occupancy of chainsaw-carved hollows by an Australian arboreal mammal is influenced by cavity attributes and surrounding habitat. *Forest Ecology and Management*. **503**, 119747.

Beyer, G.L. and Goldingay, R.L. (2006). The value of nest boxes in the research and management of Australian hollow-using arboreal marsupials. *Wildlife Research*. **33**, 161–174.

Beyer, G., Goldingay, R. and Sharpe, D. (2008). The characteristics of squirrel glider (*Petaurus norfolcensis*) den trees in subtropical Australia. *Australian Journal of Zoology*. **56**, 13–21.

Bruns, H. (1960). The economic importance of birds in forests. *Bird Study* **7**, 193–208.

Burbidge, A. H. (1985). *The Regent Parrot. A report on the breeding distribution and habitat requirements along the Murray River in south-eastern Australia.* Report Series No. 4. Australian National Parks and Wildlife Service, Canberra, ACT.

Burgess, M. (2014). Restoring abandoned coppice for birds: Few effects of conservation management on occupancy, fecundity and productivity of hole nesting birds. *Forest Ecology and Management.* **330**, 205–217.

Callan, M.N., Johnson, A. and Watson, D.M. (2023a). Influence of nest box design on internal microclimate: comparisons of plastic prototypes. *Austral Ecology.* **48**, 374–387.

Callan, M.N., Krix, D., McLean, C.M., Murray, B.R., and Webb, J.K. (2023b). Thermal profiles of chainsaw hollows and natural hollows during extreme heat events. *Biology*. **12**, 361.

Carey, A. (2002). Response of Northern Flying Squirrels to Supplementary Dens. *Wildlife Society Bulletin*. **30**, 547–556.

Carey, A.B. and Gill, J.D. (1983). Direct habitat improvements - some recent advances. In: *Snag Habitat Management: Proceedings of the Symposium.* USDA Forest Service General Technical Report RM-99 Fort Collins, Colorado. pp. 80-87.

Carey, A.B. and Sanderson, H.R. (1981). Routing to accelerate tree-cavity formation. *Wildlife Society Bulletin*. **9**, 14–21.

Carrie, N.R., Moore, K.R., Stephens, S.A. and Keith, E.L. (1998). Influence of cavity availability on red-cockaded woodpecker group size. *Wilson Bulletin*. **110**, 93–99.

Catry, I., Franco, A.M.A. and Sutherland, W.J. (2011). Adapting conservation efforts to face climate change: modifying nest-site provisioning for lesser kestrels. *Biological Conservation*. **144**, 1111–1119.

Central Coast Council (2016). *Guideline for the Relocation of Large Tree Hollows*. July 2016. Central Coast Council, Wyong.

Churchill, S.K. (2008). *Australian Bats*. Allen and Unwin, Crows Nest, Australia.

Clemann, N. and Loyn, R. (2003). *Flora and Fauna Guarantee Act 1988 Action Statement No. 116 – Barking Owl* Ninox connivens. Department of Sustainability and Environment, Victoria.

Cockle, K.L., Martin, K. and Wesołowski, T. (2011). Woodpeckers, decay, and the future of cavity-nesting vertebrate communities worldwide. *Frontiers in Ecology and the Environment*. **9**(7), 377–382.

Coelho, J.R. and Sullivan, J.B. (1994). Colonization of wildlife nest boxes by honeybee swarms. *American Bee Journal*. **134**, 697–699.

Copeyon, C.K. (1990). A technique for constructing cavities for the red-cockaded wood- pecker. *Wildlife Society Bulletin*. **18**, 303–311.

Corrigan R.M., Scrimgeour G.J. and Paszkowski C. (2011). Nest boxes facilitate local-scale conservation of common goldeneye (*Bucephala clangula*) and bufflehead (*Bucephala albeola*) in Alberta, Canada. *Avian Conservation and Ecology*. **6**, 1.

Cowan, M.A., Callan, M.N., Watson, M.J., Watson, D.M., Doherty, T.S., Michael, D.R. *et al*. (2021). Artificial refuges for wildlife conservation: what is the state of the science? *Biological Reviews*. **96**, 2735–2754.

Crawford, R.D. and O'Keefe, J.M. (2021). Avoiding a conservation pitfall: considering the risks of unsuitably hot bat boxes. *Conservation Science and Practice* 3:e412.

Deeming, D. (2011) Importance of nest type on the regulation of humidity in bird nests. *Avian Biology Research.* **4**(1), 23–31.

DCCEEW (2023). *Conservation Advice for* Lophochroa leadbeateri leadbeateri *(eastern Major Mitchell's Cockatoo).* DCCEEW, Canberra.

DCCEEW (Department of Climate Change, Energy, the Environment and Water). (2024a). *Species Profile and Threats Database SPRAT Profile:* Polytelis anthopeplus monarchoides — *Regent Parrot (eastern).* DCCEEW, Canberra. Accessed via https://www.environment.gov.au (31/01/2024).

DCCEEW (2024b). *Species Profile and Threats Database SPRAT Profile:* Nyctophilus corbeni — *Corben's Long-eared Bat, South-eastern Long-eared Bat.* DCCEEW, Canberra. Accessed via https://www.environment.gov.au (31/01/2024).

DCCEEW (2024c). *Species Profile and Threats Database SPRAT Profile:* Lophochroa leadbeateri leadbeateri — *Major Mitchell's Cockatoo (eastern), Eastern Major Mitchell's Cockatoo, Pink Cockatoo (eastern).* DCCEEW, Canberra. Accessed via https://www.environment.gov.au (31/01/2024).

Durant, R., Luck, G.W. and Matthews, A. (2009). Nest-box use by arboreal mammals in a peri-urban landscape. *Wildlife Research*. **36**, 565–573.

Edworthy, A.B. (2016). Competition and aggression for nest cavities between Striated Pardalotes and endangered Forty-spotted Pardalotes. *Condor*. **118**, 1–11.

Ellis, M.V. (2018). The presence of heartwood decay, a precursor to hollow formation, within *Eucalyptus camaldulensis* in relation to stem size. *Ecological Management and Restoration* **19**, 257–259.

Ellis, M.V., Taylor, J.E. and Rhind, S.G. (2022). Creating entrances to tree cavities attracts hollow-dependent fauna: proof of concept. *Restoration Ecology*. **30**(8), e13713.

Elston, J.J., Carney, J., Quinones, G., Sky, C., Plasse, C. and Bettinger, T. (2007). Use of novel nest boxes by carmine bee-eaters (*Merops nubicus*) in captivity*. Zoo Biology*. **26**(1), 27–39.

Emison, W.B. (1996). Use of supplementary nest hollows by an endangered subspecies of red-tailed black cockatoo. *Victorian Naturalist.* **113**, 262–263.

Ewart, K.M., Johnson, R. N., Joseph, L., Ogden, R., Frankham, G.J. and Lo, N. (2021). Phylogeography of the iconic Australian pink cockatoo, *Lophochroa leadbeateri. Biological Journal of the Linnean Society.* **132**, 704–723.

Fiehler, C.M., Tietje, W.D. and Fields, W.R. (2006). Nesting success of Western bluebirds (*Sialia mexicana*) using nest boxes in vineyard and oak-savannah habitats of California. *Wilson Journal of Ornithology*. **118**, 552–558.

Flaquer, C., Puig-Montserrat, X., López-Baucells, A., Torre, I., Freixas, L., Mas, M., Porres, X. and Arrizabalaga, A. (2014). Could overheating turn bat boxes into death traps? *Barbastella - Journal of Bat Research.* **7**(1), 39 – 46.

Geiser, F. and Ruf, T. (1995). Hibernation versus daily torpor in mammals and birds: physiological variables and classification of torpor patterns. *Physiological Zoology*. **68**, 935–966.

Gibbons, P. and Lindenmayer, D.B. (2002). *Tree Hollows and Wildlife Conservation in Australia*. CSIRO Publishing, Melbourne.

Gibbons, D., Lindenmayer, D.B., Barry, S.C. and Tanton, M.T. (2000). Hollow formation in eucalypts from temperate forests in south-eastern Australia. *Pacific Conservation Biology*. **6**, 218–228.

Gibbons, P., Lindenmayer, D.B., Barry, S.C. and Tanton, M.T. (2002). Hollow selection by vertebrate fauna in forests of south-eastern Australia and implications for forest management. *Biological Conservation*. **103**, 1–12.

Godinho, L.N., Lumsden, L.F., Coulson, G., Griffiths, S.R. (2019). Flexible roost selection by Gould's wattled bats (Chalinolobus gouldii) using bat boxes in an urban landscape. *Australian Journal of Zoology.* **67**, 269–280.

Goldingay, R.L. (2009). Characteristics of tree hollows used by Australian birds and bats. *Australian Wildlife Research*. **36**(5), 394–409.

Goldingay, R.L. (2011). Characteristics of tree hollows used by Australian arboreal and scansorial mammals. *Australian Journal of Zoology*. **59**, 277–294.

Goldingay, R.L. (2015). Temperature variation in nest boxes in eastern Australia. *Australian Mammology.* **37**(2), 225-233.

Goldingay, R.L. (2017). Does nest box use reduce the fitness of a tree-cavity dependent mammal? *Ecological Research*. **32**, 495–502.

Goldingay, R.L., Rueegger, N.N., Grimson, M.J. and Taylor, B.D. (2015). Specific nest box designs can improve habitat restoration for cavity-dependent arboreal mammals. *Restoration Ecology.* **23**, 482–490.

Goldingay, R.L., Rohweder, D. and Taylor, B.D. (2020a). Nest box contentions: are nest boxes used by the species they target? *Ecological Management and Restoration.* **21**, 115–122.

Goldingay, R.L. and Stevens, J.R. (2009). Use of artificial tree hollows by Australian birds and bats. *Wildlife Research*. **36,** 81–97.

Goldingay, R.L. and Thomas, K.J. (2021). Tolerance to high temperature by arboreal mammals using nest boxes in southern Australia. *Journal of Thermal Biology.* **98**, 102899.

Goldingay, R.L., Thomas, K.J. and Shanty, D. (2018). Outcomes of decades long installation of nest boxes for arboreal mammals in southern Australia. *Ecological Management Restoration*. **19**, 204–211.

Goldingay, R.L., Quin, D.C., Talamo, O. and Mentiplay-Smith, J. (2020b). Nest box revealed habitat preferences of arboreal mammals in box-ironbark forest. *Ecological Management and Restoration.* **21**(2), 131 – 142.

Goodenough A.E., Maitland D.P., Hart A.G. and Elliot S.L. (2008). Nestbox orientation: A species-specific influence on occupation and breeding success in woodland passerines. *Bird Study* **55**, 222–232.

Greet, J., Ashman, K.R., Watchorn, D.J., Harley, D.K.P. and Duncun, D. (2020). The vegetation structure and condition of contracting lowland habitat for Leadbeater's possum (*Gymnobelideus leadbeateri*). *Australian Mammalogy*. **43**, 344–353.

Griffiths, S. (2019). Preliminary investigations into the effectiveness of chainsaw-carved hollows. In: *Proceedings of the 20th National Street Tree Symposium*. Ed. Glenn Williams.

Griffiths, S.R., Bender, R., Godinho, L.N., Lentini, P.E., Lumsden, L.F. and Robert, K.A. (2017). Bat boxes are not a silver bullet conservation tool. *Mammal Review*. **47**, 261–265.

Griffiths, S.R., Lentini, P.E., Semmens, K. and Robert, K.A. (2023). "Set and forget" does not work when it comes to fissure roosts carved into live trees for bats. *Restoration Ecology*. **31**(1), e13751.

Griffiths, S. R., Lentini, P. E., Semmens, K., Watson, S. J., Lumsden, L. F., and Robert, K. A. (2018). Chainsaw-carved cavities better mimic the thermal properties of natural tree hollows than nest boxes and log hollows. *Forests*. **9**(5), 235.

Griffiths, S.R., Robert, K.A. and Jones, C.S. (2022). Chainsaw hollows carved into live trees provide well insulated supplementary shelters for wildlife during extreme heat. *Wildlife Research*. **49**, 596–609.

Griffiths, S.R., Rowland, J.A., Briscoe, N.J., Lentini, P.E., Handasyde, K.A., Lumsden, L.F. and Robert, K.A. (2017). Surface reflectance drives nest box temperature profiles and thermal suitability for target wildlife. PloS One 12, e0176951.

Griffiths, S.R., Semmens, K., Watson, S.J. and Jones, C.S. (2020). Installing chainsaw-carved hollows in medium-sized live trees increases rates of visitation by hollow-dependent fauna. *Restoration Ecology*. **28**, 1225–1236.

Hamerstrom, F., Hamerstrom, F.N. and Hart, J. (1973). Nest boxes: an effective management tool for kestrels. *Journal of Wildlife Management*. 400–403.

Hannan, L., Le Roux, D.S., Milner, R.N.C. and Gibbons, P. (2019). Erecting dead trees and utility poles to offset the loss of mature trees. *Biological Conservation.* **236**, 340-346.

Haramis, G.M. and Thompson, D.Q. (1985). Density-production characteristics of box-nesting wood ducks in a northern greentree impoundment. *Journal of Wildlife Management*. **49**, 429–436.

Harley, D.K.P. (2006). A role for nest boxes in the conservation of Leadbeater's Possum (*Gymnobelideus leadbeateri*). *Wildlife Research*. **33**, 385–395.

Harper, M.J., McCarthy, M.A. and van der Ree, R. (2005). The use of nest boxes in urban natural vegetation remnants by vertebrate fauna. *Wildlife Research*. **32**, 509–516.

Harrington, G. (1979). Estimation of above-ground biomass of trees and shrubs in a *Eucalyptus populnea* F. Muell. Woodland by regression of mass on trunk diameter and plant height. *Australian Journal of Botany.* **27**, 135–143.

Heard, G.W., Black, D. and Robertson, P. (2004). Habitat use by the inland carpet python (*Morelia spilota metcalfei*: Pythonidae): Seasonal relationships with habitat structure and prey distribution in a rural landscape. *Austral Ecology.* **29**(4), 446-460.

Heard, G.W., Robertson, P., Black, D., Barrow, G., Johnson, P., Hurley, V. and Allen, G. (2006). Canid predation: a potentially significant threat to relic populations of the Inland Carpet Python *Morelia spilota metcalfei* (Pythonidae) in Victoria. *The Victorian Naturalist.* **123**(2), 68 – 74.

Hingston, A.B. (2024). Buff-tailed bumblebee *Bombus terrestris* as a potential nest site competitor for the endangered forty-spotted pardalote *Pardalotus quadragintus*. *Austral Ecology*. **49**: e13475

Hipkiss T., Gustafsson J., Eklund U. and Hörnfeldt B. (2013). Is the long-term decline of boreal owls in Sweden caused by avoidance of old boxes? *Journal of Raptor Research.* **47**, 15–20.

Honey, R., McLean, C.M., Murray, B.R., Callan, M.N. and Webb, J.K. (2021). Choice of monitoring method can influence estimates of usage of artificial hollows by vertebrate fauna. *Australian Journal of Zoology.* **69**, 18–25.

Horner, G.J., Baker, P.J., Mac Nally, R., Cunningham, S.C., Thomson, J.R. and Hamilton, F. (2010). Forest structure, habitat and carbon benefits from thinning floodplain forests: managing early stand density makes a difference. *Forest Ecology and Management.* **259**, 286–293.

Hurley, V.G. (2009). *Results of nest site monitoring of active Regent Parrot (*Polytelis anthopeplus monarchoides*) nests, in the Outlet Creek system, Wyperfeld NP, during spring 2008.* Department of Sustainability and Environment, Mildura.

Hurley, V.G. and Stark, E.M. (2015). *Characteristics and uptake of simulated natural cavities for Major Mitchell's Cockatoo* (Lophochroa leadbeateri leadbeateri*) in Slender Cypress-pine.* Department of Environment, Land, Water and Planning, Mildura.

Kane, B.C. and Ryan, H. (2003). Examining formulas that assess strength loss due to decay in trees: woundwood toughness improvement in red maple (*Acer rubrum*). *Journal of Arboriculture*. **29**, 209–217.

Kavanagh, R.P., Debus, S.J.S., Rose, A.B. and Turner, R.J. (1995). Diet and habitat of the Barking Owl *Ninox connivens* in New South Wales. *Australian Bird Watcher*. **16**(4), 137-144.

Kikuchi, K., Akasaka, T., Yamaura, Y. and Nakamura, F. (2013). Abundance and use of cavity trees at the tree- and stand-levels in natural and plantation forests in Hokkaido, *Japanese Journal of Forest Research*. **18**(5), 389–397.

Kunz, T.H. and Lumsden, L.F. (2003). Ecology of cavity and foliage roosting bats. In: *Bat Ecology.* Kunz, T.H. and Fenton M.B. (eds). University of Chicago Press, Chicago, Illinois. Pages 3–89.

Lambrechts, M.M., Marrot, P., Fargevieille, A. *et al.* (2016). Nest size is not closely related to breeding success in Blue Tits: A long-term nest-box study in a Mediterranean oak habitat. *Auk*. **133**, 198–204.

Larson, E.R., Eastwood, J.R., Buchanan, K.L., Bennett, A.T.D. and Berg, M.L. (2015). How does nest-box temperature affect nestling growth rate and breeding success in a parrot? *Emu - Austral Ornithology*. **115**(3), 247–255.

Larson, E.R., Eastwood, J.R., Buchanan, K.L., Bennett, A.T.D. and Berg, M.L. (2018). Nest box design for a changing climate: the value of improved insulation. *Ecological Management and Restoration.* **19**(1), 39–48.

Law, B., Gonsalves, L., Brassil, T. and Hill, D. (2018). Does thinning homogenous and dense regrowth benefit bats? Radio-tracking, ultrasonic detection and trapping. *Diversity.* **10**(2), 45.

Law, B., Gonsalves, L, Chidel, M. and Brassil, T. (2016). Subtle use of a disturbance mosaic by the south-eastern long-eared bat (*Nyctophilus corbeni*): an extinction-prone, narrow-space bat. *Wildlife Research.* **43**(2), 153-168.

Lawton, J.A., Holland, G.J., Timewell, C., Bannon, A., Mellick, E. and Bennett, A.F. (2022). Citizen science and community action provide insights on a threatened species: nest box use by the brush-tailed phascogale (*Phascogale tapoatafa*). *Wildlife Research.* **49**(6), 513 – 528.

Le Roux, D.S., Ikin, K., Lindenmayer, D.B., Bistricer, G., Manning, A.D. and Gibbons, P. (2015). Enriching small trees with artificial nest boxes cannot mimic the value of large trees for hollow-nesting birds. *Restoration Ecology*. **24**, 252–258.

Le Roux, D., Ikin K., Lindenmayer, D. B., Bistricer, G., Manning, A. D. and Gibbons, P. (2016) Effects of entrance size, tree size and landscape context on nest box occupancy: considerations for management and biodiversity offsets. *Forest Ecology and Management*. **366**, 135–142.

Le Roux, D.S., Ikin, K., Lindenmayer, D.B., Bistricer, G., Manning, A.D. and Gibbons, P. (2016). Enriching small trees with artificial nest boxes cannot mimic the value of large trees for hollow-nesting birds. *Restoration Ecology.* **24**(2), 252–258.

Lindenmayer, D.B. (1997). Differences in the biology and ecology of arboreal marsupials in southeastern Australian forests and some implications for conservation. *Journal of Mammalogy* **78**, 1117–1127.

Lindenmayer, D.B. (2007). *The variable retention harvest system and its implications for biodiversity in the Mountain Ash Forests of the Central Highlands of Victoria. Occasional Paper No. 2, November 2007.* Fenner School of Environment and Society, The Australian National University, Canberra.

Lindenmayer, D.B., Crane, M., Blanchard, W., Okada, S. and Montague Drake, R. (2015). Do nest boxes in restored woodlands promote the conservation of hollow dependent fauna? *Restoration Ecology*. **24**, 244–251.

Lindenmayer, D.B., Crane, M., Evans, M.C., Maron, M., Gibbons, P., Bekessy, S. and Blanchard, W. (2017). The anatomy of a failed offset. *Biological Conservation*. **210**, 286–292.

Lindenmayer, D.B., Cunningham, R.B., Nix, H.A., Tanton, M.T. and Smith, A.P. (1991a). The conservation of arboreal marsupials in the montane ash forests of the Central Highlands of Victoria, south-east Australia. III. The habitat requirements of Leadbeater's Possum, *Gymnobelideus leadbeateri* McCoy and models of the diversity and abundance of arboreal marsupials. *Biological Conservation*. **56**, 295–315.

Lindenmayer, D.B., Cunningham, R.B., Donnelly, C.F., Nix, H.A. and Tanton, M.T. (1993). The abundance and development of cavities in Eucalyptus trees: a case study in the montane forests of Victoria, south-eastern Australia. *Forest Ecology and Management* **60**, 77–104.

Lindenmayer, D.B., Cunningham, R.B., Tanton, M.T., Nix, H.A. and Smith, A.P. (1991b). Predicting the abundance of hollow-bearing trees in montane ash forests of south-eastern Australia. *Australian Journal of Ecology*. **16**, 91–98.

Lindenmayer, D.B., Laurence, W.F., Franklin, J.F., Likens, G.E., Banks, S.C., Blanchard, W., Gibbons, P., Ikin, K., Blair, D., McBurney, L., Manning, A.D. and Stein, J. A. (2014). New policies for old trees: averting a global crisis in a keystone ecological structure. *Conservation Letters*. **7**, 61–69.

Lindenmayer, D.B., Welsh, A., Donnelly, C., Crane, M., Michael, D., Macgregor, C., McBurney, L., Montague, R. and Gibbons, P. (2009). Are nest boxes a viable alternative source of cavities for hollow-dependent animals? Long-term monitoring of nest box occupancy, pest use and attrition. *Biological Conservation*. **142**(1), 33–42.

Lumsden, L., Powell, C. and Cashmore, M. (2016). Successfully creating new hollows for the critically endangered Leadbeater's possum [Abstract]. In: *62nd Annual Scientific Meeting of the Australian Mammal Society. Australian Mammal Conference.* Alice Springs. pp. 58.

Lumsden, L.F., Alexander, J.S.A., Hill, F.A.R., Krasna, S.P. and Silveira, C.E. (1991). *The Vertebrate Fauna of the Land Conservation Council Melbourne – 2 Study Area. Technical Report Series No. 115.* Arthur Rylah Institute for Environmental Research, Department of Conservation and Environment, Melbourne.

Lumsden, L.F., Griffiths, S.R., Silins, J.E. and Bennett, A.F. (2020). Roosting behaviour and the tree-hollow requirements of bats: insights from the Lesser Long-eared Bat (*Nyctophilus geoffroyi*) and Gould's Wattled Bat (*Chalinolobus gouldii*) in south-eastern Australia. *Australian Journal of Zoology.* **68,** 296–306.

Macak, P.V. (2020). Nest boxes for wildlife in Victoria: an overview of nest box distribution and use. *The Victorian Naturalist*. **137**(1), 4 – 14.

Marsden, S.J. and Pilgrim, J.D. (2003). Factors influencing the abundance of parrots and hornbills in pristine and disturbed forests on New Britain, PNG. *Ibis*. **145**, 45–53.

Maziarz, M., Broughton, R.K. and Wesołowski, T. (2017). Microclimate in tree cavities and nest-boxes: implications for hole-nesting birds. *Forest Ecology and Management*. **389**, 306–313.

McComb, W.C. and Noble, R.E. (1981). Nest-box and natural-cavity use in three mid-south forest habitats. *Journal of Wildlife Management.* **45**, 93–101.

McComb, L.B., Lentini, P.E., Harley, D.K.P., Lumsden, L.F., Eyre, A.C. and Briscoe, N.J. (2022). Climate and behaviour influence thermal suitability of artificial hollows for a critically endangered mammal. *Animal Conservation*. **25**, 401-413.

McDowell, N.G., Allen, C.D., Anderson-Teixeira, K., Aukema, B.H., Bond-Lamberty, B., Chini, L, *et al*. (2020). Pervasive shifts in forest dynamics in a changing world. *Science.* **368**(6494).

McGregor, H. (2011). Large forest owls in the River Red Gum State Forests of south-western New South Wales - an account of their 2008 status. *Australian Zoologist.* **35**(3), 864 – 869.

McKenney, D.W. and Lindenmayer, D.B. (1994). An economic assessment of a nest box strategy for the conservation of an endangered species. *Canadian Journal of Forest Research.* **24**, 2012–2019.

Menkhorst, P.W. (1984). The application of nest boxes in research and management of possums and gliders. In: *Possums and Gliders*. A.P. Smith, I.D. Hume (Eds), pp. 517-525. Australian Mammal Society, Sydney.

Mering, E.D. and Chambers, C.L. (2014). Thinking outside the box: a review of artificial roosts for bats. *Wildlife Society Bulletin.* **38**, 741–751.

Milledge, D.R., Palmer, C.L. and Nelson, J.L. (1991). Barometers of change: the distribution of large owls and gliders in Mountain Ash forests of the Victorian Central Highlands and their potential as management indicators. In: *Conservation of Australia's Forest Fauna.* D. Lunney (Ed.), pp. 55–65. Royal Zoological Society of N.S.W., Sydney.

Mine, K., Yamada, A., Nanri, T., Maruyama, K., Nakamura, H. and Saigusa, M. (2014). Effect of nestbox provisioning on breeding density of dollarbirds (*Eurystomus orientalis*). *Pacific Science*. **68**, 365–374.

Moore, T., de Tores, P. and Fleming, P. A. (2010). Detecting, but not affecting, nest-box occupancy. *Wildlife Research.* **37**(3), 240-248.

Munro, H.L. and Rounds, R.C. (1985). Selection of artificial nest sites by five sympatric passerines. *Journal of Wildlife Management*. **49**, 264–276.

Nagorsen, D. (2009). *Campbell River bat project: Bat house mitigations.* British Columbia Hydro Bridge Coastal Fish and Wildlife Restoration Program.

Newton, I. (1994). The role of nest sites in limiting the numbers of hole-nesting birds – a review. *Biological Conservation.* **70**, 265–276.

O'Donnell, C.F. and Sedgeley, J.A. (1999). Use of roosts by the long-tailed bat, *Chalinolobus tuberculatus*, in temperate rainforest in New Zealand. *Journal of Mammalogy*. **80**, 913–923.

Oldroyd, B.P., Lawler, S.H. and Crozier, R.H. (1994). Do feral honey bees (*Apis mellifera*) and Regent Parrots (*Polytelis anthopeplus*) compete for nest sites? *Australian Journal of Ecology.* **19**(4), 444 – 450.

Parker, D., Roudavski, S., Jones, T. M., Bradsworth, N., Isaac, B., Lockett, M. T. and Soanes, K. (2022). A framework for computer-aided design and manufacturing of habitat structures for cavity-dependent animals. *Methods in Ecology and Evolution.* **13**, 826–841.

Payne, W., Wade, D., Hartley, I. and Shemmings-Payne, J. (2022). An evaluation of thermal imaging as a tool for assessing occupancy of enclosed nests. *Ringing & Migration.* **37**(1-2), 26 – 36.

Quin, B.R., Baker-Gabb, D.J. (1993). *Conservation and management of the turquoise parrot* Neophema pulchella *in north-East Victoria. Arthur Rylah Institute Technical Report Series No. 125.* Victorian Department of Conservation and Environment, Melbourne, Australia:

Rayner, L., Ellis, M.V., and Taylor, J.E. (2014). Hollow occurrence and abundance varies with tree characteristics and among species in temperate woodland *Eucalyptus*. *Austral Ecology* **39**,145–157.

Rhodes, M. and Jones, D. (2011). The use of bat boxes by insectivorous bats and other fauna in the greater Brisbane region. In: *The Biology and Conservation of Australasian Bats*. Law, B., Eby, P., Lunney, D. and Lumsden, L. (eds). Pages 424–442. Royal Zoological Society of New South Wales, Mosman, Australia.

Robertson, R. J. and Rendell, W. B. (2001) A long- term study of reproductive performance in tree swallows: The influence of age and senescence on output. *Journal of Animal Ecology.* **70**, 1014–1031.

Rowland, J.A., Briscoe, N.J. and Handasyde, K.A. (2017). Comparing the thermal suitability of nest-boxes and tree-hollows for the conservation-management of arboreal marsupials. *Biological Conservation.* **209**, 341–348.

Rueegger, N. (2016). Bat boxes – a review of their use and application, past, present and future. *Acta Chiropterology.* **18**, 279–299.

Rueegger, N. (2017). Artificial tree hollow creation for cavity-using wildlife – trialling an alternative method to that of nest boxes. *Forest Ecology and Management.* **405**, 404-412.

Rueegger, N. (2019). Variation in summer and winter microclimate in multi-chambered bat boxes in eastern Australia: potential eco-physiological implications for bats. *Environments.* **6**(2), 13.

Rueegger, N.N., Goldingay, R.L. and Brookes, L.O. (2013). Does nest box design influence use by the eastern pygmy-possum? *Australian Journal of Zoology.* **60**, 372–380.

Rueegger, N., Goldingay, R.L., Law, B. and Gonsalves, L. (2019). Limited use of bat boxes in a rural landscape: implications for offsetting the clearing of hollow-bearing trees. *Restoration Ecology.* **27**, 901–911.

Saenz, D., Conner, R.N., Collins, C.S. and Rudolph, D.C. (2001). Initial and long-term use of inserts by red-cockaded woodpeckers. *Wildlife Society Bulletin*. **29**, 165–170.

Salmona, J., Dixon, K.M. and Banks, S.C. (2018). The effects of fire history on hollow-bearing tree abundance in montane and subalpine eucalypt forests in southeastern Australia. *Forest Ecology and Management.* **428**, 93–103.

Saunders, D.A., Dawson, R., Mawson, P. & Cunningham, R.B. (2020a) Artificial hollows provide an effective short-term solution to the loss of natural nesting hollows for Carnaby's Cockatoo *Calyptorhynchus latirostris*. *Biological Conservation.* **245**, 1–11.

Saunders, D.A., Mawson, P.R., Dawson, R. (2020b). Predation by southwestern Carpet Python *Morelia spilota imbricata* of Carnaby's Cockatoo *Calyptorhynchus latirostris* in a breeding hollow. *Australian Zoologist*. **41**(1), 54-57.

Schölin, K.G. and Källander, H. (2011). A 64-year study of a pied flycatcher *Ficedula hypoleuca* population. *Ornis Svecica.* **21**, 79–91.

Sedgeley, J.A. (2001). Quality of cavity microclimate as a factor influencing selection of maternity roosts by a tree-dwelling bat, *Chalinolobus tuberculatus*, in New Zealand. *Journal of Applied Ecology.* **38**(2), 425–438.

Shelton, M. B., Phillips, S. S., and Goldingay, R. L. (2020). Habitat requirements of an arboreal Australian snake (*Hoplocephalus bitorquatus*) are influenced by hollow abundance in living trees. *Forest Ecology and Management*. **455**, 117675.

Shutler, D., Hussell, D.J.T., Norris, D.R. *et al.* (2012). Spatiotemporal patterns in nest box occupancy by tree swallows across North America. *Avian Conservation and Ecology.* **7**, 3.

Smith, A. (1980). *The Diet and Ecology of Leadbeater's Possum and the Sugar Glider.* PhD Thesis. Monash University, Melbourne.

Smith, A. P. and Lindenmayer, D. (1988). Tree hollow requirements of Leadbeater's possum and other possums and gliders in timber production ash forests of the Victorian central highlands. *Wildlife Research*. **15**(4), 347-362.

Smith, A. P. and Lindenmayer, D. B. (1992). Forest succession and habitat management for Leadbeater's possum in the State of Victoria, Australia. *Forest Ecology and Management*. **49**(3-4), 311-332.

Smith, G.C. and Agnew, G. (2002). The value of ''bat boxes'' for attracting hollow-dependent fauna to farm forestry plantations in southeast Queensland. *Ecological Management and Restoration.* **3**, 37–46.

Soderquist, T.R., Traill, B.J., Faris, F. and Beasley, K. (1996). Using nest boxes to survey for the Brush-tailed Phascogale *Phascogale tapoatafa*. *The Victorian Naturalist.* **113**, 256–261.

Soderquist, T., Lumsden, L. and Bennett, A. (1999). *Size does matter. Large old trees in box-ironbark forests*. Arthur Rylah Institute for Environmental Research, Heidelberg.

Strain, C., Jones, C.S., Griffiths, S.R. and Clarke, R.H. (2021). Spout hollow nest boxes provide a drier and less stable microclimate than natural hollows. *Conservation Science and Practice.* **3**, e416.

Stojanovic, D., Cook, H.C.L., Sato, C., Alves, F., Harris, G., McKernan, A., Rayner, L., Webb, M.H., Sutherland, W.J. and Heinsohn, R. (2018). Pre-emptive action as a measure for conserving nomadic species. *Journal of Wildlife Management*. **83**, 64–71.

Stojanovic, D., Owens, G., Young, C.M., Alves, F. and Heinsohn, R. (2021). Do nest boxes breed the target species or its competitors? A case study of a critically endangered bird. *Restoration Ecology.* **29**(3), e13319.

SWIFFT (State Wide Integrated Flora and Fauna Teams) (2024a). *Threatened Species Profile: Regent Parrot*. Victorian Government, Melbourne. Accessed via https://www.swifft.net.au (18/01/2024).

SWIFFT (2024b). *Threatened Species Profile: Barking Owl.* Victorian Government, Melbourne. Accessed via https://www.swifft.net.au (18/01/2024).

Taulman, J.F., Smith, K.G. and Thill, R.E. (1998). Demographic and behavioural responses of southern flying squirrels to experimental logging in Arkansas. *Ecological Applications*. **8**, 1144–1155.

Taylor, I.R., Kirsten, I. and Peake, P. (2002). Habitat, breeding and conservation of the Barking Owl *Ninox connivens* in northeastern Victoria, Australia. In: *Ecology and Conservation of Owls* (eds. Ian Newton, Rodney Kavanagh, Jerry Olsen, Iain Taylor). CSIRO, Australia.

Taylor, W.E. and Hooper, R.G. (1991). *A Modification of Copeyon's Drilling Technique for Making Artificial Red-cockaded Cavities. U.S. Forest Service General Technical Report SE-72.*

Terry, W., Goldingay, R.L. and van der Ree, R. (2021). Can chainsaw carved hollows provide an effective solution to the loss of natural tree cavities for arboreal mammals? *Forest Ecology and Management.* **490** (119122), 1 – 8.

TSSC (Threatened Species Scientific Committee) (2015). *Conservation Advice* - Nyctophilus corbeni *South-eastern Long-eared Bat.*

van der Ree, R., Bennett, A.F. and Soderquist, T.R. (2006). Nest-tree selection by the threatened brush-tailed phascogale (*Phascogale tapoatafa*) (Marsupialia: Dasyuridae) in a highly fragmented agricultural landscape. *Wildlife Research*. **33**, 113–119.

van der Ree, R. (2019). Maximising learning opportunities while replacing tree-hollows for wildlife. In: *Proceedings of the 20th National Street Tree Symposium*. Ed. Glenn Williams.

Vleck, C.M., Vleck, D., Rahn, H. and Paganelli, C.V. (1983). Nest microclimate, water-vapor conductance, and water loss in heron and tern eggs. *The Auk.* **100**(1), 76–83.

von Hartman, L. (1971). Population dynamics. In: *Avian Biology. Vol. 1*. Farner, D.S., King, J.R. (Eds.). pp. 391–459. Academic Press, London.

Wainhouse, M. and Boddy, L. (2022). Making hollow trees: inoculating living trees with wood-decay fungi for the conservation of threatened taxa – a guide for conservationists. *Global Ecology and Conservation* **33**, e01967.

Webb, J.K. and Shine, R. (1997). Out on a limb: conservation implications of tree-hollow use by a threatened snake species (*Hoplocephalus bungaroides*: Serpentes, Elapidae). *Biological Conservation.* **81**, 21 – 33.

Webster, R. (1991). *The Biology and Management of the Regent Parrot* Polytelis anthopeplus *in NSW. Species Management Report Number 4*. NSW National Parks and Wildlife Service, Hurstville, NSW.

West, P.W. (2015). *Tree and Forest Measurement*. 3rd Ed. Springer International Publishing, Switzerland.

Westerhuis, E.L., Schlesinger, C.A., Nano, C.E.M., Morton, S.R. and Christian, K.A. (2019). Characteristics of hollows and hollow-bearing trees in semi-arid river red gum woodland and potential limitations for hollow-dependent wildlife. *Austral Ecology.* **44**, 995-1004.

Wood, D., Wesley Burger Jr. L., Vilella, F. and Raulston, B. (2000). Long-Term Effects of Red-cockaded Woodpecker Cavity-Entrance Restrictors. *Wildlife Society Bulletin*. **28**, 105-109.

Wormington, K. and Lamb, D. (1999). Tree hollow development in wet and dry sclerophyll eucalypt forest in south-east Queensland, Australia. *Australian Forestry.* **62**, 336–345.

18 Supplementary Resources

The following is a list of supplementary resources that may be of assistance.

Biodiversity Conservation Trust (2020). *Biodiversity Conservation Trust – Guideline for Artificial Hollows. For Private Land Conservation Agreements.* NSW Government, Sydney.

Central Coast Council (2016). *Guideline for the Relocation of Large Tree Hollows*. July 2016. Central Coast Council, Wyong.

Leck, J. and Knight, A. R. (2022). *Guidelines for Nest Boxes and Artificial Hollows, North Coast NSW Region.* North Coast Regional Landcare. Port Macquarie