

CONSERVATION IN THE CITY: THE IMPORTANCE OF RESIDENTIAL GARDENS FOR WILDLIFE

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BSc (Hons)



This thesis is presented for the degree of Doctor of Philosophy of The
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Abstract

The replacement of natural habitat with agriculture, silviculture and urban expansion represents one of the most pervasive global threats to biodiversity. Although protected natural areas are undisputedly critical for conservation, they only cover 12% of the Earth's land surface and alone will not suffice in the battle against biodiversity loss. A significant opportunity for biodiversity conservation lies in exploiting highly modified novel ecosystems; systems that function ecologically, but have been irreversibly altered by humans from their natural state. Urban landscapes in particular could contribute to conservation outcomes because cities are often located in areas of high biodiversity and provide a structurally complex and diverse array of greenspaces.

Although residential gardens collectively comprise a major component of the total urban greenspace in many countries and a wide-range of taxa are known to occur in these areas, their value for wildlife conservation and how they could be harnessed to achieve conservation outcomes remains poorly understood. To fully understand the conservation value of gardens and how to manage these areas, we need a thorough understanding of (1) which species can prosper in gardens and how widespread they are within residential landscapes, (2) what assemblage of wildlife can be supported by gardens compared to urban remnant vegetation, (3) which factors promote the use of gardens by wildlife and how these features can be manipulated to achieve conservation outcomes, (4) how species use gardens (i.e. transient or resident use) and (5) whether residential landscapes are capable of supporting long-term persistence of populations.

This thesis addressed these key knowledge requirements through five strongly-linked research components which used both contemporary (e.g. presence, abundance and habitat use of wildlife) and historical (e.g. genetic diversity, genetic structure) lines of evidence to evaluate the value of gardens for wildlife conservation. The research was undertaken in two cities in southwest Australia using a combination of online questionnaires administered to residents, an extensive seasonal field program (trapping, spotlighting, acoustic surveys), radio-telemetry and microsatellite analyses. Mammals were used as a case study taxon because they are underrepresented in urban conservation research compared to other taxa, and they have a variety of movement capabilities (e.g. arboreal, flying, ground-dwelling) and resource requirements (e.g. folivorous, nectivorous, insectivorous). This allowed examination of whether the conservation value of gardens differed for species with differing ecological traits.

This thesis provides evidence that residential gardens can present substantial conservation opportunities. It demonstrated that some mammals can be widespread within residential landscapes, gardens can contain a considerable proportion of the regional pool of mammal diversity, and that the diversity, presence, abundance and reproductive activity of mammals in gardens can be comparable to in urban remnant vegetation. Not only were some mammals widespread and abundant, there was evidence that individuals exclusively resided within residential landscapes for several months of their life, reproduced and exploited the novel resources on offer. This ability of some species, including those that are threatened, to thrive in residential areas also appeared to support and maintain their evolutionary patterns and processes (e.g. genetic diversity, genetic structure) in city environments. There were certain garden features that influenced the presence and abundance of mammals in gardens, suggesting that if landowners can be encouraged to manage these features, conservation outcomes in residential landscapes could be realised.

These findings demonstrate that the available resources in residential gardens can support wildlife in both the short and longer-term, highlighting that residential gardens have conservation value in their own right within urban landscapes. Management initiatives that incorporate both natural urban remnants and novel habitats (e.g. gardens) in urban landscapes will likely result in better conservation outcomes than those that continue to focus only on natural areas and large urban greenspaces. This has important implications for the conservation of biodiversity globally as urban landscapes harbour numerous species worldwide (both threatened and common) and because these areas are only set to expand in the future. The inclusion of gardens in planning and management actions could play a significant role in biodiversity conservation at a global scale.

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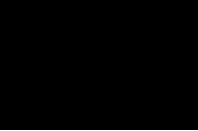
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CHAPTER 1

GENERAL INTRODUCTION



Photo: A typical residential area of Albany: one of the focus cities of this research.

Biodiversity conservation in a changing world

The conservation of biodiversity emerged as a global priority in the late 20th Century after growing recognition that human actions were increasing rates of species extinction, creating a worldwide biodiversity crisis (Pimm et al., 1995). During the 21st Century, the modification of natural habitat to make way for agricultural developments, forestry plantations and urban landscapes has been extensive and rapid, and now represents one of the most substantial and pervasive threats to biodiversity (Lindenmayer and Fischer, 2013). Although protected natural areas offer a refuge for wildlife in the face of these threats and are undisputedly critical for *in situ* conservation, they only cover 12% of the Earth's land surface (Chape et al., 2005) and alone will not suffice in the battle against biodiversity loss (Radeloff et al., 2015). Conservation efforts must, more than ever before, look for novel contemporary opportunities to protect biodiversity.

A significant opportunity for biodiversity conservation lies in exploiting the conservation value of highly modified novel ecosystems; systems that function ecologically, but have been irreversibly altered by humans from their natural state (Hallett et al., 2013; Hobbs et al., 2006). Novel ecosystems such as agricultural landscapes, cities, plantations and other non-natural habitats now dominate much of Earth's land surface and there is promising evidence that these ecosystems can play a role in the conservation of wildlife (Radeloff et al., 2015). Some novel forests, comprised primarily of exotic species, contain similar or higher tree diversity, above-ground biomass, productivity, nutrient turnover, and below-ground carbon storage relative to native forests (Mascaro et al., 2012). Agricultural landscapes can retain biodiversity and provide connective habitat for dispersing animals (Norris, 2008), and urban landscapes can sustain populations of native species (Soanes et al., 2019). As novel ecosystems continue to proliferate globally, the opportunities they present for biodiversity conservation is rapidly gaining interest (Daily, 2001; Ellis et al., 2010; Hobbs et al., 2006). The capacity of these ecosystems to support wildlife, and our ability to refocus conservation efforts to include these areas represents a substantial opportunity, and in some cases may be the last chance to save species from extinction (Soanes and Lentini, 2019). However, this opportunity may be neglected without empirical evidence of the value of novel landscapes and how these can be harnessed to achieve conservation outcomes.

Conservation opportunities within urban landscapes

Global urban land cover is expected to triple between 2000 and 2030, converting over one million km² of land into urbanised areas (Seto et al., 2012). These urban landscapes, defined by their impervious land surfaces and human-populated built-up areas, are one novel ecosystem where conserving wildlife is becoming increasingly necessary. Fortunately, urban environments offer opportunities for biodiversity conservation because cities are often located within areas of high biodiversity (Luck, 2007), the distribution of many native species overlap these areas (Ives et al., 2016; Kowarik and von der Lippe, 2018), and the public are willing to contribute to conservation efforts (e.g. Chen and Jim, 2010; Goddard et al., 2013; van Heezik et al., 2012; 2020). Urban landscapes can provide a structurally complex and diverse array of novel habitats (Aronson et al., 2017), which are often floristically rich (Threlfall et al., 2016a). These habitats, termed urban greenspaces, vary in naturalness and include private gardens, green roofs, parklands, cemeteries, golf courses, street vegetation, urban waste-lands (e.g. rubbish dumps and vacant blocks) and remnant patches of native vegetation (Aronson et al., 2017; Cilliers et al., 2013; Lepczyk et al., 2017).

Native species, including those both common (e.g. Araújo, 2003) and threatened (e.g. Ives et al., 2016), occur in all of these greenspaces (Soanes et al., 2019). Within urban greenspaces some animals can make use of the resources on offer including novel foods (e.g. human trash, supplementary feed, exotic plants), artificial shelter (e.g. under-houses spaces and roof cavities), water (e.g. ponds) and alternative structures for movement (e.g. fences and electricity power lines). In some cases, urban areas are inhabited by high densities of species that are capable of reproducing and recruiting; allowing them to seemingly thrive in urban landscapes (Hubert et al., 2011; Maclagan et al., 2018).

Which species can use urban resources and therefore occur (and sometimes thrive) in these urban habitats is largely dependent on the novel environmental features of urban areas and the biological and ecological traits (e.g. mobility, behaviour, resource requirements) of wildlife (Aronson et al., 2016; Fournier et al., 2020). Species that are food or habitat generalists, behaviourally flexible, small in body size (< 10 kg), have large litter sizes, and display bold temperaments are most likely to occur and succeed in urban landscapes (Baker and Harris, 2007; Lowry et al., 2013; Santini et al., 2019). Nevertheless, numerous taxa are capable of utilising urban areas including examples of amphibians (e.g. Parris, 2006), birds (e.g. Cannon et al., 2005; Daniels and Kirkpatrick, 2006), invertebrates (e.g. Goulson et al., 2002), fish (e.g. Gordon et al., 2009), mammals (e.g. Basham et al., 2011; Gallo et al., 2017;

Maclagan et al., 2018), plants (e.g. Smith et al., 2006) and reptiles (e.g. González-García et al., 2009). Based on the mounting evidence that urban environments can harbour a variety of wildlife, these landscapes are thought to have substantial conservation value for wildlife, prompting a call to include urban environments in conservation policy and practice (Aronson et al., 2017; Soanes and Lentini, 2019; Soanes et al., 2019).

A missed opportunity: the potential value of residential gardens for conservation

Despite the diverse array of greenspaces that support wildlife in urban landscapes, research and conservation policy have focused on the potential opportunities that ‘natural’ urban remnants offer, whilst those of smaller, more modified greenspaces have been largely overlooked in comparison (Shwartz et al., 2014; Soanes et al., 2019; Tulloch et al., 2016).

Although patches of remnant vegetation are undoubtedly critical for wildlife in urban landscapes, discounting other greenspaces such as gardens, cemeteries and street vegetation may inadvertently ignore important conservation opportunities that these other areas offer.

Residential gardens in particular may have substantial conservation value as they can collectively comprise a major component (up to 86% in León, Nicaragua) of the total urban greenspace in many countries (Goddard et al., 2010; González-García and Sal, 2008; Loram et al., 2007; Mathieu et al., 2007). Residential gardens can provide more stable and greater densities of food resources than natural habitat, offer numerous artificial shelter sites (e.g. roof cavities and under-house spaces), and can act as a refuge from the diversity of predators in natural areas (Lowry et al., 2013; Rodewald et al., 2011; Sol et al., 2013). Although a relatively new focus in urban conservation research, there is now a growing knowledge-base that some species, including those that are threatened, can attain high abundances within residential landscapes (Francis and Chadwick, 2012), and that the presence of wildlife in these areas is influenced by both local (e.g. high floral diversity and structural complexity of garden vegetation) and landscape factors (e.g. proximity to natural vegetation, city size and intensity of urbanisation) (Fontaine et al., 2016; Goddard et al., 2010). To fully understand the conservation value of gardens and whether this greenspace should be included in conservation policy we need a thorough understanding of (1) which species can prosper in gardens, (2) how widespread species are within residential landscapes, (3) which factors or mechanisms promote their use of these greenspaces and how these features can be manipulated to achieve conservation outcomes, (4) how species use residential gardens (i.e.

transient or resident use) and (5) whether these areas are capable of supporting long-term persistence.

Knowledge gaps, thesis structure and research aims

Despite the growing body of evidence that demonstrates a wide-range of taxa occur in residential gardens, large, mostly ‘natural’ greenspaces such as urban remnant vegetation are still perceived as the most important habitat for wildlife conservation within urban landscapes (Tulloch et al., 2016). This bias is reflected in urban conservation research where, compared to other greenspaces, relatively few studies (~10%, $N = 546$) have focused on the role of residential gardens for animal conservation globally, and these have predominantly concentrated on birds, invertebrates and plants (Shwartz et al., 2014). Consequently, knowledge of the value of residential gardens for wildlife conservation and how these areas can be harnessed to achieve conservation outcomes is poorly understood.

The central theme of this research addresses this key knowledge requirement by evaluating the conservation value of residential gardens using five strongly-linked research components (**Chapters 2 – 6**). Each research component addresses a critical knowledge gap that, when integrated, provides a comprehensive picture of the value of residential gardens for wildlife conservation. The research components are supported by a contextual background and justification for the research (**Chapter 1**) and general discussion (**Chapter 7**) that integrates the key findings of the research components and considers their implications for biodiversity conservation more broadly. Two appendices accompany the thesis; **Appendix 1** provides the supplementary material relevant to Chapter 4 and **Appendix 2** provides the abstracts of additional published papers I have co-authored during my candidature that are not central to my research aims but contributes to the body of knowledge that this thesis presents.

I used mammals as a case study taxon to examine the five key research components because they are underrepresented in urban conservation research compared to other taxa (Shwartz et al., 2014) and they have a variety of movement capabilities (e.g. arboreal, flying, ground-dwelling) and resource requirements (e.g. folivorous, nectivorous, insectivorous). This diversity of ecological traits and resource dependencies allowed me to consider if the conservation value of gardens differed for species with differing ecological traits. The specific knowledge gap and aim that each research component addressed is identified below.

Chapter 2

Although the presence of wildlife in gardens has been well documented, an understanding of how widespread species can be within residential landscapes and which garden features influence their presence is still in its infancy (with the exception of avian taxa, e.g. Belaire et al., 2014; Goddard et al., 2017). If species are widespread in residential landscapes and we can identify which garden features support the presence of wildlife, gardens could be managed by residents to provide wildlife with appropriate resources and therefore increase the conservation value of these greenspaces. Using an online questionnaire administered to residents of two cities in southwest Australia the aims of this research component were to:

- Establish how widespread native mammals with differing ecological traits (arboreal, semi-arboreal and ground-dwelling) were in residential landscapes.
- Identify which garden features were associated with the presence of mammal species and if these differed among species according to their habitat and resource dependencies.

Chapter 3

Large, mostly 'natural' greenspaces such as remnant vegetation are perceived as the most significant habitat for wildlife conservation within urban landscapes (Tulloch et al., 2016) despite little effort to test this assumption. Very few studies explore the value of residential gardens or other highly modified urban habitats relative to natural urban remnant vegetation, even though recent reviews and investigations stress the importance of these types of studies to inform biodiversity conservation (e.g. Maclagan et al., 2018; Shwartz et al., 2014). This represent a significant hindrance to our understanding of the conversation value of residential gardens. Using extensive trapping, spotlighting, camera trapping and acoustic survey techniques within a city in southwest Australia the aims of this research component were to:

- Describe and document the assemblage of mammals found within residential gardens relative to protected natural habitat.
- Compare the diversity, presence, abundance and reproductive activity of native mammal species between gardens and urban remnant vegetation.
- Compare the presence and abundance of introduced mammal species between gardens and urban remnant vegetation.

- Examine if the comparative presence and abundance of mammals in residential gardens and urban remnant vegetation differed for flying, ground-dwelling and arboreal native species.

Chapter 4

Some progress has been made in evaluating the mechanisms that drive the diversity, presence and abundance of wildlife within residential gardens (e.g. Baker et al., 2003; Basham et al., 2011; Daniels and Kirkpatrick, 2006; Fontaine et al., 2016; Threlfall et al., 2016b; 2017). These studies provide promising evidence that garden features could be manipulated by residents to improve conservation outcomes for fauna in gardens (Daniels and Kirkpatrick, 2006; Dique et al., 2004; Oprea et al., 2009; Patterson et al., 2017; Threlfall et al., 2016b; 2017). However, there has been little consideration of how the importance of these features vary seasonally, despite that without this information management of garden features may be misguided, or at best constrained by an incomplete understanding of the critical resources required to support wildlife throughout the year. Considering seasonality would allow for the identification of all of the features that influence the presence and abundance of animals, including those that are important in single-seasons and those that are consistent across seasons. This information would benefit decision-making through the identification of which features could be manipulated to achieve the greatest conservation outcome, providing great value to conservation if applied by wildlife-friendly gardeners. Using a seasonal cage trapping, spotlighting, camera trapping and acoustic survey regime within a city in southwest Australia the aims of this research component were to:

- Examine which garden features were associated with mammal presence and abundance and whether these varied among seasons.
- Evaluate if the seasonal importance of garden features varied among three broad mammal groups with different ecological traits (arboreal, flying and ground-dwelling).

Chapter 5

Whilst numerous examples of wildlife in residential gardens (see Goddard et al., 2010; Schwartz et al., 2014 for review) provide mounting evidence that gardens could offer conservation opportunities for biodiversity, none have demonstrated that they are capable of supporting animal residency or key ecological processes such as reproduction, recruitment and survival, essential for population persistence (Maclagan et al., 2018). If individuals are

capable of residing in residential areas, it would suggest that gardens are of sufficient habitat quality and have the necessary resources to support individuals in the longer-term (and therefore potentially reproduction) in their own right. A biological understanding of these aspects is needed to fully understand the conservation value of gardens and whether this habitat type should be included in conservation policy. Using a Critically Endangered marsupial as a case study and radio-tracking individuals within a city in southwest Australia the aims of this research component were to:

- Examine the capacity of gardens to support animal residency (i.e. exclusive use of residential habitat).
- Establish if individuals used residential areas less frequently when in closer proximity to natural remnant vegetation.
- Determine which resources were used within residential gardens and the frequency that novel resources were used compared to natural ones.
- Identify if individuals were capable of reproducing within residential landscapes.

Chapter 6

There has been a recent call for practitioners and decision makers to undertake conservation action in urban landscapes based on the contemporary presence of wildlife in urbanised habitats (Goddard et al., 2010; Soanes et al., 2019; Tulloch et al., 2016). However, the conservation value of these areas may be overestimated and conservation efforts misguided, if we fail to also consider the impacts of urbanisation on evolutionary processes that may influence the persistence of wildlife populations over longer-time scales. Urban landscapes can impede animal movement and subsequently restrict gene flow among (and within) populations resulting in loss of genetic diversity and increased genetic differentiation among populations (Johnson and Munshi-South, 2017). This can decrease the ability of species to adapt to environmental change, and can result in inbreeding depression, loss of fitness and the accumulation of deleterious mutations (Gaggiotti, 2003); increasing species vulnerability to extinction (Frankham, 2005). As urban landscapes harbour numerous threatened species worldwide (Aronson et al., 2014; Ives et al., 2016) understanding the impacts of urbanisation on both contemporary distribution and abundance, and evolutionary processes has never been more critical for the conservation of these species. Whilst there is growing evidence that some threatened species can occur in high densities, reproduce and otherwise do seemingly well in urban habitats (e.g. Maclagan et al., 2018), the impact of urbanisation on the evolutionary processes of these species has been scarcely examined. This represents a real

risk in implementing the call for practitioners and decision makers to undertake conservation action in urban environments (Goddard et al., 2010; Soanes et al., 2019; Tulloch et al., 2016). Using microsatellite analyses of a Critically Endangered marsupial within a city in southwest Australia, the aims of this research component were to:

- Investigate the impact of an urban area on the gene flow, genetic structure and genetic diversity of a highly threatened species that seemingly thrives in city landscapes (based on its contemporary presence and abundance).
- Examine if the genetic relatedness of individuals in residential landscapes differed to in urban remnant vegetation.

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CHAPTER 2

THE PREVALENCE OF MAMMALS IN RESIDENTIAL LANDSCAPES AND THE GARDEN FEATURES ASSOCIATED WITH THEIR PRESENCE



Photo: One of the garden sites surveyed for mammals in the City of Albany.

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Abstract

Urbanisation threatens biodiversity globally, yet some animal populations persist within urban landscapes. Conservation of urban wildlife has prioritised parks and remnant bushland as critical habitat and neglected the role that residential gardens offer for conservation. We explored the potential for residential gardens to assist in the conservation of mammals using an online questionnaire administered to residents of two case study cities in Australia to identify how widespread mammals in cities can be, which garden features promote mammal presence and if the features varied among species with different habitat requirements. From 649 responses we found that three mammal species with different habitat requirements occurred in residential gardens throughout the city landscape. Garden features promoting mammal presence were consistent with broad ecological and habitat requirements of each species, but differed among both species and regions. Our study suggests that residential gardens offer a valuable habitat for mammals and that garden features could be manipulated to promote use of gardens by these species. By considering gardens in urban planning and management actions aimed at conserving urban wildlife, residential gardens offer additional habitat to roadside vegetation, urban bushland and other urban greenspaces and can play a substantial role in biodiversity conservation.

Introduction

Urbanisation is rapidly replacing natural environments with modified landscapes and is considered one of the greatest threats to global biodiversity (Seto et al., 2012). However, because cities are often located in areas of high biodiversity (Luck, 2007) and overlap the distribution of many native species (Ives et al., 2016; Kowarik and von der Lippe, 2018), there is growing recognition of their potential role in biodiversity conservation (Aronson et al., 2017; Ives et al., 2016; Magle et al., 2012; Soanes and Lentini, 2019). Urban greenspaces such as remnant patches of native bushland, roadside vegetation, artificially vegetated parks, and residential gardens (Aronson et al., 2017; Cilliers et al., 2013) are all capable of supporting wildlife. The conservation value of these greenspaces depends on their ability to provide resources (i.e. food and shelter) for wildlife (Lowry et al., 2013), offer alternative habitats for individuals to reside in (Luniak, 2004), and promote gene flow of wildlife populations through connectivity of habitat patches (Braaker et al., 2017). To realise the conservation opportunity that urban areas represent, an understanding of which species use urban environments and the resources that support them is critical for the creation or management of urban greenspaces.

The majority of research that has examined the conservation value of urbanised landscapes has predominantly focused on large urban greenspaces (i.e. parks and remnant bushland) and certain taxa such as invertebrates, plants and birds (Goddard et al., 2010; Shwartz et al., 2014). This has restricted the extent and breadth of biodiversity considered in management actions that aim to create and enhance the conservation value of urban greenspaces. The supporting role of small greenspaces, such as residential gardens, has been largely overlooked in the field of conservation, despite comprising a major component of the total greenspace (as high as 86%; González-García and Sal, 2008) in urbanised landscapes (Goddard et al., 2010; Loram et al., 2007; Mathieu et al., 2007). The limited ecological research that has been conducted on wildlife in residential gardens has primarily concentrated on documenting the diversity, presence or abundance of species (e.g. Bernholt et al., 2009; Fetridge et al., 2008). Contrastingly, knowledge of the ecological mechanisms driving patterns of biodiversity in gardens (and other greenspaces) is scarce (McDonnell and Hahs, 2013), but is essential if the conservation value of these areas is to be fully realised. The mechanisms that influence biodiversity in gardens likely operate at many spatial scales, including influences of the surrounding reserve network and landscape, the attributes of the neighbourhood or suburb, and the attributes of the garden itself (Goddard et al., 2010).

Animals within urban landscapes face a myriad of challenges in the form of altered and disconnected habitat, novel resources and increased exposure to predators such as cats (*Felis catus*) and dogs (*Canis familiaris*) (McKinney 2002; Sol et al., 2013). Whether a species is able to occur in urban landscapes depends on the environmental characteristics of the urban habitat (e.g. food, water and shelter availability, suitable climate; Marzluff and Rodewald, 2008) and the behavioural or biological traits of the species (Croci et al., 2008). Animals that are food or habitat generalists, behaviourally flexible, small in body size (< 10 kg) and display bold temperaments are most likely to inhabit urban environments (Baker and Harris, 2007; Lowry et al., 2013), providing there is habitat and resources available. Whilst a species' biology cannot be modified to achieve conservation outcomes in urban areas, the environmental characteristic of gardens can. For example, Daniels and Kirkpatrick (2006) highlighted that the floristic structure and composition of gardens could be manipulated to improve their ability to support bird species. Similarly, González-García et al. (2009) demonstrated that 'patios' (i.e. private gardens) with fences and preferred food trees could enable establishment and persistence of black spiny-tailed iguana (*Ctenosaura similis*) by providing movement pathways and appropriate food in these non-natural areas. By managing gardens and other urban greenspaces so that they provide wildlife with appropriate resources they can be used to achieve substantial conservation outcomes in urban landscapes.

Mammals are often used as a flagship taxon for biodiversity conservation (Smith et al., 2012), yet they are often highly threatened by urbanisation. Despite this, very few studies have examined the role of gardens for their conservation, globally (Shwartz et al., 2014), with some notable exceptions (e.g. Dique et al., 2004; Oprea et al., 2009; Patterson et al., 2017; Threlfall et al., 2016) that provide promising evidence that gardens can play a role for at least some mammal species. For example, gardens with greater vegetation maintain bat diversity in urban centres of southeastern Brazil (Oprea et al., 2009), and greater tree coverage and more fruiting trees can increase foraging activity of urban vervet monkeys (*Cercopithecus aethiops pygerythrus*) in South Africa (Patterson et al., 2017). Given that Australia has one of the highest extinction rates for mammals in the past 200 years (Cardillo and Bromham, 2001; Woinarski et al., 2015) and 20% of threatened Australian mammals have distributions that overlap with cities (Ives et al., 2016), the country is an ideal case study to further examine the potential value of gardens for mammal conservation. Because species vary in their habitat requirements for shelter, food and structures that facilitate movement, the garden characteristics that promote mammal presence will likely vary among species. To the best of

our knowledge no study has examined the relationship between garden characteristics and the distribution of mammals with different habitat requirements in Australia.

While mammal presence can also be influenced by factors that operate at spatial scales greater than the garden (e.g. neighbouring gardens), in this study we focused on the mechanisms that operate within singular gardens. We used an online questionnaire, administered to residents of two cities in southwest Australia, to examine which garden characteristics promoted mammal presence and if the characteristics varied among species with different habitat and resource requirements. We investigated this using three mammals that occur in southwest Australia as case studies of species that have different habitat and resource requirements; the arboreal western ringtail possum (*Pseudocheirus occidentalis*), the semi-arboreal brushtail possum (*Trichosurus vulpecula hypoleucus*) and the ground-dwelling southern brown bandicoot (*Isoodon fusciventer*). We aimed to (1) establish how widespread the three mammal species were within two case study cities, (2) identify which garden characteristics promoted the presence of each species and (3) evaluate if the influence of garden characteristics on mammal presence differed between species according to their habitat and resource dependencies. A better understanding of the garden characteristics that can best support these mammal species will improve knowledge of the conservation potential of gardens and more broadly the conservation opportunity that small, private greenspaces offer for global biodiversity.

Methods

Study species

We investigated the influence of garden characteristics (as assessed by residents) on the presence and distribution of the arboreal western ringtail possum (hereafter ‘ringtail possum’), the semi-arboreal common brushtail possum (hereafter ‘brushtail possum’), and the ground-dwelling southern brown bandicoot (hereafter ‘bandicoot’). These species are nocturnal and medium-sized mammals within the critical weight range (35–5500 g) of species susceptible to population declines (Burbidge and McKenzie, 1989). All inhabit urban landscapes (Busschots et al., 2020; Eymann et al., 2006; Valentine et al., 2013; Van Helden et al., 2018), although only the brushtail possum is suggested to have adapted to urban habitats (i.e. an ‘urban adapter’; Hubert et al., 2011). These three species are relatively conspicuous and easily distinguishable mammals in urban environments and for this reason were ideal species to use in our survey of residents.

Both the ringtail and brushtail possum predominantly use canopy trees in well-connected forest habitats and feed on foliage (IUCN, 2019), although are also known to use ground habitats (Cowan, 1989; Van Helden et al., 2019). Unlike the ringtail possum that is exclusively herbivorous (Mathieson et al., 2020; Shepherd et al., 1997), the brushtail possum is omnivorous, also feeding on small mammals and eggs (Department of the Environment, 2019). The bandicoot is ground-dwelling, preferring thickly vegetated ground habitats and is omnivorous, predominantly foraging in the leaf litter and soil for insects, fungi and plant roots (IUCN, 2019; Valentine et al., 2013). Each of these three species vary in mobility; the ringtail possum is largely sedentary with home ranges often less than one hectare (Van Helden et al., 2018; Yokochi et al., 2015), whilst the bandicoot and brushtail possum are generally more mobile with larger home-range sizes (1–7 ha; DPaW, 2012b; Harper, 2005). All three species are vulnerable to predation from foxes (*Vulpes vulpes*), cats and dogs but given its ground-dwelling nature, the bandicoot is likely most susceptible (Johnson and Isaac, 2009). Each species varies in both habitat requirements and mobility, but are comparable in size and activity patterns. As a result, they were ideal case studies to investigate the importance of garden characteristics in determining the presence of animals with different habitat requirements and behaviours.

Study area

Our case study species are distributed in southwest Australia and their ranges have considerable overlap with urbanised environments (Fig. 2.1). The bandicoot and brushtail possum are widespread throughout southwest Australia with the distribution of the bandicoot extending from Guilderton north of Perth to east of Esperance, and the brushtail possum from north of Perth to east of Bremer Bay and as far inland as Laverton (DPaW, 2012a, b; Van Dyck et al., 2013; Fig. 2.1). In contrast, the ringtail possum is restricted to disjunct areas: the Swan Coastal Plain; Southern Forest Zone; and South Coast Region (DPaW, 2017; Van Dyck et al., 2013; Fig. 2.1). The distribution of each species overlaps major urbanised landscapes in the South Coast Region and Swan Coastal Plain (DPaW, 2017). For this reason we released the questionnaire within the largest city in each region; Albany (35° 01' S, 117° 53' E) and Bunbury (33° 20' S, 115° 39' E), respectively. The population of Albany and nearby surrounds is approximately 33 000 people that reside in 15 287 private dwellings, whilst Bunbury's population is approximately 32 000 people that reside in 15 406 private dwellings (ABS, 2016). Both cities are undergoing rapid urbanisation, but support multiple native animal species of which 45 in Albany and 20 in Bunbury are threatened under Australia's

Environmental Protection and Biodiversity Conservation Act 1999 (Ives et al., 2016). Although residents of Albany and Bunbury were targeted in promotion of the survey (see *Household questionnaire*), a total of 883 people from multiple urban centres in southwest Australia participated including residents of Dunsborough, Busselton, Manjimup, Augusta and Denmark (Fig. 2.1).

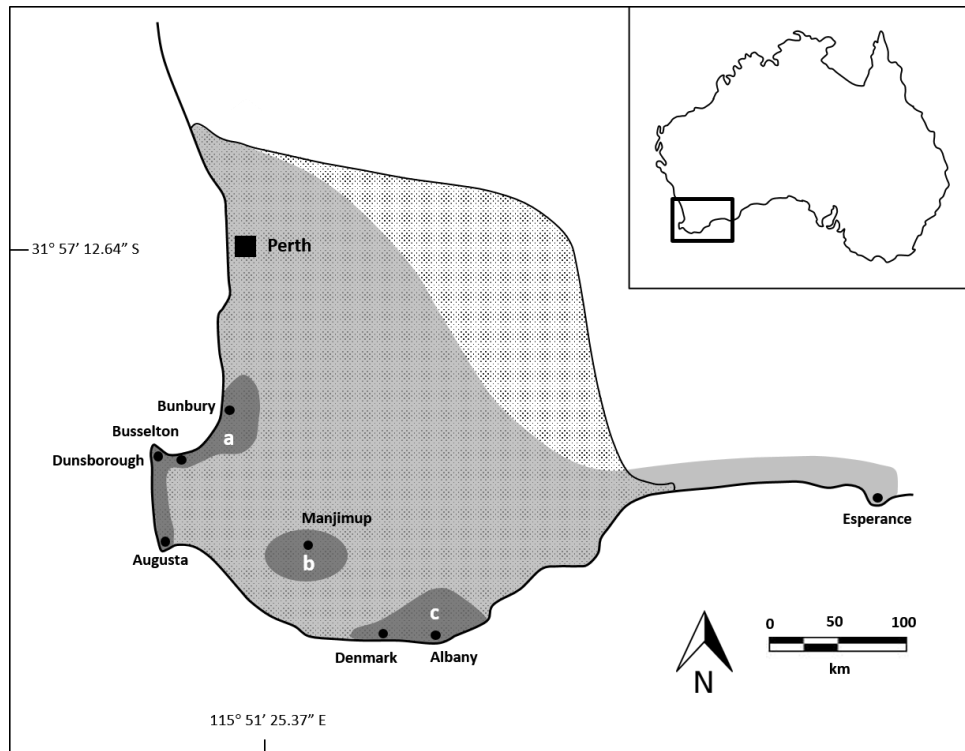


Fig. 2.1. Map illustrating the overlapping distributions of the western ringtail possum (*Pseudocheirus occidentalis*; dark grey), common brushtail possum (*Trichosurus vulpecula hypoleucus*, hatched grey) and southern brown bandicoot (*Isodon fusciventer*; light grey). The western ringtail possum is restricted to three key stronghold areas: the Swan Coastal Plain (a); Southern Forest Zone (b) and South Coast Region (c). The location of Bunbury and Albany where the questionnaire was focused, Perth Capital City and outlying population centres of Esperance, Dunsborough, Busselton, Manjimup, Augusta and Denmark are shown.

Household questionnaire

We administered an electronic questionnaire (Qualtrics, Provo, Utah, USA) from December 2017 to February 2018 in Albany, and September 2018 to December 2018 in Bunbury. We distributed a weblink to the questionnaire to a cross section of residents by promotion through social media and email. The weblink was also forwarded by local businesses and organisations to their respective networks. The questionnaire covered four key themes (1) observations of mammal wildlife, (2) certainty of observations, (3) garden and block characteristics (e.g. amount of tree cover and size of block) and (4) location of households

(Table 2.1). These questions were accompanied by a visual guide to assist people in mammal identification, and additional questions to identify basic demographic characteristics (e.g. age and gender) and whether anyone else residing in the house had or could have also completed the questionnaire (Table 2.1). We selected which garden characteristics to include in the questionnaire based on habitat dependencies of our case study species (e.g. ground-dwelling, arboreal) and other factors we thought may influence mammal presence in gardens. For these reasons, we considered the percentage of tree, shrub, ground and native cover in gardens, as well as presence of potential predators (i.e. dogs and cats), block size (i.e. an indirect indicator of proximity to humans; Baker and Harris, 2007), and distance to nearest natural bushland remnants (based on literature that emphasises the importance of natural greenspaces in urban landscapes; e.g. FitzGibbon et al., 2007; Goad et al., 2014). In the questionnaire we used broad response categories for each garden characteristic to reduce the likelihood of inaccurate reporting of garden features by residents (Table 2.1).

Data analysis

We used stepwise binomial generalised linear models (GLMs) to assess the presence or absence of ringtail possums, bandicoots, or brushtail possums relative to seven garden or block characteristics as explanatory variables (Table 2.1). Response variables were recoded as a binary variable (i.e. '1' to denote presence, and '0' for absence; Table 2.1). Explanatory variables were recoded from one to five (lowest to highest value) for distance between the household and nearest bushland remnant, percentage tree cover, percentage shrub cover, percentage open space and proportion of garden that was native based on the response categories available to respondents (Table 2.1). Property size was recoded from one to three, and presence of pets was recoded as a binary variable (i.e. '1' to denote presence of an outside cat or dog and '0' for no outside pet, Table 2.1). Prior to binomial GLM analyses, Spearman's rank correlation was applied to test for correlation between explanatory variables. No pair of variables were highly correlated ($-0.4 < r < 0.4$), and all were retained for analysis.

Any respondent that answered 'unsure' to any question could not be recoded and was removed. Questionnaire responses were also excluded from data analysis if the questionnaire was incomplete, if the respondent had indicated it was possible another household member had already completed the survey, or if the respondent was located outside of the Swan Coastal Plain and South Coast Region. We also excluded any responses that were from block sizes greater than one acre to restrict the study to households that did not potentially have

substantial remnant vegetation on their block. Although accuracy in the identification of each species by respondents was high based on ground-truthing data ($> 70\%$; R. Steven; B. Van Helden; P. Close unpublished data) any respondent that was $\leq 50\%$ certain they had correctly identified a species on their property was removed in analysis focused on that species. This resulted in variable sample sizes for analysis among the three species (Table 2.2).

For each species, four stepwise binomial GLMs were performed. A model with no interactions, and a model with two-way interactions among explanatory variables were fitted to respondents located in the South Coast Region and the Swan Coastal Plain (Table 2.2). For each region, the best model was selected based on Bayesian information criterion (BIC) rather than Akaike information criterion (AIC) as it penalises model complexity more heavily (Kuha, 2004). Where models had similar BIC values (i.e. within 3 BIC units) the simplest model was selected. Models with more than two interactions among explanatory variable were not considered because of limited plausibility and their complexity.

Based on the survey responses, we mapped (QGIS Development Team, 2019) the distribution of mammal presence in gardens to visually evaluate the distribution of mammal presence within Albany and Bunbury. The distribution of mammals in other city centres was not assessed as we considered the number of respondents too low to provide a reasonable representation of mammal presence. The number of households that had reported the presence of a mammal species was compared between study regions using a proportion test. All statistical analyses were performed in R version 3.3.1 (R Core Team, 2016) and when interpreting results, alpha was set at 0.05.

Table 2.1. Summary of key questions and possible responses from the online questionnaire that was released in Albany and Bunbury in southwest Australia. Variable type and recoding undertaken to perform binomial generalised linear models (GLMs) is also indicated. The term ‘remove’ indicates removal of survey respondents from binomial GLMs

Question themes	Questions asked	Possible responses	Recoded	Variable type
Observations of mammal wildlife	Have you seen a western ringtail possum in your garden or around your house?	Yes	1	Response
		No	0	
	Have you seen a bandicoot in your garden or around your house?	Yes	1	Response
		No	0	
	Have you seen a brushtail possum in your garden or around your house?	Yes	1	Response
		No	0	
Certainty of observations	How certain are you that it was a western ringtail possum?	Very certain (75-100% certain)	NA	Used for data curation
		Fairly certain (50-75% certain)		
		Neither certain or uncertain (50% certain)		
		Fairly uncertain (25-50% certain)		
		Very uncertain (0-25% certain)		
	How certain are you that it was a bandicoot?	Very certain (75-100% certain)	NA	Used for data curation
		Fairly certain (50-75% certain)		
		Neither certain or uncertain (50% certain)		
	How certain are you that it was a brushtail possum?	Fairly uncertain (25-50% certain)	NA	Used for data curation
Very uncertain (0-25% certain)				
Very certain (75-100% certain)				
Garden and block characteristics	Do you own pets that spend at least some of their time outside?	Yes, at least one cat	1	Explanatory
		Yes, at least one dog	1	
		Yes, at least one cat and one dog	1	
		No	0	
	What proportion of your garden is open space (<i>i.e.</i> grass, concrete, sand, dirt)?	75-100%	5	Explanatory
		50-75%		
		50%		
		25-50%		
		0-25%		
	What proportion of your garden has tree cover (from trees greater than two meters high)?	75-100%	5	Explanatory
		50-75%		
		50%		

Question themes	Questions asked	Possible responses	Recoded	Variable type
		25-50%	2	
		0-25%	1	
	What proportion of your garden has shrub cover (bush or trees less than two meters high)?	75-100%	5	Explanatory
		50-75%	4	
		50%	3	
		25-50%	2	
		0-25%	1	
	What is the proportion of your shrubs and trees that are native plants?	It is almost entirely made up of native plants (75-100% native)	5	Explanatory
		It is mostly made up of native plants (50-75% native)	4	
		About half of the garden is native (50% native)	3	
		It is mostly made up of exotic plants (25-50% native)	2	
		It is almost entirely made up of exotic plants (0-25% native)	1	
		I have none	1	
		Unsure	Removed	
	What is the distance to the nearest natural bushland area from your house?	Within sight (0-100 meters)	1	Explanatory
		Within a five minute walk (100-500m)	2	
		Within a 30 minute walk (500 m-2.5 km)	3	
		Within a 10 minute drive (2.5-10 km)	4	
		More than a 10 minute drive (greater than 10 km)	5	
	How large is your property?	Approximately an eighth of an acre (average subdivided block)	1	Explanatory
		Approximately one quarter of an acre (average suburban block)	2	
		Approximately one acre (large semi urban blocks)	3	
		Greater than an acre	Removed	
Location of household	Provide your full residential address below.	Street, suburb or city address.	NA	Used to map questionnaire respondents

Results

Garden characteristics

A total of 649 survey responses were used in the analysis, of which 302 were from the South Coast Region and 347 from the Swan Coastal Plain. Reported garden characteristics were highly variable, and all possible responses for the seven questions that focused on block or garden characteristics were represented (Table 2.1; Fig. 2.2). There was a reasonably equal number of households within categories of percentage open space (Fig. 2.2a), but the majority of households (> 70%) had low shrub and tree cover (< 50% cover) in their gardens (Fig. 2.2b, c). The percentage of shrub and tree cover in gardens that was native was variable, with a similar number of households in each category (Fig. 2.2d). Most respondents (> 80%) resided within 500 m of bushland remnants (Fig. 2.2e) in small blocks less than one quarter of an acre (Fig. 2.2f). Outside dogs or cats were reported in 62% of survey responses.

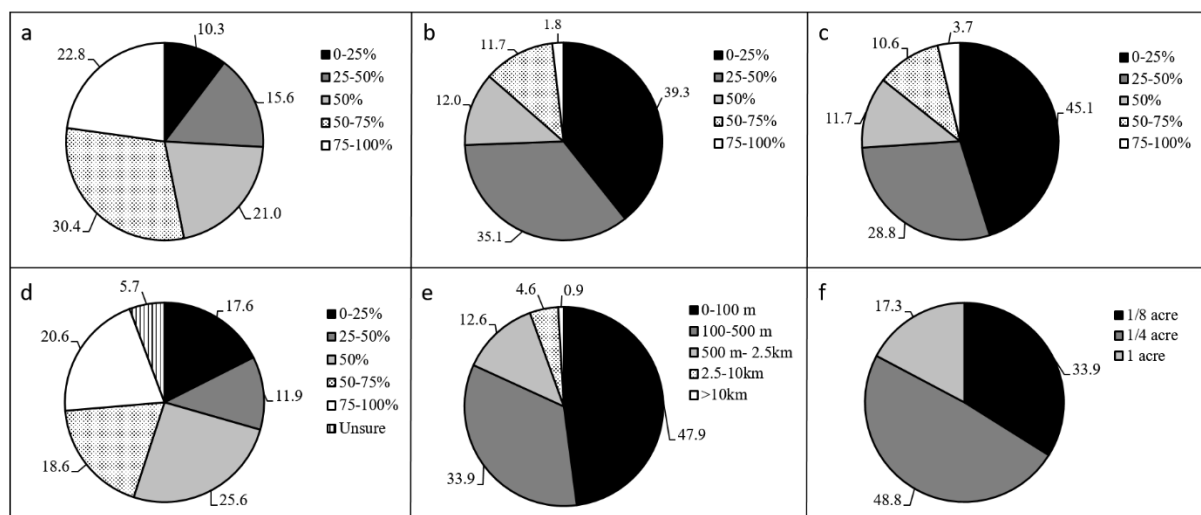


Fig. 2.2. Percentage of respondents ($N = 649$) that estimated on their property the percentage of open space (a); percentage of shrub cover (b); percentage of tree cover (c); percentage of shrub and tree cover that was native (d); distance to nearest bushland remnant (e); and size of block (f).

Mammal presence

All three mammal species were present in gardens in both study regions. Sixty five percent of respondents reported ringtail possums ($N = 635$), 27% reported bandicoots ($N = 647$), and 17% reported brushtail possums ($N = 642$) in their gardens. More than 50% of households in both study regions reported the presence of ringtail possums in their gardens, whilst bandicoots and brushtail possums were reported to be present at less than 50% of households,

regardless of region (Fig. 2.3). Both possum species were present at a higher proportion of households in the Swan Coastal Plain compared to the South Coast Region (ringtail possum: $\chi = 5.91$, $P = 0.012$; brushtail possum: $\chi = 8.21$, $P = 0.004$), whilst bandicoots were present at a higher proportion of households in the South Coast Region compared to the Swan Coastal Plain ($\chi = 104.11$, $P < 0.001$; Fig. 2.3). All three species were reported throughout suburbs of greater Albany and greater Bunbury including city centers and city edges (Fig. 2.4). There was no obvious spatial grouping or pattern of species distribution for any of the mammals in either city (Fig. 2.4).

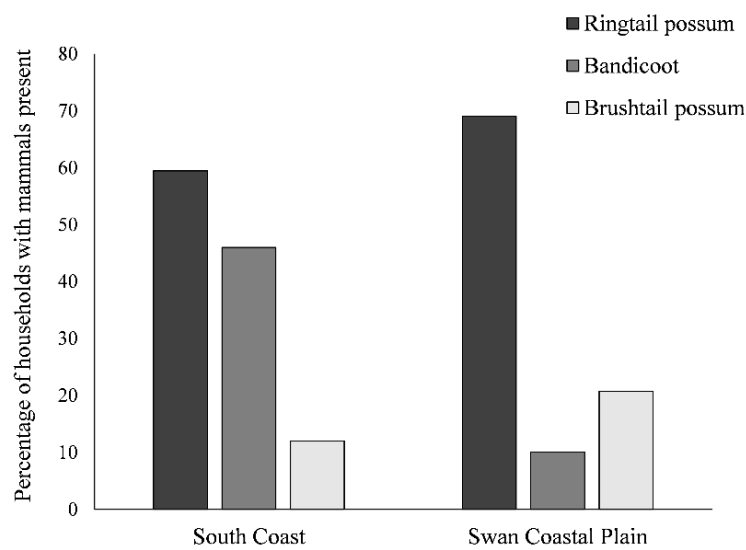


Fig. 2.3. Percentage of households that identified the presence of the western ringtail possum (*Pseudocheirus occidentalis*), southern brown bandicoot (*Isodon fusciventer*) and common brushtail possum (*Trichosurus vulpecula hypoleucus*) in the South Coast Region and Swan Coastal Plain. In the South Coast Region, $N = 296$, 300 and 300 for ringtail possums, bandicoots and brushtail possums, respectively, and in the Swan Coastal Plain, $N = 339$, 347 and 342.

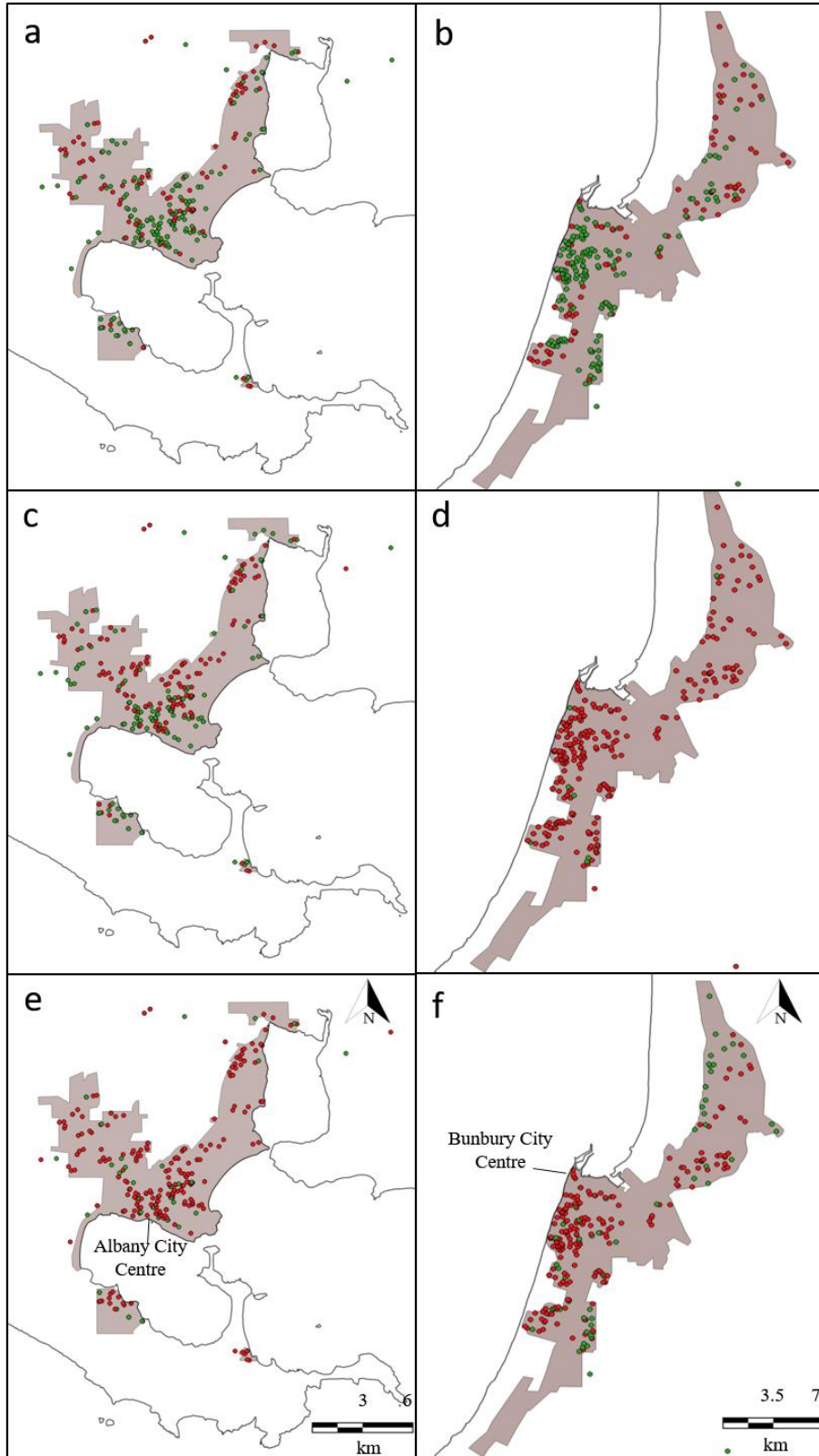


Fig. 2.4. The presence and absence of western ringtail possums (*Pseudocheirus occidentalis*) in Albany (a) and Bunbury (b); southern brown bandicoots (*Isoodon fusciventer*) in Albany (c) and Bunbury (d); and common brushtail possums (*Trichosurus vulpecula hypoleucus*) in Albany (e) and Bunbury (f) based on an online questionnaire that surveyed residents of the two cities. Green dots represent households that reported presence and red dots indicate reported absence of mammals. Grey shading denotes the urbanised areas of Albany, Bunbury and nearby surrounds. Location of city centres shown in (e) and (f).

Garden characteristics that determine mammal presence

The factors that affected mammal presence varied among the species and except for the ringtail possum, also differed between regions (Table 2.2). Tree cover availability was positively related to presence of ringtail possums in both regions ($P < 0.001$), bandicoots in the South Coast Region ($P < 0.001$), and brushtail possums in the Swan Coastal Plain ($P < 0.001$; Table 2.2). Whilst shrub cover positively influenced bandicoot presence in both regions ($P < 0.003$), the percentage of open space ($P = 0.019$), and tree cover ($P < 0.001$) positively affected bandicoot presence in the South Coast Region only (Table 2.2). For the presence of bandicoots in the South Coast Region, shrub cover negatively interacted with both open space ($P < 0.001$) and tree cover ($P = 0.002$; Table 2.2). In gardens with a higher percentage of shrub cover, the percentage of open space had a negative influence on bandicoot presence, but in gardens with low shrub cover, open space had no effect on bandicoot presence. In gardens with low shrub cover, tree cover had a strong positive influence on bandicoot presence, whilst at high shrub cover, tree cover had a strong negative influence.

The presence of pets reduced the positive influence of shrub cover on bandicoot presence in the South Coast Region ($P = 0.009$; Table 2.2). In gardens of low shrub cover pets had no influence on bandicoot presence, whilst in gardens of high shrub cover the presence of pets reduced the likelihood of bandicoot presence. The presence of pets, shrub cover and open space had no influence on either arboreal species. Larger properties were more likely to contain brushtail possums in the Swan Coastal Plain ($P < 0.001$), however had no effect in the South Coast Region nor on any other species (Table 2.2). The proportion of garden vegetation that was native and the distance to nearest bushland remnant had no effect on the presence of any species in either region (Table 2.2).

Table 2.2. Modelling results examining the relationship between seven garden or block characteristics (pets, tree cover, shrub cover, ground cover, cover that is native, distance to nearest bushland remnant and property size), and the presence or absence of three mammal species in the South Coast Region (SCR) and Swan Coastal Plain (SCP). Base model 1 represents a no interaction model between explanatory variables; Base model 2 represents a two-way interaction model among explanatory variables. The final model with lowest BIC value was selected and Δ BIC represents the decrease of BIC between the base model and the final model after stepwise binomial generalised linear regression. Significance levels of <0.001 are indicated as ‘***’, 0.001–0.01 as ‘**’, and 0.01–0.05 as ‘*’

Animal	Region	Base model	N	Δ BIC (BIC of final model)	Variables in final model (Estimate \pm SE)
Ringtail possum	SCR	1	279	26 (370)	Tree cover (0.43 \pm 0.12) ***
	SCP	1	319	32 (382)	Tree cover (0.55 \pm 0.16)***
Bandicoot	SCR	2	283	103 (360)	Pet (0.68 \pm 0.64)
					Open space (0.61 \pm 0.26)*
					Tree cover (1.19 \pm 0.30) ***
	SCP	2	326	101 (204)	Shrub cover (3.3 \pm 0.69) ***
					Pet*shrub cover (-0.81 \pm 0.31)**
					Open space*shrub cover (-0.46 \pm 0.13) ***
					Tree cover*shrub cover (-0.37 \pm 0.12) **
Brushtail possum	SCR	1	284	28 (214)	Pet (16.94 \pm 1231.87)
	SCP	1	322	22 (277)	Shrub cover (0.52 \pm 0.17)**
					Distance (-0.35 \pm 0.33)
					Pet*distance (-17.22 \pm 1231.88)
					NIL
					Tree cover (0.52 \pm 0.13)***
					Property size (1.31 \pm 0.25)***

Discussion

Urban landscapes present an opportunity for biodiversity conservation (Aronson et al., 2017; Ives et al., 2016; Soanes and Lentini, 2019). However, efforts to conserve wildlife has concentrated on remnant vegetation and public greenspaces, despite the significant contribution of residential gardens to total urban greenspace (Goddard et al., 2010; Loram et al., 2007; Mathieu et al., 2007). Against a backdrop of research that has largely examined birds and invertebrates in urban landscapes, our study assessed the presence of mammals in residential gardens and has demonstrated native, small to medium-sized mammal species can commonly occur in gardens. We found that some garden characteristics could be used to predict the presence of mammal species with both arboreal and ground-dwelling habits, albeit subject to variation among species and regions. This variation is possibly attributable to the influence of surrounding gardens which we did not assess in this study. In addition to urban remnant vegetation and public greenspaces, our study demonstrates that residential gardens offer valuable habitat to conserve wildlife in expanding urban landscapes.

Remnant vegetation in urban areas is considered critical for all three studied mammals to persist in urbanised landscapes (e.g. Harper, 2005). Although their presence in residential

gardens is well known (Hillman et al., 2017; Statham and Statham, 1997; Thompson and Thompson, 2009), our study represents the first comprehensive attempt to describe and explain their distribution in residential areas. All three species were reasonably widespread in residential areas in at least one of the cities and were reported from gardens in inner city areas and in outer city edges. This is in contrast to previous work that generally demonstrates a decrease in the abundance and presence of terrestrial fauna in highly urbanised areas compared to less urbanised regions because of increased habitat loss and reduced quality of remaining vegetation (McKinney, 2008). Our study supports the hypothesis that gardens can offer a refuge for native fauna (Goddard et al., 2010) and demonstrates that garden habitats may mitigate the effects of urbanisation for some mammal wildlife.

The value of gardens and other urban greenspaces for wildlife conservation is thought to be greatest for urban adapted or ‘synurbic’ species, a term generally reserved for species populations that have higher densities in urban areas compared to natural areas (Blair, 1996; Francis and Chadwick, 2012). The benefits that gardens can offer threatened or rare species is often overlooked as it is commonly assumed these species have specific habitat requirements (Kani, 2011), and are therefore negatively impacted by urbanisation. Interestingly, our results indicated that the Critically Endangered ringtail possum was the most commonly reported mammal in gardens (65% of gardens), whilst the brushtail possum, often termed an ‘urban adapter’ (e.g. Hubert et al., 2011) and therefore expected to be present in gardens frequently, was the least commonly reported mammal in our study (17% of gardens). The presence of ringtail possums in a range of residential gardens in two cities is consistent with recent evidence of their flexibility in habitat and resource use (Bader et al., 2019; Mathieson et al., 2020; Van Helden et al., 2018). This suggests that gardens may not only offer a conservation opportunity for common mammal species that are adapted to urban environments, but also for some of the most threatened mammals providing they are capable of flexibility in resource use and behaviour (Lowry et al., 2013; Sol et al., 2013).

Both behavioural traits (e.g. temperament, behavioural plasticity; Baker and Harris, 2007; Lowry et al., 2013) and environmental factors (e.g. floristic diversity, vegetation structure; see Goddard et al., 2010 for review) are known to influence the ability of animals to use urban habitats (Zungu et al., 2019). In this study, we focused on whether garden characteristics influenced mammal presence in residential gardens but acknowledge that other factors (e.g. supplementary feeding of wildlife) may also be influential (Fuller et al., 2008). We found that garden characteristics predicting mammal presence varied among species

which is consistent with other studies that demonstrate factors affecting presence of fauna is often species-specific (e.g. Baker and Harris, 2007; Daniels and Kirkpatrick, 2006). Our finding that garden attributes predicting the presence of brushtail possums and bandicoots differed between our study regions also suggests that important garden features cannot be generalised among cities and that additional factors are also likely to be influencing mammal presence. These factors may include natural variation in abundance of animal populations, age of city development, climatic variations and differences in urban matrix arrangement between cities; however these are yet to be investigated.

Vegetation cover was an important garden attribute for all studied mammals and corresponded to their natural habitat dependencies; the presence of both possum species was positively associated with tree cover in at least one city, whilst shrub cover was associated with bandicoot presence in both cities. This finding is congruent with the arboreal tendencies of the two possum species and ground-dwelling nature of the bandicoot (IUCN, 2019), as well as other studies that demonstrate vegetation cover is important for mammals in gardens (e.g. Oprea et al., 2009; Patterson et al., 2017; Threlfall et al., 2016). It is unclear why open space and tree cover negatively interacted with shrub cover in influencing bandicoot presence in the South Coast Region, but could be due to reduced detectability of bandicoots in heavily vegetated habitats.

Whilst all three species are vulnerable to predation from cats and dogs, it was interesting that the presence of pets in gardens only negatively affected bandicoot presence in gardens with high shrub cover (in one region). This result is somewhat counterintuitive as well vegetated ground cover may be expected to reduce the effect of pets by offering a ground-dwelling mammal an opportunity to escape predation. We suggest this effect is because bandicoots most commonly occur in gardens with high shrub cover and the presence of pets only has influence (either through predation or deterrence; Holderness-Roddam and McQuillan, 2014) on this species in these garden types. Although both possum species use ground habitats (Statham and Statham, 1997; Van Helden et al., 2019), their preference of above-ground structures such as trees, rooflines or fences may reduce their exposure and vulnerability to pets. In the Swan Coastal Plain, brushtail possums were more likely to occur on larger properties. Whether this is because larger properties reduce exposure to human-wildlife interactions that may deter the possum from the garden is unclear.

Our finding that proximity to remnant bushland had no association with mammal presence is surprising as previous studies demonstrate that the presence of mammals generally increases

with decreasing distance to natural habitats (e.g. Baker and Harris, 2007; Cavia et al., 2009). However, these studies are largely conducted in large cities that cover an extensive area of land, often have few natural bushlands, and are inhabited by millions of people (Łopucki and Kitowski, 2017). In contrast, our study cities are small in land area, have multiple small bushland remnants and are inhabited by approximately 35 000 individuals in low density living. Consequently, our finding that remnant bushland had no association with mammal presence may only occur in small cities, where green corridors that link natural non-urban vegetation with city greenspaces are more common (Łopucki and Kitowski, 2017). Whilst naturally vegetated areas are undoubtedly critical, at least in small cities, our results suggests that gardens, regardless of whether they contain native vegetation, may also provide significant habitat in their own right for some mammals.

We have highlighted that residential gardens are capable of supporting small to medium-sized mammal species, including those that are threatened, and should not be overlooked when considering the role that urban greenspaces and the general public can play in biodiversity conservation. Our results suggest that vegetation in the form of garden attributes could be manipulated by private landholders to encourage garden use by mammal species, however garden features that are important cannot be generalised among species or cities. To successfully exploit the opportunity that residential gardens can offer for native fauna conservation, species-specific knowledge of the features that encourage use by native animals, increased community awareness and participation in ‘wildlife-friendly’ gardening is required. This could be realised through planning regulations, government incentives or community group-led initiatives that promote and encourage wildlife-friendly gardens (Goddard et al., 2010). By including gardens in planning and management actions aimed at conserving urban wildlife, residential gardens offer additional habitat to parks, roadside vegetation, urban bushland and other urban greenspaces, and can play a significant role in biodiversity conservation.

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CHAPTER 3

AN UNDERRATED HABITAT: RESIDENTIAL GARDENS SUPPORT SIMILAR MAMMAL ASSEMBLAGES TO URBAN REMNANT VEGETATION



Photo: The native bush rat (*Rattus fuscipes*) captured in both remnant vegetation and residential gardens within the City of Albany.

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Abstract

Urban environments offer substantial opportunities for wildlife conservation as they commonly overlap the distribution of native species and comprise structurally complex and diverse habitats. Conservation strategies within urban environments currently focus on enhancing or maintaining natural remnant vegetation despite the variety of smaller greenspaces that are known to support wildlife. Residential gardens in particular comprise a major component of the total greenspace in urban areas and offer numerous resources that can be used by multiple taxa, and may therefore represent an important, yet unrealised opportunity for conservation. In this study we aimed to further the understanding of the potential value of gardens for biodiversity conservation by comparing the diversity, presence, abundance and reproductive activity of mammals in gardens to natural urban bushland remnants. We demonstrate that gardens support a similar diversity of native mammals as urban bushland and that the presence, abundance and reproductive activity for most native mammals was similar in both bushland and gardens. With the exception of domestic pets, the presence and abundance of introduced species was not higher in gardens compared to urban bushland. Our study highlights that the potential role of residential gardens for biodiversity conservation should not be overlooked. Given that novel urban ecosystems will continue to expand, inclusion of gardens in wildlife conservation and management initiatives is likely to greatly increase our ability to conserve wildlife in urban habitats.

Introduction

The destruction and fragmentation of natural habitats as human populations expand is one of the primary drivers of worldwide biodiversity loss (Fahrig, 2003). The resulting landscape is a mosaic of remnant vegetation surrounded by novel ecosystems; systems that function ecologically, but have been irreversibly altered by humans from their natural state (Hallett et al., 2013; Hobbs et al., 2006). Whilst novel ecosystems such as cities and agricultural land create substantial challenges for native wildlife including altered abiotic conditions (e.g. habitat loss, anthropogenic disturbances such as light and noise, presence of physical barriers) and changed community compositions (e.g. species invasions including feral predators) (Hallett et al., 2013; Hobbs et al., 2006), the distribution of native species overlaps with these ecosystems (e.g. Ives et al., 2016; Quine and Humphrey, 2010). Some animals can even prosper in these environments (e.g. Hubert et al., 2011; Ives et al., 2016; Maclagan et al., 2018), particularly if they are food or habitat generalists, behaviourally flexible, small in body size (< 10 kg), have large litter sizes, and display bold temperaments (Baker and Harris, 2007; Lowry et al., 2013; Santini et al., 2019). As novel ecosystems continue to proliferate, the role that these areas can play for biodiversity conservation is rapidly gaining interest (Daily, 2001; Ellis et al., 2010; Hobbs et al., 2006; Magle et al., 2012).

Urban environments are one novel ecosystem that arguably offer the greatest opportunity for animal conservation because cities are often located in areas of high biodiversity (Luck, 2007), overlap the distribution of many native species (Ives et al., 2016; Kowarik and von der Lippe, 2018), and can provide a structurally complex and diverse array of habitats (Aronson et al., 2017). These habitats vary in naturalness and include remnant patches of native vegetation, private gardens, street vegetation, green roofs, urban waste-lands (e.g. tips and vacant blocks) and parklands (Aronson et al., 2017; Cilliers et al., 2013; Lepczyk et al., 2017). Despite recent reviews that stress the biodiversity value of smaller greenspaces (Goddard et al., 2010; Soanes et al., 2019), research and management continue to emphasise the importance of natural remnant vegetation for conservation (Tulloch et al., 2016) because wildlife populations are assumed less likely to succeed in extensively human-dominated greenspaces (Lawson et al., 2008; Maclagan et al., 2018). This approach overlooks other greenspaces and resources in the urban landscape such as buildings, gardens, parks and street vegetation, also known to support biodiversity, and in doing so may inadvertently ignore important conservation opportunities that these other areas offer.

Residential gardens comprise a major component of the total greenspace (as high as 86%) in urbanised landscapes in many countries (Goddard et al., 2010; González-García and Sal, 2008; Loram et al., 2007; Mathieu et al., 2007). Residential gardens can provide more stable and greater densities of food resources than natural habitat, offer numerous artificial shelter sites (e.g. roof cavities and under-house spaces), and can act as a refuge from the diversity of predators in natural areas (Lowry et al., 2013; Rodewald et al., 2011; Sol et al., 2013). For example, a study in the United Kingdom estimated that residential gardens provided wildlife with 432 964 ha of habitat, ~12 million sources of supplementary food, ~5 million nest boxes and 349 ha of standing water (Davies et al., 2009). Which species can use these resources and therefore occur (and co-occur) in these highly modified habitats is largely dependent on the novel environmental features of gardens which filter (i.e. select for/against) the biological and ecological traits (e.g. mobility, behaviour, resource requirements) of wildlife (Fournier et al., 2020). For instance, the dispersal of non-flying animals into residential landscapes from natural areas is constrained by barriers that physically block or slow movement (e.g. roads, buildings, fences; Bergey, 2019; Rondinini and Doncaster, 2002), whilst the dispersal of flying animals is generally not. Even with this filtering process, numerous taxa are capable of utilising garden resources including examples of mammals (Van Helden et al., 2020); birds (Lerman and Warren, 2011; Threlfall et al., 2016); reptiles (González-García et al., 2009) and invertebrates (Jones and Leather, 2012; Threlfall et al., 2017).

Despite this growing body of evidence that demonstrates a wide-range of taxa occur in residential areas, large, mostly 'natural' greenspaces such as urban remnant vegetation are still perceived as the most significant habitat for wildlife conservation within urban landscapes (Tulloch et al., 2016). This bias is reflected in urban conservation research where, compared to other greenspaces, relatively few studies (~10%, N = 546) have focused on the role of residential gardens for animal conservation globally, and have predominantly concentrated on birds, invertebrates and plants (Shwartz et al., 2014). Of this body of literature, even fewer explore the value of gardens or other highly modified urban habitats relative to natural urban remnant vegetation, even though recent reviews and investigations stress the importance of these types of studies for biodiversity conservation (e.g. Maclagan et al., 2018; Shwartz et al., 2014). The few notable exceptions that have (see for example Basham et al., 2011; Maclagan et al., 2018), have generally focused on a single species or group (e.g. bats), but demonstrate that highly modified urban habitats can be comparable or even higher in value compared to remnant vegetation for at least certain species. Whether an

individual garden (or collective of gardens) has conservation value for multiple species and taxa with different biological and ecological attributes at the same time is still yet to be determined.

In this study we examined whether residential gardens have conservation value for a suite of species (with different biological and ecological traits) by comparing the diversity, presence, abundance and reproductive activity of wildlife in gardens to urban remnant vegetation. We explore this question using terrestrial mammals as a case study taxon because their variety of movement capabilities (e.g. arboreal, flying, ground-dwelling) and resource requirements (e.g. folivorous, nectivorous, insectivorous) may influence their presence in gardens (Fournier et al., 2020), and because they are poorly represented in urban conservation research (Shwartz et al., 2014). We predicted that the presence and abundance of mammals in gardens would be species-specific due to their different biological and ecological attributes (e.g. movement capabilities, habitat requirements and ability to tolerate urban disturbances). More specifically, given the presence of domestic predators (dogs (*Canis familiaris*) and cats (*Felis catus*)) and numerous physical barriers to ground movement in gardens (e.g. fences and buildings), we hypothesised that ground-dwelling mammals would be less frequently present and less abundant in gardens than in bushland sites. In contrast, we hypothesised that arboreal and flying mammals would be as equally present and abundant in gardens compared to bushland because of their above-ground movement capabilities and resource requirements. As gardens are one of the most extensive habitats on offer in the urban landscape, they may represent a substantial opportunity to conserve wildlife. Therefore, knowledge that furthers our understanding of the value of gardens for biodiversity is critical for the development of successful conservation strategies in urban landscapes.

Methods

Study area and study sites

This study was conducted within the urban confines (~90 km²) of the City of Albany (~4100 km²); southwest Australia (35° 01' S, 117° 53' E). We defined the 'urban confines' as any combination of land use categories identified by the City of Albany as general industry, highway commercial, hotel/motel, light industry, local and neighbourhood centre, port industry, rail, regional centre (including mixed business and mixed use) and residential (special or tourist) (Busschots et al., 2020). Situated within an agricultural landscape, Albany supports a population of approximately 31 500 people in 12 800 private dwellings and has a

population density of approximately 3.5 persons per ha (or an average of 10.5 persons per hectare when excluding land uses other than residential areas) (ABS, 2016; CoA, 2018). Although the city is rapidly expanding, there are numerous bushland remnants of varying topography, vegetation composition and disturbance within the urban landscape (between < 1 ha and 600 ha; Busschots et al., 2020), and a number of larger national parks surrounding the city boundary (i.e. within 40 km of the city center). Within these bushland environments a total of 27 native and eight introduced terrestrial mammals occur (Van Dyck et al., 2013; DBCA, 2019), of which many are known to also use residential areas (e.g. Tidemann and Flavel, 1987; Van Helden et al., 2020; Van Helden et al., 2019). Five of these native mammals are arboreal, nine are flying and 13 are ground-dwelling whilst all introduced species are ground-dwelling. This composition of mammals, comprised of species with varying movement strategies and resource requirements, made the City of Albany an ideal urban environment to test the research questions and hypotheses posed in our study.

We surveyed 52 gardens and 20 bushland remnants within urban Albany for the presence of terrestrial mammal species (Fig. 3.1). Garden sites (≤ 0.25 ha) were randomly selected from a list of residents willing to allow researchers access to their property (see Van Helden et al. 2020). Vegetation composition and structure of garden sites was highly variable, comprising both native and ornamental species (native vegetation ranged from 0 to 90%) and variable canopy cover (0 to 60%) (Van Helden et al., in press). One 0.04 ha quadrat (i.e. a typical small residential block size) was randomly located within each of the 20 bushland remnants. Bushland remnants varied in size (1.2 to 260 ha), vegetation composition (e.g. sheoak (*Allocasuarina fraseriana*), marri (*Corymbia calophylla*), jarrah (*Eucalyptus marginata*) or peppermint (*Agonis flexuosa*) woodlands) and structure (e.g. canopy cover percent ranged between 0 and 90%). Because preliminary analysis indicated that proximity to remnant bushland had no influence on the presence or diversity of mammals in gardens (see *Data analysis*) we chose to randomly select a subsample of bushland sites and garden sites with the intention of comparing an average, typical garden to an average, typical bushland site. All sites were located within the urban landscape of Albany City and were separated by a minimum distance of 100 m from other surveyed sites to ensure trappable animals (i.e. bush rats and bandicoots, for which long-range movements are rare, Fitzgibbon et al., 2007; Peakall et al., 2006) were unlikely to move between sites within one night.

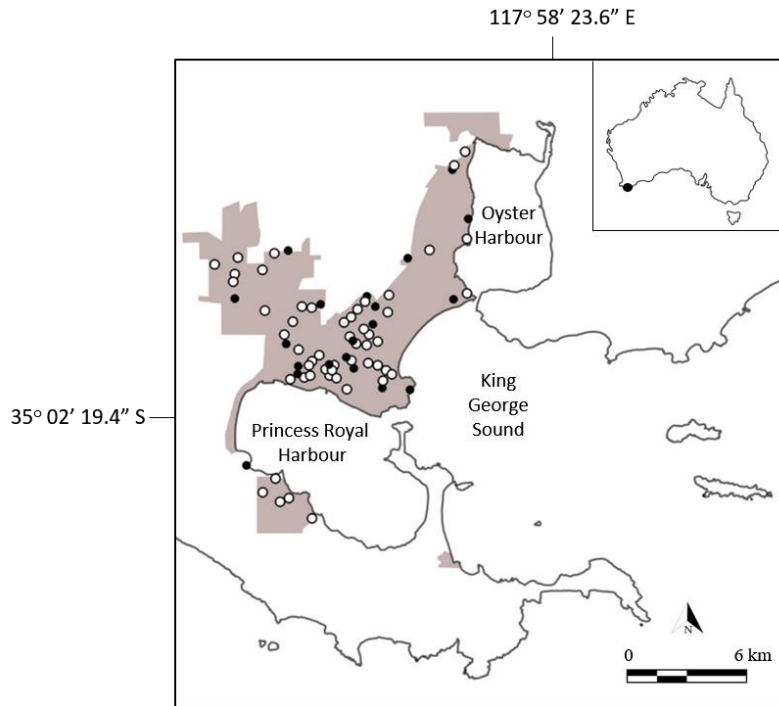


Fig. 3.1. The location of the 20 bushland (black) and 52 garden (white) study sites within the urban area (light grey) of the City of Albany in southwest Australia.

Mammal surveys

All 72 sites were surveyed for the presence of terrestrial mammals in autumn (April/May 2018), winter (July/August 2018), spring (October/November 2018) and summer (January/February 2019). We used a combination of trapping, spotlighting and acoustic methods to increase the likelihood of detecting a representative record of the mammal assemblage. We assumed no difference in the detectability of animals in bush and garden sites for bats (acoustic detection) and ground-dwelling mammals (equal baited trap effort), and implemented an exhaustive spotlight search approach (see [Spotlight surveys](#)) to reduce the potential of differing detectability of arboreal species among survey sites. It was reasonable to assume that detectability of arboreal mammals was equal between garden and bushland sites given that the bushland vegetation in Albany is generally low in height (< 15 m) with a relatively sparsely vegetated canopy, allowing a thorough search of all canopy habitat in both site types. In each season, every site was surveyed using each method on three nights within one week (12 nights total over four seasons for each site). Observational surveys (see [Acoustic surveys](#) and [Spotlight surveys](#)) were conducted on different nights to trapping surveys (see [Camera trapping](#) and [Trapping](#)) to ensure presence of researchers in the field did not impact upon trap success. Up to 12 sites were surveyed concurrently (i.e. on

each night), allowing all 72 sites to be sampled within a six week period in each season. The survey order of sites was randomised in each season to minimise temporal and spatial effects on the presence and abundance of mammals. Trapping at one bushland site was discontinued after the first sampling event due to evidence of trap tampering, but spotlighting and acoustic surveys were continued. This resulted in different sample sizes for the methods used.

Acoustic surveys

We determined the presence of bats at all sites using an Echo Meter Touch 2 Handheld Detector (Wildlife Acoustics Inc., Maynard, USA) for 5 min on each sampling night. Sonograms of recorded echolocation calls were analysed with Kaleidoscope Pro Analysis Software (Wildlife Acoustics Inc., Maynard, USA) to discriminate bat species based on previously described call features including shape, and frequency peak, start and finish (Titley Scientific, 2009). Call sequences that could not be confidently assigned to a species (e.g. too short, faint call phrase, feeding buzz) were removed from analyses (7% of call sequences, $N = 787$). Unidentifiable call sequences were present in both garden and bushland sites suggesting that habitat type had little influence on the quality of recordings. Three *Nyctophilus* spp. that potentially occur in the Albany area could not be differentiated based on call sequences (Titley Scientific, 2009), and were therefore grouped as one taxon (i.e. *Nyctophilus* spp.).

Spotlight surveys

All sites were searched by the same observer between last light and midnight with a 350 lumen spotlight at a standardised search rate of 5 min per 400 m². This allowed the area to be exhaustively searched whilst reducing the likelihood that individuals could move into or out of the area. We recorded the species and number of individuals seen for each sampling event.

Camera trapping

We deployed two passive infrared motion-sensing cameras (ScoutGuard 560K-18M HD; ScoutGuard Australia, Queensland, Australia) without a lure at all garden sites and 19 of the 20 bushland sites. Cameras were programmed to take three successive photos when triggered on medium sensitivity between 14:00 and 10:00 h. Cameras were mounted on stakes and positioned 40 cm above the ground with a minimum of 3 m between the camera and nearest obstruction within the field of view. All photos were examined and all species present were recorded. All photos where species identification was not possible (e.g. obscured view of animal or animal too far away for visual identification) were removed from analyses.

Trapping

At all garden sites and 19 of the 20 bushland sites we strategically deployed two medium-sized Sheffield cage traps (220 × 220 × 450 mm) and two Elliot traps baited with universal bait (peanut butter, sardines and oats) in shaded and weather-protected areas of the sites. Where possible (i.e. not rock underground), one pitfall trap was also deployed with 1.5 m of drift fence either side of the pitfall. This resulted in the installation of a pitfall trap at 48 garden sites and 11 bushland sites. All traps were placed at least 10 m apart and opened within 3 h prior to sunset and cleared the following morning within 4 h of sunrise. All animals captured were identified to species, sexed, assigned to a reproductive status if female (e.g. gravid or non-gravid for female rodents, or presence of pouch young for female marsupials) and temporarily marked on the tail with non-toxic pen ink to identify recaptures within a season before being released at their point of capture.

Data analysis

We compared the diversity, presence, abundance and reproductive activity of mammals between bushland and garden sites in *R* 3.6.1 (R Core Team, 2019) with the package ‘lme4’ (Bates et al., 2014). All *P*-values less than 0.05 were considered statistically significant. Prior to analyses we considered if proximity of bushland to garden sites was a confounding factor that influenced the presence and diversity of mammals in gardens (and therefore the random selection of garden and bushland sites, see *Mammal surveys*). For all species that were captured in gardens we assessed if distance to nearest bushland remnant predicted their presence in gardens using a binomial generalised linear model (GLM). We also examined if distance to nearest bushland remnant predicted species richness within gardens using a Poisson GLM. As distance to nearest bushland remnant had no effect on the presence of any species or species richness ($P > 0.05$) we were confident that this landscape feature was not a confounding effect and therefore did not consider it in further analyses.

We used species richness (i.e. number of species) calculated for each site over the entire sampling period to determine the diversity of mammals. This metric of diversity was used as the inability to derive robust abundance estimates from camera traps made other measures of diversity such as the Shannon-Wiener Diversity Index (Shannon, 1948) unsuitable. We used data collected from all survey methods, but excluded sites without pitfall traps (nine bushland sites and four gardens sites) from this analysis as these sites had reduced mammal detection effort. Species richness of native and introduced mammals, and the proportion of mammal

species that were native were calculated for gardens and bushland sites, and compared using quasi-Poisson generalised linear models due to evidence of over and under dispersion. We used Analysis of Similarity (ANOSIM) in PRIMER V. 7 (Clarke and Gorley, 2015) based on Bray-Curtis distances to evaluate if the assemblage of mammal species differed between gardens and bushland for all species, and for only native species. When assemblage differences were detected we conducted Similarity Percentages Analysis (SIMPER) in PRIMER V. 7 (Clarke and Gorley, 2015) to identify the species contributing most to the dissimilarity in assemblage between site types.

Mammal presence was determined as the proportion of sites at which a species or group of species (i.e. arboreal, flying, and ground-dwelling) was detected. Data from acoustic surveys for flying mammals, spotlight surveys for arboreal mammals (except the honey possum (*Tarsipes rostratus*) where only pitfall trap data were used), and trapping and camera trapping for ground-dwelling and introduced mammals were used. The presence of individual species and mammal groups were first compared between bushland and gardens, and among seasons using a Fisher's exact test. For species or mammal groups where presence varied among seasons, post-hoc Mann-Whitney tests with Bonferroni corrections were used to compare presence between gardens and bushland in each season. For species or mammal groups where presence did not vary among seasons, the data was pooled across seasons and the proportion of sites was compared using a Fisher's exact test. We also evaluated the consistency of site use by species by calculating the proportion of sites where a species was detected in all four seasons. We compared consistency of detection between bushland and gardens for each species using a Fisher's exact test. Comparisons of consistency in site use was possible for only eight of the detected mammal species as the remaining species were not detected in all four seasons at either bushland or garden sites.

Trap success was used as a measure of relative abundance for ground-dwelling animals. This method is a well-known and widely used estimator of relative abundance (Seber, 1973).

Recaptures of individual animals within seasons was low (9%, $N = 464$) and thus were not excluded from the data when estimating trap success. We used the mean count of individuals from the three survey nights to calculate the density of arboreal species at each site for each season. Only data collected from spotlight surveys and trapping surveys (not including cameras) were used to estimate abundance of mammal species. The abundance of bat species could not be determined from acoustic surveys. We compared the abundance of five mammal species, that occurred in both garden and bush sites, between site types in each season using

quasi-Poisson generalised linear models (GLMs) due to evidence of over and under dispersion.

We compared the difference in reproductive activity between bushland and gardens for the southern brown bandicoot (*Isoodon fusciventer*) as all other animals could not be reliably assigned to a reproductive status (i.e. from observational surveys), or because captures were too few for the remaining species. Only data collected from trapping surveys (not cameras) were used to evaluate reproductive activity in bandicoots. We compared the proportion of female bandicoots with pouch young among seasons using a Fisher's exact test which indicated season had no effect on reproductive status. For this reason we then pooled the data across seasons and compared the difference in reproductive activity of bandicoots between bushland and gardens using a proportion test.

Results

Mammal assemblage and richness

A total of 21 mammal species were detected during the sampling period of which 14 were native species (Table 3.1). Of the native species recorded, three were arboreal, seven were flying and four were ground-dwelling (Table 3.1). All seven introduced mammals were ground-dwelling (Table 3.1). These total numbers represented 56% of native mammals ($N = 25$) and 88% of introduced mammals ($N = 8$) known to occur in natural, non-urban environments (when *Nyctophilus* spp. counted as one species) within 40 km of the City of Albany (Van Dyck et al., 2013; DBCA, 2019). Gardens contained 44% ($N = 25$) of the native mammals known to occur in natural habitats, whilst urban bushland remnants contained 48% ($N = 25$). Sixty-four percent of native mammal species were detected in both bushland and garden sites whereas all introduced species were detected in both site types (Table 3.1). There was no difference between gardens and bushland in terms of the species richness of native mammals (t-value = -1.74 , $P = 0.088$, quasi-Poisson GLM; Fig. 3.2a), introduced mammals (t-value = 0.46 , $P = 0.65$, quasi-Poisson GLM; Fig. 3.2b), or the proportion of species that were native (t-value = -1.23 , $P = 0.229$, quasi-Poisson GLM; Fig. 3.2c).

Table 3.1. Mammal species detected (Y) and not detected (N) in bushland and gardens during surveys conducted between April 2018 and February 2019 in the City of Albany. Mammal groups abbreviated as: arboreal (A), flying (F), and ground-dwelling (G). * Grey-bellied dunnart formerly *Sminthopsis griseoventer*.

Scientific name	Common name	Group	Bush	Garden
<u>Native</u>				
<i>Austronomus australis</i>	White-striped free-tailed bat	F	Y	Y
<i>Chalinolobus gouldii</i>	Gould's wattled bat	F	Y	Y
<i>Chalinolobus morio</i>	Chocolate wattled bat	F	Y	Y
<i>Falsistrellus mackenziei</i>	Western false pipistrelle	F	Y	Y
<i>Isodon fusciventer</i>	Southern brown bandicoot	G	Y	Y
<i>Macropus fuliginosus</i>	Western grey kangaroo	G	Y	N
<i>Nyctophilus</i> spp.	Long-eared bats	F	Y	Y
<i>Ozimops kitcheneri</i>	South-western free-tailed bat	F	N	Y
<i>Pseudocheirus occidentalis</i>	Western ringtail possum	A	Y	Y
<i>Rattus fuscipes</i>	Western bush rat	G	Y	Y
<i>Sminthopsis fuliginosus</i> *	Grey-bellied dunnart	G	Y	N
<i>Tarsipes rostratus</i>	Honey possum	A	Y	N
<i>Trichosurus vulpecula hypoleucus</i>	Common brushtail possum	A	N	Y
<i>Vespadelus regulus</i>	Southern forest bat	F	Y	Y
<u>Introduced</u>				
<i>Canis familiaris</i>	Dog	G	Y	Y
<i>Felis catus</i>	Cat	G	Y	Y
<i>Mus musculus</i>	House mouse	G	Y	Y
<i>Oryctolagus cuniculus</i>	Rabbit	G	Y	Y
<i>Rattus norvegicus</i>	Brown rat	G	Y	Y
<i>Rattus rattus</i>	Black rat	G	Y	Y
<i>Vulpes vulpes</i>	Fox	G	Y	Y

The assemblage of native mammal species did not differ between gardens and bushland ($R = 0.02$, $P = 0.401$, ANOSIM). However, the mammal assemblage differed between site type when introduced species were included (50% dissimilarity, $R = 0.30$, $P = 0.001$, ANOSIM). Domestic pets (cats and dogs) and black rats (*Rattus rattus*) were the introduced species that contributed most to dissimilarity in species assemblage between gardens and bushland (10.5%, 8.5% and 9.8% respectively); unsurprisingly, domestic pets were more commonly detected at garden sites whilst the black rat was more commonly detected at bushland sites (Fig. 3.4).

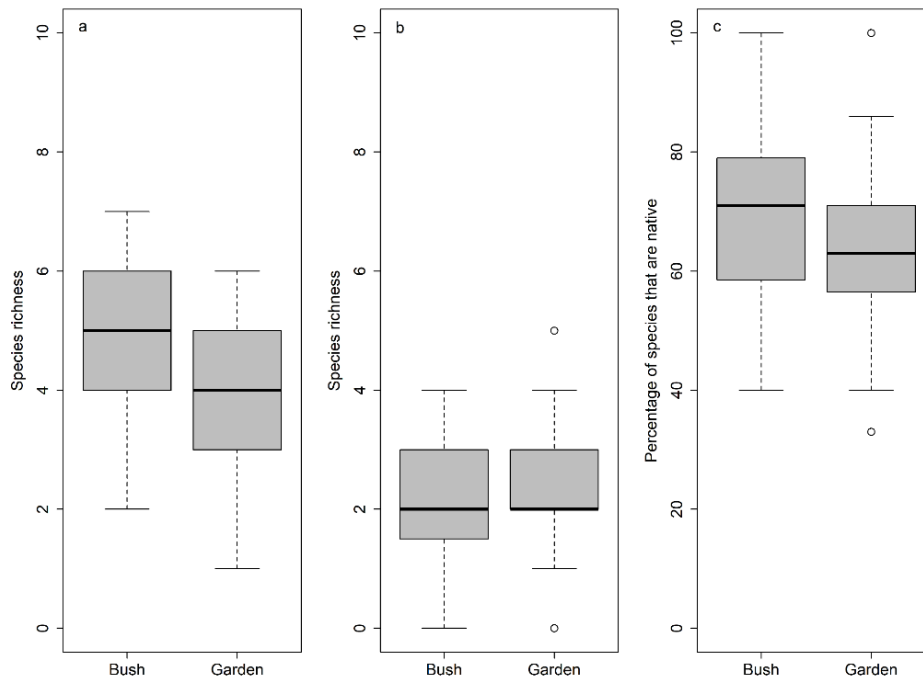


Fig. 3.2. Native mammal richness (a), introduced mammal richness (b), and the proportion of mammal species that were native (c) recorded in bushland and gardens between April 2018 and February 2019 in the City of Albany. The grey boxes represent the interquartile range (IQR), the black lines represent medians in box plots, and the whiskers represent the maximum and minimum value within $1.5 \times$ IQR of the upper and lower quartile, respectively. Values outside of these maximum and minimum values are denoted with \circ .

Presence of mammal groups

All mammal groups (arboreal, flying, ground-dwelling and introduced species) were recorded from at least 50% of both bushland and garden sites (Fig. 3.3). Season had no influence on the presence of any mammal group ($P > 0.05$, Fisher's exact test). Native ground-dwelling mammals was the only group where presence varied between site type, with these more commonly detected in bushland compared to gardens ($P = 0.012$, Fisher's exact test; Fig. 3.3).

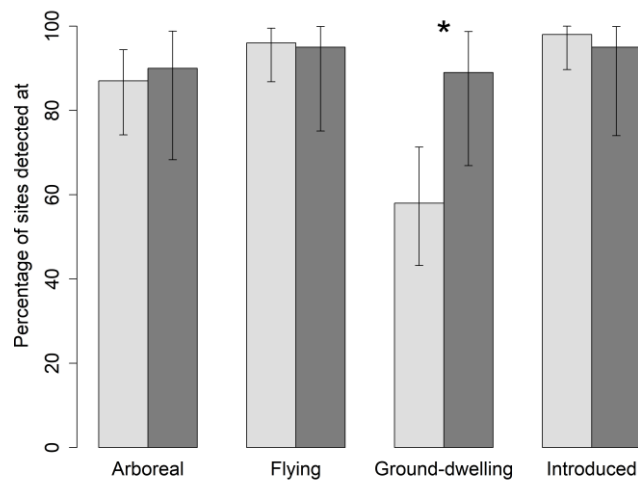


Fig. 3.3. Proportion of garden sites (light grey) and bushland sites (dark grey) where native arboreal, native flying, native ground-dwelling and introduced ground-dwelling mammals were present based on surveys undertaken between April 2018 and February 2019 in the City of Albany. Significant differences ($P < 0.05$) between site type are denoted by '*'. Whiskers on bars represent Clopper-Pearson 95% confidence intervals.

Presence of individual mammal species

Three species of bats, the southern brown bandicoot and the western ringtail possum (*Pseudocheirus occidentalis*) were commonly detected and were present in over half of all surveyed bushlands and gardens (Fig. 3.4). All native arboreal and most native flying species were present in garden sites as frequently as in bushland sites ($P > 0.05$, Fisher's exact test; Fig. 3.4a, b). The Long-eared bat (*Nyctophilus* spp.) was the only flying mammal present more frequently in bushland sites (25% of sites, $N = 20$) than in gardens sites (4% of sites, $N = 52$) ($P = 0.015$, Fisher's exact test; Fig. 3.4b). Except for the southern brown bandicoot which was found as commonly in gardens (54% of sites, $N = 52$) as in bushland (79% of sites, $N = 20$) ($P = 0.062$; Fisher's exact test), the remaining three native ground-dwelling species were present in more bushland sites than garden sites ($P < 0.05$, Fisher's exact test; Fig. 3.4c).

The presence of native mammal species did not vary among seasons ($P > 0.05$; Fisher's exact test) with the exception of the chocolate wattled bat (*Chalinolobus morio*) ($P = 0.008$; Fisher's exact test). In autumn this species was only present in gardens (17% of gardens, $N = 52$), whereas in all other seasons this bat was present in 10 to 25% of bushland sites ($N = 20$) and in less than 6% of gardens ($N = 52$). Within each season, presence of the chocolate wattled bat did not differ between gardens and bushland ($P > 0.05$, Mann-Whitney U test).

Domestic pets (dogs and cats) were present at more garden sites than bush sites whilst the introduced black rat and fox (*Vulpes vulpes*) were present at more bush sites than garden sites ($P < 0.05$, Fisher's exact test; Fig. 3.4d). The presence of other introduced species did not differ between gardens and bushland ($P > 0.05$, Fig. 3.4d). The presence of introduced mammals did not vary among seasons for any species ($P > 0.05$; Fisher's exact test).

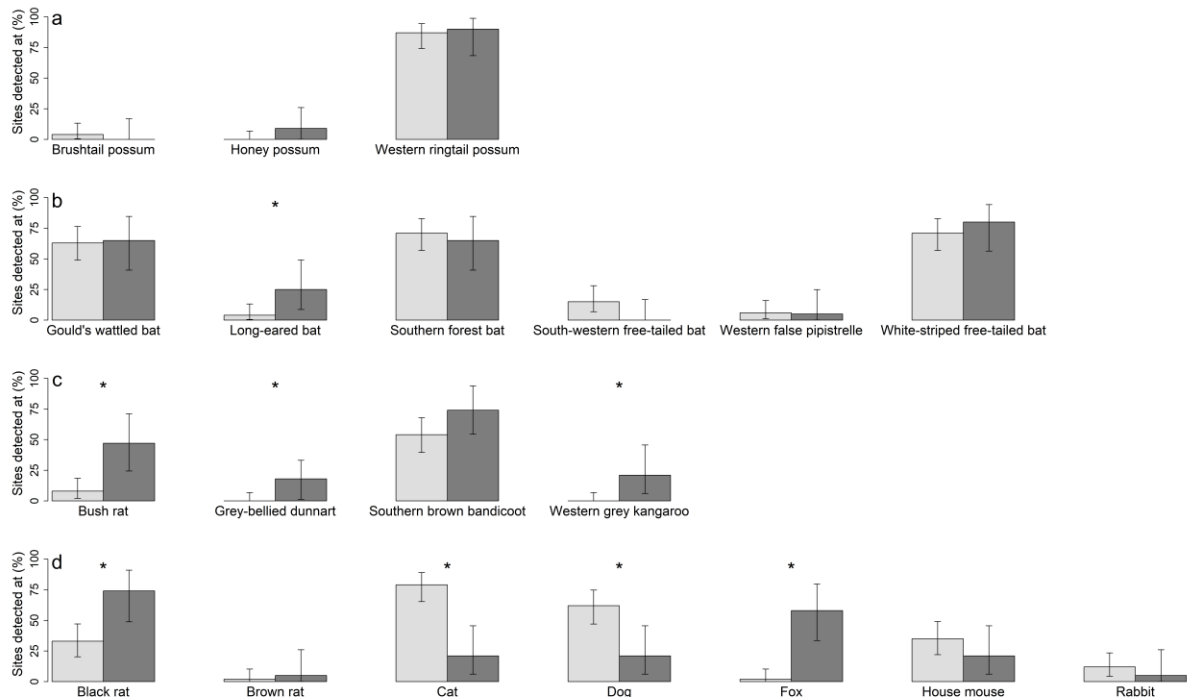


Fig. 3.4. The proportion of garden sites (light grey) and bushland sites (dark grey) at which native arboreal mammals (a), native flying mammals (b), native ground-dwelling mammals (c), and introduced ground-dwelling mammals (d) were present during surveys conducted between April 2018 and February 2019 in the City of Albany. The chocolate wattled bat is excluded from this figure as its presence at bushland and garden sites was influenced by the season of sampling. Significant differences ($P < 0.05$) between site type are denoted by ‘*’. Whiskers on bars represent Clopper-Pearson 95% confidence intervals.

Consistency of site use among seasons

Eight mammal species were present in all seasons at either bush or gardens (or both) sites (Table 3.2), although the percentage of sites at which native mammals were detected in all seasons was generally low ($< 50\%$ of sites; Table 3.2). Whilst the southern brown bandicoot and western ringtail possum were present in all seasons at both bush and garden sites, western bush rats (*Rattus fuscipes*) were present in all seasons only at bushland sites, and the southern forest bat (*Vespadelus regulus*) was present in all seasons only at garden sites (Table 3.2). Despite this, the small sample sizes (see Table 3.2) resulted in the finding that the proportion

of sites at which mammals were present in all seasons did not differ between bush and garden sites for any native species ($P > 0.05$, Fisher's exact test).

None of the introduced mammals were present in all seasons at bushland sites, however black rats, dogs, cats and rabbits (*Oryctolagus cuniculus*) were present in all seasons at garden sites (Table 3.2). Dogs were present in all seasons at garden sites more frequently than at bushland sites ($P = 0.012$; Fisher's exact test), whereas the presence of other introduced species in all seasons did not differ between site type ($P > 0.05$, Fisher's exact test; Table 3.2).

Table 3.2. The proportion of sites where mammal species were detected in all four seasons during surveys conducted between April 2018 and February 2019 in the City of Albany. Only mammals that were detected in all four seasons in at least one of the site types are displayed. ‘*N*’ represents the total number of sites at which a species was detected in at least one season.

Species	Garden sites (<i>N</i>)	Bush sites (<i>N</i>)
<u>Native</u>		
Western bush rat	0% (4)	56% (9)
Southern brown bandicoot	46% (28)	47% (15)
Southern forest bat	3% (37)	0% (13)
Western ringtail possum	36% (45)	22% (18)
<u>Introduced</u>		
Black rat	6% (17)	0% (14)
Cat	15% (41)	0% (4)
Dog	72% (32)	0% (4)
Rabbit	33% (6)	0% (1)

Abundance of mammal species

Abundance estimates in bushland and gardens for native species were possible for the western bush rat, southern brown bandicoot and western ringtail possum. The relative abundance of southern brown bandicoots did not differ between gardens and bush in any season ($P > 0.05$, quasi-Poisson GLM; Fig. 3.5a). Bush rats were detected only from bush sites in winter and summer, and their abundance was higher in bushland compared to gardens in autumn (t-value = -3.58 , $P < 0.001$, quasi-Poisson GLM) and spring (t-value = -4.52 , $P < 0.001$, quasi-Poisson GLM) (Fig. 3.5b). The abundance of western ringtail possums only differed between bush and garden sites in winter, during which their abundance was higher in bushland (t-value = -3.02 , $P = 0.004$, quasi-Poisson GLM; Fig. 3.5c).

Abundance estimates for introduced mammal species were possible for the introduced black rat and house mouse (*Mus musculus*). Relative abundance of black rats was significantly higher in bushland than in gardens in summer (t-value = -2.12 , $P = 0.038$, quasi-Poisson

GLM), but did not differ between site types in any other season (Fig. 3.5d). The abundance of house mice did not differ between gardens and bushland in any season ($P > 0.05$, quasi-Poisson GLM; Fig. 3.5e).

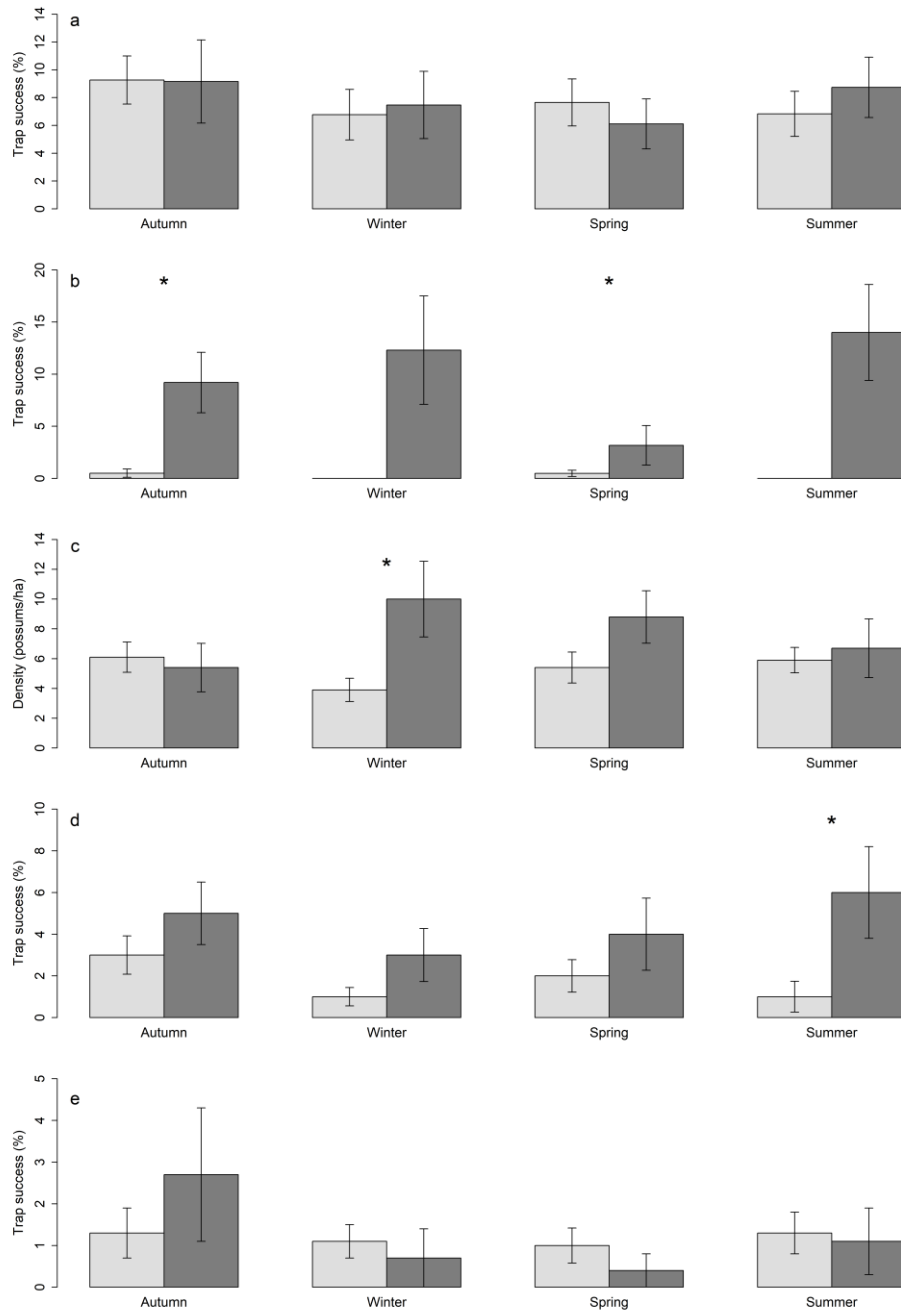


Fig. 3.5. Mean relative abundance (trap success) of southern brown bandicoots (a), mean relative abundance (trap success) of western bush rats (b), mean density of western ringtail possums (possums/ha) (c), mean relative abundance (trap success) of black rats (d), and mean relative abundance (trap success) of house mice (e) in garden sites (light grey) and bushland sites (dark grey) based on surveys conducted between April 2018 and February 2019 in the City of Albany. Significant differences ($P < 0.05$) between site type are denoted by '*'. Whiskers on bars represent the standard error of the mean.

Reproductive activity of southern brown bandicoots

A total of 116 female southern brown bandicoots were captured during the sampling period. Of the 82 individuals captured in gardens, 54% had pouch young compared to 38% ($N = 34$) captured in bushland. There was no evidence that presence of pouch young varied among seasons ($P = 0.413$, Fisher's Exact test) or between site type ($\chi = 0.54$, $P = 0.464$, proportion test). The median number of pouch young carried by female bandicoots was two in both bushland (Interquartile range = 0.5) and gardens (IQR = 2).

Discussion

There is a growing body of evidence demonstrating that urban landscapes offer substantial opportunities for biodiversity conservation (Aronson et al., 2017; Soanes and Lentini, 2019), yet gardens, the most extensive component of these environments, have been largely disregarded in urban conservation research (Shwartz et al., 2014). Our study is one of the first to evaluate the diversity, presence and abundance of a suite of mammal species in gardens relative to urban bushland; one novel urban habitat that's value has already been well established (Daily, 2001). We demonstrated that gardens supported a similar diversity and composition of native mammals as urban bushland and almost half of all native mammals found in natural areas of the study region. We also showed that the presence, abundance and reproductive activity of native mammals was similar between bushland and gardens for most native mammal species detected. Although the presence of domestic pets was higher in gardens, the presence and abundance of other introduced species that may compete with (e.g. black rat) or predate on (e.g. fox) native mammals was not higher in gardens compared to bushland. As hypothesised, native ground-dwelling mammals were generally found to be less abundant and less common in gardens than in bushland, albeit with some exceptions (e.g. the southern brown bandicoot). Our findings add to a growing list of wildlife, including birds (e.g. Daniels and Kirkpatrick, 2006), invertebrates (e.g. Goulson et al., 2002; Threlfall et al., 2017) and plants (e.g. Smith et al., 2006) that occur within residential gardens, and further develops this knowledge base by providing evidence that the diversity and abundance of some taxa (with differing ecological traits) can be similar between gardens and remnant vegetation.

Gardens supported a substantial proportion of native mammal diversity found in natural areas of the study region (44%), including threatened species such as the Critically Endangered western ringtail possum (*Biodiversity Conservation Act 2016*). This result is even more

noteworthy given that urban bushland sites contained only 48% of all native mammals, and natural, protected areas of the region, such as Two Peoples Bay Nature Reserve, contain only 68% (S. Comer unpublished data). This is consistent with natural spatial variations in species richness and species' distributions that commonly result in an incomplete representation of regional biodiversity in any given area (Vázquez and Gaston, 2004). Whether the mammal composition detected in gardens in this study is representative of the species composition that historically occurred in the region prior to urbanisation is unclear. Regardless, our results highlight gardens can contain a substantial proportion of current native diversity and thus, could make a significant contribution to mammal conservation. Our finding supports the ideas of Fischer and Lindenmayer (2006) who present the 'continuum model' of species distributions in human-modified landscapes. This model suggests that species' distributions are driven by gradients of environmental factors such as food and shelter, rather than by human-defined habitat types (e.g. gardens, remnant vegetation). Conservation initiatives that incorporate these ideas by considering both natural (i.e. remnants) and novel (e.g. residential) areas as an integrated habitat, will likely result in better conservation outcomes than those that continue to focus only on predefined 'natural' areas within urban landscapes.

Whilst our finding that multiple mammal species and groups are frequently detected in gardens supports recent evidence of considerable overlap of native species with cities (Ives et al., 2016; Van Helden et al., 2020), it contrasts with the commonly held perception that the value of gardens is greatest for urban adapted or 'synurbic' species; a term generally reserved for species populations that have higher densities in urban areas compared to natural areas (Francis and Chadwick, 2012). Interestingly, of the 11 native mammals we detected in gardens, only the brushtail possum (*Trichosurus vulpecula hypoleucus*) is widely coined an 'urban-adapted species' (e.g. Hubert et al., 2011), yet was one of the least commonly detected mammals in gardens (4% of garden sites), whilst a Critically Endangered possum, five species of bat and the southern brown bandicoot were detected as frequently in gardens as in bushland. This suggests that the potential value of gardens may not be limited to common mammal species that are adapted to urban environments, but could apply to a range of species, including those that are threatened, providing they are capable of some flexibility (as seen in the western ringtail possum, Bader et al., 2019; Mathieson et al., 2020) in resource use and behaviour (Lowry et al., 2013; Sol et al., 2013).

Ground-dwelling species were less likely to be detected in gardens than bushland sites in our study and this may be due to the greater presence of domestic pets (i.e. cats and dogs) which

predate and deter animals (Holderness-Roddam and McQuillan, 2014) or the presence of man-made structures that act as barriers to movement of ground-dwelling species (e.g. impermeable fences, buildings and roads) in residential areas (e.g. Rondinini and Doncaster, 2002). The flying nature of bats and preference of above-ground structures such as trees, roof-lines or fences for arboreal species likely reduces the influence of ground-based barriers and the species' vulnerability to pets. This potentially explains why their presence was not also lower in gardens. Interestingly, although ground-dwelling native species were generally more commonly detected in bushland and in higher abundances than in gardens, there was evidence of some exceptions to this trend (i.e. the southern brown bandicoot). Consistent with other studies on bandicoots that demonstrate its frequent occurrence and high abundance in gardens and other non-remnant urban habitats (Hillman et al., 2017; Maclagan et al., 2018), the southern brown bandicoot was detected as commonly and at similar abundances in gardens as in bushland. This suggests whilst gardens generally have lower abundances of ground-dwelling species compared to remnant habitats, there are species-specific responses to gardens that need to be considered and in some cases, gardens may still be useful habitat for these species despite the presence of pets.

Our study demonstrated that native mammals were detected in all seasons as frequently in gardens as in bushland, suggesting shelter, food or movement opportunities are available to mammal wildlife in gardens throughout the year. Whilst this finding is consistent with other studies which highlight that urban habitats can provide stable and abundant resources (Lowry et al., 2013), it is unclear from our study whether it is the same individuals that are present in all seasons, and therefore what their level of dependence is on these garden habitats. For example, individuals may reside within garden habitats all year round, access gardens to supplement resources obtained in natural habitat, or simply transit these habitats during movement (e.g. for dispersal or resource acquisition) (e.g. Caryl et al., 2013; Davison et al., 2009; Fitzgibbon et al., 2011). All of these types of use offer conservation opportunities within urban landscapes, particularly if wildlife is capable of long-term exclusive use of gardens. Which of these is the case for mammals found in our study is uncertain, but worthy of investigation to further examine the role of gardens for mammal conservation.

Although the diversity and presence of our study species in gardens was unrelated to proximity of remnant habitat, other landscape and local factors such as the degree of urbanisation (Fontaine et al., 2016), and the availability of shelter (Moretto and Francis, 2017) and food (Fontaine et al., 2016; González-García et al., 2009) can influence the

presence of wildlife in gardens. In light of our study and others that demonstrate the value of gardens for biodiversity conservation (e.g. Goddard et al., 2010; Soanes et al., 2019), further examination of the local factors that influence animal presence has merit. Such knowledge will allow residents that are both willing (e.g. Goddard et al., 2013; van Heezik et al., 2012) and capable of manipulating garden features (e.g. Daniels and Kirkpatrick, 2006; Goddard et al., 2010; Van Helden et al., 2020) to contribute conservation outcomes.

Lastly, one of the greatest concerns for wildlife in urban areas is that gardens and other novel habitats could represent ecological ‘traps’ and ‘sinks’, where although a species occurs and may even actively select that habitat, their survival and reproductive rates are too low to sustain population viability over time (Battin, 2004; Donovan and Thompson III, 2001). Whilst our paper does not directly address this concern, our finding that the reproductive activity of bandicoots did not differ between bushland and gardens is consistent with other papers focused on bandicoots in urban habitats (e.g. Hillman et al., 2017) and provides some optimism that gardens may be capable of supporting viable populations of at least some mammal species. Research that examines body condition, demographic features, survival and reproduction in garden habitats would significantly advance our understanding of the risks associated with animals using residential environments and the long-term viability of animal population in these areas.

Our study is one of the first to evaluate the conservation value of garden habitats relative to natural urban remnant vegetation for a suite of mammal species. We have demonstrated that the diversity, presence, abundance and reproductive activity of some taxa can be similar between gardens and remnant vegetation, confirming that gardens can provide an extensive habitat for many animals with differing ecological attributes, including those that are threatened. Knowledge of the conservation value of residential gardens will be further developed by studies that determine the mechanisms that influence the patterns of biodiversity in gardens, the habitat use of gardens (i.e. transient or resident use) and the long-term viability of animals in gardens. Current conservation approaches that focus on urban remnants at the expense of smaller greenspaces such as gardens, need rapid re-assessment. Given that novel urban ecosystems will continue to expand, inclusion of gardens in wildlife conservation and management actions is likely to greatly increase our ability to conserve wildlife in urban habitats.

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CHAPTER 4

MANAGING GARDENS FOR WILDLIFE: FEATURES THAT PREDICT MAMMAL PRESENCE AND ABUNDANCE IN GARDENS VARY SEASONALLY



Photo: Pitfall trap in a surveyed residential garden within the City of Albany.

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Abstract

Residential gardens in urban areas offer substantial opportunities for biodiversity conservation. To realise this opportunity, a comprehensive understanding of which species prosper in gardens, which mechanisms promote their use of these greenspaces and how these mechanisms can be manipulated by residents to achieve conservation outcomes is needed. Whilst substantial progress has been made in evaluating the mechanisms that influence the diversity, presence and abundance of wildlife in gardens, how these mechanisms vary seasonally is largely unknown. This study investigated whether the features influencing the presence and/or abundance of arboreal, flying and ground-dwelling mammals in gardens varied seasonally using acoustic, trapping and spotlighting surveys undertaken in four consecutive seasons (2018/19) in 52 gardens in southwest Australia. As animal behaviour and resource availability varies seasonally, we predicted that garden features influencing the presence and abundance of mammals in gardens would also vary seasonally. For arboreal and ground-dwelling mammals we found that some garden features (e.g. canopy cover and presence of dogs, respectively) were important in more than one season, however none were important in all seasons with most only being influential in one. No garden features predicted the presence of flying mammals in more than one season. Our results suggest that sampling in a single or few seasons may fail to identify features influencing wildlife at times other than in the season of sampling, or alternatively, may overemphasise factors that do not operate across all seasons. Studies that consider the mechanisms on a seasonal basis will provide a more comprehensive list of features that influence the presence and abundance of animals in gardens. We suggest it is both those features that support wildlife in multiple or all seasons, and those that support critical life history events in a single season that could contribute to conservation if manipulated by residents appropriately.

Introduction

There is growing recognition that urban landscapes offer substantial opportunities for biodiversity conservation (Aronson et al., 2017; Soanes and Lentini, 2019; Soanes et al., 2019). As global urban land cover is expected to triple between 2000 and 2030, converting over one million km² of land into urbanised areas (Seto et al., 2012), the ability to conserve wildlife within these areas is becoming increasingly important. These landscapes comprise an assortment of structurally complex, albeit novel, habitats that wildlife can utilise (Aronson et al., 2017) and can harbour numerous species, both common (e.g. Araújo, 2003) and threatened (e.g. Ives et al., 2016). These urban habitats, commonly termed urban greenspaces, can be both natural and artificial and include private gardens, green roofs, parklands, cemeteries, golf courses, street vegetation, urban waste-lands (e.g. rubbish tips and vacant blocks) and remnant patches of vegetation (Aronson et al., 2017; Cilliers et al., 2013; Lepczyk et al., 2017). Whilst remnant vegetation has traditionally been the focus of conservation initiatives in urban landscapes, the potential conservation value of smaller greenspaces such as private gardens and green roofs is rapidly gaining interest as these could be manipulated by residents to achieve conservation outcomes in urban areas (Belaire et al., 2014; Goddard et al., 2010; Soanes et al., 2019; Threlfall et al., 2016; 2017; Tulloch et al., 2016). The conservation benefits that wildlife-friendly gardeners can provide depends on a robust ecological knowledge of the garden features that support urban wildlife.

Residential gardens represent a major component of the total urban greenspace (as high as 86%) in both developing and developed countries (Goddard et al., 2010; González-García and Sal, 2008; Loram et al., 2007; Mathieu et al., 2007) and are used by numerous taxa including birds (Daniels and Kirkpatrick, 2006), mammals (Van Helden et al., 2020a; 2020b), reptiles (Koenig et al., 2001) and invertebrates (Ramírez-Restrepo et al., 2017). Investigation of (1) which species can prosper in gardens, (2) which factors or mechanisms promote their use of these greenspaces and (3) how these factors can be manipulated to achieve conservation outcomes in residential gardens will all contribute to a better understanding of the potential conservation value that these habitats offer for biodiversity.

Substantial progress has been made in evaluating the diversity, presence and abundance of wildlife within residential gardens (e.g. Bernholt et al., 2009; Fetridge et al., 2008; Van Helden et al., 2020b), and the mechanisms that drive these patterns (e.g. Baker et al., 2003; Basham et al., 2011; Daniels and Kirkpatrick, 2006; Fontaine et al., 2016; Threlfall et al., 2016; 2017; Van Helden et al., 2020a). Whilst high floral diversity and structural complexity

of garden vegetation are two commonly identified features thought to promote animal diversity and abundance in gardens (see Goddard et al., 2010 for review), numerous other mechanisms, that operate at both landscape and local scales, are known to influence patterns of biodiversity (e.g. Adams et al., 2020; Baker et al., 2003; Bergerot et al., 2011; Daniels and Kirkpatrick, 2006; Fontaine et al., 2016). Landscape factors such as the degree of urbanisation (Fontaine et al., 2016), the environmental suitability of the area (e.g. altitude, Daniels and Kirkpatrick, 2006) and proximity to remnant vegetation (i.e. accessibility of gardens) (Baker et al., 2003; Basham et al., 2011) can influence the presence of a species in a garden. Local scale factors such as the availability of shelter (Moretto and Francis, 2017), food and water (Fontaine et al., 2016; González-García et al., 2009), and an absence of predators (Baker et al., 2003; Van Helden et al., 2020a) can also influence the presence and abundance of species in gardens. The effects of these mechanisms and the interactions between them are complex, with the relative importance of factors varying among species (Daniels and Kirkpatrick, 2006; Fontaine et al., 2016; Threlfall et al., 2016) and even among cities for the same species (Van Helden et al., 2020a).

Despite the extensive and continually growing knowledge-base that illustrates that these mechanisms are complex, how these mechanisms vary seasonally is largely unknown (although see Adams et al., 2020 as an exception). To date, studies that have investigated the mechanisms driving patterns of biodiversity in gardens have generally sampled their target species in one season (e.g. Basham et al., 2011), or if they have sampled across multiple seasons, have either combined data for analysis or considered season only as an independent effect (e.g. Baker et al., 2003; Daniels and Kirkpatrick, 2006; Patterson et al., 2018). There are other examples where the season of data collection was not recorded (e.g. in the case of citizen-derived data, Baker and Harris, 2007; Belaire et al., 2014; González-García et al., 2009; Van Helden et al., 2020a). In all of these studies, how the relative importance of these mechanisms varies among seasons was not considered, despite that this information could provide great value to conservation if applied by wildlife-friendly gardeners.

The distribution, abundance and behaviour of animals is well known to vary seasonally (Dayton, 2008); food sources of animals often show significant temporal fluctuations (such as fruiting trees for monkeys, Patterson et al., 2017), animals have seasonal diets (e.g. grizzly bears, Nielsen et al., 2010), resource requirements differ during breeding seasons (e.g. birds, Unno, 2002) and activity patterns can vary with seasonal changes in climate (e.g. frogs become active after rain, Paltridge and Southgate, 2001). Given that the presence, abundance

and activity of animals vary seasonally, often in response to seasonal fluctuations in their resources (Dayton, 2008), it would be reasonable to assume that the factors that drive the abundance and presence in gardens may also vary seasonally. Whilst multiple studies provide promising evidence that garden features could be manipulated to improve conservation outcomes for fauna in gardens (e.g. Daniels and Kirkpatrick, 2006; Dique et al., 2004; Oprea et al., 2009; Patterson et al., 2017; Threlfall et al., 2016; 2017; Van Helden et al., 2020a), considering seasonality will allow for the identification of all of the features that influence the presence and abundance of animals, including those that are important in single-seasons and those that are consistent across seasons. This will better identify those features which could be manipulated to achieve the greatest conservation outcome and in which season garden manipulations will have the most benefit.

In this study, we examined which garden features promoted mammal presence and abundance and whether these varied among seasons for three broad mammal groups with different resource requirements (arboreal, flying and ground-dwelling). We used the City of Albany, southwest Australia, as a case study city because fauna of the region exhibit strong seasonality in their diet, behaviour, abundance and reproductive biology (e.g. Bradshaw et al., 2007; Hosken et al., 1998; Wayne et al., 2005) in response to a strongly seasonal Mediterranean climate. Whilst both local and landscape factors will undoubtedly play a role in determining the presence and abundance of animals in gardens, in this study we chose to focus on garden features (i.e. local factors) that promote mammal presence and abundance as these are the factors that could be modified by residents to achieve conservation outcomes in urban areas. Based on the seasonal variations in diet, behaviour and resource selection of animals in natural environments, we hypothesised that the garden features that influence the presence and abundance of mammals would vary seasonally. More specifically, we predicted that the importance of fruiting and seed-bearing food resources (e.g. fruiting trees and vegetable patches) would vary seasonally due to their seasonal changes in availability, and that water resources would be more influential in the drier seasons (summer) than wetter seasons (winter). In contrast, we hypothesised that the presence of predators (e.g. dogs and cats) and competitors (e.g. rats), the accessibility of gardens (e.g. fence permeability and distance to nearest bushland) and the resources that provide protection and shelter (e.g. canopy cover, understorey cover, roof cavities, under-house spaces) would be influential in multiple (if not all) seasons.

Methods

Study area and study sites

This study was conducted within the urban confines (~ 90 km²) of the City of Albany (~ 4100 km²), located in southwest Australia (34° 58' S, 117° 52' E) in a globally recognised biodiversity hotspot (Myers et al., 2000). This region is characterised by a Mediterranean climate, with cool wet winters (Jun–Aug) and hot dry summers (Dec–Feb), making it an ideal city to test whether the garden features that influence mammal assemblages vary seasonally. Urban Albany supports a population of approximately 31 500 people in 12 800 private dwellings (generally detached houses) and has a low population density of approximately 3.5 persons per ha in 60 suburbs (or an average of 10.5 persons per hectare when excluding land uses other than residential areas) (ABS, 2016; CoA, 2018). We defined the ‘urban confines’ of Albany as any combination of land use categories identified by the City of Albany as general industry, highway commercial, hotel/motel, light industry, local and neighbourhood centre, port industry, rail, regional centre (including mixed business and mixed use) and residential (special or tourist) (Busschots et al., 2020). Residential gardens in Albany usually consist of both a front (i.e. garden between the street and house) and back garden, which are highly variable in size, and vegetation composition and structure (Van Helden et al., 2020a). Although the city is rapidly expanding, there are numerous bushland remnants of varying topography, vegetation composition and disturbance within the urban landscape (between < 1 ha and 600 ha (Busschots et al., 2020; Van Helden et al., 2018)), and a number of larger national parks surrounding the city boundary (i.e. within 40 km of the city center). Numerous arboreal, flying and ground-dwelling mammals occur within both bushland and residential environments of urban Albany (Van Dyck et al., 2013; DBCA, 2019; Van Helden et al., 2020b).

We surveyed 52 gardens within urban Albany for the presence of terrestrial mammal species (Fig. 4.1). Garden sites (≤ 0.25 ha) were randomly selected from a list of residents willing to allow researchers access to their property (obtained from Van Helden et al., 2020a). All sites were separated by a minimum distance of 100 m from other surveyed sites to ensure spatial independence among sites for trappable animals (i.e. bush rats and bandicoots, for which long-range movements are rare, FitzGibbon et al., 2007; Peakall et al., 2006).

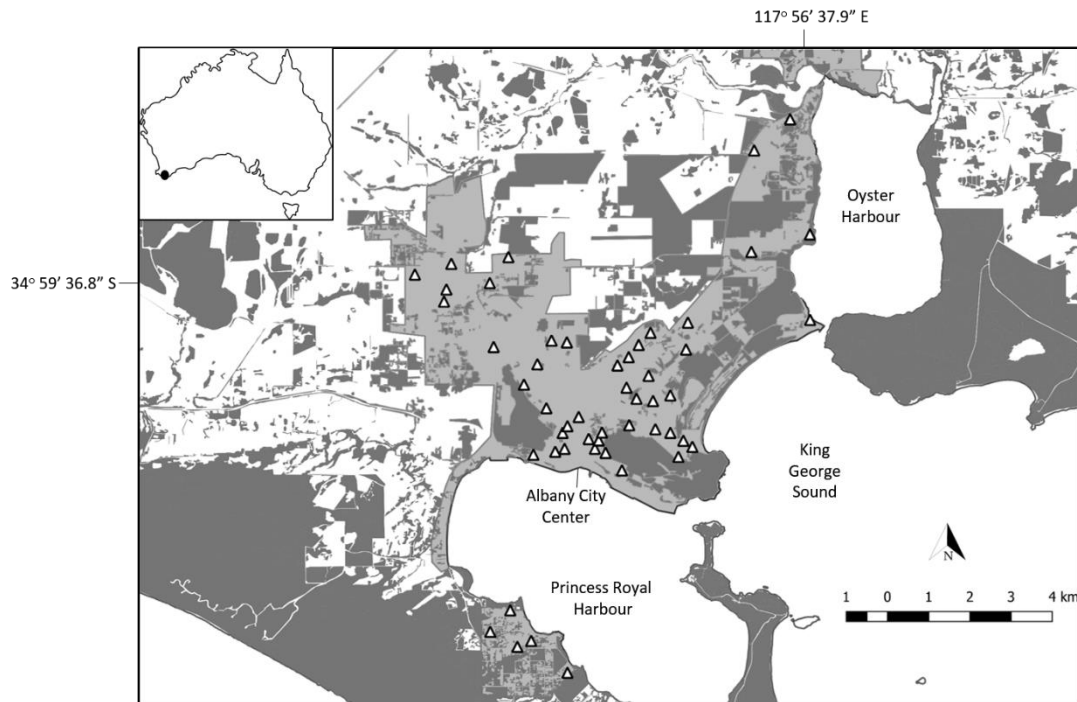


Fig. 4.1. The location of 52 residential garden study sites (Δ) within the urban area (light grey) of the City of Albany in southwest Australia. Natural remnant vegetation is also shown (dark grey). Remnant vegetation layer sourced from Sandiford and Barrett (2010) and Busschots et al. (2020).

Mammal surveys

The methods used to survey for the presence and abundance of terrestrial mammals have been previously described in detail by Van Helden et al. (2020b). In summary, we surveyed for the presence and abundance of terrestrial mammals in autumn (April/May 2018), winter (July/August 2018), spring (October/November 2018) and summer (January/February 2019) using equal detection effort per garden. Climatic conditions (i.e. temperature and rainfall) in each survey season was broadly typical for Albany (i.e. cool and wet in winter, and hot and dry in summer). In each garden ($N = 52$) we set two Elliot traps, two Sheffield cage traps ($220 \times 220 \times 450$ mm), two passive infrared motion-sensing cameras (ScoutGuard 560K-18M HD; ScoutGuard Australia, Queensland, Australia) without a lure and one pitfall trap (with 1.5 m of drift fence either side of the pitfall) to detect ground-dwelling mammals. Elliot and cage traps were baited with universal bait (peanut butter, oats and sardines). Cameras were mounted on stakes and positioned 40 cm above the ground and placed a minimum of 3 m from the nearest obstruction to standardise the field of view among sites. Cameras were programmed to take three consecutive photos when triggered on medium sensitivity between 14:00 and 10:00 h, whilst all other traps were opened within three hours prior to sunset and cleared the following morning within four hours after sunrise. Exhaustive patch spotlight

surveys (with a 350 lumen spotlight) of gardens were conducted with a standardised search effort of five minutes per 400 m² to detect arboreal mammals. Acoustic surveys, consisting of a five minute observation period in each garden using an Echo Meter Touch 2 Handheld Detector (Wildlife Acoustics Inc., Maynard, USA), were used for the detection of flying species. Both spotlight surveys and acoustic surveys were undertaken between last light and midnight on all survey nights.

In every season, each site was surveyed using each survey method on three nights within one week (i.e. 12 nights total over four seasons for each site). All acoustic, trapping and spotlight surveys were undertaken by the same observer. Trapping surveys were conducted on different nights to acoustic and spotlighting surveys so that presence of researchers in the field did not impact upon trap success. All 52 sites were sampled within a six week period in each season (up to nine sites were surveyed on each night). We randomised the survey order of sites in each season and on each night to minimise spatial and temporal effects on the presence and abundance of mammals.

Data processing

Sonograms of recorded echolocation calls (acquired from acoustic surveys) were analysed with Kaleidoscope Pro Analysis Software (Wildlife Acoustics Inc., Maynard, USA). Bat species were discriminated based on previously described call features including shape, and frequency peak, start and finish (Tittley Scientific, 2009) by one common investigator. Call sequences that could not be confidently assigned to a species (e.g. too short, faint call phrase, feeding buzz) were removed from analyses (6% of call sequences, $N = 549$). Photos obtained from cameras were visually examined by one common observer and all species present at each site in each season were recorded. All photos where species identification was not possible (e.g. obscured view of animal or animal too far away for visual identification) were removed from analyses. We recorded the species captured during trapping surveys (Elliot, cage and pitfall) and sighted during spotlight surveys at each site in each season.

As vegetation in gardens was generally sparse, we assumed no difference in the detectability of bats (acoustic detection) among gardens. We mitigated the potential differences in detectability of ground-dwelling species by using multiple complimentary detection methods (camera, cage, pitfall and Elliot trapping) and similar fields of view for camera traps. We implemented an exhaustive spotlight search approach to reduce the potential of differing detectability of arboreal species among sites. It was reasonable to assume that detectability of

arboreal mammals was equal among gardens given that the canopy vegetation in gardens was generally sparse and low in height (<15m), allowing a thorough search of all canopy habitat at all sites.

Garden surveys

At each site we collected information on garden and property features that may influence the presence or abundance of mammals in gardens (Table 4.1) based on relevant literature and the resource requirements of mammals within Albany. These features comprised four broad categories related to shelter availability, food and water availability, interspecific interactions with introduced species and accessibility of the garden by animals (Table 4.1). We also included distance to nearest natural remnant as an explanatory variable to consider its potentially confounding influence on mammal presence and abundance in gardens (e.g. gardens close to remnant vegetation may contain more wildlife than those far from natural habitat regardless of garden features). The distance from each property to nearest remnant vegetation (> 1 ha) was calculated using Quantum Geographic Information System (QGIS; version 3.4.4 with GRASS 7.4.4; QGIS Development Team, 2019). We used 1 ha of vegetation as this approximates the smallest home range of any of our study species (i.e. western ringtail possum; Van Helden et al., 2018). The presence of dogs, cats and potential introduced competitors (rats, rabbits and mice for ground-dwelling mammals, and only rats for arboreal mammals) in each garden was derived from the trapping data (camera, Elliot, cage and pitfall) at each site. All remaining variables were collected during garden surveys undertaken between April and May 2018. The percentage cover of the canopy (> 2 m), understorey (< 2 m), grass (lawn) and open space (lawn and hard surfaces) were estimated visually. These percentages were multiplied by the property size to obtain estimates of the available cover in m².

Data analysis

We undertook analyses for the five species that were detected in at least one garden in all seasons (although all were detected in ≥ 5 gardens in each season). These species included the arboreal western ringtail possum (*Pseudocheirus occidentalis*), the ground-dwelling southern brown bandicoot (*Isodon fusciventer*) and the flying Gould's wattled bat (*Chalinolobus gouldii*), southern forest bat (*Vespadelus regulus*) and white-striped free-tailed bat (*Austronomus australis*). We examined which garden features influenced the presence and abundance of arboreal and ground-dwelling mammals, but only examined presence for

flying mammals due to the inability to derive abundance estimates from acoustic surveys. The presence of flying mammals was identified from the acoustic surveys, the presence and abundance of the western ringtail possum was determined from the spotlight surveys and the presence of the southern brown bandicoot was determined by trapping (Elliot and cage) and camera surveys (i.e. if any of these survey methods detected the species it was recorded as present). Estimates of abundance for the southern brown bandicoot were derived only from cage and Elliot trapping. Trap success for each site in each season was used as an estimate of relative abundance (Seber, 1973) for the southern brown bandicoot whilst the number of individuals per ha was calculated as a measure of relative abundance for the western ringtail possum. We used the mean count of individuals from the three spotlight survey nights and the total number of captures from the trapping surveys in each season to calculate the relative abundance of arboreal and ground-dwelling species in each season, respectively. The presence or absence of a species at each site were recoded as binary variables (i.e. '1' to denote presence, and '0' for absence).

We examined which garden features influenced the presence and abundance of species within four broad categories (i.e. shelter, food/water, interspecific interactions and accessibility) (Table 4.1) using a generalised linear model (GLM) for each category. To investigate which garden features influenced the presence/absence of a species we used binomial GLMs for each species in each season as there was no evidence of overdispersion ($P > 0.05$). To determine which garden features were significant in explaining the abundance of ground-dwelling and arboreal species we used quasi-Poisson GLMs for each species in each season as there was evidence of over dispersion for all species. Explanatory variables included in GLMs for each species were species-specific and reflected their general resource and habitat requirements (Table 4.1).

Based on previous simulation studies that identified the number of samples per explanatory variable should be at least ten (Peduzzi et al., 1996), we used a maximum of five explanatory variables (given our sample size of 52) in each regression model without interactions. For ground-dwelling and arboreal species, this analysis was undertaken for all four seasons (autumn, winter, spring and summer) of presence and abundance data to evaluate if the garden features that determine presence/absence or abundance of mammal species varied seasonally. For flying mammals, we excluded the winter season in analyses as detections of all bat species at sites was generally low (< 5 garden sites) due to the hibernation tendencies of temperate bats during colder weather (Wilkinson and South, 2002). We chose to analyse

data for each season rather than including season as an additional explanatory variable in a single model (e.g. as seen in Patterson et al., 2018) because (1) this allowed greater interpretive power including the ability to compare our results to previous studies that based sampling on a single season and (2) to avoid overspecification bias.

Prior to GLM analyses, Spearman's rank correlation was applied to test for correlation between explanatory variables included in the same model. As no variables included together in any of the GLMs were highly correlated ($-0.7 < r < 0.7$, $N = 52$), all variables were retained for analyses. We performed all GLM analyses in R 3.6.1 (R Core Team, 2019) and all P -values less than 0.05 were considered statistically significant.

Table 4.1. Garden or property features recorded during garden surveys and their inclusion in generalised linear models (GLMs) for each mammal group.

Mammal group	Variable group	Variables included in GLMs	Values
Arboreal	Shelter	Canopy cover	m ²
		Understorey cover	m ²
		Roof access	1: no gaps or spaces, 2: spaces available but access restricted, 3: multiple roof spaces
	Food/water	Canopy cover	m ²
		Presence of permanent water (ponds/water bowls, bird baths)	1: Yes, 0: No
		Percentage of vegetation which was native	Percentage
		Presence of fruiting plants (fruit in spring and/or autumn)	1: Yes, 0: No
		Plant richness	1: < 5 species, 2: 5-10 species, 3: > 10 species
	Interspecific interactions	Presence of dogs	1: Yes, 0: No
		Presence of cats	1: Yes, 0: No
Presence of potential introduced competitors (rats)		1: Yes, 0: No	
Accessibility	Distance to nearest patch of remnant vegetation	m	
Flying	Shelter	Canopy cover	m ²
		Understorey cover	m ²
		Presence of chimneys (duct that transfers smoke from an inside fireplace to sky)	1: Yes, 0: No
		Roof access	1: no gaps or spaces, 2: spaces available but access restricted, 3: multiple roof spaces
	Food/water	Total open space (lawn and pavement)	m ²
		Presence of permanent water (ponds/water bowls, bird baths)	1: Yes, 0: No
		Percentage of vegetation which was native	Percentage
		Plant richness	1: < 5 species, 2: 5-10 species, 3: > 10 species
	Interspecific interactions	NIL	NA
	Accessibility	Distance to nearest patch of remnant vegetation	m
Ground-dwelling	Shelter	Understorey cover	m ²
		Percentage of house with under-house space	Percentage
	Food/water	Presence of permanent ground water (ponds/water bowls)	1: Yes, 0: No
		Grass cover (lawn)	m ²
		Percentage of vegetation which was native	Percentage
	Presence of vegetable patch	1: Yes, 0: No	

Mammal group	Variable group	Variables included in GLMs	Values
		Plant richness	1: < 5 species, 2: 5-10 species, 3: > 10 species
	Interspecific interactions	Presence of dogs	1: Yes, 0: No
		Presence of cats	1: Yes, 0: No
		Presence of potential introduced competitors (rats, rabbits and/or mice)	1: Yes, 0: No
	Accessibility	Distance to nearest patch of remnant vegetation	m
		Fence permeability	1: impermeable, 2: limited holes under fence or infrequent gaps, 3: multiple holes or no fence

Results

Mammal presence and abundance

The western ringtail possum and southern brown bandicoot were frequently detected in all seasons, whilst the three bat species were detected most commonly in spring and summer (Fig. 4.2). The trap success of southern brown bandicoots in each garden ranged between $6.8 \pm 1.8\%$ (mean \pm SE) in winter and $9.3 \pm 1.7\%$ in autumn, whilst the abundance of western ringtail possums ranged between 3.9 ± 0.8 individuals/ha in winter and 6.2 ± 1.0 individuals/hectare in autumn ($N = 52$; Fig. 4.3). For a full description of all other mammal species detected in gardens see Van Helden et al. (2020b).

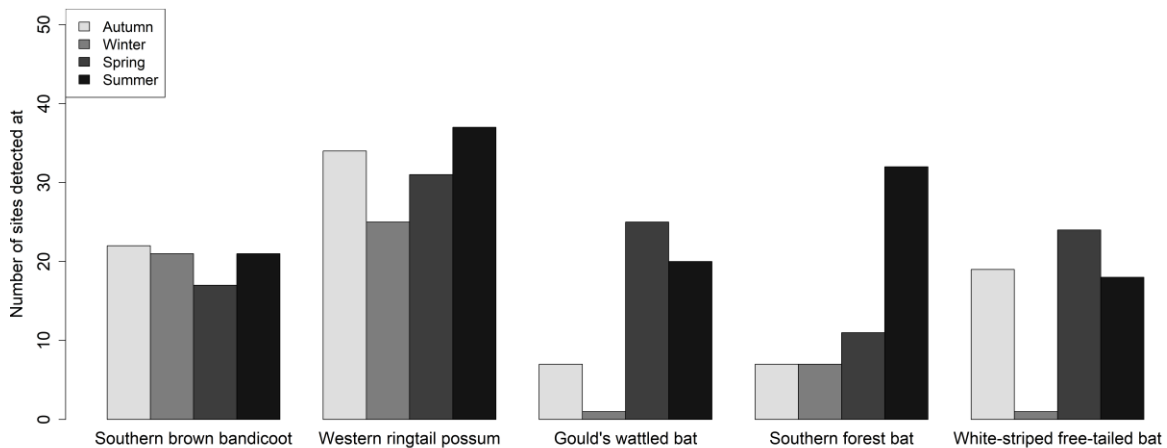


Fig. 4.2. The number of garden sites at which each species was detected in each season between April 2018 and February 2019 in the City of Albany. The total number of garden sites surveyed in each season was 52.

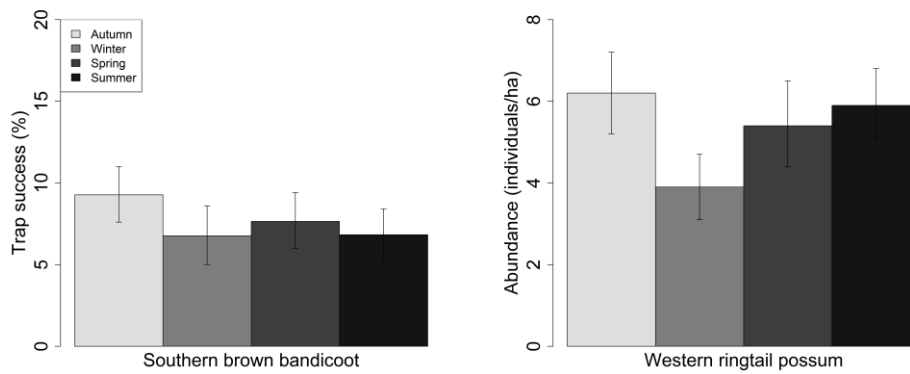


Fig. 4.3. The mean trap success (% , $N = 52$) of southern brown bandicoots and mean abundance (individuals/ha, $N = 52$) of western ringtail possums in gardens in each season between April 2018 and February 2019 in the City of Albany. Whiskers represent the standard error of the mean. Figure has been modified from Van Helden et al. (2020b).

Garden features

Garden and property features were highly variable among sites (Fig. 4.4). Of the 52 garden properties surveyed, 77% had a chimney, 77% had fruiting plants, 52% had a vegetable patch, 62% had a ground water source and 50% had an elevated water source. Dogs were detected at 62% of gardens, cats at 79% of gardens and potential introduced competitors (rabbits, rats and mice combined) at 62% of gardens. Introduced rats were detected at 33% of gardens. Roof spaces were highly available at 24 of the houses, restricted at 19 houses and inaccessible at 9 houses. Fences surrounding the surveyed properties were highly permeable at 31 of the houses, restricted at 17 houses and impermeable at 4 houses. Plant richness was high in 19 gardens, moderate in 22 gardens and low in 11 gardens.

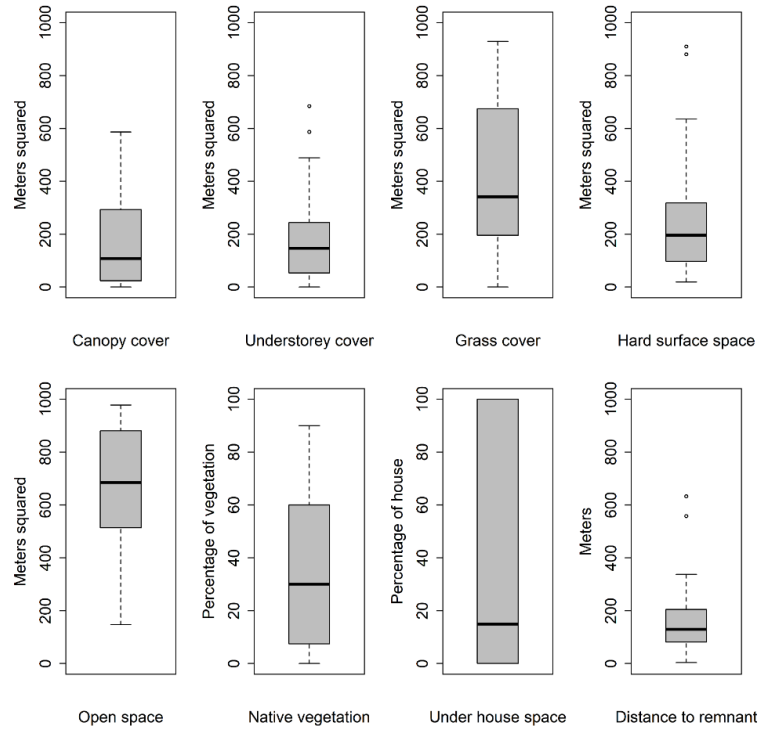


Fig. 4.4. Variability in vegetated cover (canopy, understorey, grass, native vegetation), hard surfaces and open space, under-house space and distance to closest patch of remnant vegetation at 52 garden sites based on surveys undertaken in April and May 2018 in the City of Albany. The grey boxes represent the interquartile range (IQR), the black lines in box plots represent medians, and the whiskers represent the maximum and minimum value within $1.5 \times \text{IQR}$ of the upper and lower quartile, respectively. ‘o’ represents values outside of these maximum and minimum values.

Garden features affecting mammal presence

A total of six garden features influenced the presence of mammal species, of which only one influenced a species in more than one season (Table 4.2). No garden features influenced the presence of any mammal species consistently in all four seasons (Table 4.2).

The presence of the western ringtail possum was influenced by two garden features, of which only one (canopy cover) was influential in more than one season (Table 4.2). The presence of western ringtail possums was positively related to canopy cover in autumn ($P = 0.049$), spring ($P = 0.009$) and summer ($P = 0.019$; Table 4.2). In autumn, their presence was also negatively related to plant richness ($P = 0.044$; Table 4.2). In winter, presence of the possum was not influenced by any of the garden features collected in this study ($P > 0.05$).

The presence of southern brown bandicoots was influenced by one garden feature in one season (Table 4.2). In winter, southern brown bandicoot presence was positively influenced by the presence of vegetable patches in gardens ($P = 0.007$; Table 4.2). In all other seasons

none of the garden features collected in this study influenced southern brown bandicoot presence ($P > 0.05$; Table 4.2).

The presence of bats was influenced by four garden features; none of which were influential in more than one season (Table 4.2). The presence of Gould's wattled bat in autumn was negatively related to distance to nearest bushland remnant ($P = 0.017$; Table 4.2). In all other seasons none of the garden features collected in this study influenced the presence of Gould's wattled bat ($P > 0.05$). In summer, the presence of the southern forest bat was positively related to the availability of roof cavities ($P = 0.026$; Table 4.2), however in autumn and spring none of the garden features collected in this study influenced the presence of the southern forest bat ($P > 0.05$). Presence of the white-striped free-tailed bat was negatively related to the availability of understorey cover in autumn ($P = 0.035$) and to plant richness ($P = 0.036$) in spring (Table 4.2). In summer, no garden features influenced the presence of white-striped free-tailed bats ($P > 0.05$).

Garden features affecting mammal abundance

A total of six garden features influenced the abundance of mammal species, of which two influenced a species in more than one season (Table 4.2). No garden features influenced the abundance of any mammal species consistently in all four seasons (Table 4.2).

The abundance of the western ringtail possum was related to four garden features, of which only one (canopy cover) was influential in more than one season (Table 4.2). The abundance of western ringtail possums was positively related to canopy cover in winter ($P = 0.006$) and summer ($P = 0.005$). In winter, the abundance of western ringtail possums was also positively related to the availability of roof spaces ($P = 0.009$) and the presence of rats ($P = 0.04$). In spring, the abundance of western ringtail possums was negatively related to the presence of dogs ($P = 0.003$), while in autumn none of the garden features influenced the abundance of western ringtail possums ($P > 0.05$; Table 4.2).

The abundance of the southern brown bandicoot was influenced by three garden features, of which only one (presence of dogs) was influential in more than one season (Table 4.2). The abundance of southern brown bandicoots was negatively correlated with the presence of dogs in winter ($P = 0.008$), spring ($P = 0.020$) and summer ($P = 0.025$; Table 4.2). In winter, the abundance of southern brown bandicoots was also negatively related the presence of introduced competitors ($P = 0.015$; Table 4.2). Southern brown bandicoots were more abundant in autumn when vegetable patches were present in gardens ($P = 0.049$; Table 4.2).

Table 4.2. Summary of garden features that were significantly related the presence and abundance of mammal species in each season of sampling. The parameter estimates (\pm SE) for each significant variable are shown and significance at the 5% level and 1% level is denoted by ‘*’ and ‘**’, respectively. Full details of results are supplied in Appendix 1. Parameters have been abbreviated to denote canopy cover (canopy), understorey cover (understorey), plant richness (plantrich), presence of a vegetable patch (veg), roof access (roof), presence of a dog (dog), distance to nearest patch or remnant vegetation (distance), presence of competitors (competitor) and presence of rats (rat).

Species	Variable Group	Parameters	Autumn	Winter	Spring	Summer
<u>Presence</u>						
Southern brown bandicoot	Food/water	Veg	-	1.96 (0.730)**	-	-
Western ringtail possum	Shelter	Canopy	0.01 (0.003)*	-	-	0.01 (0.004)*
	Food/water	Canopy	0.01 (0.004)*	-	0.01 (0.003)**	-
Gould’s wattled bat	Accessibility	Plantrich	-1.37 (0.680)*	-	-	-
		Distance	-0.02 (0.010)*	NA	-	-
Southern forest bat	Shelter	Roof	-	NA	-	1.10 (0.500)*
White-striped free-tailed bat	Shelter	Understorey	-0.01 (0.003)*	NA	-	-
	Food/water	Plantrich	-	NA	-1.29 (0.620)*	-
<u>Abundance</u>						
Southern brown bandicoot	Food/water	Veg	0.85 (0.420)*	-	-	-
	Interspecific interactions	Dog	-	-1.43 (0.520)**	-1.04 (0.430)*	-1.12 (0.490)*
		Competitor	-	-	-1.22 (0.480)*	-
Western ringtail possum	Shelter	Canopy	-	0.01 (0.001)**	-	0.01 (0.001)**
		Roof	-	0.56 (0.220)**	-	-
	Food/water	Canopy	-	-	-	0.01 (0.001)*
	Interspecific interactions	Dog	-	-	-	-1.18 (0.380)**
		Rat	-	-	0.79 (0.37)*	-

Discussion

The creation of wildlife-friendly residential gardens may offer substantial opportunities for biodiversity conservation (Goddard et al., 2010; Soanes et al., 2019). Whilst multiple studies provide promising evidence that garden features could be manipulated by residents to improve conservation outcomes for fauna in gardens (e.g. Daniels and Kirkpatrick, 2006; Dique et al., 2004; Oprea et al., 2009; Patterson et al., 2017; Threlfall et al., 2016; 2017; Van Helden et al., 2020a), there has been little consideration of how the importance, and therefore the conservation benefits, of these features varies seasonally. Our study is one of the first to evaluate if the mechanisms that drive the assemblage, presence and abundance of vertebrate fauna in gardens varies among seasons. In support of our hypothesis, we found that the garden features that influenced the presence and abundance of arboreal and ground-dwelling mammals differed among seasons. Similarly, the garden features that predicted the presence of flying species only operated in one season. Contrary to our predictions, seasonal variation in the importance of garden features was not limited to fruit and seed-bearing food resources but also applied to more stable, and consistently present features such as vegetation cover, roof cavities and the presence of dogs. These results suggests that while single-season studies will provide some insight into the broad mechanisms that drive animal presence in gardens, they may fail to identify factors influential at times other than in the season of sampling or alternatively, overemphasise factors that do not operate across all seasons. Studies that consider the mechanisms on a seasonal basis will better identify all the features influencing the presence and abundance of animals in gardens, as well as identify factors that are consistent across seasons. This will better inform conservation initiatives within gardens as it establishes in which season manipulations will have the most benefit, and which features will likely have the greatest conservation outcomes. We suggest it is both those features that support wildlife in multiple or all seasons, and those that support critical life history events in a single season that could contribute to conservation if manipulated by residents appropriately.

Although there was a high degree of variability in the relative importance of garden features among seasons, there were some features that influenced the presence and abundance of particular species in more than one season. For example, we found that canopy cover influenced the presence and abundance of western ringtail possums in three seasons (autumn, spring and summer) and two seasons (winter and summer), respectively (see also Van Helden et al., 2020a); consistent with the arboreal tendency of the species (Jones et al., 1994).

Similarly, the presence of dogs negatively influenced the abundance of southern brown bandicoots in three seasons (winter, spring and summer), supporting the contention that the presence and abundance of ground-dwelling mammals are limited in residential gardens by predators (e.g. cats (Baker et al., 2003) and dogs (Van Helden et al., 2020a)). These findings are consistent with previous evidence that the garden features that predict fauna presence and abundance is species-specific and often resource driven (e.g. Baker and Harris, 2007; Daniels and Kirkpatrick, 2006; Threlfall et al., 2016; Van Helden et al., 2020a) and also demonstrates that only some of these garden features act in multiple seasons (as seen in insects; Adams et al., 2020). When considering conservation initiatives within residential gardens, we suggest that it is these factors that operate in multiple seasons, which could be prioritised for management or manipulation, as they are likely to be more consistently important for wildlife. For example, in the context of our study, resident education programs or incentive-based policies could be used to provide information and incentivise land-owners to engage in wildlife-friendly pet ownership (e.g. keep pets indoors at night during active periods of the southern brown bandicoot) and to plant suitable tree species that provide shelter and food for the Critically Endangered western ringtail possum (*Biodiversity Conservation Act 2016*).

We found multiple factors only influenced mammal presence or abundance in a single season (83% of identified features that influenced presence and 67% of identified features that influenced abundance). This is unsurprising given that the presence, abundance and activity of animals vary seasonally, often in response to seasonal fluctuations in their resources and resource requirements (Dayton, 2008). For instance, the presence of vegetable patches in gardens positively influenced the abundance of southern brown bandicoots only in autumn, and roof cavities positively influenced the abundance of western ringtail possums only in winter. Vegetable patches are commonly harvested in autumn, leaving exposed soil containing an abundance of soil fauna; likely an ideal resource for the omnivorous digging southern brown bandicoot (Valentine et al., 2013). Similarly, in winter, when environmental conditions are colder, wetter and windier, the presence of roof cavities offer substantial protection from the elements which may explain why the abundance of western ringtail possums was positively related to roof cavities only in winter. For some species, seasonal resources such as these may be critical for key demographic and ecological processes (e.g. for reproduction during breeding seasons) that maintain populations in the longer-term. Failing to identify these important single-season resources may represent a substantial hindrance to the conservation of these species. Our finding that multiple garden features only have influence

in a single season exemplifies the dangers of assessing predictors of wildlife presence and abundance based on one or few seasons of sampling.

Whilst previous studies have extensively furthered our understanding of which garden features could be manipulated to achieve conservation outcomes, these studies have predominantly sampled in one or few seasons (e.g. Baker et al., 2003; Basham et al., 2011; Daniels and Kirkpatrick, 2006). These studies may have therefore overemphasised certain mechanisms that operate only in that season, and missed others operating in unsampled seasons. For example, based on sampling conducted within autumn and winter (October – February) in north-west Bristol (England), Baker et al. (2003) found that the abundance of cats and the distance to nearest patch of vegetation influenced the abundance of the wood mouse (*Apodemus sylvaticus*) whilst other factors such as food availability (including fruit/seed-bearing plants) did not. Given the wood mouse is primarily granivorous but exhibits seasonal variations in diet (Khammes and Aulagnier, 2007), these fruit/seed-bearing plants may be more influential in determining wood mouse abundance in the unsampled seasons (spring and summer), yet cannot be determined based on the timing of sampling used for this study. In contrast, based on sampling undertaken in spring and summer in Hobart (Australia), Daniels and Kirkpatrick (2006) found that percentage of deciduous tree cover positively influenced the silver eye (*Zosterops lateralis*), common blackbird (*Turdus merula*) and European goldfinch (*Carduelis carduelis*). In this case, the importance of deciduous tree cover may be overemphasised given that these trees shed their leaves during autumn and are therefore unlikely to provide much shelter or food resources during the colder, unsampled seasons. These examples, and the evidence we present of seasonal variation in the importance of garden features, highlights the benefits of examining the mechanisms in all seasons, not only to identify all important features, but to also ensure others are not overemphasised.

Although we identified numerous garden features that are important for mammal wildlife and show that their importance varies seasonally, we did not consider the influence of landscape-scale features in our study which may explain why no garden features influenced the presence or abundance of mammals in some seasons. While there is promising evidence that wildlife-friendly gardening can be beneficial regardless of landscape-scale factors (e.g. level of surrounding urbanisation, Fontaine et al., 2016), it is important to recognise that the occurrence of wildlife in gardens likely also depends on broader landscape factors (such as city size and intensity of urbanisation), the biological and ecological traits of species (e.g. mobility, behaviour, resource requirements) and complex interactions among all of these

(Fournier et al., 2020). How local-scale features (including the seasonal importance of features), landscape-scale factors and the biological traits of animals interact to determine the occurrence of wildlife in gardens is worthy of further investigation.

Whilst our study furthers the understanding of the mechanisms that drive animal abundance and presence in gardens, there are a number of knowledge gaps that need further consideration to fully understand the implications and application of our findings for wildlife conservation. Firstly, it is unclear from our study the level of dependence species have on these features as animals may be capable of modifying their behaviour to utilise other resources in seasons when preferred ones are not available. Secondly, we conducted this study in one year, on one taxonomic group, in one geographic region with pronounced seasonal differences. Further research that investigates (1) whether seasonal drivers of mammal presence and abundance is consistent among years, (2) whether seasonality is important for other taxonomic groups (e.g. birds, reptiles, amphibians) and (3) if the mechanisms that influence fauna presence and abundance also differs seasonally in regions with less pronounced seasonal differences (e.g. tropical regions) would better inform when seasonality should be considered in these types of mechanistic studies. Investigation of these remaining knowledge gaps warrant serious consideration and would further inform decision-making regarding wildlife conservation within residential landscapes.

Using mammals as a case study taxon we exemplify that in addition to variation among cities (Van Helden et al., 2020a), species (Daniels and Kirkpatrick, 2006) and taxa (Fontaine et al., 2016; Threlfall et al., 2016), the mechanisms that influence the presence and abundance of wildlife in gardens can also vary seasonally. Our results suggest that the evaluation of local factors driving patterns of biodiversity in gardens, should consider seasonality in their sampling, or at the very least, exercise caution in the interpretation of results based on one (or few) seasons. Identifying which garden features consistently influence patterns of biodiversity among seasons, or are critical for key demographic processes in a single season, may be particularly important as these are the features that, if manipulated by residents, will likely have the greatest conservation benefit in urban landscapes. By including gardens in planning and management actions based on knowledge of which garden features enhance native urban biodiversity, and in which season, residential gardens could play a significant role in biodiversity conservation.

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CHAPTER 5

CRITICALLY ENDANGERED MARSUPIAL CALLS RESIDENTIAL GARDENS HOME



Photo: A radio-tracked western ringtail possum (*Pseudocheirus occidentalis*), affectionately known as ‘Lilly’ because of her love of lilly pillly trees (*Syzygium smithii*), and her companion on a garden fence.

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Abstract

Urbanisation is one of the biggest global threats to biodiversity. However, urban areas can offer opportunities for biodiversity conservation because of the assortment of resources and numerous taxa that occur in these habitats. While the use of gardens by animals is well documented, the ability of native fauna to exclusively use (i.e. reside in) them is largely uninvestigated and therefore constrains our current understanding of their conservation value. We aimed to determine whether individuals of a threatened species can exclusively reside in gardens for several months of their life, and the frequency that novel resources (i.e. non-natural) are used compared to natural ones. We predicted that individuals captured in gardens would use both bushland and residential habitat, and use natural resources more frequently than novel resources because of the assumption that remnant habitat is more suitable than modified gardens. Using the Critically Endangered western ringtail possum (*Pseudocheirus occidentalis*) as a case study, we captured 20 individuals in gardens and radio-tracked them for a period of 95 ± 3.3 days to determine their habitat use. Contrary to our predictions, for the duration of our study individuals exclusively resided within gardens, irrespective of proximity to bushland, and used novel resources more frequently than natural resources for all behaviours. We also found that over half (53%) of all the captured female possums had pouch young, and at the conclusion of the study, male possums were on average 43 g heavier. Our results exemplify that gardens can be of sufficient quality to support individuals, and that these modified habitats should be incorporated into conservation policy in urban landscapes. Given the extensive number of threatened species located within cities globally, our finding that a threatened species can reside exclusively within gardens further highlights the conservation opportunity that residential landscapes and cities offer wildlife.

Introduction

The expansion of urban areas has caused a global decline in biodiversity (McKinney, 2002; 2008). This is primarily attributable to habitat loss, which is one of the leading causes of species decline (Czech et al., 2000; Seto et al., 2012). However, the distributions of many native species overlap with urban areas (Ives et al., 2016; Kowarik and von der Lippe, 2018) and some of these species have altered their behaviour to adapt to these highly modified landscapes (Davison et al., 2009; Sol et al., 2013). There is growing evidence that a variety of taxa including birds (e.g. Cannon et al., 2005; Daniels and Kirkpatrick, 2006), invertebrates (e.g. Goulson et al., 2002), mammals (e.g. Gallo et al., 2017; Maclagan et al., 2018; Van Helden et al., 2020a; 2020b), plants (e.g. Smith et al., 2006) and reptiles (e.g. González-García et al., 2009) occur within urban landscapes. As this knowledge base grows, so does the recognition that urban landscapes offer substantial opportunities for biodiversity conservation (Aronson et al., 2017; Soanes and Lentini, 2019; Soanes et al., 2019).

Urban landscapes comprise different greenspaces including urban remnant vegetation, private gardens, street vegetation, green roofs, urban waste-lands (e.g. rubbish dumps and vacant blocks) and parklands (Aronson et al., 2017; Cilliers et al., 2013; Lepczyk et al., 2017). Not only do wildlife occur in urban remnant habitats (e.g. Deikumah and Kudom, 2010; Garden et al., 2007), they also occur in highly modified greenspaces (e.g. Daniels and Kirkpatrick, 2006; Gallo et al., 2017; Le Roux et al., 2018; Van Helden et al., 2020a; 2020b). Despite this, policies and strategies for conservation within urban landscapes have focused on the potential opportunities that ‘natural’ remnants offer, while those of smaller, modified greenspaces have been largely overlooked (Soanes et al., 2019; Tulloch et al., 2016). Although remnant vegetation is undoubtedly critical for wildlife in urban landscapes, conservation strategies that take advantage of the presence of animals and plants in small, modified greenspaces may result in better conservation outcomes (Soanes et al., 2019).

Residential gardens are one component of the urban landscape that can collectively represent up to 86% of the total urban greenspace (Goddard et al., 2010; González-García and Sal, 2008; Loram et al., 2007; Mathieu et al., 2007). Not only do gardens comprise a highly diverse floristic assemblage (Threlfall et al., 2016a), they also provide wildlife with an assortment of resources (Davies et al., 2009), including novel foods (e.g. human trash, supplementary feed, exotic plants), artificial shelter (e.g. under-house spaces and roof cavities) and alternative structures for movement (e.g. fences and electricity power lines). The ability of animals to use the resources within residential gardens has long been

established, with evidence of animals foraging in gardens (e.g. badgers *Meles meles*, Davison et al., 2009), and using artificial structures as both movement pathways (e.g. use of fences by spiny-tailed iguana *Ctenosaura similis*, González-García et al., 2009), and rest sites (e.g. use of roof cavities and nest boxes by brushtail possums *Trichosurus vulpecula*, Eymann et al., 2006). These and numerous other examples (see Goddard et al., 2010; Shwartz et al., 2014 for review) provide mounting evidence that residential gardens offer conservation opportunities for wildlife. However, they do not demonstrate that garden habitats are capable of supporting animal residency or key ecological processes such as reproduction, recruitment and survival, essential for population persistence (Maclagan et al., 2018). A biological understanding of these aspects is needed to fully understand the conservation value of gardens and whether this habitat type should be included in conservation policy.

Animal residency (i.e. exclusive use of a habitat) is one metric that indicates habitat is of sufficient quality to consistently support wildlife rather than only providing short-term movement corridors and transient resources (Maclagan et al., 2018). Despite this, the ability of individuals to reside exclusively in garden habitats (i.e. without using remnant vegetation) is largely unknown. For example, only a single study (the blue-tongued lizard, *Tiliqua scincoides*; Koenig et al., 2001) demonstrating that individuals of an animal species can exclusively reside within residential gardens was identified in recent global reviews of urban conservation research (Goddard et al., 2010; Shwartz et al., 2014; Soanes et al., 2019). Studies that examine the behaviour of animals within urban environments generally report some dependence on, or use of remnant vegetation when using residential areas (e.g. Davison et al., 2009; Fitzgibbon et al., 2011; Van Helden et al., 2018). However, most individuals examined in these studies were sourced from non-garden habitats (e.g. remnant vegetation; Fitzgibbon et al., 2011; Van Helden et al., 2018), and therefore it is possible that other, unstudied individuals could reside entirely within the residential landscape. If individuals are capable of residing in these areas, it would suggest that residential gardens are of sufficient habitat quality and have the necessary resources to support individuals in the long-term (and therefore potentially reproduction) in their own right. Thus, further studies that investigate the capacity of individuals to exclusively reside within residential gardens is critical to further establish the value of residential gardens for biodiversity conservation.

In this study, we examined whether individuals of a Critically Endangered arboreal marsupial were capable of exclusively residing within residential gardens over several months of their life. We selected the western ringtail possum (*Pseudocheirus occidentalis*) as a case study

because of its common occurrence in residential habitats (Thompson and Thompson, 2009; Van Helden et al., 2019; 2020a; 2020b) and flexible use of resources (Bader et al., 2019; Mathieson et al., 2020; Van Helden et al., 2018). These factors support the notion that residential areas may offer substantial conservation opportunities for the species, yet currently its presence in urban landscapes is assumed dependent on urban remnant vegetation (DPaW, 2017). Given existing evidence of the dependence of individuals (including the western ringtail possum) on urban remnant vegetation when using residential areas (e.g. Davison et al., 2009; Fitzgibbon et al., 2011; Van Helden et al., 2018), we predicted that western ringtail possums captured in residential gardens would use both urban remnant vegetation and residential habitat. More specifically, we hypothesised that residential areas would be used less frequently when in closer proximity to natural bushland areas because of the assumption that the resources and habitat found in remnant vegetation are more suitable than the highly modified residential habitat. We also aimed to determine which resources were used within residential gardens and the frequency that novel resources were used compared to natural ones. Understanding the capacity of gardens to support animal residency will greatly improve knowledge of the conservation opportunity that gardens represent for biodiversity.

Methods

Study species

The western ringtail possum is a Critically Endangered (*Biodiversity Conservation Act 2016, IUCN Red List*) folivorous, arboreal marsupial that weighs approximately 1 kg at maturity and exhibits strong site fidelity with home ranges generally less than 1 ha (Van Helden et al., 2018; Yokochi et al., 2015). Although commonly observed in close proximity with other individuals, the species exhibits territoriality (Ellis and Jones, 1992) and is thought to live for 4-5 years (Wayne et al., 2005). The western ringtail possum is capable of breeding all year round (generally with litter sizes of one), although breeding peaks in April and July, and September and November (Jones et al., 1994). The species was historically distributed over much of the southwest corner of Australia; however, since European colonisation the geographic range of this species has declined by 90% primarily due to agricultural and urban expansion (Jones, 2004). The species is now largely restricted to three known disjunct populations, all of which at least partially overlap urban centres (DPaW, 2017). Despite its apparent vulnerability to urbanisation, the species is flexible in its diet (Mathieson et al., 2020) and behaviour (Bader et al., 2019; Van Helden et al., 2018; 2019), which likely

contributes to its frequent use of residential gardens, other modified urban greenspaces and remnant vegetation in city centres (Thompson and Thompson, 2009; Van Helden et al., 2019; 2020a; 2020b). These behavioural attributes and habitat preferences made the western ringtail possum an ideal subject species to test our research hypotheses.

Study area and study sites

We conducted this study in the City of Albany (34°58'S, 117°52'E) (Fig. 5.1), located in one of the three remaining stronghold regions for the western ringtail possum (DPaW, 2017). Situated within an agricultural landscape, Albany is a small city supporting approximately 31 500 people that reside in 12 800 private dwellings that extend over an area of 90 km² (population density of ~ 3.5 persons per ha; CoA, 2018). Although the city is rapidly urbanising (both in terms of urban infill and expansion of the city), there are multiple bushland remnants scattered throughout the built-up environment (i.e. areas with infrastructure) that range in size from less than 1 to 600 ha (Busschots et al., 2020; Van Helden et al., 2018; Fig. 5.1). Many of these remnants support multiple native animals including the western ringtail possum (Mathieson et al., 2020, Van Helden et al., 2020b). We undertook this study in 16 private gardens, all < 0.2 ha and located within 3 km of Albany City center (Fig. 5.1). We selected sites based on whether western ringtail possums had been previously detected in the garden (derived from Chapter 3) and where the resident was willing to allow researchers to collar and track possums for an extended period of time. The properties varied in vegetation cover and composition, and ranged in proximity to bushland remnants from 5 m (i.e. bordered remnant vegetation) to 750 m (i.e. a substantially greater distance than what is likely traversed by western ringtail possums, see Van Helden et al., 2018; Fig. 5.1).

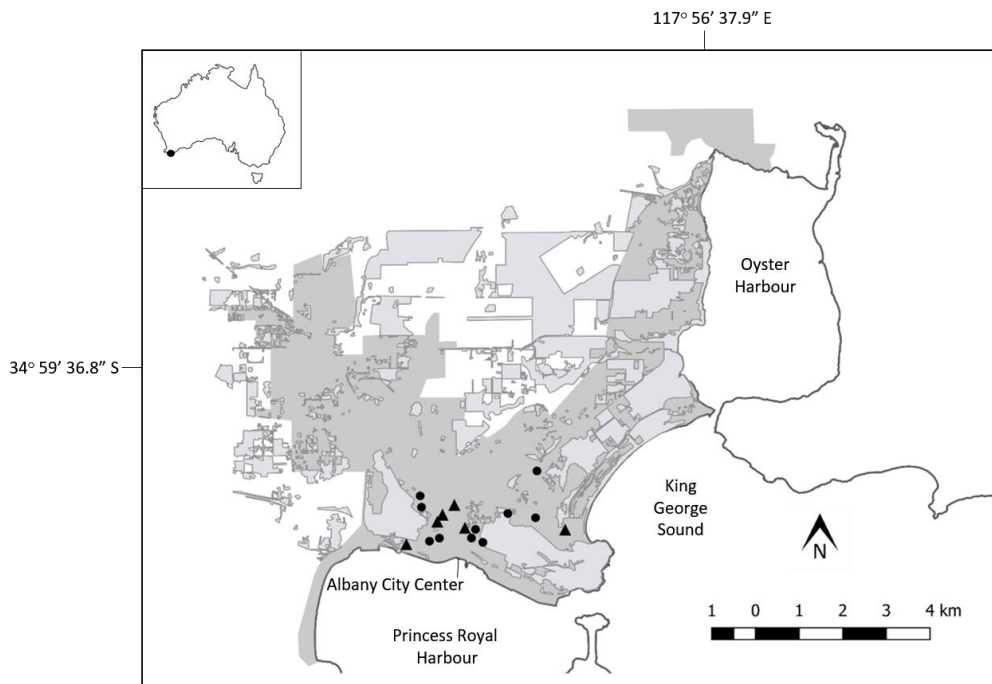


Fig. 5.1. The location of the 16 residential gardens (black dots and triangles) where 20 western ringtail possums (*Pseudocheirus occidentalis*) were captured for radio-tracking and the subset of sites where habitat surveys were undertaken (black triangles). The location of natural remnant vegetation (light grey) within the urban area (dark grey) of the City of Albany in southwest Australia is also shown. Remnant vegetation layer sourced from Sandiford and Barrett (2010) and Busschots et al. (2020). The urban area of Albany comprised any land-use category defined by the City of Albany as: general industry, highway commercial, hotel/motel, light industry, local and neighbourhood centre, port industry, rail, regional centre (including mixed business and mixed use) and residential (special or tourist).

Possum capture and collaring

We captured 28 western ringtail possums in private gardens in Albany between 28 May and 5 June 2019 using medium-sized Sheffield cages (220 × 220 × 450 mm) baited with almond meal soaked in strawberry essence. We strategically positioned traps on movement pathways by securing cage traps to fence lines and tree branches. We baited and opened traps between 16:00 and 17:00 h and then cleared and closed traps the following morning within 2 h of sunrise. We restrained each captured animal by hand inside a dark, cotton bag to minimise distress and then recorded sex, morphometric measurements and body weight. Only adult individuals (≥ 800 g) without pouch young greater than 1.5 cm were considered suitable for collaring. As a result, we fitted 20 of the 28 captured individuals with a collar that contained a VHF radio-transmitter (M1640 Mammal Collar, Advanced Telemetry Systems, Gold Coast, Australia) and then released them at the point of capture. After at least one animal had been collared at a site on any given night, we removed the traps. At four properties two individuals were captured on the same night, both of which were collared. This resulted in a total of 20

western ringtail possums (11 females, 9 males) fitted with radio-transmitters in 16 separate gardens.

Radio-tracking

We located collared possums between 30 May and 9 September in 2019 by radio-tracking using a scanning receiver and a three-element Yagi antenna (Titley Scientific, Brendale, Australia). Wherever possible we tracked animals to their exact location (i.e. homed in) and recorded their position (latitude and longitude; Garmin, GPSMAP 64, USA), structure use (e.g. species of tree, roof, fence), behaviour (e.g. foraging, day-time resting, moving) and if resting, the type of refuge used. Plants were identified to species where possible, however at times could only be identified to genus or family. Structures used by possums were grouped into four broad categories: ‘non-indigenous plants’, ‘indigenous plants’, ‘man-made structures’ and ‘ground’. A fifth category, ‘unknown plants’, was created for plants that could not confidently be classified as indigenous or non-indigenous. When dreys (i.e. possum nests) were used, we recorded the plant that the drey was constructed in. Nesting material for dreys could not be determined because they were frequently inaccessible (i.e. too high). Indigenous plants were defined as any species naturally occurring in the Albany region. We chose not to classify native plants as any plant endemic to Australia because the western ringtail possum is a specialist folivore only distributed within southwest Australia and has therefore not been exposed to the flora found in other regions of Australia. When animals could not be tracked to their exact location (e.g. where permission to access a property had not been obtained) we determined its location using triangulation from three nearby locations within a 5-min time interval.

We obtained one location of each animal on 4 days every week; three of which were night-time locations and one of which was a day-time location. Tracking continued until a minimum of 50 locations of each animal were achieved as at least 50 locations are required to minimise area bias and variance in home-range size estimates (Seaman et al., 1999). We also confirmed 50 locations were sufficient to robustly estimate home-range size for western ringtail possums by initial analysis that indicated minimal increase in home-range size and stabilisation of variance (< 3% change) after ~ 40 locations. At the end of the sampling period, all collared animals were re-trapped (see *Possum capture and collaring*), or hand-captured, and their collars were removed before animals were weighed and released at their point of capture.

Home-range estimation

We used a 95% kernel density estimate (KDE) to determine home-range size (Worton, 1989) in RANGES 9 (Kenward et al., 2014) with fixed kernels as they are considered more accurate and less biased than adaptive kernels for 95% KDEs (Seaman et al., 1999; Seaman and Powell, 1996). We used KDEs of home-range size as this method is less susceptible to unpredictable biases and outliers than other methods such as the minimum convex polygon method (Börger et al., 2006; Hansteen et al., 1997). Fifty locations could not be achieved for five collared individuals due to mortality (predation or road strike), or the premature removal of collars because of neck rubbing. These five possums were not included in home-range analyses. Home-range sizes for the remaining 15 possums were estimated based on an average number of 52 ± 0.5 locations (\pm SE) that spanned a mean of 95 ± 3.3 days for each individual.

We used both triangulated locations and exact animal locations for the estimation of home-range size. The average percentage of locations used in home-range analysis that were obtained from triangulation was $25 \pm 4.4\%$. For animals located by triangulation, we defined their location as the midpoint of the error polygon (Heezen and Tester, 1967), which was calculated using the plugin ‘AniMove’ (Bocacci et al., 2018) in Quantum Geographic Information System (QGIS Development Team, 2019). The mean error polygon for locations obtained from triangulation was 0.02 ± 0.003 ha and no locations had an error polygon greater than 0.03 ha. We corrected any obvious GPS errors in handheld locations and removed all triangulation locations that had more than one divergent bearing. This resulted in the removal of nine locations (1% of all locations).

We checked for autocorrelation in animal locations using the Schoener index (Schoener, 1981; Swihart and Slade, 1985) calculated in RANGES 9. This indicated that more than 100 h were required to reach independence between locations for each animal. To investigate this further, on a subset of six individuals (five females and one male) we undertook a short intensive radio-tracking period where we obtained a location of each animal every 15 min throughout the night. From these data the time to independence was calculated to be less than 2.5 h for five of the six animals. Given these results and that accuracy and precision of home-range estimates improve with more locations regardless of the increase in autocorrelation (De Solla et al., 1999), we retained all locations in home-range analyses.

Habitat surveys

We determined the preference for non-indigenous or indigenous plant species for six individual possums (five females and one male; Fig. 5.1). These were the only individuals that had enough foraging locations (> 10 each) to robustly compare their choice of forage plants with the availability of those plants. We overlaid the home-range boundary of each possum on a satellite image of the area and then accessed each of the properties within the home ranges and ground-truthed the vegetation seen on the satellite image in June 2020. While at each property, all the vegetation cover within the home range was classified as non-indigenous or indigenous to the Albany region. The proportion of vegetation cover that was indigenous was then calculated for the home range of each possum by dividing the total amount of vegetation cover that was indigenous by the total amount of vegetation cover present within the home range. There was no structural change in the vegetation at any site between the time of habitat survey and the time of radio-tracking.

Data analysis

We performed statistical analyses in R 3.6.1 (R Core Team, 2019) with the package ‘RVAideMemoire’ (Hervé, 2013). We compared home-range size of possums between male and female possums using a Mann-Whitney test, because data were non-normal despite attempts to normalise (e.g. square-root and \log_{10} transformations). Chi-squared tests were used to determine whether there were differences in the frequencies of observed use of refuge types and structures. When significant differences in frequencies were detected among structures or refuge types we conducted pairwise comparison using Bonferroni corrections to identify which structures and refuge types differed from each other. To determine if the consumption of novel resources (i.e. non-indigenous plants) reflected the availability of novel resources in residential gardens, we compared the number of sightings where each possum ($N = 6$) foraged on non-indigenous plants to the expected number of forage events on non-indigenous plants using a chi-squared test. The expected number of forage events was calculated based on the available non-indigenous vegetation cover in the home range of each possum. Without preference, the number of forage events on non-indigenous plants would be proportional to the availability of non-indigenous vegetation cover. Any species of plant that could not confidently be classified as indigenous or non-indigenous were excluded from these analyses. All P -values less than 0.05 were considered statistically significant.

Results

Possum capture, garden use and home-range size

Of the 28 western ringtail possums captured in residential gardens, 17 were female and just over half (9) of these had pouch young. On average (\pm SE) captured adult individuals ($N = 26$) weighed 1276 ± 25.5 g (males = 1222 g, $N = 9$; females = 1299 g, $N = 17$). At the conclusion of the study, eight of the collared individuals had gained weight, the weight of two were unchanged and three had lost weight (although this loss was ≤ 25 g). For male individuals (for which the presence or absence of pouch young did not affect changes in weight), there was an average weight gain of 43 ± 16.5 g and none of the individuals had lost weight ($N = 5$).

We obtained a total of 846 locations from 20 western ringtail possums over a period of 103 days. For the duration of the study all individuals exclusively resided within residential gardens or street verges, with no locations recorded from remnant bushland (Fig. 5.2). Therefore, proximity to remnant bushland had no influence on the proportion of observations located within residential habitats. Average home-range size of western ringtail possums was 0.51 ± 0.15 ha (min = 0.08 ha, max = 2.3 ha, $N = 15$) and was significantly larger for males (0.88 ± 0.06 ha, $N = 6$) than females (0.26 ± 0.06 ha, $N = 9$, $P = 0.018$). The home range of each possum overlapped with an average of 11 ± 2.5 gardens but some individuals used as few as two gardens while one individual used 41 gardens. Six of the 15 animals (40%) crossed at least one road during the study; however, no animal crossed more than two different roads (Fig. 5.2).



Fig. 5.2. Example home ranges (95% Kernel density estimate) of three of the studied western ringtail possums (*Pseudocheirus occidentalis*) captured in residential gardens in the City of Albany (each individual is represented by a different letter). Home ranges exemplify (1) no use of remnant habitat, irrespective of proximity to remnant bushland, (2) the crossing of roads by some individuals and (3) the range in number of gardens used by possums.

Structure use and behaviour

A total of 649 exact locations were recorded, of which the structure use and behaviour could be determined for 91% and 80% of sightings respectively. For those locations where the structure used was identified ($N = 588$), 75% of possum locations was on non-indigenous plants, 11% was on indigenous plants, 7% was on man-made structures, 1% was on the ground and 6% was on a plant for which the native status could not be determined. Possums were observed to use a variety of man-made structures including fences, buildings, power lines and roofs.

Of the locations where behaviour was identified ($N = 516$), 35% was foraging, 36% was day-time resting and 29% was a night-time non-foraging activity (i.e. movement, grooming, interactions with other individuals, sitting or defecating). Possums used non-indigenous plants more frequently than they used indigenous plants, man-made structures and the ground for each behavioural category ($160 < \chi^2 < 303$, d.f. = 3, $P < 0.001$, Fig. 5.3). Man-made structures were used during night-time non-foraging activities (20.5% of non-foraging locations), and occasionally as daytime refuges (8.1% of day-time resting locations; Fig. 5.3).

Non-foraging activities occurred on man-made structures more frequently than indigenous plants ($P = 0.006$) and ground habitat ($P < 0.001$; Fig. 5.3).

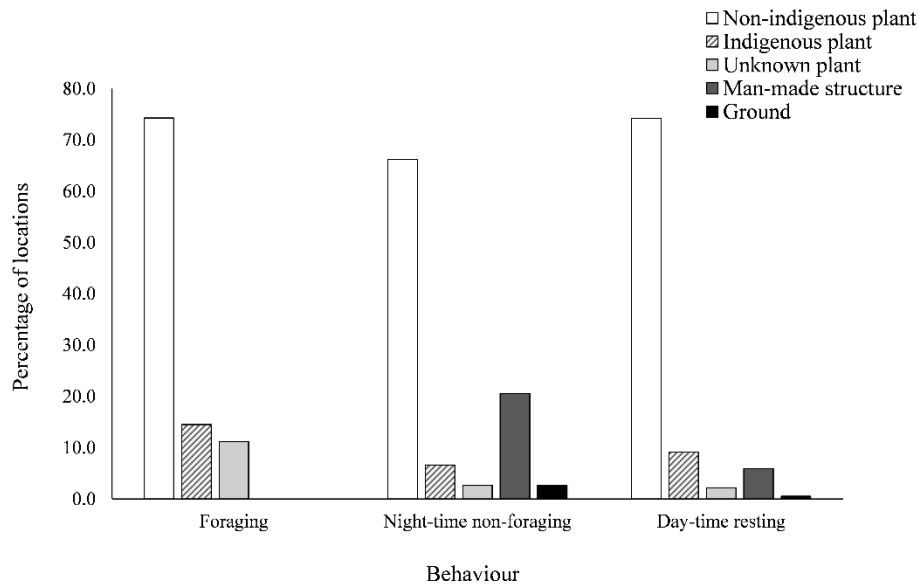


Fig. 5.3. Types of structures used within residential gardens by western ringtail possums (*Pseudocheirus occidentalis*) when exhibiting different behaviours. ‘Night-time non-foraging’ behaviours include moving, grooming, interactions with other individuals, sitting and defecating. ‘Man-made structure’ includes fences, buildings, power lines and roofs. Plants that could not be confidently categorised as non-indigenous or indigenous were grouped as ‘Unknown plant’.

Foraging

Vegetation within the home range of possums comprised primarily non-indigenous plants ($88 \pm 3.1\%$ of plants, mean \pm SE, $N = 6$). Non-indigenous and indigenous plants were foraged by possums in an equal proportion to their availability ($\chi^2 = 2.0$, $P = 0.849$, $N = 6$). Possums ($N = 20$) foraged on a total of 33 flora taxa and numerous additional unidentified non-indigenous species (Table 5.1). Of the 33 identifiable taxa, only eight were indigenous to the Albany region (Table 5.1). New Zealand Christmas trees (*Metrosideros excelsa*), lilly pilly trees (*Syzygium smithii*) and avocado trees (*Persea americana*) were the most commonly foraged plant species and were consumed by multiple individuals (Table 5.1).

Table 5.1. Non-indigenous and indigenous flora foraged by radio-tracked western ringtail possums (*Pseudocheirus occidentalis*). The number of times each flora species was foraged by possums (Frequency) and the percentage of locations this represented (%), and the number of individual possums observed to consume each species (Possum) in residential gardens in the City of Albany are shown. Flora that could not be confidently categorised as non-indigenous or indigenous are listed as 'Unknown'.

Status	Scientific name	Common name	Frequency (%)	Possum
Non-indigenous	<i>Bambusa sp.1</i>	Bamboo	1(0.6)	1
	<i>Callistemon spp.</i>	Bottlebrush	5(2.8)	3
	<i>Camellia sp.1</i>	Camellia	1(0.6)	1
	<i>Coprosma spp.</i>	Coprosma	8(4.5)	4
	<i>Cotoneaster spp.</i>	Cotoneaster tree	8(4.5)	2
	<i>Dombeya sp.1</i>	Dombeya	1(0.6)	1
	<i>Ficus macrophylla</i>	Morton Bay Fig tree	1(0.6)	1
	<i>Jacaranda spp.</i>	Jacaranda	2(1.1)	2
	<i>Malus spp.</i>	Crab apple tree	2(1.1)	1
	<i>Metrosideros excelsa</i>	New Zealand Christmas tree	27(15.3)	7
	<i>Olea europaea</i>	Olive tree	1(0.6)	1
	<i>Persea americana</i>	Avocado tree	18(10.2)	4
	<i>Pinus spp.</i>	Pine	3(1.7)	2
	<i>Plumbago spp.</i>	Plumbago	3(1.7)	1
	<i>Prunus armeniaca</i>	Apricot tree	2(1.1)	1
	<i>Prunus triloba</i>	Ornamental plum	2(1.1)	1
	<i>Pyrus nivalis</i>	Snow pear	1(0.6)	1
	<i>Rosa spp.</i>	Rose	2(1.1)	2
	<i>Schinus molle</i>	Pepper tree	1(0.6)	1
	<i>Syzygium smithii</i>	Lilly pilly	20(11.4)	4
	<i>Tecoma capensis</i>	Cape honeysuckle	2(1.1)	2
	<i>Ulmus parvifolia</i>	Chinese elm	2(1.1)	1
	Unidentified nonindigenous	20(11.4)	7	
Indigenous	<i>Agonis flexuosa</i>	Peppermint tree	12(6.8)	5
	<i>Allocasuarina fraseriana</i>	Western sheoak	2(1.1)	1
	<i>Banksia praemorsa</i>	Cut-leaf banksia	1(0.6)	1
	<i>Corymbia calophylla</i>	Marri	1(0.6)	1
	<i>Corymbia ficifolia</i>	Red flowering gum	2(1.1)	1
	<i>Eucalyptus conferruminata</i>	Bushy yate	6(3.4)	1
	<i>Eucalyptus leucoxydon</i>	Yellow gum	1(0.6)	1
	<i>Hakea laurina</i>	Pincushion hakea	1(0.6)	1
	Unknown	<i>Eucalyptus sp. 1</i>	Eucalypt	10(5.7)
<i>Grevillia sp.1</i>		Grevillia	1(0.1)	1
<i>Melaleuca spp.</i>		Paperbark	6(3.4)	5

Day-time refuges

Day-time refuge sites were identified for 63% ($N = 186$) of day-time possum locations. From these locations ($N = 121$), collared animals were recorded in five refuge types (dreys, hollows, ground nests, tree forks and roof/building cavities; Fig. 5.4). Non-indigenous vegetation was used for rest sites more frequently than indigenous vegetation ($P < 0.001$, Table 5.2) and man-made structures ($P < 0.001$). Man-made structures were used as frequently as indigenous vegetation for rest sites ($P > 0.05$). Possums used dreys ($n = 96$) more frequently than other refuge types ($\chi^2 = 281$, d.f. = 4, $P < 0.001$, Fig. 5.4). Building cavities were used as frequently as tree forks ($P > 0.05$), but were used more frequently than

hollows and ground nests ($P = 0.039$; Fig. 5.4). Hollows and ground nests were used only once and were recorded in indigenous plant species (Table 5.2). Collectively, 17 non-indigenous flora taxa (with additional unidentified non-indigenous species) and four indigenous taxa were used for day-time refuges (Table 5.2). Lilly pilly trees, ornamental plums (*Prunus triloba*) and peppermint trees (*Agonis flexuosa*) were the most commonly used plant species for day-time refuges.

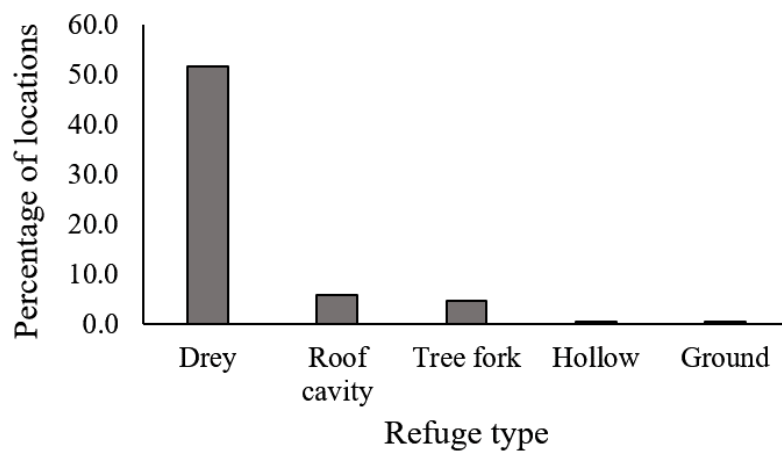


Fig. 5.4. Day-time refuge types used within residential gardens by western ringtail possums (*Pseudocheirus occidentalis*).

Table 5.2. Non-indigenous and indigenous flora used as day-time refuge sites by radio-tracked western ringtail possums (*Pseudocheirus occidentalis*). The number of times each flora species was used (Frequency) and the percentage of locations this represented (%), and the number of individual possums observed to rest in each species (Possum) in residential gardens in the City of Albany are shown. Flora taxa that could not be confidently categorised as non-indigenous or indigenous are listed as ‘Unknown’

Type	Status	Scientific name	Common name	Frequency (%)	Possum
Drey	Non-indigenous	<i>Arecaceae spp.</i>	Palm tree	2 (1.9)	1
		<i>Bambusa sp.1</i>	Bamboo	9 (8.4)	1
		<i>Callistemon spp.</i>	Bottlebrush	4 (3.7)	2
		<i>Cotoneaster sp.1</i>	Cotoneaster tree	1 (0.9)	1
		<i>Lonicera sp.1</i>	Honeysuckle	1 (0.9)	1
		<i>Metrosideros excelsa</i>	New Zealand Christmas tree	6 (5.6)	2
		<i>Morus sp.1</i>	Mulberry tree	5 (4.7)	1
		<i>Olea europaea</i>	Olive tree	3 (2.8)	2
		<i>Pinus sp.1</i>	Pine	1 (0.9)	1
		<i>Plumbago spp.</i>	Plumbago	8 (7.5)	1
		<i>Prunus triloba</i>	Ornamental plum	11 (10.3)	1
		<i>Pyrus nivalis</i>	Snow pear	1 (0.9)	1
		<i>Schinus molle</i>	Pepper tree	1 (0.9)	1
		<i>Syzygium smithii</i>	Lilly pilly	15 (14.0)	4
		<i>Tecoma capensis</i>	Cape honeysuckle	9 (8.4)	3
			Unidentified non-indigenous	4 (3.7)	3
			Indigenous	<i>Agonis flexuosa</i>	Peppermint tree
		<i>Banksia praemorsa</i>	Cut-leaf banksia	1 (0.9)	1
	Unknown	<i>Melaleuca spp.</i>	Paperbark	1 (0.9)	1
Ground	Indigenous	<i>Lepidosperma sp.1</i>	Sword rush	1 (0.9)	1
Hollow	Indigenous	<i>Corymbia calophylla</i>	Marri	1 (0.9)	1
Tree fork	Non-indigenous	<i>Citrus limon</i>	Lemon tree	1 (0.9)	1
		<i>Persea americana</i>	Avocado tree	1 (0.9)	1
		<i>Plumbago sp.1</i>	Plumbago	1 (0.9)	1
			Unidentified non-indigenous	5 (4.7)	3
	Unknown	<i>Melaleuca sp.1</i>	Paperbark	1 (0.9)	1

Discussion

Residential landscapes are thought to offer conservation opportunities for wildlife because numerous taxa occur in, and use the novel resources that these landscapes provide (Goddard et al., 2010; Ives et al., 2016; Soanes et al., 2019). However, to fully understand the conservation value of gardens, additional ecological knowledge of whether gardens can support animal residency and key demographic processes such as reproduction and survivorship is needed (Maclagan et al., 2018). Our study examined one of these key aspects. To the best of our knowledge, our study is the first attempt to explicitly investigate if residential gardens are capable of supporting native mammal residency, and one of few to examine native animal residency in gardens more broadly (although see Koenig et al., 2001). Contrary to our hypotheses, our results demonstrated that (1) for several months of their life, individual adult western ringtail possums exclusively resided within garden habitats regardless of their proximity to natural remnant vegetation, and (2) novel resources were used

more frequently than natural resources for foraging, non-foraging night-time activities and day-time refuges in these urban landscapes. Using a Critically Endangered arboreal marsupial as a case study, our results exemplify that residential gardens can exclusively support some adult individuals (including threatened species) within urban landscapes for at least several months of their life. This supports the argument that residential gardens have conservation value in their own right. Management initiatives that incorporate both natural (i.e. remnants) and novel (i.e. residential) habitats in urban landscapes may result in better conservation outcomes than those that continue to focus only on natural areas. This has important implications for the conservation of animal wildlife globally as urban landscapes harbour numerous threatened species worldwide (Aronson et al., 2014; Ives et al., 2016). In Australia alone, the distribution of nearly half of all animals of national conservation significance (46%, $N = 428$) overlap with cities, and all cities on the continent ($N = 99$) contain threatened species (Ives et al., 2016). Given the extensive number of species of conservation interest located within cities, our finding that gardens have the capacity to support native animal residency (at least for several months of their life) further highlights the substantial conservation opportunity that residential landscapes and cities more broadly offer for wildlife.

Our result of exclusive use of gardens highlights that the resources in residential gardens can be sufficient to support an individual's presence, irrespective of the surrounding natural remnant habitat. This is further supported by our result of small home ranges (0.51 ha) compared to those reported in natural bushland areas in the same season and same population of western ringtail possums (0.88 ha, Van Helden et al., 2018), given that smaller home ranges typically indicate better quality habitat (e.g. Bjørneraas et al., 2012). These findings are consistent with recent evidence that the presence of the western ringtail possum in residential areas is unrelated to proximity of remnant vegetation (Busschots et al., 2020; Van Helden et al., 2020a), but contrasts with extensive evidence that individuals depend on urban remnant vegetation when using residential areas (e.g. Davison et al., 2009; Fitzgibbon et al., 2011; Van Helden et al., 2018). One explanation for this contrasting result is that most studies that examine habitat use of animals within gardens have sourced individuals from non-garden habitats (e.g. Fitzgibbon et al., 2011; Van Helden et al., 2018), or have studied species with wide-ranging home ranges (e.g. Janko et al., 2012). For these individuals, resources in residential gardens likely supplement those obtained from remnant vegetation. Our study differed in that we examined a relatively sedentary species, with a comparatively small home

range (Van Helden et al., 2018; Yokochi et al., 2015) that we sourced from residential gardens. Although remnant vegetation is undoubtedly critical for some species, our results suggest that there may be individuals of species, that current methods largely neglect to sample, that obtain all their resource needs from residential landscapes and can reside exclusively within these areas. This may be particularly true for species with strong site fidelity and small home ranges as seen in the western ringtail possum and the blue-tongued lizard (*Tiliqua scincoides*; Koenig et al., 2001). Studies that investigate the capacity of other species and taxa to reside within residential environments long-term should be undertaken, and will greatly improve the understanding of the value of residential gardens for biodiversity conservation.

The availability of resources in a landscape can determine the spatial distributions of wildlife as animals tend to congregate in areas of high resource quality and quantity (as seen in large herbivore grazers, e.g. Bailey et al., 1996). We speculate that the ability of western ringtail possums to exclusively use residential areas for the duration of the study, and therefore obtain all their resource needs from these habitats, may be a result of: (1) the floristic diversity of indigenous and non-indigenous vegetation in gardens and/or (2) the artificially elevated nutrient and moisture content of vegetation which results from artificial watering and the application of fertilisers by residents. It is unclear from our study whether individual western ringtail possums are residing within residential areas because gardens offer foraging opportunities that are equal, or even better than urban remnant habitats, or because nearby remnants are already saturated in terms of population size and/or depleted of resources. Whether gardens are used by individuals preferentially to natural habitat or simply out of necessity warrants consideration to further shed light on the conservation value of gardens for wildlife.

The lack of foraging preference for indigenous species and the frequent use of novel resources was an interesting finding given that the western ringtail possum is thought to be a relatively specialist folivore and threatened by urbanisation (DPaW, 2017; Jones et al., 1994). Our finding is also unexpected based on the assumption that urban areas and novel resources are generally unsuitable for habitat specialists and those sensitive to habitat fragmentation (Kowarik, 2011). However, it supports recent evidence that the presence of western ringtail possums in gardens is unrelated to the nativeness of the vegetation (Van Helden et al., 2020a) and that the species is flexible in its behaviour and diet (Bader et al., 2019; Mathieson et al., 2020; Van Helden et al., 2018). Our findings contribute more evidence to recent suggestions

that the value of gardens is not limited to common species that are adapted to urban environments (e.g. Maclagan et al., 2018; Van Helden et al., 2020a; 2020b), but applies to a range of species, including those that are threatened, providing they are capable of some flexibility in resource use and behaviour (Lowry et al., 2013; Sol et al., 2013).

While this study demonstrates that residential areas are capable of supporting native animal residency with no association to nearby natural habitat, it is still unclear if individuals are capable of exclusively residing within gardens for the entirety of their life (~ 4-5 years for western ringtail possums). We recognise that remnant vegetation may be critical for processes operating at longer time-scales (e.g. recruitment, dispersal, colonisation), that sub-adult individuals may behave differently to adults, and that the suitability of residential gardens to support resident populations of wildlife may fluctuate seasonally and annually (Maclagan et al., 2018). We also acknowledge that the consumption of novel resources and additional threats in residential areas such as vehicle strike and predation from domestic pets (dogs and cats) may impart some risks (e.g. Murray et al., 2016). Indeed, even during our study one collared individual was killed from road strike and two were predated by pets. Lastly, animal occurrence and short-term residency within residential gardens is not indicative of long-term viability of populations within urban landscapes. For example, gardens may represent ecological ‘traps’ or ‘sinks’, where low survival and reproductive rates impede population viability over time (Battin, 2004; Donovan and Thompson III, 2001). Although we did not directly address these concerns in our paper, over half (53%, $N = 17$) of all the female possums we captured in gardens had pouch young and on average male individuals gained 43 g over the duration of the study. This provides some optimism that gardens may be capable of supporting animal residency and viable populations of at least some species long-term. Studies that examine long-term residency, body condition, survival and reproduction of individuals in garden habitats would further advance our understanding of these concerns in urban landscapes (Johnson, 2007; Van Horne, 1983).

Given our findings, conservation initiatives that promote the participation in wildlife-friendly gardening may not only increase native urban biodiversity (Goddard et al., 2010), but also assist populations to persist within urban landscapes. There is promising evidence that gardens could be manipulated by residents to achieve conservation outcomes in urban areas (Adams et al., 2020; Daniels and Kirkpatrick, 2006; Goddard et al., 2010; Soanes et al., 2019; Threlfall et al., 2016b; 2017; Van Helden et al., 2020a) and that residents are willing to engage in these activities (e.g. Goddard et al., 2013; van Heezik et al., 2012). Residents are

capable of achieving conservation outcomes in numerous ways including the planting and maintenance of suitable habitat structures, reducing the exposure of wildlife to domestic predators (dogs and cats) and provision of resources (e.g. water and rest sites such as nest boxes; Ikin et al., 2015). Management strategies that improve public awareness and provide incentives to engage in these types of activities will improve our ability to conserve wildlife within urban landscapes (Ikin et al., 2015).

Using a Critically Endangered arboreal marsupial as a model species, our study demonstrates that adult individuals can exclusively reside within residential gardens for several months of their life, with seemingly no dependence on nearby natural habitat. Our findings add to a growing body of evidence highlighting that gardens and other highly modified habitats can offer substantial conservation opportunities to wildlife, and that these habitats should not be neglected in favour of assumed, 'better quality' natural remnant habitat. Research that investigates the ability of individuals to reside long-term, and the transferability of our findings to other species and taxa would further advance our current understanding of the value of residential areas for wildlife conservation. As residential gardens provide an extensive greenspace that wildlife can both use and reside in, conservation and management initiatives that include gardens and other small, modified greenspaces may result in better conservation outcomes in urban landscapes.

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CHAPTER 6

THE IMPACT OF URBANISATION ON THE GENETICS OF A THREATENED SPECIES THAT SEEMINGLY THRIVES IN URBAN LANDSCAPES



Photo: An adult and juvenile western ringtail possum (*Pseudocheirus occidentalis*) hanging out in a peppermint tree (*Agonis flexuosa*).

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Abstract

As modified landscapes continue to replace natural ecosystems worldwide, there is an increasing need to conserve wildlife in novel habitats. The contemporary presence of wildlife in cities, and the ability of some to thrive in these areas, provides promise that urban landscapes offer opportunities for biodiversity conservation. However, failure to consider processes that operate over longer-time scales (e.g. gene flow, genetic structuring), and their subsequent influence on population persistence, may result in misinterpreting the true conservation value of these landscapes. In this study, we used Bayesian clustering and spatial autocorrelation analyses to investigate the impact of an urban landscape on the gene flow and genetic diversity of a Critically Endangered marsupial, the western ringtail possum (*Pseudocheirus occidentalis*), that seemingly thrives in city landscapes. We found that genetic differentiation between identified genetic clusters was low (pairwise $F_{ST} \leq 0.05$) and gene diversity within the city ($H_e = 0.61-0.69$) was comparable with that reported from populations in natural parts of the species range ($H_e = 0.56-0.79$). Genetic correlation between individuals spanned a greater distance in residential areas (> 3300 m) than bushland areas (< 600 m) suggesting that the movement of genes within the residential landscape may be greater than in bushland habitat. These findings suggest that urbanisation has had minimal negative impact on the population genetics of a Critically Endangered marsupial. This highlights that city environments are capable of supporting the evolutionary processes of at least some threatened species, and provides further support that urban areas have conservation value for wildlife.

Introduction

Modified landscapes are being created worldwide as anthropogenic activity such as urbanisation and deforestation continue to rapidly disfigure natural areas (Lugo, 2011). Whilst conservation approaches once prioritised natural areas for the protection of wildlife (Wuerthner et al., 2014), there is now recognition that modified landscapes could also play a role in conservation, and that neglecting these areas may represent a missed opportunity in the battle against biodiversity loss (Radeloff et al., 2015). Urban environments offer considerable opportunities for conservation because cities are often located in areas of high biodiversity (Luck, 2007) and can provide a structurally complex and diverse array of habitats (Aronson et al., 2017), in which numerous species, including those that are threatened, occur (e.g. Ives et al., 2016; Soanes and Lentini, 2019). In some cases, these areas are inhabited by high densities of species that are capable of reproducing and residing; allowing them to seemingly thrive in urban landscapes (Hubert et al., 2011; Maclagan et al., 2018; Van Helden et al., 2020c). Based on this contemporary ecological evidence, urban landscapes are thought to have substantial conservation value for wildlife, prompting a call to include urban environments in conservation policy and practice (Aronson et al., 2017; Soanes and Lentini, 2019; Soanes et al., 2019).

Whilst the contemporary presence of wildlife in cities provides promise that urban landscapes have value for biodiversity conservation, failure to consider the impacts of urbanisation on evolutionary processes that influence the persistence of wildlife populations over longer-time scales, may result in misinterpreting the true conservation value of these novel landscapes. Urbanisation can impact upon evolutionary processes through its influence on gene flow, genetic drift, natural selection and mutation rates among wildlife populations (see Miles et al., 2019; Rivkin et al., 2019 for review). This can decrease the ability of species to adapt to environmental change, and can result in inbreeding depression, loss of fitness and the accumulation of deleterious mutations (Gaggiotti, 2003); increasing the species vulnerability to extinction (Frankham, 2005). For instance, genetic heterozygosity positively correlates with population fitness (Reed and Frankham, 2003) and is generally lower in threatened taxa compared to non-threatened taxa (on average 35% lower) (Spielman et al., 2004). While at face-value species may be thriving in urban areas, if underlying changes to evolutionary processes are impacting the capacity of wildlife to persist in the long-term (Frankham et al., 2002), the conservation value of urbanised habitats may be overestimated. To gain a more complete picture of the impact of urbanisation on wildlife and the true conservation value of

these landscapes, the impact of urban areas on both ecological patterns and evolutionary processes is needed.

The impact of urbanisation on evolutionary processes is by no means consistent across taxa. Urban landscapes can impede animal movement and subsequently restrict gene flow among (and within) populations resulting in loss of genetic diversity and increased genetic differentiation among populations (Johnson and Munshi-South, 2017). For other species, urbanisation has seemingly little impact on genetic structure or diversity (e.g. Atterby et al., 2015; Khimoun et al., 2020), and in some cases urban areas can even facilitate gene flow and high genetic diversity (e.g. Miles et al., 2018a; 2018b; Tang et al., 2015). The differential patterns of genetic drift and gene flow among species is thought to be driven by an interaction between their biological attributes (e.g. life-history, dispersal ability, physiology) and the environmental conditions of the city (e.g. intensity of urbanisation, population size) (Miles et al., 2019). For example, species that are poor dispersers and sedentary are expected to be more sensitive to urbanisation, exhibiting a greater loss of genetic diversity and gene flow than more-mobile species with larger dispersal ranges (Johnson and Munshi-South, 2017; Miles et al., 2019). Whilst this hypothesis is supported by multiple case studies of species with limited dispersal (Hitchings and Beebee, 1997; Munshi-South and Kharchenko, 2010; Noël et al., 2007), there are also examples of highly mobile species that are negatively impacted by urbanisation (e.g. moose (*Alces alces*); Wilson et al., 2015). This highlights that the factors that influence species responses to urbanisation is complex, and difficult to generalise among species even with similar biological attributes.

As urban landscapes harbour numerous threatened species worldwide (Aronson et al., 2014; Ives et al., 2016) understanding the impacts of urbanisation on both contemporary distribution and abundance, and evolutionary processes has never been more critical for the conservation of these species. For example, in Australia alone, the distribution of nearly half of all animals of national conservation significance (46%, $N = 428$) overlap with cities, and all cities on the continent ($N = 99$) contain, or are likely to contain threatened species (Ives et al., 2016). Whilst there is growing evidence that some threatened species can occur in high densities, reproduce and otherwise do seemingly well in urban habitats (Maclagan et al., 2018; Van Helden et al., 2020b; 2020c), the impact of urbanisation on the evolutionary processes of these species has been scarcely examined. This represents a real risk in implementing the call for practitioners and decision makers to undertake conservation action in urban environments (Goddard et al., 2010; Soanes et al., 2019; Tulloch et al., 2016).

The Critically Endangered western ringtail possum (*Pseudocheirus occidentalis*) (*Biodiversity Conservation Act 2016, IUCN Red List*) represents a prime example of a highly threatened species that is thought to thrive in urban areas (Van Helden et al., 2020a; 2020b; 2020c) despite little knowledge of the impact of urbanisation on its genetic diversity and structure. Although a reasonably specialist folivore that feeds primarily on understorey or canopy vegetation, the possum exhibits flexibility in both its diet (Mathieson et al., 2020) and behaviour (Bader et al., 2019; Van Helden et al., 2018; 2019). These traits are thought to contribute to the species' abundant and widespread presence in residential landscapes where it makes use of the available novel resources (e.g. fences, roofs, exotic plants) (Thompson and Thompson, 2009; Van Helden et al., 2019; 2020a; 2020b; 2020c). The western ringtail possum has a very sedentary nature with unusually small home ranges (generally < 1 ha), that tend to be smaller in modified urban habitats (residential areas) than natural bushland habitats (Chandler et al., 2020; Van Helden et al., 2018; 2020c; Yokochi et al., 2015). While the species is capable of dispersing over distances equivalent to several home ranges where possum density is high (DPaW, 2017), individuals typically disperse over short distances and establish home ranges adjacent to or overlapping the natal range (Jones et al., 1994). While these latter ecological traits (e.g. sedentary nature and short dispersal range) suggest that the possum may be negatively impacted by a city landscape as other species with similar traits have been (e.g. Noël et al., 2007), its behavioural flexibility and widespread presence in urban areas may suggest otherwise. Although urban infrastructure such as roads and artificial waterways have been shown to impede the species' movement (Yokochi et al., 2015), there is yet to be a study that explicitly examines the genetic structure of the western ringtail possum within a city landscape.

Using the western ringtail possum as a case study species, we aimed to investigate the impact of an urban area on the gene flow, genetic structure and genetic diversity of a highly threatened species that seemingly thrives in city landscapes (based on its contemporary presence and abundance). Despite its short dispersal distances (Jones et al., 1994), sedentary nature and vulnerability to urban threats (e.g. road strike and predation from pets) (DPaW, 2017), we predicted that the urban landscape would have minimal negative impact on its genetic structure and diversity because of its widespread presence in urban areas and ability to use urban infrastructure and resources (e.g. fences, power lines, roofs, exotic plants). More specifically, we hypothesised that there would be multiple genetic clusters present within the urban environment because of the philopatric nature of the species, but that the genetic

differentiation between clusters would be low due to adequate gene flow. We also predicted that the genetic diversity of western ringtail possums within our urban study area would be similar to previously studied populations (and other mammal species) in natural bushland areas, and that the spatial genetic structure of the species would be similar in residential habitat compared to urban bushland habitat. Despite being essential to evaluate the conservation value of urban areas for wildlife, our study is one of few to consider the longer-term impacts of urbanisation on threatened species that seemingly thrive in city landscapes.

Methods

Study species

The Critically Endangered western ringtail possum is a small (~ 1kg), arboreal marsupial that is now largely restricted to three known disjunct populations in southwest Australia (South Coast, Swan Coastal Plain and Southern Forest population), all of which at least partially overlap urban centres (DPaW, 2017). Within two of its remaining populations (South Coast and Swan Coastal Plain) the majority of the species distribution overlaps with urbanised habitats and other non-protected areas (DPaW, 2017). The western ringtail possums occurs in both remnant natural habitat and highly modified habitats such as residential gardens, road verges and parks within urban landscapes (Busschots et al., 2020; Thompson and Thompson, 2009; Van Helden et al., 2020a; 2020b). Individuals are thought to live for up to five years (Wayne et al., 2005), with young dispersing at eight to 12 months old and becoming reproductively active at less than 12 months of age (DPaW, 2017). The species has exhibited relatively rapid genetic subdivision across its range (< 200 years and over distances as small as 30 km) (Wilson, 2009), and consistent with its philopatric nature, shows positive spatial genetic structuring at distances as small as 600 m within continuous bushland habitats (Yokochi et al., 2016).

Study location

We conducted this study within the urban confines (~ 90 km²) of the City of Albany (~ 4300 km²; 34° 58' S, 117° 52' E); a city that is located within one of the three remaining populations of the western ringtail possum in southwest Australia. We defined the 'urban confines' as any combination of land use categories identified by the City of Albany as general industry, highway commercial, hotel/motel, light industry, local and neighbourhood centre, port industry, rail, regional centre (including mixed business and mixed use) and residential (special or tourist) (Busschots et al., 2020). Albany was founded in the early 1800s

by European settlers and currently supports a population of approximately 31 500 people in 12 800 private dwellings and has a population density of approximately 3.5 persons per ha (or an average of 10.5 persons per hectare when excluding land uses other than residential areas) (ABS, 2016; CoA, 2018). Although the city is rapidly expanding, it is relatively ‘green’ with numerous bushland remnants of varying topography, vegetation composition and disturbance within the urban landscape (between < 1 ha and 600 ha (Busschots et al., 2020; Van Helden et al., 2018)), and has a number of larger national parks and reserves surrounding the city boundary (i.e. within 40 km of the city center) (Fig. 6.2a).

Sampling and microsatellite genotyping

We obtained ear tissue samples from 138 live-captured ($N = 20$), dead ($N = 102$), injured or sick ($N = 16$) western ringtail possums (58 females, 64 males, 16 unknown sex) found at various locations (recorded as latitude and longitude) within urban ($N = 95$), semi-rural ($N = 8$) and bushland remnant ($N = 35$) areas of Albany between July 2016 and November 2018. Ear tissue was stored in Longmire’s solution at room temperature until DNA extraction. Individuals were genotyped for 12 species-specific polymorphic, dinucleotide microsatellite loci (A1, A127, A106, A6, A2, D113, A122, D104, C111, A119, D114, B104) previously isolated by Wilson et al. (2009). DNA was extracted using the QIAGEN DNeasy Blood and Tissue kit (Qiagen, Hilden, Germany). Samples were quantified using a NanoDrop ND-1000 spectrophotometer (Thermo Fisher Scientific Inc, Massachusetts, USA) and diluted to 10ng/μl. The twelve microsatellite loci were amplified using the QIAGEN Multiplex PCR kit as described in Wilson et al. (2009). PCR products were analysed on an ABI3730XL Sequencer using Genescan-500 LIZ dye standard (Applied Biosystems Inc, California, USA) and scored using GeneMarker V1.91 (SoftGenetics, LLC, USA). Samples were randomised on the genotyping plate to avoid genotyping bias (Meirmans, 2015). One of the 138 samples failed to amplify and was removed from subsequent analyses.

Genetic structure

We used two Bayesian approaches to evaluate the genetic structure and optimal number of genetic clusters (K) of western ringtail possums in Albany; one based only on genetic data and one based on both genetic and geographic information. We first used STRUCTURE V. 2.3.4 (Pritchard et al., 2000) to assign individuals to a genetic cluster based on allele frequencies that minimised departures from Hardy-Weinberg equilibrium and linkage disequilibrium (Pritchard et al., 2000) without *a priori* knowledge of geographic location. We

used options for correlated allele frequencies (Falush et al., 2003) and genetic admixture among populations because we had no evidence of discrete populations based on the locations of sampled individuals. To infer the number of genetic clusters, ten replicate runs for each value of $K = 1$ to $K = 15$ with burn-in periods of 100 000 and Markov chain Monte Carlo (MCMC) lengths of 500 000 were used to calculate the log posterior probabilities of data [$\ln P(D)$] based on Pritchard et al. (2010) suggested settings. We then determined the most likely value of K by using the ΔK method described by Evanno et al. (2005). Based on the 10 replicates of the most likely K value, western ringtail possums were assigned to a cluster if they had a mean probability of membership to that cluster (q -value) ≥ 0.8 . Individuals with a q -value between 0.19 and 0.79 were considered mixed or hybrid animals. We then used Geneland (Guillot et al., 2005a; 2005b; 2008) and R V. 3.6.1 (R Core Development Team, 2016) to determine the number of genetic clusters based on both genetic and spatial data (i.e. geographic location of each genotype). With a minimum $K = 1$ and maximum $K = 15$, the MCMC analysis was run 10 times (100 000 iterations, 100 thinning) with 10 m uncertainty of geographical coordinates (to account for GPS error), using the correlated allele frequencies model to infer the most likely value of K (determined as the modal K value). Assignment of genotypes into clusters was determined from the run with the the highest average posterior probability as suggested by The Geneland Development Group and Guillot (2012).

We tested for correlation between pairwise genetic and geographic distances (i.e. isolation by distance, IBD) using mantel tests (Mantel, 1967) with 999 permutations performed in GenAIEx V. 6.5 (Peakall and Smouse, 2006; 2012). We conducted mantel tests within each genetic cluster identified by STRUCTURE and Geneland (that had sample sizes ≥ 15), and across all sampled genotypes.

Genetic diversity

We used the program MicroChecker V. 2.2.3 (Van Oosterhout et al., 2004) to test for null alleles, large allele dropout and stuttering for each loci (1000 iterations) across all samples and within the broadest groupings of inferred genetic clusters (i.e. STRUCTURE clusters, see *Landscape-scale genetic structure*). We eliminated two loci (A1 and A122) that showed consistent evidence of null alleles across the entire sample of genotypes and within each genetic cluster. All further analyses were undertaken using the remaining 10 loci and the previously described genetic structure analyses (see *Genetic structure*) were repeated after

eliminating the two problem loci. All reported results are based on the 10 loci. Each locus across all samples and within each cluster was tested for departures from Hardy-Weinberg equilibrium with Bonferroni adjustment using GenAIEx V. 6.5. We also tested for linkage disequilibrium between all pairs of loci in each cluster and across all samples with a Bonferroni adjustment using Genepop V. 4.2 (Rousset, 2008).

To evaluate the genetic diversity of western ringtail possums we calculated the mean number of alleles (N_a) and mean observed and expected heterozygosity (H_o and H_e) across all loci for each cluster and for the entire sample using GenAIEx V. 6.5. We also used GenAIEx V. 6.5 to calculate the number of private alleles (A_p) for each cluster. Because the observed number of alleles in a cluster is dependent on sample size, we calculated the allelic richness for each cluster adjusted to the lowest common sample size (A_r) using FSTAT V 2.9.4 (Goudet, 1995). We calculated the inbreeding coefficient (F_{IS}) for each cluster and for the entire sample in FSTAT V. 2.9.4. We calculated pairwise F_{ST} values between each genetic cluster using GenAIEx V. 6.5.

Spatial autocorrelation

To compare the spatial genetic structure of western ringtail possums in residential areas to natural habitat we examined the pairwise genetic correlation of individuals using two spatial autocorrelation approaches described by Peakall et al. (2003). We first examined how the pairwise genetic correlation (r) of individuals changed at increasing distance increments of 200 m (e.g. 0-200, 200-400, 400-600 m) using 999 bootstraps and 999 permutations (hereafter referred to as 'spatial autocorrelation (SA) analysis'). We selected a distance class size of 200 m to maximise the sample size in each class whilst still examining the structure at a fine scale. We also used the multiple distance class (MDC) approach which calculates r between individuals with increasing size of distance class (e.g. 0-100, 0-200, 0-300 m). The distance class at which r no longer significantly differs from zero represents the distance at which positive spatial genetic structure can be detected (Peakall et al., 2003). We performed MDC analyses each with 999 permutations, 999 bootstraps and even distance class sizes of 100 m. We undertook SA analyses and MDC analyses on a subset of individuals sampled from continuous bushland habitat (Mount Clarence [$N = 28$]) and two subsets of individuals located within continuous residential habitat (Urban one: $N = 28$, and Urban two: $N = 25$). We restricted all analyses to a distance of 3.3 km as this was the maximum distance between individuals within the bushland habitat. For SA analyses we only present data between 0 – 3

km as estimates of r for distance classes above 3 km was based on too few permutations for robust estimates. Mount Clarence was the only continuous bushland habitat within our study area that had enough samples to conduct MDC and SA analyses. We used only two urban subsets of data as they were the only subsets of individuals with comparable sample sizes (i.e. ≥ 25 samples) where pairwise distances between individuals were only calculated in residential habitat (i.e. no distances were calculated across agricultural or bushland areas).

Results

Genotyping data quality

All 10 loci from the 137 genotyped western ringtail possums were polymorphic with 3-14 alleles per locus. There was no evidence of stuttering or large allelic drop out at any loci. Null alleles were detected at several of the 10 loci (A127, A106, A6, D113, D104, A119, B104), significant departures from Hardy-Weinberg equilibrium were apparent for five of the ten loci (A127, A106, A2, D113, D114) and significant linkage disequilibrium was evident for three (6.7%) of the loci combinations (A127/A106, A2/B104, D104/B104) for the entire sample. When these analyses were undertaken within the broadest inferred genetic clusters (STRUCTURE clusters, see *Landscape-scale genetic structure*), evidence of null alleles was detected at one locus in both Cluster A (A6) and Cluster B (A106). There were significant deviations from Hardy-Weinberg equilibrium at two loci in Cluster A (A127, A6), but none in Cluster B. Similarly, four locus combinations showed evidence of linkage disequilibrium in Cluster A (A127/A6, A6/D104, D113/D104, D113/D114) whilst there was no evidence of linkage disequilibrium in Cluster B. As there was no consistent pattern of null alleles, linkage disequilibrium or Hardy-Weinberg disequilibrium for any particular locus in both genetic clusters, all ten loci were retained for further analyses.

Landscape-scale genetic structure

Based on evaluation of ΔK , STRUCTURE identified two genetic clusters (hereafter labelled Cluster A and B) of western ringtail possums within Albany (Fig. 6.1) Forty-seven (34 %) individuals belonged to Cluster A, 52 (38%) belonged to Cluster B and 38 (28 %) were mixed or hybrid animals ($0.19 < q < 0.79$). The high number of mixed individuals was consistent with the low level of genetic differentiation between the two clusters (pairwise $F_{ST} = 0.054$), indicating reasonable admixture (i.e. gene flow) between clusters. Despite the exclusion of spatial data in the STRUCTURE analysis, there was a strong geographic pattern in the distribution of individuals belonging to either Cluster A or B (Fig. 6.2b). Cluster A

occupied a broader geographical area than Cluster B and included individuals distributed throughout the residential suburbs of Albany and in semi-rural areas outside of the city (Fig. 6.2b). Members of Cluster B were predominantly distributed on or within 1.5 km of Mount Clarence; one of the largest remaining natural bushland remnants within urban Albany (Fig 2b). Only four individuals belonging to Cluster B were found outside this area and interestingly these individuals were located within 500 m of wildlife carers that are known to release rehabilitated western ringtail possums either on or nearby to their property. Western ringtail possums of mixed genotype ($0.19 < q < 0.79$) were scattered throughout most of the study area and overlapped with both clusters (Fig. 6.2b).

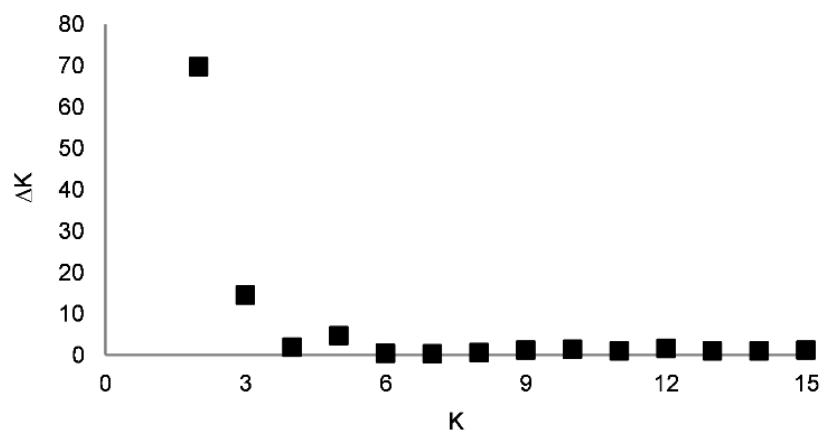


Fig. 6.1. Calculated ΔK values (Evanno et al., 2005) for $K = 2$ to $K = 15$, derived from the STRUCTURE analysis of 137 western ringtail possums (*Pseudocheirus occidentalis*) within urban Albany. Peak ΔK value occurred at $K = 2$, indicating that two genetic clusters are the optimal number of population clusters within Albany.

With the inclusion of spatial information, the clustering program Geneland identified $K = 10$ as the most likely number of genetic clusters of western ringtail possums in Albany (hereafter labelled as Clusters 1 - 10). Four of these clusters (1, 5, 7, 10) consisted of ≥ 15 individuals, whilst the remaining six clusters had ≤ 5 individuals (Table 6.1). Most of the Geneland clusters were spatially grouped with the exception of Cluster 2 and 5 which were less geographically restricted than the other clusters (Fig. 6.2c). Geneland Cluster 7 was consistent with STRUCTURE Cluster B, with individuals belonging to this cluster predominantly distributed on or within 1.5 km of Mount Clarence (Fig. 6.2b, 2c). Clusters 1, 3, 4, 9 and 10 generally formed small, geographically separated groups in the residential

landscape, whilst Clusters 6 and 8 were geographically grouped in semi-rural areas of Albany (Fig 2c).

There was evidence of IBD when all genotypes were included ($r = 0.27$, $P = 0.001$). Within STRUCTURE clusters, IBD was only detected in Cluster A ($r = 0.36$, $P = 0.001$), whilst in Geneland clusters there was only evidence of IBD in Cluster 10 ($r = 0.32$, $P = 0.01$).

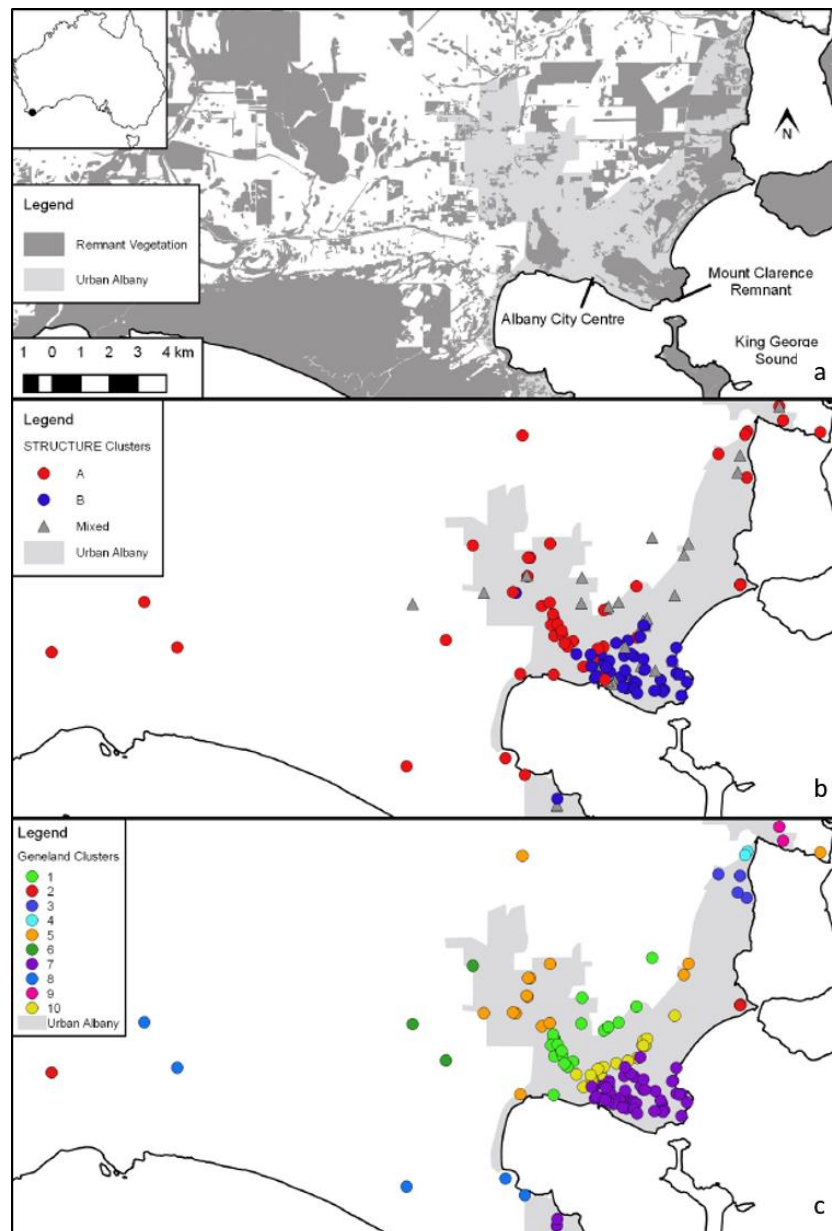


Fig. 6.2. The study area including urban Albany, bushland remnants and location of Mount Clarence remnant (a); and the location of genetic clusters of western ringtail possums (*Pseudocheirus occidentalis*) derived from the STRUCTURE clustering program ($K = 2$, $N = 137$; b) and Geneland clustering program ($K = 10$, $N = 137$; c). Remnant vegetation layer sourced from Busschots et al. (2020) and Sandiford and Barrett (2010).

Genetic diversity

Overall, the population of western ringtail possums in Albany was moderately diverse (mean $H_e = 0.704$, $H_o = 0.62$, $N_a = 8.4$) with moderate levels of inbreeding ($F_{IS} = 0.121$). Genetic diversity was moderate for both STRUCTURE clusters, although H_e , A_r and levels of inbreeding was higher in Cluster A than Cluster B (Table 6.1). Both clusters contained private alleles however there were substantially more in Cluster A. Genetic diversity and genetic differentiation between clusters was calculated only for the four Geneland clusters with large enough sample sizes ($N \geq 15$) to provide reasonable estimates (Table 6.1). There was low levels of genetic differentiation between the four Geneland clusters (pairwise $F_{ST} < 0.05$; Table 6.2) and genetic diversity was moderate for all Geneland clusters, albeit slightly higher for Cluster 1 and 5 than for Cluster 7 and 10 (Table 6.1). Each cluster contained a small number of private alleles and inbreeding coefficients were generally low for all four clusters (Table 6.1).

Table 6.1. Summary of genetic diversity of western ringtail possums (*Pseudocheirus occidentalis*) according to genetic clusters identified by both STRUCTURE and Geneland. N = number of genotypes per cluster; N_a = mean number of alleles (\pm S.E); A_r = mean allelic richness (\pm S.E) adjusted to a minimum common sample size of 46 for STRUCTURE clusters and 15 for Geneland clusters; A_p = number of private alleles; H_o = mean observed heterozygosity (\pm S.E); H_e = mean expected heterozygosity (\pm S.E); F_{IS} = inbreeding coefficient. Geneland cluster 2, 3, 4, 6, 8 and 9 are not shown because they had sample sizes ≤ 5 .

Program	Cluster	N	N_a	A_r	A_p	H_o	H_e	F_{IS}
Structure	A	47	7.8 ± 0.89	7.79 ± 2.81	25	0.634 ± 0.05	0.712 ± 0.05	0.12
	B	52	5.6 ± 0.70	5.56 ± 2.18	3	0.605 ± 0.06	0.643 ± 0.07	0.07
Geneland	1	20	5.8 ± 0.61	5.5 ± 0.59	1	0.665 ± 0.07	0.670 ± 0.07	0.03
	5	15	5.9 ± 0.67	5.9 ± 0.67	2	0.687 ± 0.05	0.693 ± 0.04	0.04
	7	59	6.3 ± 0.63	5.2 ± 0.53	4	0.609 ± 0.06	0.656 ± 0.04	0.08
	10	21	5.2 ± 0.47	4.9 ± 0.45	1	0.610 ± 0.05	0.668 ± 0.05	0.11

Table 6.2. Pairwise genetic differentiation between western ringtail possum (*Pseudocheirus occidentalis*) genetic clusters identified by Geneland.

Cluster	1	5	7	10
1	-			
5	0.035	-		
7	0.047	0.040	-	
10	0.040	0.032	0.019	-

Spatial autocorrelation

Fine-scale spatial genetic structure was detected in both bushland habitat (Fig. 6.3a, 4a) and residential areas (Fig. 6.3b, 3c, 4b, 4c). SA analyses indicated significant positive genetic correlation up to 200 m in both bushland and residential habitats (Fig. 6.3). In the bushland habitat r intercepted zero at a much shorter distance (384 m) compared to both residential areas (790 m and 1429 m) and was significantly negative at a shorter distance (800 m) compared to both residential areas (2800 m) (Fig. 6.3). There was a general decline in r between 200 and 3000 m in both residential areas, whereas in bushland areas there was a decline between 200 and 800 m before little change between 1000 and 3000 m.

MDC analyses indicated the detectable extent of positive spatial genetic structure of western ringtail possums was up to 600 m in natural bushland habitat (Fig. 6.4a), and exceeded 3.3 km in residential habitat (Fig. 6.4b, 4c). Individuals in bushland environments also exhibited consistently lower genetic correlation values than both residential environments for each distance class size (Fig. 6.4).

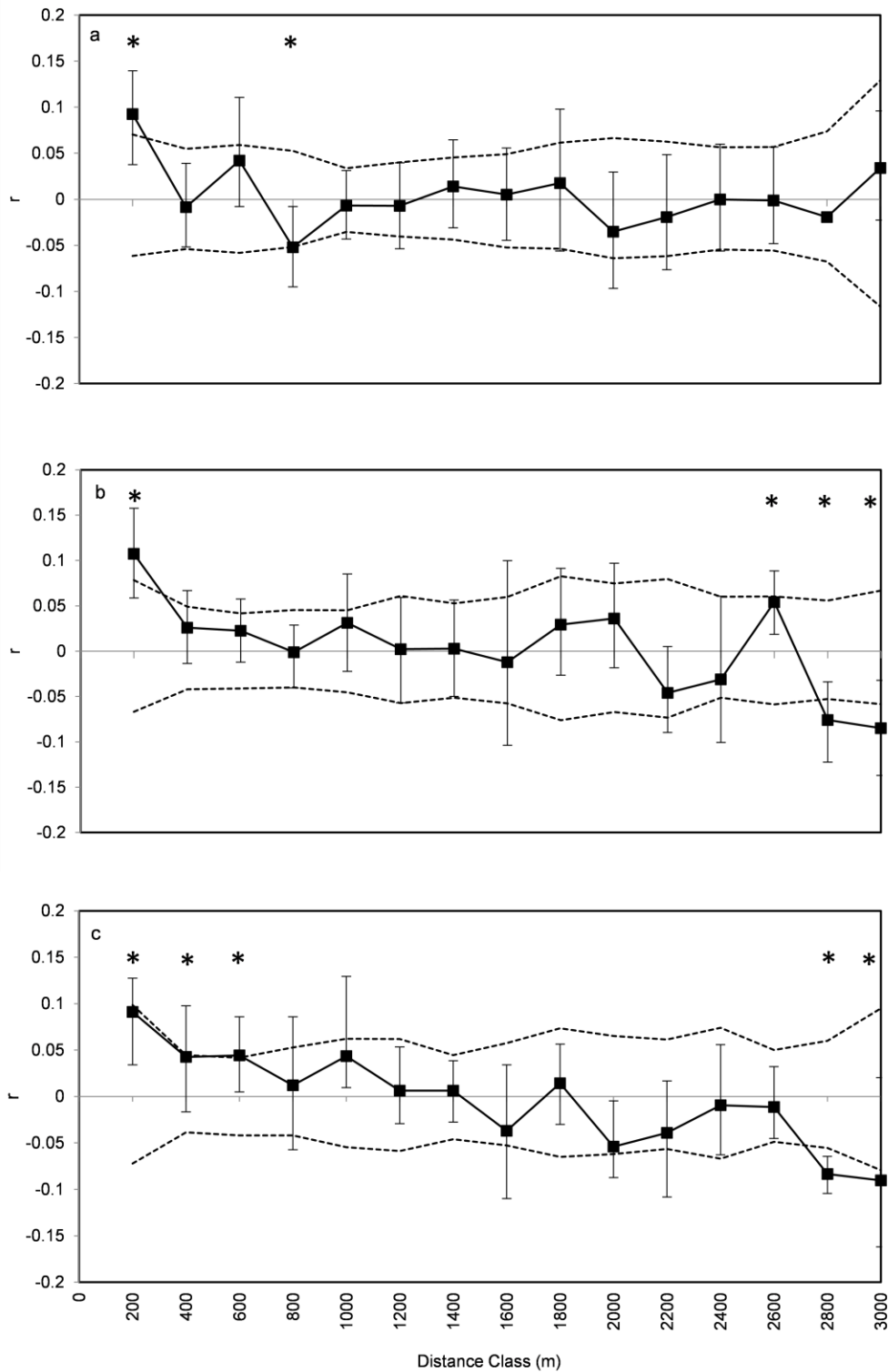


Fig. 6.3. The genetic correlation (r , black squares) of western ringtail possums (*Pseudocheirus occidentalis*) at incremental distance classes of 200 m in natural bushland habitat (a [$N = 28$]) and residential habitat (b [$N = 28$]) and c [$N = 25$]) within urban Albany. Bars on graph represent the 95% confidence interval about r as determined by bootstrapping. Dotted lines represent the 95% confidence interval of zero genetic correlation for each distance class. Distance classes where r significantly differs from zero ($P < 0.05$) are denoted by ‘*’.

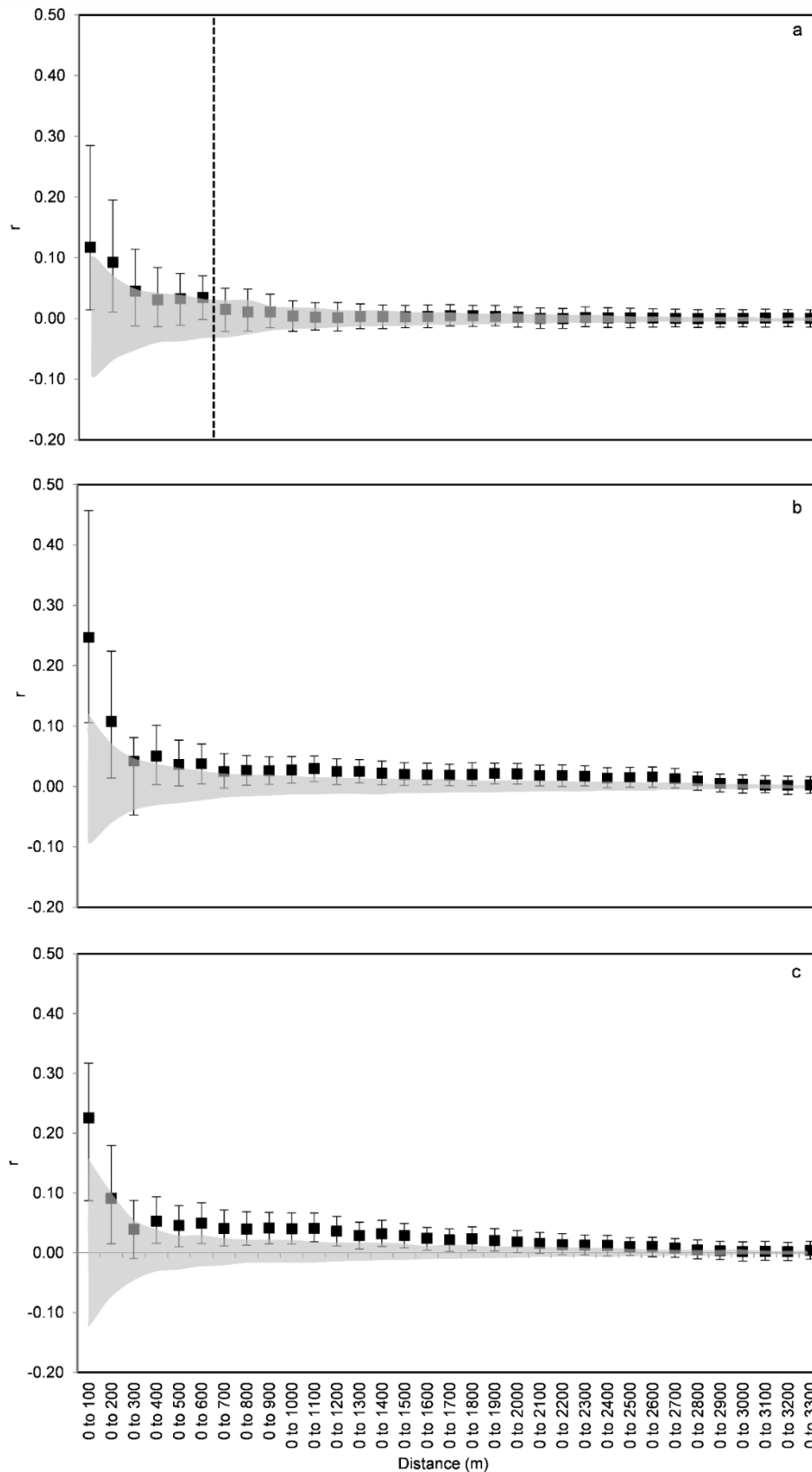


Fig. 6.4. The genetic correlation (r , black squares) of western ringtail possums (*Pseudocheirus occidentalis*) at increasing distance class sizes in natural bushland habitat (a [$N = 28$]) and residential habitat (b [$N = 28$]) and c [$N = 25$]) within urban Albany. Bars on graph represent the 95% confidence interval about r as determined by bootstrapping. Grey shading represents the 95% confidence interval of zero genetic correlation for each distance class. The broken line represents the extent of spatial correlation (m) determined as the last class size where r significantly differs from zero ($P < 0.05$).

Discussion

Based on contemporary ecological evidence, urban landscapes are thought to have substantial conservation value for wildlife, prompting a call to include urban environments in conservation policy and practice (Aronson et al., 2017; Soanes and Lentini, 2019; Soanes et al., 2019). However, the conservation value of these areas may be overestimated and conservation efforts misguided if we fail to also consider the impacts of urbanisation on evolutionary processes that may influence the persistence of wildlife populations over longer-time scales. In this study we investigated the impact of urbanisation on the population genetics of a Critically Endangered species that seemingly thrives in city landscapes (Van Helden et al., 2020b; 2020c) to better establish the value of urban areas for threatened species conservation. In support of our predictions and consistent with the widespread presence of western ringtail possums in residential areas (Thompson and Thompson, 2009; Van Helden et al., 2020a; 2020b), we found genetic differentiation between identified clusters was low, with limited evidence of inbreeding in any cluster, and that the gene diversity ($H_e = 0.61-0.69$) within clusters was comparable with that reported from populations in natural bushland in other parts of the species range ($H_e = 0.60-0.79$, Wilson, 2009; $H_e = 0.56-0.64$, Yokochi et al., 2016). We also found that the positive genetic correlation between individuals spanned a far greater distance in residential areas than bushland areas suggesting that the movement of genes within residential areas may be greater than in remnant natural habitat. These findings suggest that urbanisation has had minimal negative impact on the genetic structure, gene flow and genetic diversity of the species within a city landscape despite its vulnerability to the threats associated with urbanisation (DPaW, 2017) and propensity for limited dispersal (Jones et al., 1994). Our findings highlight that city environments are capable of supporting the evolutionary processes of at least some threatened wildlife. This provides further support that urban areas have conservation value for threatened species and that urban areas should be included in conservation initiatives.

Although genetic structuring of western ringtail possums was evident within Albany, the genetic subdivision between clusters was low (pairwise $F_{ST} \leq 0.05$) compared to the genetic subdivision of other mammals in urban landscapes (up to 0.18 in squirrel gliders (*Petaurus norfolcensis*), Goldingay et al. (2013); up to 0.15 in Koalas (*Phascolarctos cinereus*), Lee et al. (2010); up to 0.36 in prairie dogs (*Cynomys ludovicianus*), Magle et al. (2010); and up to 0.15 in quenda (*Isoodon fusciventer*), Ottewell et al. (2019)). This suggests reasonable gene

flow throughout the urban Albany landscape and that the fragmentation of natural habitat and the creation of infrastructure such as fences, roads and buildings that commonly impede the movement of other animals (Holderegger and Di Giulio, 2010; Johnson and Munshi-South, 2017) have had little negative impact on the genetic structure of the western ringtail possum. We found limited evidence of reduced gene flow in the urban landscape despite a clear genetic IBD effect among individuals within the study area, consistent with the short dispersal range of the western ringtail possum (Jones et al., 1994). IBD within a data set can result in the detection of artificial genetic clusters by clustering programs (Guillot et al., 2009), and therefore some caution should be used in inferring the true number of clusters within Albany based on the two structuring programs. In particular, given the presence of IBD and high genetic spatial autocorrelation at short distances, it is likely that the 10 genetic clusters that Geneland identified represent clines of genetic differentiation driven by reproduction between close neighbours. However, the consensus of one genetic cluster located within and immediately surrounding Mount Clarence and lack of Hardy-Weinberg disequilibrium within the STRUCTURE clusters provided some confidence that at least two genetic clusters of western ringtail possums exist in Albany. It is unclear why a genetic grouping of individuals has formed in and around Mount Clarence. Interestingly, the mount is one of the largest remaining natural bushland remnants (~260 ha) within urban Albany and the grouping corresponds with the historical spatial extent of the remnant in the 1850s (Dowson 2009, and photos therein). Therefore, it is possible that the genetic cluster may relate to the historical extent of vegetation on Mount Clarence.

Our study is the first to assess the genetic diversity of western ringtail possums within its southern-most stronghold population (the South Coast region) and the first to examine the species genetics within a city landscape. The genetic diversity of the western ringtail possum within the City of Albany was comparable with many small Australian marsupials sampled from natural habitat including southern brown bandicoots (*Isodon obesulus*) (Zenger et al., 2005), northern bettongs (*Bettongia tropica*) (Pope et al., 2000), western barred bandicoots (*Perameles bougainville*) (Smith and Hughes, 2008) and tiger quolls (*Dasyurus maculatus*) (Firestone et al., 1999). The genetic diversity and level of inbreeding within Albany was also comparable to other populations of the western ringtail possum predominantly sampled from relatively large remnant habitat (Wilson, 2009; Yokochi et al., 2016). Whether the genetic diversity of western ringtail possums in our study areas reflects the genetic diversity of populations found in extensive areas of natural habitat across its range remains unclear, but

worthy of further investigation. Regardless, for two of the three remaining populations of this species, such habitat is limited and a significant proportion of the remaining population overlaps urban landscapes, adding weight to the comparisons we provide. Our result of comparable genetic diversity provides additional evidence that the urban landscape has had minimal negative impact on the evolutionary processes of the western ringtail possum.

The positive spatial genetic structure we observed over short distances in both bushland and residential areas is consistent with the findings of previous studies investigating fine-scale structure of the possum (Yokochi et al., 2016), and the limited dispersal, small home ranges and sedentary nature of the species (DPaW, 2017, Jones et al., 1994; Van Helden et al., 2018; 2020c; Yokochi et al., 2015). Unexpectedly, we found positive spatial genetic correlation between individuals over a greater spatial extent in residential habitat than natural habitat. These preliminary results suggests a subset of individuals within residential areas may be moving or dispersing over greater distances, resulting in positive spatial genetic autocorrelation at a larger scale in residential areas than bushland areas. We speculate that the greater movement of individuals in residential areas could be due to (1) sub-adults being forced to disperse further from their natal range because of the patchy distribution of resources in residential areas that may already be fully exploited by individuals (DPaW, 2017), (2) the linear nature of infrastructure in residential landscapes (e.g. fences) and lack of continuous canopy habitat creates movement pathways that facilitate greater horizontal movement, or (3) the same behavioural attributes that allow individuals to persist within residential landscapes (e.g. a bold temperament) also predispose individuals to disperse over greater distances (see Lowry et al., 2013 for review). The pattern of fine-scale genetic structure may also substantially differ between males and females (as seen in bush rats (*Rattus fuscipes*), Peakall et al., 2003), which we were unable to investigate in our study due to small sample sizes. Further research that examines which factors drive the observed patterns in genetic correlation, including the influence of sex, resource availability, habitat structure and animal temperament warrants investigation.

Our results from multiple analytical approaches demonstrate that urbanisation appears to have had limited negative impact upon the gene flow, genetic diversity and fine-scale genetic structuring of the western ringtail possum in Albany City. This result contrasts with many previous studies that demonstrate the gene flow and genetic diversity of wildlife is often reduced in urban areas (Johnson and Munshi-South, 2017; Miles et al., 2019), particularly for those that disperse over short distances (e.g. Hitchings and Beebee, 1997; Noël et al., 2007).

There are many possible explanations for why the genetic structure of the western ringtail possum was inconsistent with other species that have short dispersal ranges. First, although the species is largely sedentary, the western ringtail possum possesses other ecological attributes such as behavioural and dietary flexibility (Bader et al., 2019; Mathieson et al., 2020; Van Helden et al., 2018; 2019) which likely contributes to its frequent and abundant presence within residential areas (Van Helden et al., 2020b) and may assist in maintaining better gene flow and genetic diversity than expected. Second, the urban confines of Albany is reasonably small (~ 90 km²), and Albany City is a young (~ 200 years old) and ‘green’ city with generally well vegetated gardens (Van Helden et al., 2020a) and an extensive network of remnant bushland (Busschots et al., 2020). It is possible that the intensity of urbanisation is too low, city size too small or city too young to observe a detrimental impact on the species. Nonetheless, Albany City is sufficiently old to detect changes in genetic structure based on the rapid mutation rate of microsatellites (Schlötterer, 2000). Further studies that explore how city size and age, urban intensity and human population size influence species evolutionary responses to urbanisation would greatly improve our ability to identify in what circumstances cities can be least detrimental and assist in the conservation of wildlife.

We have demonstrated that a city landscape has had little negative impact on the genetic diversity and structure of a Critically Endangered species that appears to thrive in urban areas. Using the western ringtail possum as a case study species, our results highlight that some threatened species, including those with short dispersal ranges, can occur within urban landscapes with seemingly little detriment to evolutionary processes. Whether this is only applicable for species (or populations) that occur in small, young and/or green cities, or depends on the species’ capability of behavioural flexibility to maintain their presence and abundance in urban landscapes warrants further consideration. These findings provide support that urban landscapes offer conservation opportunities for wildlife (including those that are threatened) as these areas can support both contemporary populations of wildlife and evolutionary processes that are essential to support wildlife longer-term. Further research that examines which features within a city maintains gene flow and genetic diversity and which species can occur within urban landscapes long-term, will greatly assist in the evaluation of the conservation value of these highly modified habitats.

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CHAPTER 7

GENERAL DISCUSSION



Photo: The perfect fieldwork mobile: 'Rover' the Kombi!

Key findings and implications for biodiversity conservation

As pristine, natural areas continue to disappear globally, there is a growing necessity to consider the role of novel ecosystems such as urban landscapes in biodiversity conservation. This thesis aimed to evaluate the value of residential gardens for wildlife conservation using both contemporary and historical lines of evidence. The research presented in this thesis addressed five key knowledge requirements needed to understand the value of gardens for wildlife: (1) how widespread can wildlife be in residential landscapes, (2) how the diversity, presence, abundance and reproductive activity of wildlife in gardens compares to natural urban remnant vegetation, (3) which garden features are associated with the presence of wildlife, (4) whether animals are capable of exclusively residing in residential areas and (5) whether the evolutionary processes of species that thrive in urban areas are impacted by urbanisation.

This thesis has demonstrated that residential gardens present substantial conservation opportunities for some mammal species in at least some cities. It has demonstrated that some mammals can be widespread within residential landscapes (Chapter 2, Van Helden et al., 2020a), gardens can contain a considerable proportion of the regional pool of mammal diversity, and that the diversity, presence, abundance and reproductive activity of mammals in gardens can be comparable to urban remnant vegetation (Chapter 3, Van Helden et al., 2020b). Not only are some mammals widespread and abundant, there is evidence that individuals can exclusively reside within residential landscapes for several months of their life, reproduce and exploit the novel resources on offer (Chapter 5, Van Helden et al., 2020c). This ability of some species, including those that are threatened, to seemingly thrive in residential areas also appears to support and maintain their evolutionary patterns and processes (e.g. genetic diversity, genetic structure) in city environments (Chapter 6, Van Helden et al., in review). The contemporary and historical evidence presented in this thesis provides great promise that residential gardens are valuable for wildlife conservation. These findings demonstrate that the available resources in residential gardens can support wildlife in both the short (Chapters 2-4) and longer-term (Chapters 5-6), highlighting that residential gardens have conservation value in their own right within urban landscapes. Management initiatives that incorporate both natural (i.e. urban remnants) and novel (i.e. residential) habitats in urban landscapes will likely result in better conservation outcomes than those that continue to focus only on natural areas and large urban greenspaces.

Given that gardens comprise a major component of the urban greenspace (Goddard et al., 2010; González-García and Sal, 2008; Loram et al., 2007; Mathieu et al., 2007) and provide habitat for numerous species (Fetridge et al., 2008; Goddard et al., 2010; Van Helden et al., 2020a; 2020b), the management of these areas to foster biodiversity could substantially improve conservation outcomes. Conservation initiatives that increase resident's participation in wildlife-friendly gardening may not only increase contemporary native urban biodiversity (Goddard et al., 2010), but may also assist populations to persist within urban landscapes in the longer-term. There is evidence in this thesis (Chapter 2, Van Helden et al., 2020a; Chapter 4, Van Helden et al., in press) and other studies that certain garden features, often consistent with the ecological requirements of species, influence the presence and abundance of wildlife in these greenspaces (Adams et al., 2020; Daniels and Kirkpatrick, 2006; Goddard et al., 2010; Soanes et al., 2019; Threlfall et al., 2016; 2017). If landowners can be encouraged to manage these features to promote wildlife in their gardens, conservation outcomes in urban areas could be realised.

Fortunately, residents are willing to engage in wildlife-friendly gardening activities (e.g. Goddard et al., 2013; van Heezik et al., 2012; 2020) and the manipulation of garden features has been shown to increase biodiversity (Gaston et al., 2005). Residents are capable of achieving conservation outcomes in numerous ways including retaining and enhancing vegetation to create suitable habitat structure, avoiding or reducing the application of pesticides and herbicides, reducing the exposure of wildlife to domestic predators (dogs and cats) and providing stable resources for wildlife throughout the year (e.g. water, supplementary feed and rest sites such as nest boxes) (Ikin et al., 2015; Selinske et al., 2020). Management strategies that improve public awareness of the value of gardens for biodiversity, and provide incentives to engage in wildlife-friendly gardening will improve our ability to conserve wildlife within urban landscapes (Goddard et al., 2010; Ikin et al., 2015).

The research of this thesis adds to a growing list of native species (both animals and plants) that are known to occur in urban landscapes, and draws attention to an often forgotten urban greenspace that could have substantial value for biodiversity if incorporated into conservation policy. This has important implications for the conservation of biodiversity globally as urban landscapes harbour numerous species worldwide (both threatened and common) (Araújo, 2003; Aronson et al., 2014; Ives et al., 2016; Kowarik and von der Lippe, 2018; Schwartz et al., 2002), and because these areas are only set to expand in the future (Seto et al., 2012).

Based on this and the findings in this thesis, the inclusion of gardens in planning and

management actions could play a significant role in biodiversity conservation at global scales.

Considerations and future research directions

Whilst this thesis has demonstrated that residential landscapes offer considerable opportunities for wildlife conservation, it is important to take into account that cities also present threats to biodiversity and challenges for wildlife management. These include disturbances and threats associated with urbanisation, issues associated with human-wildlife conflict, and the spatial scale at which management of gardens is likely most effective.

Balancing the opportunities and threats in residential landscapes

Native animals face a myriad of challenges within residential landscapes including exposure to domestic predators (e.g. dogs and cats), diseases (e.g. *Toxoplasma gondii*) (Stanley et al., 2015), fragmented habitat (Kowarik, 2011), noise and light disturbances (Fuller et al., 2007; Kempenaers et al., 2010), road strike (Coffin, 2007), chemical pollution, and poor diet quality (Isaksson, 2015). Although these threats do not preclude conservation within residential landscapes, research that identifies the best way to minimise these and how to implement management initiatives to combat their effects is needed to best exploit the conservation opportunity that gardens represent.

The presence of wildlife in residential gardens also poses a unique challenge for management in that wildlife will undoubtedly interact with humans to some degree (Soulsbury and White, 2016), which in some cases may deter residents from fostering wildlife in their gardens. Wildlife in gardens can result in property damage, physical injury, disease transmission and economic costs (see Soulsbury and White, 2016 for review). Whilst these concerns can be well founded, they are often perceptions among people rather than direct human-wildlife conflicts (Peterson et al., 2010). Information to correct ill-informed perceptions of risk, and how to mitigate or avoid the real risks associated with urban wildlife in residential gardens may increase uptake of wildlife-friendly gardening. Whilst human-wildlife conflict was once the focus of research on human-wildlife interactions, there is now a growing body of evidence demonstrating that conserving biodiversity in urban areas can have health benefits for people (Miller, 2005; Turner et al., 2004) including on physical health (e.g. Mitchell and Popham, 2008), psychological health (e.g. Taylor et al., 2015) and quality of life (e.g. Luck et al., 2011). Educating residents on these health benefits and the ecosystem services that urban

wildlife can provide (Lowenstein et al., 2015; Matteson and Langellotto, 2009) could be used to increase the willingness of residents to implement conservation initiatives in their gardens.

Issues of scale and transferability

A key challenge for the conservation of biodiversity in gardens is that wildlife-friendly gardening is probably most effective if conservation initiatives are implemented at scales greater than individual gardens (e.g. at the neighbourhood or suburb scale) (Aronson et al., 2017; Goddard et al., 2010; Lepczyk et al., 2017). This is because the factors that determine the presence of wildlife in gardens are complex and operate across multiple spatial and temporal scales (Fournier et al., 2020). For example, the ability of wildlife to access residential gardens can depend on the surrounding landscape, and may also vary according to the ecological traits of the species, such as its mobility (Fournier et al., 2020). The difficulty is that residential gardens are individually owned, and generally cannot be collectively managed to create suitable habitat at broader spatial scales than an individual garden (Aronson et al., 2017). One way to mitigate this problem is to increase the uptake of wildlife-friendly gardening by as many residents as possible to increase patches of suitable habitat throughout the urban landscape. This approach relies on a thorough understanding of the factors that influence a residents decision to implement conservation initiatives, which are often complex and depend on numerous economic, cultural and social influences (see Aronson et al., 2017 for review). The next step for the successful conservation of wildlife in gardens is to understand the factors that influence the management decisions of residents and develop appropriate incentives or awareness campaigns to foster wildlife-friendly gardening. Understanding the influence of city size, age and shape on biodiversity is also needed (Magle et al., 2019). Cities come in all shapes and sizes and range from small towns or villages, to megacities. An understanding of how city size, age and greenness influences the biodiversity within urban areas is still in its infancy, although there is some evidence that species richness and the rate of diversity declines as cities grow and age (Łopucki et al., 2020; McDonnell and Hahs, 2008). Whether residential landscapes have similar conservation value for wildlife as shown in this thesis, in larger, less green and more densely populated cities warrants further consideration. Regardless, given that small cities (i.e. cities with a population < 100 000 and an urban area < 100 km²) are more common than large cities in terms of both total number and land area globally (Łopucki and Kitowski, 2016), the finding of this research that residential areas can have high conservation value for wildlife is likely applicable to a large proportion of current and emerging urban landscapes, worldwide.

Further research that investigates the transferability of the findings of this thesis to other taxonomic groups with differing biological and ecological traits (e.g. size, mobility, behaviour, resource requirements) is worthy of investigation. Evidence from this research (Chapters 2-4) and other work (Crocì et al., 2008; Fournier et al., 2020) has highlighted that the response of wildlife to urbanisation and urban greenspaces is often species-specific and difficult to generalise among taxa. Studies that examine whether other taxonomic groups can also be supported by residential gardens in a similar capacity (i.e. high abundance, high genetic diversity, and evidence of residency) as seen for mammals in this research would further advance our understanding of the conservation value of residential landscapes. This research direction warrants serious consideration given that the occurrence of numerous other taxa in residential landscapes, including examples of birds (Daniels and Kirkpatrick, 2006), reptiles (Koenig et al., 2001) and invertebrates (Ramírez-Restrepo et al., 2017), provides optimism that residential landscapes offer substantial value for these taxa too.

Conclusion

This thesis has exemplified that residential gardens have substantial conservation value for wildlife using mammals as a case study taxon. The research has highlighted that: wildlife can be widespread and abundant in residential landscapes; individuals can reside exclusively within these areas; the evolutionary processes of wildlife can be supported in urban landscapes; and garden features could potentially be manipulated by residents to achieve conservation outcomes in these areas. Current conservation approaches that focus on urban remnants or other larger greenspaces at the expense of smaller greenspaces such as gardens, need rapid re-assessment. Management approaches that incorporate all of the urban greenspaces within novel urban ecosystems, and considers the interaction among them, will likely result in better conservation outcomes than those that continue to focus only on a select few. To best exploit residential gardens, we require a careful approach that considers the challenges associated with conserving in residential gardens including the exposure to urban threats, human-wildlife conflict and the best way to harness the ‘power of the people’. Given that novel urban ecosystems will continue to expand, inclusion of gardens in wildlife conservation and management actions is likely to greatly increase our ability to conserve wildlife on a global scale.

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APPENDIX 1

SUPPLEMENTARY MATERIAL ASSOCIATED WITH CHAPTER 4



Photo: A southern brown bandicoot (*Isodon fusciventer*) captured during fieldwork.

Appendix 1: Results of generalised linear models (GLMs) for Chapter 4 used to examine the relationship between garden features and the presence and abundance of mammal species according to season and variable group in Albany City. Analysis is based on data collected between April 2018 and February 2019. Binomial GLMs were used to examine the relationship between garden features and mammal presence while quasi-Poisson models were used to examine the relationship between garden features and mammal abundance. All models did not consider interactions among explanatory variables and analyses were not undertaken for bats in winter. The parameter estimates (SE) (where possible rounded to two d.p) and *P* value (3 d.p) are shown and significance at the 5% level is denoted by grey shading. Species have been abbreviated to denote southern brown bandicoot (SBB), western ringtail possum (WRP), Gould’s wattled bat (GWB), southern forest bat (SFB) and white-striped free-tailed bat (WSFB). Parameters have been abbreviated to denote canopy cover (canopy), understorey cover (understorey), grass cover (grass), total open space (open), plant richness (plantrich), presence of a vegetable patch (veg), presence of fruiting plants (fruit), roof access (roof), proportion of vegetation that is native (native), proportion of house with under-house space (underhouse), presence of a dog (dog), presence of a cat (cat), presence of rats (rat), fence permeability (fence), presence of elevated water source (ewater), presence of ground water source (gwater), presence of any type of water source (water), distance to nearest patch or remnant vegetation (distance), presence of a chimney (chimney) and presence of competitors (competitor).

Response	Species	Variable Group	Parameters	Autumn	Winter	Spring	Summer
Presence	SBB	Shelter	Understorey	0.001 (0.002), 0.556	0.003 (0.002), 0.190	0.002 (0.002), 0.247	0.004(0.002), 0.095
			Underhouse	0.01 (0.01), 0.085	0.01 (0.01), 0.192	0.01 (0.01), 0.146	0.01 (0.01), 0.091
		Food/water	Gwater	-0.58 (0.63), 0.356	0.39 (0.72), 0.590	0.76 (0.71), 0.282	-0.03 (0.66), 0.960
			Grass	0.001 (0.001), 0.708	-0.002 (0.001), 0.200	-0.00002 (0.002), 0.991	-0.0001 (0.002), 0.977
			Native	0.004 (0.01), 0.651	-0.01 (0.01), 0.341	-0.01 (0.01), 0.395	-0.0003 (0.01), 0.978
			Veg	0.75 (0.60), 0.210	1.96 (0.73), 0.007	0.25 (0.65), 0.694	0.48 (0.62), 0.441
			Plantrich	0.44 (0.53), 0.413	0.47 (0.55), 0.390	0.73 (0.54), 0.175	1.05 (0.56), 0.061
			Interspecific interactions	Dog	-1.23 (0.63), 0.052	-0.99 (0.63), 0.116	-1.03 (0.68), 0.127
		Cat	-0.16 (0.76), 0.835	-0.18 (0.77), 0.815	-0.67 (0.81), 0.409	0.43 (0.80), 0.591	
		Competitor	-0.06 (0.60), 0.924	-0.57 (0.60), 0.337	-0.84 (0.62), 0.178	-0.59 (0.60), 0.329	
		Accessibility	Fence	0.98 (0.54), 0.068	0.32 (0.48), 0.512	0.51 (0.52), 0.332	0.43 (0.48), 0.377
			Distance	-0.001 (0.002), 0.780	-0.002 (0.002), 0.348	0.00001 (0.002), 0.999	-0.0001 (0.002), 0.951
			Canopy	0.01 (0.003), 0.049	0.002 (0.002), 0.496	0.01 (0.002), 0.058	0.01 (0.004), 0.019
		WRP	Shelter	Understorey	-0.001 (0.003), 0.790	0.0005 (0.002), 0.846	0.002 (0.003), 0.499
	Roof			0.26 (0.42), 0.531	0.75 (0.40), 0.060	0.30 (0.41), 0.468	0.67 (0.49), 0.169
	Canopy			0.01 (0.004), 0.049	0.001 (0.002), 0.777	0.01 (0.003), 0.009	0.01 (0.004), 0.086
	Food/water		Water	0.45 (0.82), 0.578	0.91 (0.70), 0.195	0.48 (0.72), 0.507	-0.61 (0.85), 0.473
			Native	0.02 (0.01), 0.208	-0.002 (0.01), 0.875	-0.01 (0.01), 0.324	0.02 (0.02), 0.242
			Fruit	1.70 (1.01), 0.092	-0.59 (0.84), 0.487	0.26 (0.91), 0.773	-0.30 (1.10), 0.785
			Plantrich	-1.37 (0.68), 0.044	0.51 (0.50), 0.310	-0.52 (0.55), 0.348	-0.01 (0.64), 0.991
	Interspecific interactions		Dog	-0.70 (0.65), 0.281	-0.28 (0.61), 0.651	-0.72 (0.64), 0.258	-1.07 (0.71), 0.133
Cat			-1.43 (0.90), 0.111	-0.50 (0.74), 0.494	0.01 (0.74), 0.989	-1.23 (0.91), 0.175	
Rat			0.58 (0.67), 0.382	0.38 (0.61), 0.536	-0.02 (0.62), 0.977	0.64 (0.71), 0.372	
GWB	Accessibility	Distance	-0.001 (0.002), 0.986	0.001 (0.002), 0.775	-0.0001 (0.002), 0.947	0.0001 (0.002), 0.957	
		Canopy	0.003 (0.003), 0.342	NA	-0.001 (0.002), 0.631	-0.001 (0.002), 0.589	
		Understorey	-0.002 (0.003), 0.626	NA	0.0004 (0.002), 0.875	0.002 (0.002), 0.356	

Response	Species	Variable Group	Parameters	Autumn	Winter	Spring	Summer
Abundance	SFB	Food/water	Roof	-0.30 (0.60), 0.620	NA	0.18 (0.38), 0.630	0.30 (0.39), 0.448
			Chimney	-0.26 (0.93), 0.779	NA	-0.13 (0.67), 0.843	-0.72 (0.69), 0.294
			Water	0.02 (0.96), 0.986	NA	0.19 (0.68), 0.779	-0.22 (0.71), 0.761
			Native	-0.0001 (0.02), 0.996	NA	-0.0003 (0.01), 0.976	0.02 (0.01), 0.061
			Plantrich	0.16 (0.82), 0.848	NA	-0.15 (0.56), 0.795	0.29 (0.60), 0.633
			Open	0.002 (0.003), 0.445	NA	0.001 (0.002), 0.779	0.002 (0.002), 0.354
		Interspecific interactions	NA	NA	NA	NA	NA
		Accessibility	Distance	-0.02(0.01), 0.017	NA	0.003 (0.002), 0.276	-0.004 (0.003), 0.128
		Shelter	Canopy	-0.001 (0.003), 0.969	NA	0.002 (0.003), 0.433	-0.003 (0.003), 0.260
		Understorey	0.003 (0.003), 0.385	NA	-0.0002 (0.003), 0.955	0.002 (0.003), 0.523	
		Roof	0.24 (0.56), 0.660	NA	0.30 (0.47), 0.526	1.10 (0.50), 0.026	
		Chimney	0.65 (1.17), 0.577	NA	-0.92 (0.76), 0.225	1.08 (0.73), 0.140	
	Food/water	Water	-1.40 (1.03), 0.174	NA	-1.10 (0.83), 0.184	-0.30 (0.73), 0.676	
		Native	0.01 (0.02), 0.631	NA	-0.01, (0.01), 0.483	0.01 (0.01), 0.256	
		Plantrich	0.55 (0.88), 0.531	NA	-0.51 (0.70), 0.465	0.29 (0.59), 0.616	
		Open	-0.001 (0.003), 0.682	NA	-0.003 (0.002), 0.247	0.001 (0.002), 0.542	
		Interspecific interactions	NA	NA	NA	NA	
		Accessibility	Distance	-0.01 (0.01), 0.135	NA	-0.01 (0.005), 0.077	-0.002 (0.002), 0.393
	WSFB	Shelter	Canopy	0.003 (0.002), 0.272	NA	0.002 (0.002), 0.352	-0.0002 (0.002), 0.947
		Understorey	-0.01 (0.003), 0.035	NA	-0.002, (0.002), 0.330	0.002 (0.002), 0.391	
		Roof	0.08 (0.40), 0.834	NA	0.42 (0.39), 0.281	-0.05 (0.41), 0.91	
		Chimney	0.34 (0.74), 0.647	NA	0.20 (0.68), 0.769	1.21 (0.85), 0.154	
		Food/water	Water	-0.60 (0.71), 0.369	NA	-0.80 (0.73), 0.272	1.03 (0.87), 0.236
			Native	-0.01 (0.01), 0.572	NA	-0.02 (0.01), 0.112	-0.01 (0.01), 0.272
Plantrich	-0.76 (0.60), 0.204		NA	-1.29(0.62), 0.036	-0.64 (0.64), 0.316		
SBB	Interspecific interactions	Open	-0.001, (0.002), 0.709	NA	-0.003 (0.002), 0.103	-0.003 (0.002), 0.179	
		NA	NA	NA	NA		
		Accessibility	Distance	0.001 (0.002), 0.814	NA	0.001 (0.002), 0.792	-0.001 (0.002), 0.772
	Shelter	Understorey	0.002 (0.001), 0.144	0.002 (0.001), 0.220	0.001 (0.001), 0.347	0.001 (0.001), 0.567	
		Underhouse	0.005 (0.004), 0.236	0.01 (0.01), 0.080	0.01 (0.005), 0.122	0.01 (0.01), 0.214	
		Food/water	Gwater	-0.38 (0.40), 0.347	-0.11 (0.56), 0.839	0.005 (0.54), 0.992	-0.03 (0.57), 0.955
Grass	-0.001 (0.001), 0.578		-0.001 (0.001), 0.264	-0.0004 (0.001), 0.726	-0.001 (0.001), 0.266		
Native	0.001 (0.01), 0.932		-0.01 (0.01), 0.170	-0.003 (0.01), 0.773	-0.01 (0.01), 0.490		
Interspecific interactions	Veg	0.85(0.42), 0.049	1.05 (0.60), 0.088	0.08 (0.49), 0.879	0.77 (0.56), 0.177		
	Plantrich	0.23 (0.33), 0.487	0.33 (0.45), 0.474	0.56 (0.43), 0.203	0.31 (0.45), 0.494		
	Dog	-0.74 (0.40), 0.069	-1.43(0.52), 0.008	-1.04 (0.43), 0.020	-1.12 (0.49), 0.025		
	Cat	0.22 (0.57), 0.702	-0.23 (0.61), 0.704	-0.52 (0.49), 0.298	-0.71 (0.53), 0.190		
	Competitor	-0.38 (0.37), 0.310	-1.22 (0.48), 0.015	-0.77 (0.40), 0.061	-0.75 (0.45), 0.099		

Response	Species	Variable Group	Parameters	Autumn	Winter	Spring	Summer
		Accessibility	Fence	0.79 (0.41), 0.056	0.56 (0.53), 0.294	0.52 (0.42), 0.221	0.71 (0.49), 0.155
	WRP	Shelter	Distance	-0.001 (0.002), 0.601	0.0003 (0.002), 0.875	0.0005 (0.002), 0.775	-0.002 (0.002), 0.445
			Canopy	0.001 (0.001), 0.396	0.003(0.001), 0.006	0.001 (0.001), 0.339	0.003(0.001), 0.005
		Food/water	Understorey	0.001 (0.001), 0.666	-0.002 (0.001), 0.089	0.0002 (0.001), 0.912	-0.001 (0.001), 0.509
			Roof	0.20 (0.22), 0.373	0.56 (0.22), 0.009	0.21 (0.26), 0.420	0.15 (0.18), 0.427
			Canopy	0.001 (0.001), 0.634	0.003 (0.002), 0.095	0.002 (0.002), 0.225	0.002(0.001), 0.049
		Interspecific interactions	Water	1.05 (0.53), 0.053	0.12 (0.49), 0.806	0.23 (0.47), 0.620	-0.22 (0.32), 0.482
			Native	0.003 (0.01), 0.615	-0.01 (0.01), 0.294	-0.01 (0.01), 0.165	-0.0005 (0.01), 0.941
			Fruit	0.17 (0.46), 0.708	-0.30 (0.55), 0.586	0.32 (0.64), 0.622	0.18 (0.41), 0.658
			Plantrich	0.07 (0.29), 0.806	0.06 (0.33), 0.846	0.24 (0.32), 0.453	0.25 (0.22), 0.270
			Dog	0.01 (0.36), 0.986	-0.51 (0.40), 0.205	-1.18 (0.38), 0.003	-0.33 (0.31), 0.296
		Cat	-0.04 (0.44), 0.921	-0.67 (0.46), 0.148	-0.62 (0.44), 0.163	-0.34 (0.37), 0.357	
		Rat	0.54 (0.34), 0.122	0.79 (0.37), 0.037	-0.05 (0.37), 0.887	0.36 (0.30), 0.236	

APPENDIX 2

RELATED RESEARCH PUBLISHED DURING CANDIDATURE



Photo: One of most vegetated residential gardens surveyed for mammals in the City of Albany.

Is the presence of a threatened arboreal mammal in residential areas related to remnant habitats?

Austral Ecology (2020)

doi.org/10.1111/aec.12953

Busschots, M. B., Close, P. G., Van Helden, B. E., Speldewinde, P. C.

Abstract

Remnant natural habitat is assumed critical for supporting threatened animals within urban landscapes because these species generally have specific habitat requirements and typically respond poorly to anthropogenic disturbances. However, evidence that demonstrates some threatened species can occur, persist and even prosper in highly modified areas with seemingly little dependence on remnant vegetation challenges the role remnant vegetation is perceived to play in sustaining threatened species in urban landscapes. In this study, we tested the assumption that the presence of a threatened species in modified areas of an urban environment was dependent on remnant vegetation using the Critically Endangered western ringtail possum (*Pseudocheirus occidentalis*) as a case study. We predicted that the presence of possums in highly modified habitats would be positively correlated with proximity to remnant vegetation. Using spotlight surveys of 195 transects to determine species occurrence on residential streets, we found that the presence of the western ringtail possum was not related to remnant vegetation within the immediate surrounds, nor the distance from remnant habitats (neither large nor small). Our results promote the suggestion that highly modified habitats in urban landscapes can contribute conservation outcomes and that their potential role in wildlife conservation warrants serious consideration.

2d or not 2d? Three-dimensional home range analysis better represents space use by an arboreal mammal

Acta Oecologia (2020)

doi.org/10.1016/j.actao.2020.103576

Chandler, C. J., Van Helden, B. E., Close, P. G., Speldewinde, P. C

Abstract

For animals which utilise vertically distributed habitat resources, home ranges quantified in two-dimensions (2D) likely misrepresent space use and constrain interpretations of species' ecology and behaviour. Three-dimensional (3D) home range estimation has proved useful for understanding space use over relatively large vertical ranges (e.g. 100's of meters), yet none have applied a 3D approach to describe space use over narrower vertical ranges (e.g. meters or 10's of metres) typical for many arboreal animals. This study demonstrates 3D home range estimation for the arboreal western ringtail possum (*Pseudocheirus occidentalis*) using temporally resolute locations (15 min intervals) of five radio collared animals. We hypothesised that, due to the use of vertically distributed habitat resources, 3D home ranges would be more complex, spanning a range of heights, and would subsequently be larger in size compared to those estimates derived from typical 2D analysis. A comparison of 50% (core) and 95% (total) 2D kernel density and 3D kernel density utilisation distributions reveal structurally complex home ranges that span the entire vertical range of vegetated habitat. Estimates of the total 3D home range size were significantly greater than those derived from 2D analysis, and the difference in size was positively related to the vertical range of available habitat. This study establishes that 3D home range analysis can reveal information on vertical space use of arboreal species over relatively small vertical ranges, and is likely to be valuable in understanding the use and identification of key habitats and resources, as well as interactions among sympatric species.

New evidence of unexpectedly high animal density and diet diversity will benefit the conservation of the critically endangered western ringtail possum

Austral Ecology (2020)

doi.org/10.1111/aec.12874

Mathieson, T. J., Close, P. G., Van Helden, B. E., Speldewinde, P. C., Comer, S. J.

Abstract

A comprehensive and contemporary understanding of habitat and resource requirements has been critical to the conservation of multiple taxa and ecosystems globally. Until recently, much of the ecological knowledge that contributes to conservation priorities and strategies for the Critically Endangered western ringtail possum (*Pseudocheirus occidentalis*) was largely derived from decades-old observations in peppermint (*Agonis flexuosa*) and marri-jarrah (*Corymbia calophylla* and *Eucalyptus marginata*) woodlands in the northern parts of the species range. These observations do not account for more recent evidence of their flexible use of habitat resources in other regions of its range. This may represent a significant conservation opportunity for the species through the identification of additional habitats that warrant protection. In a region where knowledge of their ecology is scarce, we used scat analysis and quantitative spotlighting to determine the diet and density of western ringtail possums in three vegetation types: peppermint, sheoak (*Allocasuarina fraseriana*) and marri-eucalypt (*C. calophylla*, *E. marginata* and *Eucalyptus staerii*) woodlands. Given the species' reported dependence on peppermint woodlands and dominant canopy species for food sources, we hypothesised that western ringtail possums would be most abundant in this habitat type and that their diet would comprise the foliage of few (≤ 2 species) canopy species. We found western ringtail possums consumed a higher diversity of plant species than expected (8–14), exhibited dietary preference for non-dominant canopy species and were present in all sampled vegetation types at substantially higher densities than previously recorded for the region (as high as 17 possums ha⁻¹). Our results confirm (i) the western ringtail possum is flexible in its use of habitat resources and (ii) the significant conservation value of sheoak and marri-eucalypt woodlands in the southernmost portion of its distribution.

Going to ground: Implications of ground use for the conservation of an arboreal marsupial

Australian Mammalogy (2019)

doi.org/10.1071/AM18053

Van Helden, B. E., Close, P. G., Stewart, B. A., Speldewinde, P. C., Comer, S. J.

Abstract

On the basis of previous observations, the critically endangered western ringtail possum (*Pseudocheirus occidentalis*) has been described as strictly arboreal. Using motion-sensing cameras placed in bushland remnants and residential gardens, we demonstrate that the species uses the ground during its nocturnally active period. To acknowledge that ground use may make the species more susceptible to terrestrial predators than previously thought, we propose that the species should not be described as ‘strictly’ arboreal.

Sheoak woodlands: A newly identified habitat for western ringtail possums

Journal of Wildlife Management (2019)

doi.org/10.1002/jwmg.21686

Bader, J. C., Van Helden, B. E., Close, P. G., Speldewinde, P. C., Comer, S. J.

Abstract

Although critically endangered western ringtail possums (*Pseudocheirus occidentalis*) have been recorded from a variety of vegetation types in southwestern Australia, the extent to which many of these vegetation types are occupied by the species remains unknown. We conducted spotlight surveys for western ringtail possums between March and April 2018 in Albany, southwestern Australia, to determine the species' occupancy in 2 vegetation types. Using occupancy models, we demonstrated that sheoak (*Allocasuarina fraseriana*) woodlands, previously unrecognised as providing western ringtail possum habitat, support the species, although the median probability of their occupancy was lower than in marri (*Corymbia calophylla*) and eucalypt (*Eucalyptus marginata* and *E. staeri*) woodlands. Use of trees and other habitat components by western ringtail possums varied within and between vegetation types. Sheoak woodlands are likely critical for western ringtail possums and given the apparent flexibility in vegetation types used by the species, investigation of the potential value of other vegetation types for this species has conservation merit.

**Use of urban bushland remnants by the western ringtail possum
(*Pseudocheirus occidentalis*): Short-term home-range size and habitat use
in Albany, Western Australia**

Australian Mammalogy (2018)

doi.org/10.1071/AM17026

Van Helden, B. E., Speldewinde, P. C., Close, P. G., Comer, S. J.

Abstract

Management of wildlife in habitats fragmented by urbanisation requires an understanding of a species' habitat use. Known populations of the critically endangered western ringtail possum (*Pseudocheirus occidentalis*) are largely restricted to bushland remnants in rapidly urbanising areas of south-western Australia. Habitat use is thought to be driven by nutritional content, structure and connectivity of canopy vegetation. At the southernmost extent of the species' range, habitat use is largely unknown, although it is expected to be different from previous descriptions due to differences in vegetation characteristics. We used VHF and global positioning system tracking collars to determine short-term home-range size, diurnal refuge use and night-time tree use of western ringtail possums in bushland remnants within Albany City. Possums had small home ranges (0.88 ha) that were negatively correlated with percentage canopy cover; used a variety of daytime refuges (predominantly dreys); and preferentially utilised marri (*Corymbia calophylla*) and jarrah (*Eucalyptus marginata*) at night. These results confirm that differences in habitat use among populations exist and suggests that the species is reasonably flexible in its use of habitat. Management of western ringtail possums needs to be population specific and will benefit from further examination of habitat use in the variety of occupied habitats.