Managing gardens for wildlife: Features that predict mammal presence and abundance in gardens vary seasonally

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Abstract. Residential gardens in urban areas offer substantial opportunities for biodiversity conservation. To realize this opportunity, a comprehensive understanding of which species prosper in gardens, which mechanisms promote their use of these green spaces and how these mechanisms can be manipulated by residents to achieve conservation outcomes, is needed. While 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 garden 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 southwestern Australia. As animal behavior 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 overemphasize 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 influencing 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.

Key words: conservation; mammals; mechanisms; novel ecosystems; residential gardens; seasonality; urban ecology; urbanization.

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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 urbanized 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 utilize (Aronson et al. 2017), and can harbor
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 wastelands (e.g., rubbish tips and vacant blocks), and remnant patches of vegetation (Cilliers et al. 2013, Aronson et al. 2017, Lepczyk et al. 2017). While 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 (Goddard et al. 2010, Tulloch et al. 2016, Soanes et al. 2019). 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 (Loram et al. 2007, Mathieu et al. 2007, González-García and Sal 2008, Goddard et al. 2010) and are used by numerous taxa including birds (Daniels and Kirkpatrick 2006), mammals (Van Helden et al. 2020a, b), 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 green spaces, 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., Fetridge et al. 2008, Bernholt et al. 2009, Van Helden et al. 2020a), and the mechanisms that drive these patterns (e.g., Baker et al. 2003, Daniels and Kirkpatrick 2006, Basham et al. 2011, Fontaine et al. 2016, Van Helden et al. 2020b). While 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., Baker et al. 2003, Daniels and Kirkpatrick 2006, Bergerot et al. 2011, Fontaine et al. 2016, Adams et al. 2020). Landscape factors such as the degree of urbanization (Fontaine et al. 2016), the environmental suitability of the area (e.g., altitude, Daniels and Kirkpatrick 2006), or 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 (González-García et al. 2009, Fontaine et al. 2016), and an absence of predators (Baker et al. 2003, Van Helden et al. 2020b) 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), and even among cities for the same species (Van Helden et al. 2020b).

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). Alternatively, in some studies, season of data collection was not recorded (e.g., in the case of citizen-derived data, Baker and Harris 2007, González-García et al. 2009, Belaire et al. 2014, Van Helden et al. 2020b). In all of these studies, how the relative importance of these mechanisms vary 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 behavior 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. While multiple studies provide promising evidence that garden features could be manipulated to improve conservation outcomes for fauna in gardens (e.g., Dique et al. 2004, Daniels and Kirkpatrick 2006, Oprea et al. 2009, Patterson et al. 2017, Van Helden et al. 2020b), considering seasonality will allow for the identification of all 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 inform decision-making as to which features 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, southwestern Australia, as a case study city because fauna of the region exhibits strong seasonality in their diet, behavior, abundance, and reproductive biology (e.g., Hosken et al. 1998, Wayne et al. 2005, Bradshaw et al. 2007) in response to a strongly seasonal Mediterranean climate. While 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, behavior, and resource selection of animals in natural environments, we hypothesized 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 hypothesized 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 southwestern Australia (34°58’ S, 117°52’ E) in a globally recognized biodiversity hotspot (Myers et al. 2000). This region is characterized by a Mediterranean climate, with cool wet winters (June–August) and hot dry summers (December–February), 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). 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 (B. Van Helden, unpublished data). 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; 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. 2020a).

We surveyed 52 gardens within urban Albany for the presence of terrestrial mammal species (Fig. 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. 2020b). All sites 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, Peakall et al. 2006, FitzGibbon et al. 2007) were unlikely to move between sites within one night. We focused on maintaining independence between sites for these species because their capture at one site would preclude their capture at another site on the same night.

**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. (2020a). 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. In each garden (N = 52), we set two Elliot traps, two Sheffield cage traps (220 × 220 × 450 mm), two passive infrared motion-sensing camera traps (ScoutGuard 560K-18 M 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 standardize the field of view among sites. Camera traps were programmed to take three successive photographs when triggered on medium sensitivity.

![Fig. 1. The location of the 52 residential garden study sites (Δ) within the urban area (light gray) of the City of Albany in southwestern Australia. Natural remnant vegetation is also shown (dark gray). Remnant vegetation layer sourced from Sandiford and Barret (2010) and Busschots et al. (2021).](image-url)
between 14:00 and 10:00 h, while 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 standardized 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, Maynard, Massachusetts, 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 the 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 randomized the survey order of sites in each season and on each night to minimize spatial and temporal effects on the presence and abundance of mammals.

Data processing
Sonograms of recorded echolocation calls (acquired from acoustic surveys) were analyzed with Kaleidoscope Pro Analysis Software (Wildlife Acoustics, Maynard, Massachusetts, USA). Bat species were discriminated based on previously described call features including shape, and frequency peak, start and finish (Titley 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). Photographs obtained from camera traps were visually examined by one common observer, and all species present at each site in each season were recorded. All photographs 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 is 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 between gardens given that the canopy vegetation in gardens is generally sparse and low in height (<15 m), 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 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 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). 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) was 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

**Table 1. Garden or property features recorded during garden surveys and their inclusion in generalized linear models (GLMs) for each mammal group.**

| Mammal group | Variable group | Variables included in GLMs | Values |
|--------------|----------------|---------------------------|--------|
| Arboreal     | Shelter        | Canopy cover m²           |        |
|              |                | Understorey cover m²      |        |
|              |                | Roof access 1: no gaps or spaces; 2: spaces available but access restricted; 3: multiple roof spaces |        |
|              | Food/water     | Canopy cover m²           |        |
|              |                | 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 m²           |        |
|              |                | Understorey cover m²      |        |
|              |                | Presence of chimneys (duct that carries smoke from a 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) m² |        |
|              |                | Presence of permanent water (ponds/water bowls, bird baths) 1: Yes 0: No |        |
|              |                | Percentage of vegetation which was native Percentage |        |
|              | Interspecific interactions | NIL | NA |
| Ground-     | Accessibility  | Distance to nearest patch of remnant vegetation m |        |
| dwelling     | Shelter        | Understorey cover m²      |        |
|              |                | Percentage of house with under-house space Percentage |        |
|              | Food/water     | Presence of permanent groundwater (ponds/water bowls) 1: Yes 0: No |        |
|              |                | Grass cover (lawn) m² |        |
|              |                | Percentage of vegetation which was native Percentage |        |
|              |                | Presence of vegetable patch 1: Yes 0: No |        |
|              | 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 |    |
each season). These species included the arboreal western ringtail possum (*Pseudocheirus occidentalis*), the ground-dwelling southern brown bandicoot (*Isoodon fusciventer* Gould) 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 the cage trapping, Elliot trapping, and camera trapping 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 for the southern brown bandicoot while 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 was recorded 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 1) using a generalized 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 quasipoisson GLMs for each species in each season as there was evidence of overdispersion for all species. Explanatory variables included in GLMs for each species were species-specific and reflected their general resource and habitat requirements (Table 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 whether the garden features that determine the 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 analyze 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. No variables included together in any of the GLMs were highly correlated (−0.7 < r < 0.7, N = 52), so 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.

**RESULTS**

*Mammal presence and abundance*

The western ringtail possum and southern brown bandicoot were frequently detected in all seasons, while the three bat species were detected most commonly in spring and summer (Fig. 2). The trap success of southern brown bandicoots in each garden ranged between 6.8 ± 1.8% (mean ± SE) in winter and 9.3 ± 1.7% in autumn, while the abundance of western ringtail possums ranged between 3.9 ± 0.8 individuals/hectare in winter and 6.2 ± 1.0 individuals/hectare in winter.
hectare in autumn (N = 52; Fig. 3). For a full description of all other mammal species detected in gardens, see Van Helden et al. (2020a).

**Garden features**

Garden and property features were highly variable among sites (Fig. 4). Of the 52 garden properties surveyed, 77% had a chimney, 77% had fruiting plants, 52% had a vegetable patch, 62% had a groundwater 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

Fig. 2. The number of garden sites in which each species was detected in each season between April 2018 and February 2019. Total number of garden sites surveyed in each season was 52.

Fig. 3. The mean trap success (%; N = 52) of southern brown bandicoots and mean abundance (individuals per hectare, N = 52) of western ringtail possums in gardens in each season between April 2018 and February 2019. Whiskers represent the standard error of the mean. Figure has been modified from Van Helden et al. (2020a).
houses. Plant richness was high in 19 gardens, moderate in 22 gardens, and low in 11 gardens.

**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 2). No garden features influenced the presence of any mammal species consistently in all four seasons (Table 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 2). The presence of western ringtail possums was positively related to canopy cover in autumn ($P = 0.049$), spring

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**Fig. 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. The gray 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. Circles represent outlier values outside of these maximum and minimum values.
Table 2. Summary of garden features that were significantly related the presence and abundance of mammal species according to season.

| Species | Variable group | Parameters | Autumn | Winter | Spring | Summer |
|---------|----------------|------------|--------|--------|--------|--------|
| SBB     | Food/water     | Veg        | ...    | 1.96 (0.73)†| ...    | ...    |
| WRP     | Shelter        | Canopy     | 0.006 (0.003)†| ...    | ...    | 0.01 (0.004)†| ...    |
| Food/water | Canopy     | ...    | ...    | 0.007 (0.004)†| ...    | 0.008 (0.003)†| ...    |
| GWB     | Accessibility  | Distance   | −1.37 (0.68)†| ...    | ...    | ...    |
| SFB     | Shelter        | Roof       | ...    | NA     | ...    | ...    |
| WSFB    | Shelter        | Understorey| −0.007 (0.003)†| ...    | NA     | 1.10 (0.50)†| ...    |
| Food/water | Canopy     | ...    | ...    | NA     | −1.29 (0.62)†| ...    |

Notes: Full details of results are supplied in Appendix S1: Table S1. Species have been abbreviated to SBB (southern brown bandicoot), WRP (western ringtail possum), GWB (Gould’s wattled bat), SFB (southern forest bat), and WSFB (white-striped free-tailed bat). 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). The parameter estimates (± SE) for each significant variable are shown.
† Significance at the 5% level.
‡ Significance at the 1% level.

(P = 0.009), and summer (P = 0.019; Table 2). In autumn, their presence was also negatively related to plant richness (P = 0.044; Table 2). In winter, the 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, which was only influential in one season (Table 2). In winter, southern brown bandicoot presence was positively influenced by the presence of vegetable patches in gardens (P = 0.007; Table 2). In all other seasons, none of the garden features collected in this study influenced southern brown bandicoot presence (P > 0.05; Table 2).

The presence of bats was influenced by four garden features, none of which were influential in more than one season (Table 2). The presence of Gould’s wattled bat in autumn was negatively related to distance to nearest bushland remnant (P = 0.017; Table 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 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). The 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 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 2). No garden features influenced the abundance of any mammal species consistently in all four seasons (Table 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 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 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 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 2). In winter, the presence of introduced competitors was also negatively related to the abundance of southern brown bandicoots ($P = 0.015$; Table 2). Southern brown bandicoots were more abundant in autumn when vegetable patches were present in gardens ($P = 0.049$; Table 2).

**Discussion**

The creation of wildlife-friendly residential gardens may offer substantial opportunities for biodiversity conservation (Goddard et al. 2010, Soanes et al. 2019). While multiple studies provide promising evidence that garden features could be manipulated by residents to improve conservation outcomes for fauna in gardens (e.g., Dique et al. 2004, Daniels and Kirkpatrick 2006, Oprea et al. 2009, Patterson et al. 2017, Van Helden et al. 2020b), 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 whether the mechanisms that drive the assemblage, presence, and abundance of vertebrate fauna in gardens vary 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, the 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 suggest 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, overemphasize 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 garden manipulations will have the most benefit, and which manipulations 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. 2020b), 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. 2020b). 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., Daniels and Kirkpatrick 2006, Baker and Harris 2007, Van Helden
et al. 2020b), 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 prioritized 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 incentivize landowners 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 offers 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 long term, and thus, 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 are only influential in a single season exemplifies the dangers of assessing predictors of wildlife presence and abundance based on one or few seasons of sampling.

While 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, Daniels and Kirkpatrick 2006, Basham et al. 2011). These studies may have therefore over-emphasized 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 northwest 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) while 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 silvereye (Zosterops lateralis), common blackbird (Turdus merula), and European goldfinch (Carduelis carduelis). In this case, the importance of deciduous tree cover may be over-emphasized 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, highlight the benefits of examining the mechanisms in all seasons, not only to identify all important features, but also to ensure others are not over-emphasized.

Although we highlight numerous garden features that are important for mammal wildlife and that these vary 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 (e.g., in autumn for southern brown bandicoot presence and western ringtail possum abundance). While there is promising evidence that wildlife-friendly gardening can be beneficial regardless of landscape-scale factors (e.g., level of surrounding urbanization; Fontaine et al. 2016), it is important to recognize that the occurrence of wildlife in gardens likely also depends on broader landscape factors (such as city size and intensity of urbanization), the biological and ecological traits of species (e.g., mobility, behavior, 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.

While 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. First, it is unclear from our study the level of dependence species have on these features (i.e., animals may be capable of modifying their behavior to utilize other resources in seasons when preferred ones are not available). Second, 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 differ 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. 2020b), species (Daniels and Kirkpatrick 2006), and taxa (Fontaine 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 benefits 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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LITERATURE CITED

Adams, B. J., E. Li, C. A. Bahlai, E. K. Meineke, T. P. McGlynn, and B. V. Brown. 2020. Local- and landscape-scale variables shape insect diversity in an urban biodiversity hot spot. Ecological Applications 30:e02089.
Araújo, M. B. 2003. The coincidence of people and biodiversity in Europe. Global Ecology and Biogeography 12:5–12.
Aronson, M. F., C. A. Lepczyk, K. L. Evans, M. A. Goddard, S. B. Lerman, J. S. Maclvor, C. H. Nilon, and T. Vargo. 2017. Biodiversity in the city: key challenges for urban green space management. Frontiers in Ecology and the Environment 15:189–196.
Australian Bureau of Statistics [ABS]. 2016. Region data summary. http://www.abs.gov.au/
Baker, P. J., R. J. Ansell, P. A. Dodds, C. E. Webber, and S. Harris. 2003. Factors affecting the distribution of small mammals in an urban area. Mammal Review 33:95–100.
Baker, P. J., and S. Harris. 2007. Urban mammals: What does the future hold? An analysis of the factors affecting patterns of use of residential gardens in Great Britain. Mammal Review 37:297–315.
Basham, R., B. Law, and P. Banks. 2011. Microbats in a ‘leafy’ urban landscape: Are they persisting, and what factors influence their presence? Austral Ecology 36:663–678.
Belaire, J. A., C. J. Whelan, and E. S. Minor. 2014. Having our yards and sharing them too: the collective effects of yards on native bird species in an urban landscape. Ecological Applications 24:2132–2143.
Bergerot, B., B. Fontaine, R. Julliard, and M. Baguette. 2011. Landscape variables impact the structure and composition of butterfly assemblages along an urbanization gradient. Landscape Ecology 26:83–94.
Bernholt, H., K. Kehlenbeck, J. Gebauer, and A. Buerkert. 2009. Plant species richness and diversity in urban and peri-urban gardens of Niamey, Niger. Agroforestry Systems 77:159.
Biodiversity Conservation Act. 2016. Western Australia. Department of Biodiversity, Conservation and Attractions (DBCA). 2019. Nature map. https://naturemap.dba.wa.gov.au/
Blette, J. A., C. J. Whelan, and E. S. Minor. 2014. Hav-
I. Distribution and Habitat. Wildlife Research 21:175–187.

Khammes, N., and S. Aulagnier. 2007. Diet of the wood mouse, Apodemus sylvaticus in three biotopes of Kabylie of Djurdjura (Algeria). Folia Zoologica 56:243–252.

Koenig, J., R. Shine, and G. Shea. 2001. The ecology of an Australian reptile icon: How do blue-tongued lizards (Tiliqua scincoides) survive in suburbia? Wildlife Research 28:214–227.

Lepczyk, C. A., M. F. J. Aronson, K. L. Evans, M. A. Koenig, J., R. Shine, and G. Shea. 2001. The ecology of urban domestic gardens (X): the extent & structure of the resource in five major cities. Landscape Ecology 22:601–615.

Mathieu, R., C. Freeman, and J. Aryal. 2007. Mapping private gardens in urban areas using object-oriented techniques and very high-resolution satellite imagery. Landscape and Urban Planning 81:179–192.

Moretto, L., and C. M. Francis. 2017. What factors limit bat abundance and diversity in temperate, North American urban environments? Journal of Urban Ecology 3:1–9.

Myers, N., R. A. Mittermeier, C. G. Mittermeier, G. A. B. da Fonseca, and J. Kent. 2000. Biodiversity hotspots for conservation priorities. Nature 403:853.

Nielsen, S. E., G. McDermid, G. B. Stenhouse, and M. S. Boyce. 2010. Dynamic wildlife habitat models: Seasonal foods and mortality risk predict occupancy-abundance and habitat selection in grizzly bears. Biological Conservation 143:1623–1634.

Oprea, M., P. Mendes, T. B. Vieira, and A. D. Ditchfield. 2009. Do wooded streets provide connectivity for bats in an urban landscape? Biodiversity and Conservation 18:2361–2371.

Paltridge, R., and R. Southgate. 2001. The effect of habitat type and seasonal conditions on fauna in two areas of the Tanami Desert. Wildlife Research 28:247–260.

Patterson, L., R. Kalle, and C. Downs. 2017. A citizen science survey: perceptions and attitudes of urban residents towards vervet monkeys. Urban Ecosystems 20:617–628.

Patterson, L., R. Kalle, and C. Downs. 2018. Factors affecting presence of vervet monkey troops in a suburban matrix in KwaZulu-Natal, South Africa. Landscape and Urban Planning 169:220–228.

Peakall, R., D. Ebert, R. Cunningham, and D. Lindenmayer. 2006. Mark-recapture by genetic tagging reveals restricted movements by bush rats (Rattus fuscipes) in a fragmented landscape. Journal of Zoology 268:207–216.

Peduzzi, P., J. Conlato, E. Kemper, T. R. Holford, and A. R. Feinstein. 1996. A simulation study of the number of events per variable in logistic regression analysis. Journal of Clinical Epidemiology 49:1373–1379.

QGIS Development Team. 2019. QGIS Geographic Information System Open Source Geospatial Foundation Project. https://qgis.osgeo.org

R Core Team. 2019. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Ramirez-Restrepo, L., S. Koi, and I. MacGregor-Fors. 2017. Tales of urban conservation: Eumaeus butterflies and their threatened cycad hostplants. Urban Ecosystems 20:375–378.

Sandiford, E., and S. Barrett. 2010. Albany regional vegetation survey, extent type and status. Funded by the Western Australian Planning Commission, South Coast Natural Resource Management Inc. and City of Albany for the Department of Environment and Conservation.

Sato, K. C., B. Guneralp, and L. R. Hutrya. 2012. Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. Proceedings of the National Academy of Sciences of the United States of America 109:16083–16088.

Soanes, K., and P. E. Lentini. 2019. When cities are the last chance for saving species. Frontiers in Ecology and the Environment 17:225–231.

Soanes, K., M. Sievers, Y. E. Chee, N. S. G. Williams, M. Bhardwaj, A. J. Marshall, and K. M. Parris. 2019. Correcting common misconceptions to inspire conservation action in urban environments. Conservation Biology 33:300–306.

Titley Scientific. 2009. Bat calls of southwest Western Australia. Field guide presented at Anabat Techniques Workshop, Manjimup, Western Australia.

Tulloch, A. I. T., M. D. Barnes, J. Ringma, R. A. Fuller, and J. E. M. Watson. 2016. Understanding the importance of small patches of habitat for conservation. Journal of Applied Ecology 53:418–429.

Unno, A. 2002. Tree species preferences of insectivorous birds in a Japanese deciduous forest: the effect of different foraging techniques and seasonal change of food resources. Ornithological Science 1:133–142.

Valentine, L. E., H. Anderson, G. E. S. Hardy, and P. A. Fleming. 2013. Foraging activity by the southern brown bandicoot (Isodon obesulus) as a mechanism
for soil turnover. Australian Journal of Zoology 60:419–423.

Van Dyck, S., I. Gynther, and A. Baker. 2013. Field companion to the mammals of Australia. Third edition. New Holland Publishers, London, Sydney, Auckland, Australia.

Van Helden, B. E., P. G. Close, and R. Steven. 2020b. Mammal conservation in a changing world: Can urban gardens play a role? Urban Ecosystems 23:555–567.

Van Helden, B. E., P. G. Close, B. A. Stewart, P. C. Speldewinde, and S. J. Comer. 2020a. An underrated habitat: Residential gardens support similar mammal assemblages to urban remnant vegetation. Biological Conservation 250:108760.

Wayne, A. F., J. F. Rooney, C. G. Ward, C. V. Vellios, and D. B. Lindenmayer. 2005. The life history of Pseudocheirius occidentalis (Pseudocheiridae) in the jarrah forest of south-western Australia. Australian Journal of Zoology 53:325–337.

Wilkinson, G. S., and J. M. South. 2002. Life history, ecology and longevity in bats. Aging Cell 1:124–131.

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