Vegetation cover and not size of remnants determines composition and diversity of ground-dwelling arthropods in native vegetation remnants

JOSLYN BERKELAAR [1*], PETER SPELDWINDE [1] & CLAIRE FARELL [2]

[1] *Centre of Excellence in Natural Resource Management, University of Western Australia, Albany, WA 6330 Australia*

[2] *University of Melbourne, Parkville, VIC 3010 Australia*

* *corresponding author:* ✉ *joslyn.berkelaar@uwa.edu.au*

Human urban populations continually grow and expand around the globe, and the urban footprint can directly and indirectly have deleterious effects on biodiversity of native flora and fauna through fragmentation. This study examined whether remnant area and habitat type between urban remnants affected arthropod biodiversity. Eighteen remnants within urban areas of a growing city in the South-western Australian Global Biodiversity Hotspot were surveyed using pitfall traps for ground-dwelling arthropods. Contrary to our hypothesis that arthropod diversity would increase in larger remnants, we found that size of remnant habitats had no effect on arthropod diversity; rather habitat composition had a much greater influence on arthropod diversity. Although remnant size had no significant effect on arthropod diversity, larger remnants supported a greater diversity of species that utilise the same type of resources, known as functional guilds. In our study we found that phytophagous (herbivores) and parasitoid functional guilds were more abundant in larger fragments, while the habitat structure and cover in each remnant affected scavengers, detritovores and pollinators. The abundance of angiosperms in remnants increased arthropod pollinator diversity, while increased sedge (Cyperaceae) cover decreased pollinator diversity. Interestingly, an increase in tree and leaf-litter cover decreased the number of detritivores collected. As all sites were identified as "ecologically functional" with maintenance of biogeochemical cycling, this is likely to closely reflect the arthropod diversity in Albany's remnants and would have outweighed the effects of remnant size on diversity. This concludes that healthy habitat patches of all sizes are useful to maintain arthropod populations.

KEYWORDS: Vegetation Composition; Arthropods; Functional Guilds; Urban Remnants; Island Biogeography; Habitat Matrix.

INTRODUCTION

Human populations are increasingly being concentrated in urban areas around the globe, with 70% of the world's population projected to live in urban areas by 2050 (United Nations 2008). For example, in Australia, 83% of the population currently live in cities and towns (ABS 2012). As human populations are increasingly concentrated into urban areas, native vegetation is fragmented by development and infrastructure and remnant habitats in cities can be some of the last remaining examples of ecosystems that once covered vast areas of the landscape (e.g. Sydney's Eastern Suburbs Banksia Shrub and Melbourne's Western Basalt Plains Grasslands (McDonnell 2007)). Consequently, urban areas can become a mix of infrastructure, humanity and remnant ecosystems.

Urbanisation fragments native vegetation resulting in vegetation patches that range in size, shape and connectivity. Native vegetation is considered vital for maintaining ecosystem processes within urbanised environments as vegetation controls temperatures, erosion, water runoff, nutrient cycling, air quality and provides habitat for wildlife (Dodds, Wilson *et al.* 2008). Fragmentation of native vegetation and the introduction of urban infrastructure, such as roads and drains modifies these processes (Alberti, Marzluff *et al.* 2003) and reduces the area available for remnant flora and fauna communities (Oliver, Hong-Wa *et al.* 2011). Urbanisation is considered a major anthropogenic risk to biodiversity (FitzGibbon, Putland, & Goldizen 2007) and the preservation of remaining habitat remnants has been identified as being critical for maintaining urban biodiversity (Bennett & Gratton 2012). To this end, ecological studies, worldwide, are establishing relationships between landscape structure, urban development and the persistence of native species (Alberti 2005).

Fragmentation and the introduction of barriers confines biota to isolated pockets or patches within an urban landscape, increasing the likelihood of species confined to only a few small remnants (Parker & Mac Nally 2002). For these species, the ability for species to move between disconnected habitats becomes vital for maintaining demographic and genetic stability of populations (Magle, Theobald & Crooks 2009). Additional influences on species survival in fragmented urban patches include their isolation, dispersal ability, ability to survive stochastic events and the habitat health of the remnant (Drinnan 2005; Niemela 1999). Species movement between patches is a function of their dispersal ability and patch isolation (Braaker *et al.* 2014; Magle *et al.* 2009). Urban development generally increases the unusable habitat for native species around habitat remnants (Bennett & Gratton 2012) resulting in the isolation of remnant vegetation patches. The survival of a species, during stochastic events in small remnant habitat patches, may depend on its mobility and its ability to colonise other fragments and establish and maintain breeding population (Abensperg-Traun & Smith 1999).

Vegetation structure and the health of a patch will influence its biological value. The greater diversity of habitats within an area increases the number of species (Hortal, Triantis *et al*. 2009). Community composition of a patch tends to be altered by urbanisation through the introduction of 'edge effects' (Marcantonio, Rocchini, Geri, Bacaro & Amici 2013; Porensky & Young 2013). Edge effects change the natural patterns of wind, light, temperature across the landscape (Porensky & Young 2013) and can also facilitate the introduction of exotic weed and pest species (Bolger, Suarez *et al.* 2000); the effect of which is expected to be greater in smaller and more irregularly shaped remnants (Porensky & Young 2013). Consequently, urbanisation often results in simplification of vegetation structural diversity within patches (Bryne 2007). Therefore, maintaining vegetation structure and health will provide a diversity of habitat types and have a positive influence on species survival (Cook 2002) and contribute to maintaining key biophysical processes such as: preventing soil erosion, reducing flooding and protecting water quality (Naiman & Decamps 1990).

To understand the effects of urbanisation and fragmentation on ecosystem functioning, ground-dwelling arthropod diversity and composition have often been studied (Bennett & Gratton 2013; Kowarik 2011; McIntrye, Rango, Fagan & Faeth 2001; Philpott *et al*. 2014). Arthropods provide a useful model for investigating the effects of urbanisation as they represent the most diverse taxon in most ecosystems and are vitally important to the health of the natural environment (Bolger *et al.* 2000; McIntyre 2000). In ecosystems, ground-dwelling arthropods help break down and redistribute nutrients into the soil (Didham, Ghazoul, Stork, & Davis 1996; Bolger, Suarez *et al.* 2000; McIntyre, Rango *et al*. 2001), pollinate flora (Didham, Ghazoul *et al.* 1996), biologically control the rate at which plants and pest species grow and multiply (Bolger *et al.* 2000; Bennett & Gratton 2012) and are the basis for many food webs (Bolger *et al*. 2000). In urban landscapes the relationship between arthropod diversity and fragmentation characteristics such as fragment size and connectivity is not a linear relationship but instead dependent on thresholds or minimum remnant size (Drinnan 2005). Urbanisation alters arthropod abundance and diversity through changes in land use, habitat structure and climate (Bennett & Gratton 2013; McIntyre *et al.* 2001; Philpott *et al.* 2013). For example, Gibbs and Hochuli (2002) found that anthropogenic disturbances such as habitat fragmentation alters arthropod assemblages, with opportunistic species, particularly spiders and wasps from higher trophic levels, becoming more common in smaller habitats. In terms of land use change, Bennett and Gratton (2012) found that changing from rural to urban land use negatively affected parasitic Hymenoptera abundance and diversity. In this same study, parasitoid abundance increased at a local scale as the floral diversity increased within urban sites (Bennett & Gratton 2012).

Conserving biodiversity within our cities is a global issue that commonly focuses on establishing protected natural areas and linking corridors (Hostetler *et al.* 2011). To assess whether urban remnant planning needs to be based on size of remnants or on the vegetation composition of each remnant, the effects of both patch size and habitat composition and health on arthropod diversity and abundance were examined in remnant native vegetation patches. This study focused on habitat patches in a growing regional city situated in south-western Australia, within a global biodiversity hotspot. We hypothesized that: 1) the size of vegetation remnants would have a significant positive influence on the ground dwelling arthropod composition and abundance; and 2) more diverse vegetation structure and composition would increase the arthropod diversity in remnants.

METHODS

Site description and selection

Vegetation structure and arthropod biodiversity of individual remnant patches within the City of Albany, Western Australia (population approximately 35,500 [City of Albany 2007]) was investigated. The City of Albany is a growing regional centre situated within the South-western Australian Biodiversity Hotspot. By the beginning of the 21st century, 63% of the vegetation within the City of Albany local government area had been cleared for agriculture and urban growth (City of Albany 2007). Within the Albany region over 800 flora species have been recorded in a variety of vegetation types (Heath, Low Woodland, Scrub, Reed swamps, Woodland/Forest, Scrub-Heath) (Sandiford & Barrett 2010). A total of 38 reserves within the city boundaries had been established for the preservation of vegetation, historical values and significant wetland habitats. From the 38 reserves in the City of Albany, we excluded wetland reserves from our study and those reserves sharing common boundaries were amalgamated; restricting the total number of reserves in the study to 18 (Figure 1). The majority of the landscape between reserves was urban residential, commercial and light industrial.

Vegetation structure, composition and Landscape measures

The 18 remnant reserves ranged in size from 0.26 – 219.8 ha, and were then classified into three arbitrary size categories 'large' (> 75 ha), 'medium' (6–74 ha), and 'small' (< 5.5 ha). The area (ha) of each reserve was calculated using ArcGIS (ESRI, version 8). Remnants were also classified initially into four distinct habitat types based on vegetation structural composition following Specht's (1970) classification; granite outcrop, shrubland, heathland, forest. In order to classify the structure and composition of vegetation within each of the 18 vegetation remnants, Landscape Function Analysis (Tongway & Hindley 2005) and the Landscape Organisation Index (Tongway & Hindley 2005) were used. Landscape Function Analysis (LFA) indirectly determines ecosystem function and resource capture by calculating the proportion of patch (indicating potential resource capture) to inter-patch (indication

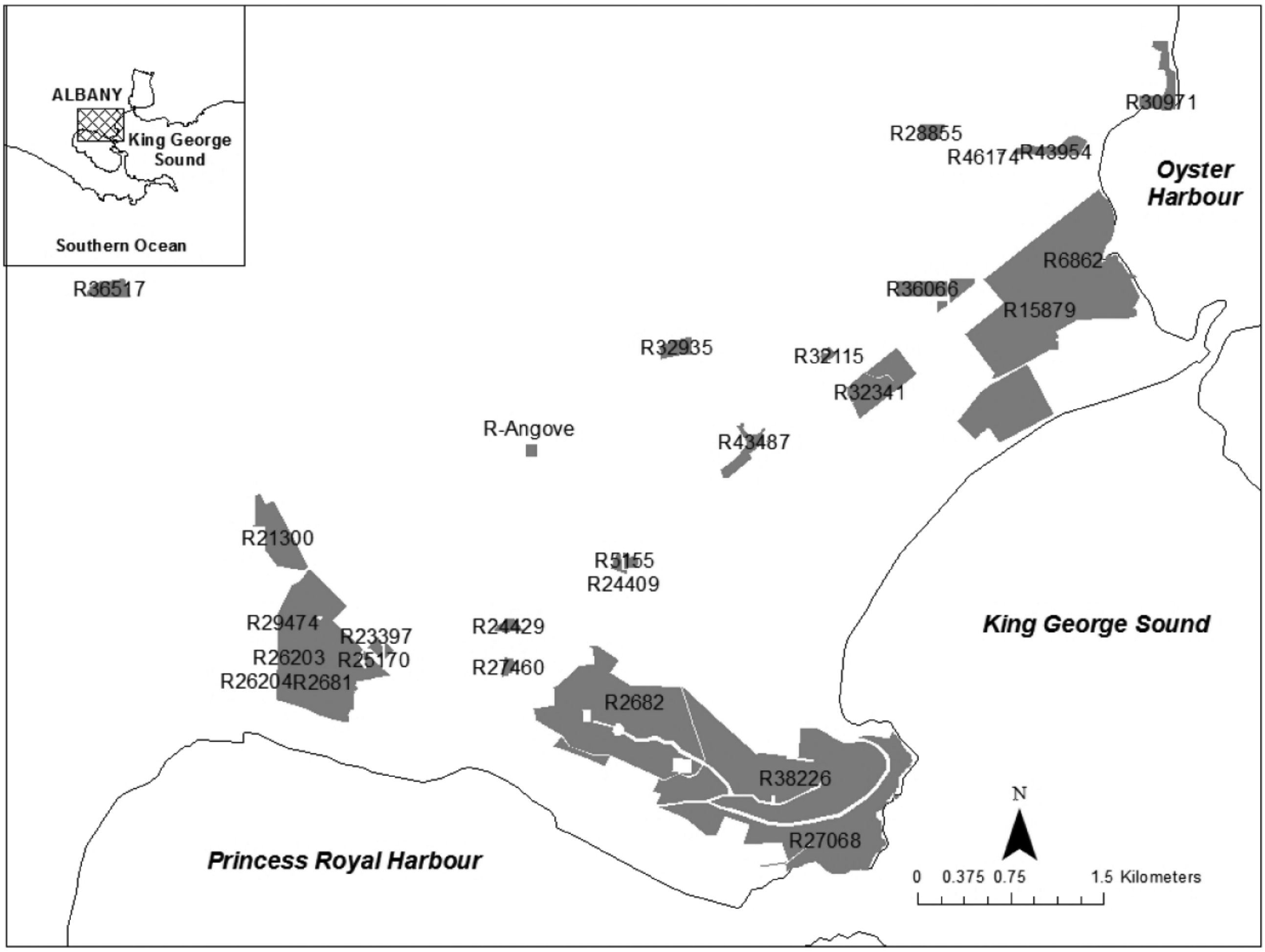

Figure 1. Remnant Bushland reserves in the city of Albany, WA.

potential resource loss) areas along transects. These simple indicators (patch and inter-patch zones) have been used to assess how well an ecosystem works as a biogeochemical system, i.e. whether resources are lost or retained by the system (Ampt 2007). It has been used around Australia on agriculture, orchards, and rangelands and provides useful comparison between different land-uses, on similar land types (Ampt 2007). In this analysis, patches are landscape features in contact with the ground (e.g. rocks, leaf-litter, low shrubs, stems, trunks and tussocky grasses), which prevent loss of water and nutrient resources from the system (Ludwig & Tongway 1997). Landscape Function Analysis determines the percentage cover of all vegetative life-forms as well as the entire patch and inter-patch zones along each transect. The patch and inter-patch data is then used to calculate a Landscape Organisation Index (Tongway & Hindley 2005).

The Landscape Organisation Index was calculated as the sum of all patch lengths along the transect divided by the total length of transect. The index places a landscape site on a continuum between highly functional where all resources are likely to be retained (LO = 1) and dysfunctional where resources are lost (LO = 0) (Tongway & Ludwig 1997). For example, a totally bare transect would have an index of 0 (zero) or if the transect was entirely covered by resource trapping patches (e.g., forest with closed understorey or continuous leaf-litter cover) the index would be 1.

For determination of the Landscape Organisation Index data was collected along two 20 metre long transects established in the centre of each fragment (Tongway & Hindley 2005). Location in the centre of remnants minimised edge effects which may have confounded interpretation of the data (Bolger, Suarez *et al*. 2000). Each transect was positioned parallel to the slope, in the observed direction of surface water flow and the location and orientation of each transect was recorded. All vegetative life-forms that intersected transects on a vertical plane were recorded. The length (extent of vegetation along the transect tape measured in metres), height (m) and percentage cover (1 m^2 quadrats) of vegetation patches (tree, shrub, herb, weed, grass and sedge), and the length (m) and percentage cover (1 m^2 quadrats) of non-vegetation groundcover patches (cryptogam, woody debris and leaf-litter) were measured. The length (extent along the transect tape, metres) of bare ground and rocks defined as inter-patches was also recorded. One person collected all estimated percentage cover (canopy and non-vegetation components) visually to avoid multi-observer bias in the measurements.

Arthropod Collection

Three pitfall traps (500 ml plastic specimen jars filled with 2.5 cm of ethylene glycol) were established along each transect (i.e. 6 per site) Gibb & Hochuli (2002). Traps were positioned at 0, 10 and 20 m from the origin of each transect (McIntyre *et al*. 2001). All traps remained closed for at least a week after deployment of trap lines to ensure arthropods were trapped under the same conditions for all transects and to avoid digging-in effects (Gibbs and Hochuli 2002). Traps were open for seven days during March, 2011. At the end of this period, samples were preserved in ethyl alcohol (70% ethanol) for later identification. Samples were sorted with the aid of a stereo dissecting light microscope and all arthropods and accidental captures were counted and identified to Family or Order level (CSIRO 1991), then morphospecies; differentiating arthropods within orders due to their external appearance, and functional guild. Spiders were identified to family using Schimming (2010); and ants were identified to subfamily using Shattuck (1999) and functional guild using Andersen and Majer (2004).

Statistical Analysis

Univariate data analysis was conducted using Genstat 12.1.0 (2009, VSN International Ltd.) and multivariate data analysis using Primer 6 (2001, Primer-E Ltd). Initially, remnants were characterised by area, habitat classification (Specht 1970) and Landscape Organisation Indices. A univariate approach was used to compare all vegetation characteristics (cover and vegetation life-form) observed among remnant area and habitat type. Differences in Landscape Organisation indices among remnants were determined using Analysis of Variance (ANOVA) and Tukey's studentised test (significance level set to α = 0.05). Using the Bray-Curtis Distance Matrix; Principle Component Analysis (PCA) was used to examine patterns in vegetation composition and canopy cover among sites. Arthropod data was pooled from all traps along both transects in each site. The richness and diversity (Shannon-Weiner Index) of both species and functional groups of arthropods among remnants was calculated. The Shannon-Weiner Index was calculated to take into account the species richness and the proportion of each species within a study site, as both of these factors influence diversity. All arthropod data were Log (x+1) transformed to satisfy assumptions of normality and heteroscedasticity. A similarity matrix between samples was constructed using the Bray-Curtis Similarity Index for community structure based on arthropod diversity.

The algorithm SIMPER (Primer) was used to identify the species and vegetation variables that contributed most to the similarities and difference between remnants. These variables were then analysed univariately to determine whether they had any significant influence on the functional guilds found in each of the remnants (regression analysis and ANOVA). The relationship between vegetation and biodiversity composition was examined using linear regression and ANOSIM to calculate whether there were any significant differences between patch vegetation structure or quality (habitat classification, vegetation characteristics and Landscape Organisation Index) and arthropod richness and diversity.

RESULTS

Habitat Data

The 18 remnants varied in size, with three 'large' (> 75 ha), 3 'medium' (> 6 ha), and 12 'small' (< 5.5 ha) patches. Seven habitat classes were identified; Closed Forest, Low Forest, Low Open Forest, Woodland, Shrubland,

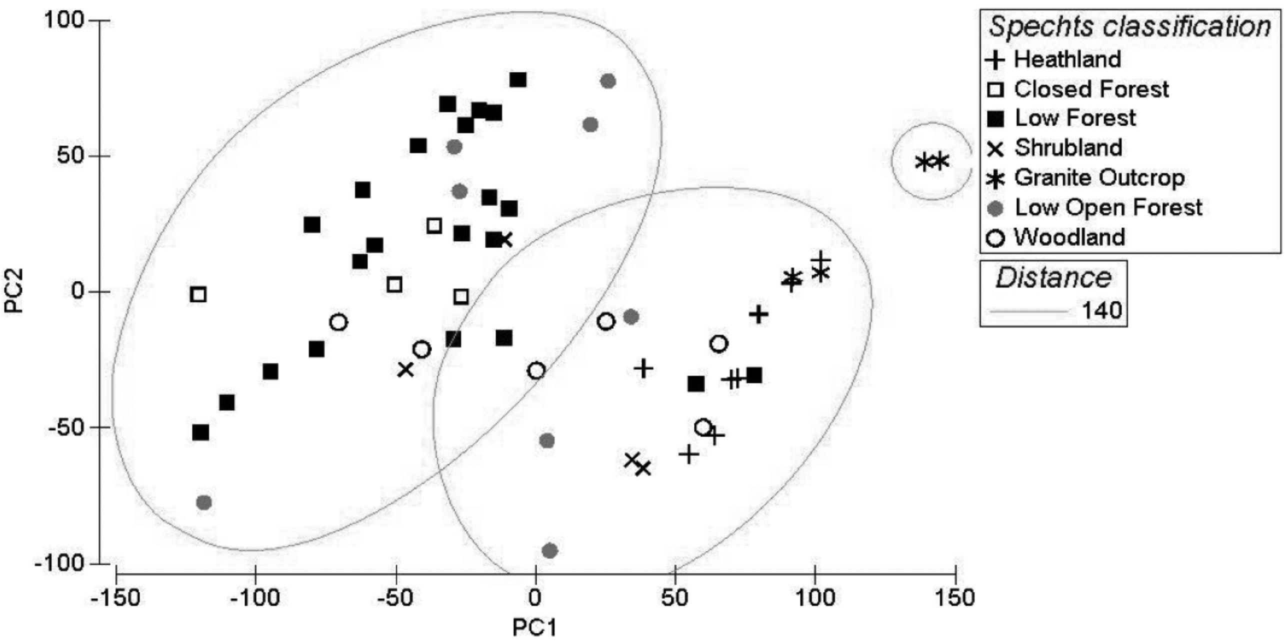

Figure 2. Principle Component Analysis (PCA) of remnants based on vegetative classification (Spechts 1970) and objective clusters (circles) using dissimilarity matrix at Level 140.

Heathland and Granite Outcrop. Landscape Organisation indices for these habitat classes ranged from 0.48 (Granite outcrop) to 1.0 (Closed Forest). For the remaining habitat classes Landscape Organisation Indices were all greater than 0.78: Heathland (0.78), Low Forest (0.97), Low Open Forest (0.92), Woodland (0.92) and Shrubland (0.95). Remnant area had no significant effect on the Landscape Organisation Index of patches (ANOVA; $P > 0.05$)

Principle Component Analysis (PCA) identified substantial overlap in vegetation cover and life-form among the habitat classes identified using Specht's (1970) classification (Figure 2). The first two principle components together explained 72.7% of the variation among sites. Objective classification using dissimilarity matrix at level 140 combined habitat classifications into three broad vegetation structural types. Group 1: High Vegetative Structure comprised Closed Forest, Low Forest, and most Low Open Forest; Group 2: Low Vegetative Structure comprised predominately heathland, shrubland, and most woodland; and Group 3: Granite Vegetative Structure compromised solely of granite outcrop habitat (Figure 2). Of the 11 vegetation life-form or cover variables measured, PCA showed that five variables (percentage cover of canopy, trees, shrubs, leaf-litter and sedges along each transect) had the greatest influence on differences between sites (Table 1).

Arthropod Data

In this study a total of 10 352 arthropod individuals were collected, representing 163 morphospecies, 29 orders and 9 functional groups (Table 2). The most diverse order was Hymenoptera with 37 different taxa, and the most abundant species was Collembola *Entomobryidae* sp.. Species diversity did not vary between remnants of different size (P = 0.7) (Figure 3), nor was there a significant difference due to ecological function (Landscape Organisation Index) of remnant sites (ANOVA, P = 0.98) (Figure 4). Although total species diversity did not vary between remnants of different sizes, when considered separately, members

Table 1. Eigenvalues for Component 1(PC1) and Component 2 (PC2). The highlighted values are those that contributed most to the differences between remnants (Figure 2)

Variable	PC1	PC2
Canopy cover	-0.722	-0.436
Tree	-0.423	0.346
Shrub	-0.226	-0.308
Weeds	-0.002	-0.008
Herb	-0.042	0.029
Cryptogram	0.132	-0.150
Woody debris	-0.103	-0.007
Leaf-litter	-0.447	0.428
Inter patch	0.131	-0.015
Sedge	0.039	-0.623

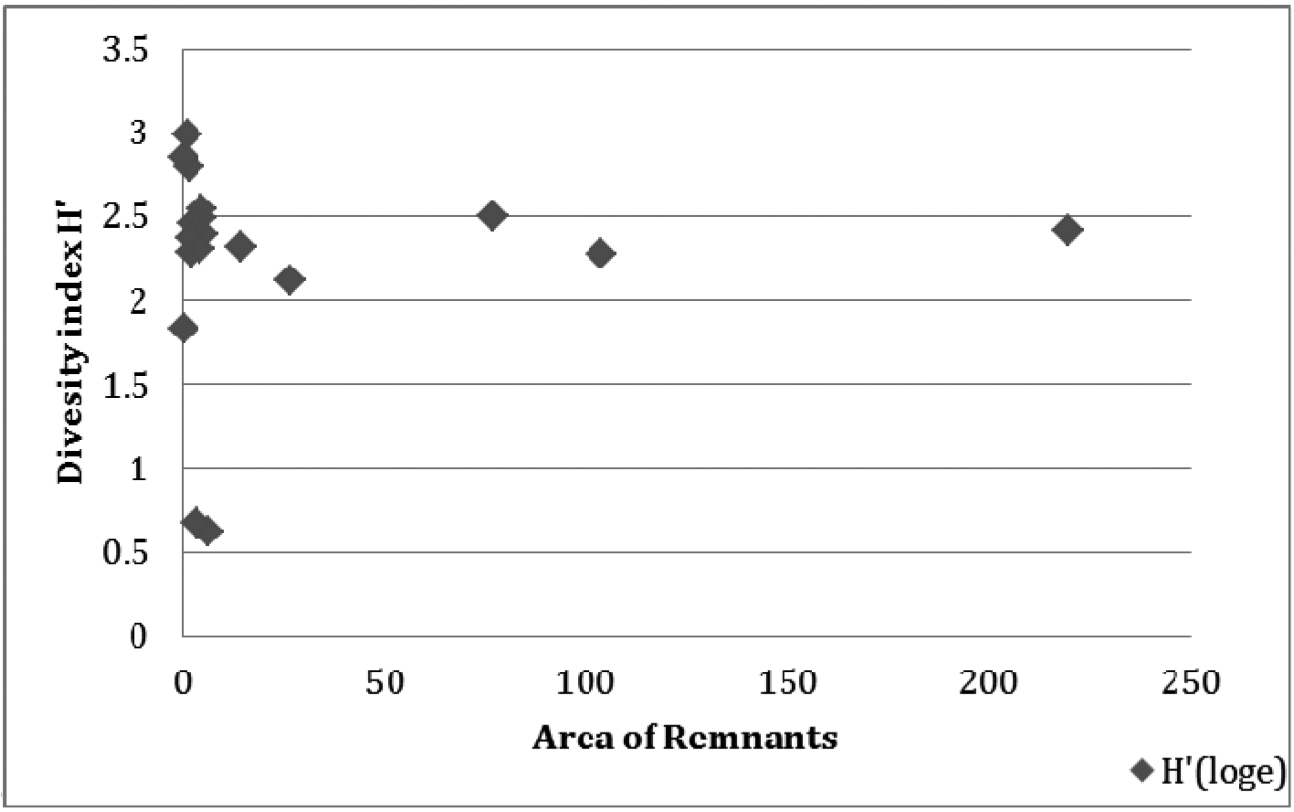

Figure 3. Effect of remnant size on diversity of arthropods in remnants surveyed within the City of Albany, Western Australia; $R^2 = 0.007$ showing that area has no effect on arthropod diversity.

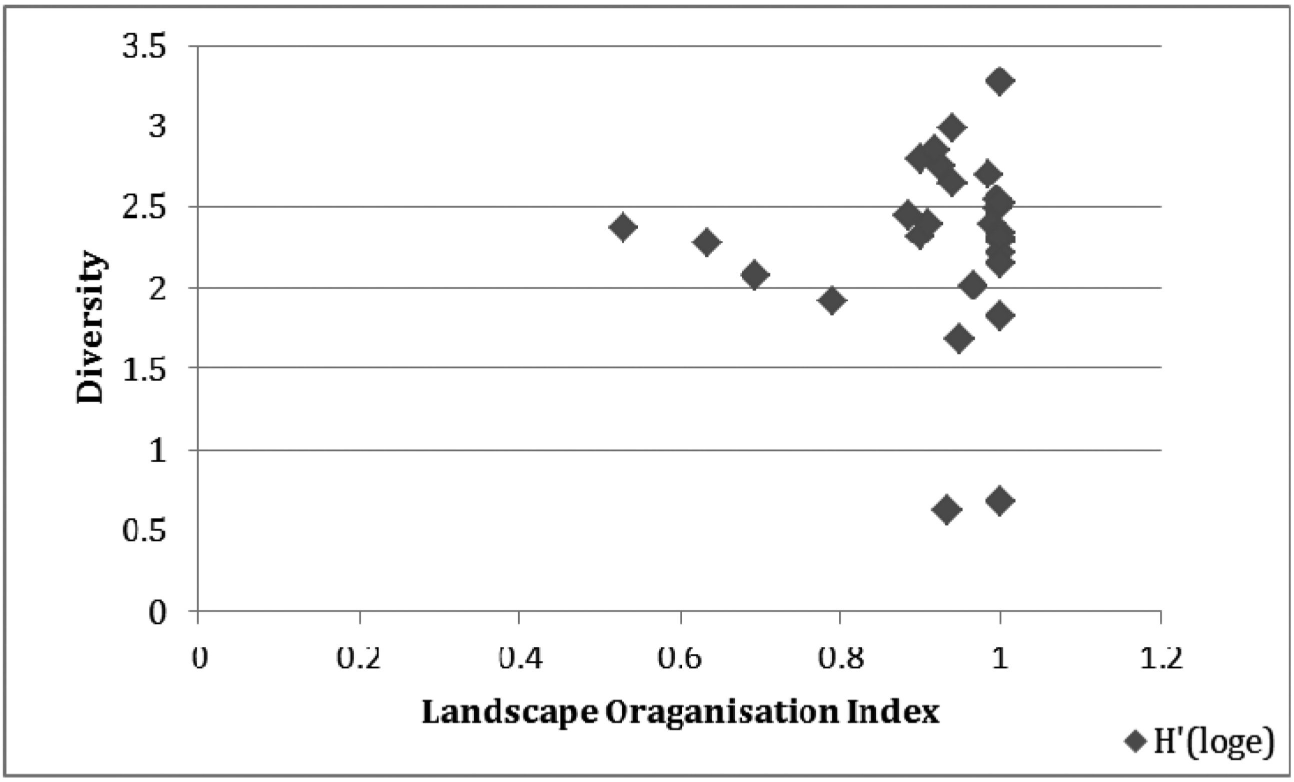

Figure 4. Relationship between Landscape Organisation Index (LOIndex) and diversity of Arthropods in remnants surveyed within the City of Albany, Western Australia (R^2 <0.01).

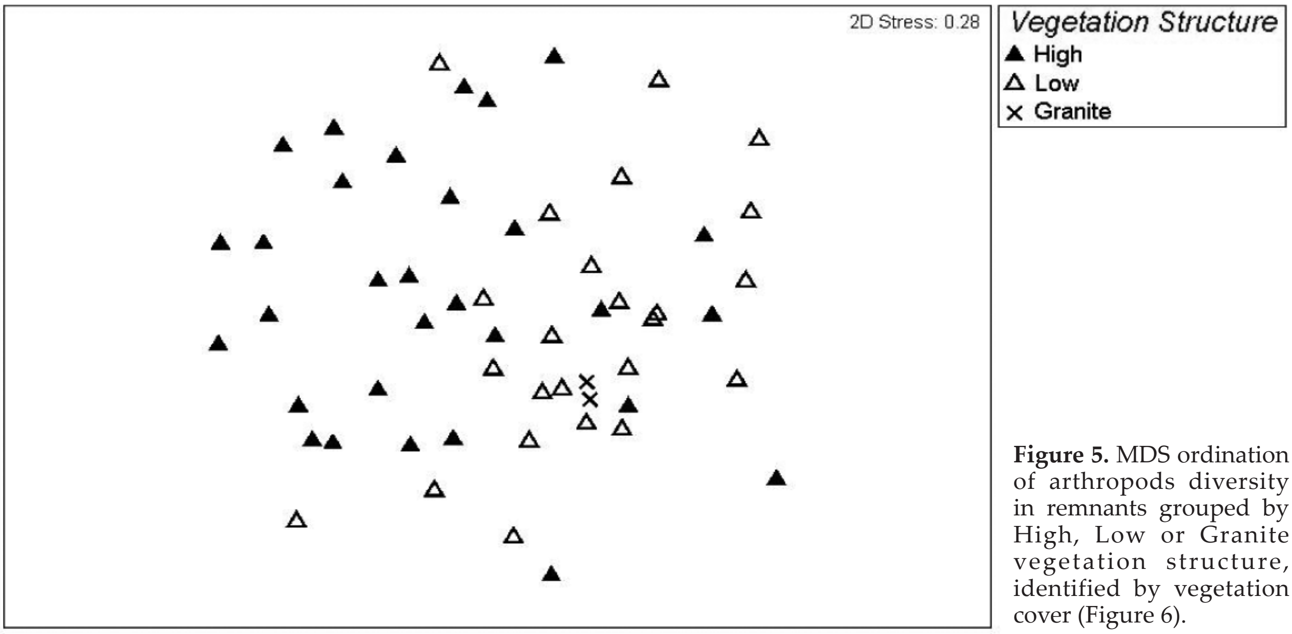

Figure 5. MDS ordination of arthropods diversity in remnants grouped by High, Low or Granite vegetation structure, identified by vegetation cover (Figure 6).

of phytophagous and parasite functional guilds were significantly more abundant in larger remnants (ANOVA; *P* = 0.03 and 0.02, respectively).

There was no relationship found between species richness and Landscape Organisation Indices (P>0.05) nor was there any difference in richness due to area of remnant (P>0.05). There was no significant difference in arthropod diversity among habitat classifications (ANOVA; *P* = 0.7), and only weak differentiation among vegetation structural groups (ANOSIM; Global R = 0.086, *P* = 0.009) (Figure 5). The species which contributed most to the similarities and differences within habitat groups were: Collembola, Formicidae and Amphipoda (SIMPER analysis).

The vegetation life-form and cover variables with the greatest influence on vegetation structural or habitat differences (Table 1) showed a non-significant influence on arthropod species diversity (Figure 5) but had a

Table 2. Arthropod Orders collected organized according to functional guilds.

Arthropod Orders	Predator	Phytophagous	Scavanger	Parasite	Ditritivore	Coprophagous	Omnivore	Pollinator	Forager
Araneae	14	0	0	0	0	0	0	0	0
Acarina	22	0	0	0	0	0	0	0	0
Amphipoda	0	0	1	0	0	0	0	0	0
Blattodea	0	0	2	0	0	0	0	0	0
Chilopoda	1	0	0	0	0	0	0	0	0
Coleoptera	5	9	5	0	3	1	0	0	0
Collembola	0	0	0	0	3	0	0	0	0
Dermaptera	0	0	0	0	0	0	3	0	0
Diplopoda	0	0	1	0	0	0	0	0	0
Diptera	1	0	0	3	0	1	6	2	0
Formicidae	0	0	0	0	0	0	0	0	7
Hemiptera	0	7	0	1	0	0	0	0	0
Hymenoptera	2	0	0	3	0	0	0	25	0
Isopoda	0	0	0	0	1	0	0	0	0
Isoptera	0	0	0	0	1	0	0	0	0
Lepidoptera	0	4	0	0	0	0	0	2	0
Nematodorpha	0	0	0	1	0	0	0	0	0
Neuroptera	1	0	0	0	0	0	0	0	0
Oligochaeta	0	0	0	0	1	0	0	0	0
Opiliones	0	0	0	0	0	0	3	0	0
Orthoptera	0	5	0	0	0	0	0	0	0
Phasmatoda	0	1	0	0	0	0	0	0	0
Phthiraptea	0	0	0	1	0	0	0	0	0
Pseudoscorpions	1	0	0	0	0	0	0	0	0
Psocoptera	0	0	1	0	0	0	0	0	0
Scorpion	1	0	0	0	0	0	0	0	0
Stephanocircidae	0	0	0	1	0	0	0	0	0
Thysanoptera	0	0	0	0	2	0	0	0	0
Thysanura	0	0	0	0	0	0	1	0	0

significant impact on the functional guilds found within the sites. While scavenger abundance was greater in remnant habitats with greater total canopy-cover, tree-cover and leaf-litter (ANOVA; $P = 0.005$, 0.0004, and 0.007 respectively), these variables had a significantly negative impact on the abundance of detritivores collected (tree-cover; ANOVA; $P < 0.001$) and leaf-litter (ANOVA; $P = 0.009$)). Pollinators were more abundant in remnants with greater tree-cover (ANOVA; $P = 0.05$) but decreased in those habitats with greater sedge-cover (ANOVA; $P = 0.02$). An increase in shrub-cover along transects had a positive significant influence on parasitic (ANOVA; $P = 0.008$) and coprophagous guilds (ANOVA; $P = 0.006$).

DISCUSSION

This study focused on the biodiversity of arthropods found in urban remnants within a growing urban centre. Contrary to our hypothesis that arthropod diversity would increase in larger remnants, we found that size of remnant habitats had no effect on arthropod diversity. Although arthropod diversity has generally been found to increase in larger remnants (Bolger *et al*. 2000; Faeth & Kane 1978; Yamaura *et al.* 2008) other studies have also found no relationship between species richness and urban fragmentation (Gibb & Hochuli 2002; Oliver *et al.* 2011; Parker & MacNally 2002). For example, Parker and MacNally (2002) found that grassland invertebrates in south-eastern Australia did not respond to habitat loss or habitat fragmentation when mowing decreased available habitat by 60 and 90%. In our study, although we found that arthropod diversity did not increase, phytophagous (herbivores) and parasitoid functional guilds were more abundant in larger fragments, while the habitat structure and cover in each remnant affected scavangers, detritovores and pollinators. Gibb and Hochuli (2002) also found that species richness was not greater in large than small fragments, rather assemblage composition responded to fragmentation. They found that generalist species were more abundant in smaller remnants, while predators and parasitoids were negatively affected by fragmentation (Gibb & Hochuli 2002). On the other hand Christie, Cassis and Hochuli (2010) found that trees in larger habitat patches supported fewer arboreal arthropods compared to tree in edges and small urban remnants. They also found a shift in functional grouping, with a greater number of herbivore invertebrates in small remnants and edge sites than interior sites (Christie *et al.* 2010).

There are a number of possible reasons for the absence of size effects on arthropod diversity in our study; firstly, the critical reserve size for arthropods could potentially be smaller than the remnant areas surveyed or that the number of remnants in each size class might not have been large enough to detect differences in arthropod diversity. Yamaura *et al.* (2008) surveyed 48 sites ranging in size from 2.4 to 296 ha and Bolger *et al.* (2000) studied 40 remnants ranging in size from 0.3 to 296 ha, while our study surveyed 18 remnants ranging in size from 0.25 to 219.8 ha. Another reason for the absence of any significant effect of remnant size on arthropod diversity in our study, is that, in the city of Albany, terrestrial remnant vegetation 'islands' are mostly surrounded by domestic gardens and green verges, which are landscapes that can be potentially utilised by arthropods as corridors (McIntyre 2000), essentially negating island and isolation effects as conceived in the Theory of Island Biogeography set out by MacArthur and Wilson (1963 and 1967). Key findings from a range of studies have found that the floral diversity and structural complexity of domestic gardens is an important predictor of arthropod abundance and diversity (Goddard *et al.* 2010).

Another more important factor that may have removed the effect of remnant size on arthropod diversity is ecosystem function of remnants. In our study, we used Landscape Function Analysis (LFA) to indirectly determine ecosystem function and resource capture by calculating the proportion of patch (resource capture) to inter-patch (resource loss) zones (Landscape Organisation Index). While Landscape Organisation Indices can range from 0.0 – 1.0 (Tongway & Hindley 2005), values in our study ranged from 0.48 – 1.0. This shows that for almost all the sites studied, that resource capture is greater than resource loss, implying that nutrients and water are conserved by the native vegetation rather than being lost from the system. As all sites were 'ecologically functional' with maintenance of biogeochemical cycling, this is likely to closely reflect the arthropod diversity in Albany's remnants and would have outweighed the effects of remnant size on diversity (McKinney 2008).

Although remnant size had no significant effect on arthropod diversity, larger remnants supported a greater diversity of species that utilise the same type of resources, known as functional guilds (Gardener, Cabido *et al.* 1995). In particular, larger fragments supported a greater abundance of phytophagous and parasitic arthropods. Larger fragments support greater vegetation, which increases the abundance of phtophagous arthropods (Bennett & Gratton 2013). As birds and mammals are more diverse in larger habitat fragments (Drinnan 2005; FitzGibbon, Putland *et al.* 2007) this would also increase the number of suitable hosts for arthropod parasites. Increased parasite abundance was also associated with increased shrub-cover along transects. Further, shrub-cover could also have increased the number of flowering species in these remnants and consequently the number of suitable host bird and mammal species for parasitoid arthropods. These results were also found by Bennett and Gratton (2012) who found a positive relationship between parasitoid abundance and flower diversity.

Vegetation structural diversity is generally considered important for maintaining the health of native vegetation (Cook 2002; Brodie 2003). Structural diversity also increases the possibility of greater habitat diversity; likely yielding increased diversity and greater possibilities for species survival (Cook 2002). Studies have shown that ground-dwelling and soil-arthropods are strongly influenced by habitat structure (Bryne 2007), for example Loyola and Martins (2008) found a positive correlation between structural heterogeneity, tree abundance and shrub height and Hymenoptera richness and abundance. Floral diversity particularly increases resources for pollinator arthropods (Hodge, Marshall *et al.* 2010). Jaganmohan, Vailshery and Nagendra (2013) found that in Bangalore, India the number of insect orders increased as the number of tree, herb and shrub species increase. In our study, vegetation structure greatly affected pollinator abundance, with increased tree-, shrub-, and herb-cover increasing pollinator abundance, while increased sedge-cover decreased pollinator abundance. This is not surprising as most species of trees, shrubs and herbs within the studied remnants were flowering plants (angiosperms) and other studies have found arthropod pollinators are predominately found on angiosperms, which provide them with food resources (Herrera & Pellmyr 2002). Sedges on the other hand, are mostly wind pollinated (Herrera & Pellmyr 2002) and an increase in sedge cover relative to angiosperm cover could lead to a decrease in diversity of pollinating arthropods, due to decreased food supply. This is consistent with other studies, for example, Hennig and Ghazoul (2012) found that floral abundance has a positive effect on bee diversity and Potts, Vulliamy, Dafina, Ne'eman and Willmer (2003) found that floral richness is highly correlated with bee species richness.

Vegetation cover and the presence of leaf-litter also influenced functional guild abundance. While other studies have found that detritivores increase in forest habitats (Bryne 2007); increased tree- and leaf-litter cover along transects in our study significantly reduced the number of detritivores. An increase in total vegetation canopy-cover, tree-cover, and leaf-litter cover was also associated with an increase in the number of scavengers, which also live in the leaf-litter feeding on dead plant material and animals (CSIRO 1991). As detritivore abundance is related to leaf-litter and soil moisture content, a decrease in detritivores collected may also be due to the lower than average annual rainfall that Albany has received in the 3 years prior to our study (< 700 mm compared to the expected average of 930 mm) (Bureau of Meterology 2011). Norton, Thomson, Williams and McDonnell (2014) also found that dry microclimate conditions had an effect on the arthropods collected, with more arthropods collected in grassland plots than bare patches. An increase in shrub-cover was accompanied by an increase in the numbers of coprophagous, or dung eating (Angel & Wicklow 1975), arthropods. As hypothesised for parasitiod numbers, increased shrub-cover may increase the number of vertebrates that live in or visit these remnants, increasing the food supply for coprophagous arthropods.

There are several conclusions that can be drawn from this study of remnants within a growing urban centre; firstly, the size of remnant had no significant effect on either ecological functioning or on arthropod or functional guild diversity. This lack of effect could be due to the fact that our remnant scale was either too big or too small to observe any significant differences or due to the high ecological functioning of these remnants; although additional studies should be carried out to test this further. Secondly, arthropod diversity and presence or absence of functional groups was highly influenced by vegetation structural composition within remnants, with percentage total canopy-, tree-, shrub-, sedge- and leaf-litter- cover being the most important habitat variables determining diversity.

Consequently, in urban areas where vegetation remnants are highly functional, a diversity of vegetation life-forms and habitat structure may be more important than remnant size for maintaining arthropod diversity. This may lead to planning in urban areas to focus on the maintenance of habitat health and structural diversity when planning their green areas and corridors for native biodiversity.

ACKNOWLEDGEMENTS

Thank you especially to Dr Paul Close for many hours of advice, constructive comments and manuscript discussion. We would like to thank all the field work volunteers and Caroline Vesteeg and Geraldine Jannike for their advice and knowledge in arthropod sampling and identifying.

REFERENCES

Abensperg-Traun M & Smith G T 1999. How small is too small for small animals? Four terrestrial arthropod species in different-sized remnant woodlands in agricultural Western Australia. *Biodiversity and Conservation* **8,** 709–726.

ABS 2012. Reflecting a Nation: Stories from the 2011 Census, 2012–2013. cat. no. 2071.0 (Australian Bureau of Statistic, Canberra).

Alberti M 2005. The Effects of Urban Patterns on Ecosystem Function. *International Regional Science Review* **28** (2), 168–192.

Alberti M, Marzluff J M, Shulenberger E, Bradley G *et al*. 2003. Integrating Humans into Ecology: Opportunities and Challenges for Studying Urban Ecosystems. *Bioscience* **53** (12), 1169–1179.

Andersen A N & Majer J D 2004. Ants show the way Down Under: invertebrates as bioindicators in land management. *Frontiers in Ecology and the Environment.* **2,** 291–298.

Ampt P 2007. Landscape Function Analysis (LFA) and its potential contribution to conservation-orientated land management *Biodiversity Extinction Crisis Conference – A Pacific Response*.

Angel K & Wicklow D T 1975. Relationships between Coprophilous Fungi and Fecal Substrates in a Colorado Grassland. *Mycologia* **67**(1), 63–74.

Bennett A B & Gratton C 2012. Local and Landscape scale variables impact parasitoid assemblages across an urbanization gradient. *Landscape and Urban Planning* **104**, 26–33.

Bennett A B & Gratton C C 2013. Floral diversity increases beneficial arthropod richness and decreases variability in arthropod community composition. *Ecological Applications*. **23**, 86–95.

Bolger D T, Suarez A V, Crooks K R, Morrison S A *et al.* 2000. Arthropods in Urban Habitat Fragments in Southern California: Area, Age, and Edge Effects. *Ecological Applications* **10**(4), 1230–1248.

Braaker S, Ghazoul J, Obrist M K & Moretti M 2014 Habitat connectivity shapes urban arthropod communities – the key role of green roofs. *Ecology.* **95,** 1010–1021.

Brodie L 2003. Protecting remnant bush on your land. NPWS, NSW National Parks and Wildlife Service.

Bryne L B 2007. Habitat structure: A fundamental concept and framework for urban soil ecology. *Urban Ecosystems* **10,** 255–274.

Bureau Of Meterology 2011. Climate of Albany. Retrieved 9/06/2011, from http://www.bom.gov.au/wa/albany/climate.shtml.

Christie F J, Cassis G & Hochuli D F 2010. Urbanization affects the trophic structure of aboreal arthropod communities *Urban Ecosytems* **13,** 169–180.

City Of Albany 2007. Albany Local Planning Strategy. Albany.

Cook E A 2002. Landscape structure indices for assessing urban ecological networks. *Landscape and Urban Planning* **58**, 269–280.

CSIRO (1991). *The Insects of Australia*. Melbourne, Australia Melbourne University Press.

Didham R K, Ghazoul J, Stork N E & Davis A J 1996. Insects in fragmented forests: a functional approach. *Tree* **11**(6), 255–260.

Dodds W K, Wilson K C, Rehmeier R L, Knight G L *et al.* 2008. Comparing Ecosystem Goods and Services Provided by Restored and Native Lands. *Bioscience* **58**(9), 837–845.

Drinnan I N 2005. The search for fragmentation thresholds in a Southern Sydney Suburb. *Biological Conservation* **124,** 339–349.

Faeth S H & Kane T C 1978. Urban Biogeography: City Parks as Islands for Diptera and Coleoptera." *Oceologia* **32**, 127–133.

Fitzgibbon S I, Putland D A & Goldizen A W 2007. The importance of functional connectivity in the conservation of a ground-dwelling mammal in an urban Australian landscape. *Landscape Ecology* **22,** 1513–1525.

Gardener S M, Cabido M R, Valladares G R & Diaz S 1995. The influence of habitat structure on arthropod diversity in Argentine semi-arud Chaco forest. *Journal of Vegetation Science* **6**, 349–356.

Gibbs H & Hochuli D 2002. Habitat fragmentation in an urban environment: large and small fragments support different arthopod assemblages. *Biological Conservation* **106**, 91–100.

Goddard M A, Benton T G & Dougill A J 2010. Beyong the garden fence: landscape ecology of cities. *trends in Ecology and Evolution.* **25**, 202.

Hennig E I & Ghazoul J 2012. Pollinating animals in the urban environment. *Urban Ecosystems* **15,** 149–166.

Herrera C M & Pellmyr O Eds. 2002. *Plant – Animal Interactions: an evolutionary approach.* Oxford, UK, Blackwell Publishing.

Hodge S, Marshall S A, Oliver H, Berry J *et al*. 2010. A preliminary survey of the insects collected using mushroom baits in native and exotic New Zealand woodlands. *New Zealand Entomologist* **33**, 43–54.

Hortal J, Triantis K A, Meiri S, Thebault E *et al.* 2009. Island species richness Increases with Habitat Diversity. *The American Naturalist* **174**(6), 205–217.

Hosttetler M, Allen W & Meurk 2011. Conserving urban biodiveristy? Creating green infrastructure is only the first step. *Landscape and Urban Planning* **100,** 369+371.

Jaganmohan M, Vailshery L S & Nagendra H 2013. Patterns of Insect Abundance and distribution in Urban Domestic Gardens in Bangalore, India. *Diversity* **5,** 767–778.

Kowarik I 2011. Novel ecosystems, biodiversity, and conservation. *Environmental Pollution.* **159,** 1974–1983.

Loyola R D & Martins R P 2008. Habitat structure components are effective predictors of trap-nesting Hymenoptera diversity. *Basic and Applied Ecology* **9**, 735–742.

Ludwig J A & Tongway D 1997. A Landscape Approach to Rangeland Ecology. In: *Function and Management: Principles from Australia's Rangelands*, LUDWIG J A, TONGWAY D, FREUDENBERGER D, NOBLE D & HODGINSON K (Eds.), Landscape Ecology, CSIRO Publishing, Melbourne, Australia, pp: 121–131.

MacArthur R H & Wilson E O 1963. An equilibrium theory of insular zoogeography. *Evolution* **17**, 373–387.

MacArthur R & Wilson E O 1967. *The theory of island biography*. Princeton, Princeton University Press.

Magle S B, Theobald D M & Crooks K R 2009. A comparison of metrics predicting landscape connectivity for a highly interactive species along an urban gradient in Colorado, USA. *Landscape Ecology* **24**, 267–280.

McDonnell M J 2007. Restoring and managing biodiversity in an urbanizing world filled with tensions. *Ecological Management and Restoration* **8**(2), 83–84.

McIntyre N E 2000. Ecology of Urban Arthropods: A Review and a Call to Action. *Ecology and Population Biology* **93**(4), 825–835.

McIntyre N E, Rango J, Fagan W F & Faeth S H 2001. Ground arthropod community structure in a heterogeneous urban environment. *Landscape and Urban Planning* **52,** 257–274.

McKinney M L 2008. Effects of urbanization on species richness: A review of plants and animals. *Urban Ecosystems* **11,** 161–176.

Naiman R J & Decamps H Eds. 1990. The ecology and management of aquatic ectones: *Man and the Biosphere series,* vol 4. Paris, UNESCO.

Niemela J 1999. Ecology and urban planning. *Biodiversity and Conservation* **8**, 119–131.

Norton B A, Thomson L J, williams N S G & McDonnell M J 2014. The effect of urban ground covers on arthropods: An experiment. *Urban Ecosystems* **17,** 77–99.

Oliver A J, Hong-wa C, Devonshire J, Olea K R *et al.* 2011. Avifana richness enhanced in large isolated urban parks. *Landscape and Urban Planning* **102**, 215–225.

Parker M & Mac Nally R 2002. Habitat loss and the habitat fragmentation threshold: an experimental evaluation of impacts of richness and total abundances using grassland invertebrates. *Biological Conservation* **105,** 217–229.

Philpott S M, Cotton J, Bichier P, Friedrich R L *et al.* 2014. Local and landscape drivers of arthropod abundance, richness and trophic composition in urban habitats *Urban Ecosystems* **17,** 513–532.

Porensky L M & Young T P 2013. Edge-effet interations in fragemented and patchy landscapes *Conservation Biology* **27,** 509–519.

Potts S G, Vulliamy B, Dafni a, Ne'eman G *et al.* 2003 Response of plant-pollinator communities to fire: changes in diversity, abundance and floral reward structure. *Oikos* **101,** 103–112.

Sandiford E M & Barrett S 2010. *Albany Regional vegetation survey: Extent, Type and Status*. A project funded by the Western Australian Planning Commision, South Coast Natural Resource Management Inc. and City of Albany for the Department of Environment and Conservation, Western Australia.

Schimming L 2010. S pider Eye Arrangement. *Bug Guide.*

Shattuck S 1999. *Australian Ants, Their Biolgy and Identification*. Melbourne, Victoria, Australia, CSIRO Publishing.

Specht R L 1970. *Vegetation in The Australian Envirionment*. G. W. Leeper. Melbourne, CSIRO and Melbourne Universtiy Press.

Tongway D J & Hindley N L 2005. *Landscape Function Analysis: Procedures for Monitoring and Assessing Landscapes – with special reference to Minesites and Rangelands*. Canberra, CSIRO Sustainable Ecosystems.

United Nations 2008. World Urbanization Prospects: The 2007 Revision Population Database. Retrieved 10/6/2011, from http://esa.un.org/unup/.

Yamaura Y, Kawahara T, Iida S & Ozaki K 2008. Relative Importance of the Area and Shape of Patches to the Diversity of Multiple Taxa. *Conservation Biology* **22**(6), 1513–1522.