

Green roof and ground-level invertebrate communities are similar and are driven by building height and landscape context

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Abstract

Green roofs are increasingly promoted for urban biodiversity conservation, but the value of these novel habitats is uncertain. We aimed to test two hypotheses: (i) green roofs can support comparable invertebrate family and order richness, composition and abundances to ground-level habitats and (ii) green roofs planted with native species from local habitats will support a richer invertebrate community at family and order level than other green roofs. We sampled the invertebrate community on green roofs dominated by native grassland or introduced succulent species in Melbourne, Australia, and compared these to the invertebrate community in ground-level sites close by, and sites with similar vegetation types. The only significant differences between the invertebrate communities sampled on green roofs and ground-level habitats were total abundance and fly family richness, which were higher in ground-level habitats. Second hypothesis was not supported as invertebrate communities on green roofs supporting a local vegetation community and those planted with introduced *Sedum* and other succulents were not detectably different at family level. The per cent cover of green space surrounding each site was consistently important in predicting the richness and abundance of the invertebrate families we focused on, while roof height, site age and size were influential for some taxa. Our results suggest that invertebrate communities of green roofs in Melbourne are driven largely by their surrounding environment and consequently the effectiveness of green roofs as invertebrate habitat is highly dependent on location and their horizontal and vertical connection to other habitats.

Key words: green infrastructure, pollinators, urban bees, arthropods, urban greening

Introduction

Increasing urbanisation is associated with the loss of urban green space as natural areas and parks are often destroyed by new or infill developments (Jim 2004). However, in many cities the area of green or vegetated roofs, a new and elevated form of urban green space, is rapidly increasing. The area of green roofs in some cities is already substantial. For example, Stuttgart has more than 200 ha of green roofs, Dusseldorf has 73 ha

(Holzmüller 2009), Zurich 87 ha (Mayrand and Clergeau 2018), Tokyo 55 ha (Carter and Fowler 2008) and Paris 44 ha (Mayrand and Clergeau 2018). It is therefore not surprising that green roofs are increasingly being considered as sites for urban biodiversity conservation. Cities are places of frequent disturbance, which can prevent the successional development of plant communities, remove important habitats, feeding and breeding resources and disrupt animal behaviour, making otherwise

suitable habitat inhospitable. The relative isolation of a green roof may allow for plant species and animals sensitive to disturbance, or prone to competitive exclusion, to persist. Green roofs have therefore been proposed as components of urban biosphere networks (Kim 2004), as sites that could support metapopulations of endangered butterflies in urban areas (Snep, WallisDeVries, and Opdam 2011) and replacement habitats for biodiverse areas destroyed by development (Lorimer 2008). They are also included in biodiversity policy recommendations and strategies in cities around the world, for example London (Greater London Authority 2010), Toronto (Torrance et al. 2013) and Basel (Kazmierczak and Carter 2010).

The majority of green roofs being constructed in cities are classified as extensive green roofs because they have relatively shallow, low nutrient, growing media and typically require low levels of management input. This design limits the types of vegetation that can be grown to low growing, often drought-tolerant species and prevents the establishment of taller vegetation. This means that most green roofs cannot have a complex vegetation structure of multiple strata, which at ground level are consistently associated with higher biodiversity levels (Tews et al. 2004; Threlfall et al. 2017). Green roofs specifically designed to improve biodiversity outcomes are often planted with species native to the local area, because it is assumed native plants will attract more fauna species than exotic plants. For example, 60% of the 105 green roof papers, Butler, Butler, and Orians (2012) identified as advocating the use of native plant species cited provision of habitat as a reason for doing so. Policies in cities, such as Basel and Toronto also encourage the planting of native species to increase the biodiversity value of green roofs (Kazmierczak and Carter 2010; Torrance et al. 2013), and green roof researchers often use a habitat-template approach to identify potentially suitable local native species for evaluation (Lundholm 2006; Sutton et al. 2012; Van Mechelen, Dutoit, and Hermy 2014). Other design features utilised to increase the biodiversity on green roofs include using a variety of different substrate materials, including transplanting ground-level brownfield habitats to green roofs, introducing microtopographic variation to create different habitats and incorporating structural diversity in the form of rocks, tiles or branches (Gedge et al. 2010).

Invertebrates are one of the most frequently reported groups of organisms to utilise green roofs. Numerous studies have surveyed invertebrates on green roofs and commonly record a diversity of orders, including Araneae (spiders), Coleoptera (beetles), Collembola (springtails), Lepidoptera (butterflies) and Hymenoptera (bees, wasps and ants) (Coffman and Davis 2005; Gedge and Kadas 2005; Baumann 2006; Brenneisen 2006; Kadas 2006; Colla, Willis, and Packer 2009; MacIvor and Lundholm 2011; Schindler, Griffith, and Jones 2011; Tonietto et al. 2011; Madre et al. 2013; Rumble and Gange 2013; Braaker et al. 2014; Blank et al. 2017; Joimel et al. 2018). Green roofs also have the potential to support pollinators as suggested by Tonietto et al. (2011), Ksiazek et al. (2014) and MacIvor, Ruttan, and Salehi (2015), all of whom found that green roofs are frequently utilised by a diversity of bee species. This is of interest given concerns about recent pollinator decline and our continued reliance on them for pollination services (Potts et al. 2010; Hall et al. 2017).

Many studies of green roof biodiversity have been conducted in isolation. Few researchers have compared green roofs to bare, non-vegetated conventional roofs (e.g. a negative control) (Williams, Lundholm, and Scott Macivor 2014) presumably because there is little insight to be gained. The studies that have

found green roofs to be more biodiverse than conventional roofs (Pearce and Walters 2012; Partridge and Clark 2018; Belcher et al. 2019). Somewhat surprisingly, relatively few studies have undertaken more conservation relevant comparison of invertebrate communities on extensive green roofs to those found in nearby equivalent ground-level habitats that have similar vegetation (for exceptions, see Ksiazek, Fant, and Skogen 2012; Braaker et al. 2014, 2017). There has also been little evaluation of green roofs specifically designed for biodiversity (Butler, Butler, and Orians 2012; Williams, Lundholm, and Scott Macivor 2014). The exceptions are Kadas (2006) who found that brownfield sites at ground level had higher invertebrate diversity than sedum green roofs and green roofs designed to replicate the brownfield sites, although these were relatively new at the time of the surveys. Pearce and Walters (2012) also found significantly higher bat feeding activity over biodiverse roofs compared to succulent and conventional roofs in London, perhaps reflecting increased invertebrate prey abundance.

The aim of this article is to test two hypotheses, namely that: (i) green roofs support family- and order-level richness, composition and abundance of invertebrates comparable with ground-level habitats and (ii) green roofs designed specifically to support native organisms support greater family- and order-level richness and abundance of invertebrates than green roofs not specifically designed for this purpose. To evaluate these hypotheses, we compared the invertebrate community on green roofs with communities from nearby ground-level vegetation and also ground-level vegetation with similar composition to the roofs. Because there had been no prior published studies of invertebrates on Australian green roofs, we chose to examine the broad range of invertebrates occurring on them. We compared the abundance and richness of invertebrates at order level and family level (Coleoptera, Diptera, Hemiptera and Hymenoptera). We focussed particularly on pollinator species (bees and hoverflies) due to their importance for ecosystem services and as a means of comparison with northern hemisphere green roof invertebrate studies.

Another aim is to determine the properties of the green roofs that were most important in explaining any observed differences in the invertebrate assemblage between sites. This could provide information critical for the effective siting and design of the future green roofs and enable the development of general ecological design principles. We investigated five factors likely to influence invertebrate composition. Per cent cover of vegetation in the area surrounding the roof (Smith et al. 2006), green roof age (Kadas 2006; Sattler et al. 2010), vegetation cover on the roof and roof size were hypothesised to positively influence invertebrate richness and abundance, while building height was hypothesised to have negative effect (MacIvor 2016).

Methods

Study sites

Six extensive green roofs in Melbourne, Australia, were available for this study (Supplementary Table S1). Melbourne (37° 49' S; 144° 58' E) has a Mediterranean-type climate, with warm, dry summers and cool, wet winters, with an average monthly rainfall of 54.1 mm (Australian Bureau of Meteorology 2018). The Australian green roof industry is under-developed compared to that of Europe and North America (Williams, Rayner, and Raynor 2010) and as a result there were relatively few extensive green roofs available to sample (<20).

The selected green roofs were either planted with a range of native forb and grass species from the critically endangered grasslands to the north and west of Melbourne (sites 1, 2 and 3), which are considered a habitat template for south-eastern Australian green roofs (Williams, Rayner, and Raynor 2010; Rayner et al. 2016), or with succulents, primarily introduced *Sedum* species (sites 4, 5 and 6) (Supplementary Table S1). All green roofs sampled had a scoria (lava rock)-based growing medium, similar to that described by Farrell et al. (2012), that was less than 300 mm deep. All roofs were un-irrigated. They were also relatively new, with the youngest constructed in January 2013, and the oldest in 2006 (Supplementary Table S2).

Each green roof was paired with two types of ground-level site, a 'companion' site and a 'nearby' site, to compare the invertebrate communities (Supplementary Fig. S1 and Supplementary Table S1). Companion sites were garden beds of similar sizes and had similar vegetation to their matched green roof with many of the same grassland or succulent species planted at both the roof and its companion site. They were as close as possible to their green roof; all were located within 900 m, except for sites 5 and 6 where the closest suitable sites were 7.2 km and 6 km away, respectively. We restricted sampling to vegetated areas of roofs for three reasons: (i) we are interested in whether these new structures contain invertebrates; (ii) unvegetated roof area provide extremely limited resources and would only contain taxa fleetingly resident; and (iii) access to vegetated roof areas is constrained by tight health and safety regulation, and access to unvegetated areas was not possible at most of our sites. As this is a natural experiment utilising existing sites, vegetation composition between green roofs and companion sites could not be identical. Grassland green roofs and companion sites had considerably more native plant species than the succulent green roofs and their companion sites. Nearby sites were vegetated areas close to the green roof—no more than 200 m away—and of similar size. They were predominantly small parks containing mown grass with scattered shrubs and few trees. By pairing each green roof with two ground-level sites, we were able to investigate the relative importance of vegetation proximity and similarity to the structure of invertebrate communities on green roofs. This design led to data collection at 18 sites clustered around six green roofs (six green roofs; six companion sites; six nearby sites).

Five site characteristics that have previously been shown to influence invertebrate diversity were also measured, either in the field or via aerial imagery using a Geographic Information System (Supplementary Table S1). The vegetation cover in a 100-m radius around each site was calculated by using 100 points in i-Tree Canopy (USDA Forest Service 2013) a web-based tool that facilitates measurement of landscape characteristics using Google Map aerial photos. Roof age ranged from 1 to 7 years. Age was based on the year of construction and landscaping of the site and was determined by consulting records held by local councils and building owners or managers. Roof height ranged from 3.3 to 45 m and was measured using a clinometer. Vegetation cover on the roof was the average cover of three 50 cm × 50 cm quadrats, appropriate for low vegetation, placed using a stratified random technique. The size of each green roof was measured in the field to the nearest square metre and vegetation cover was similar across the roofs.

Invertebrate sampling

Invertebrate communities were surveyed using two methods: (i) pitfall traps, which collect ground-active taxa (Luff 1975) and (ii)

pan traps, which are widely used for sampling agricultural pests (Southwood 1978), and to sample the abundance and diversity of a variety of insect communities (Leong and Thorp 1999) predominantly sample flying invertebrates. This combination of methods allowed us to survey diurnal and nocturnal invertebrates (Hill 2005). The sampling design aimed to capture a comparative sample between sites without assessing the full assemblage composition within sites.

Three 30-ml tube pitfall traps were installed in a line 50 cm from each other at each site. Each trap was placed in a polyvinyl chloride tube, with the upper edge of the pipe bevelled, so that the test tube opening was flush with the soil surface. To limit the 'digging-in effect' (increased catch due to the disturbance of digging the pitfall trap hole) (Digweed et al. 1995) these were inserted 1 week prior to each trapping period and were stopped. Traps were filled with 15 ml of ethanol and ethylene glycol to preserve invertebrates and left open for 7 days in autumn (19–26 April) and spring (23–30 September). The same trap locations were used during both trapping periods.

Three coloured pan traps (white, blue and yellow bowls, 15 cm in diameter, 4 cm deep) were placed haphazardly throughout each site for a 24-h period. Traps were half-filled with water and a drop of detergent to help break surface tension. Pan traps were left open for six 24-h periods at each site; three times in autumn (2–11 April), and three times in spring (30 September–3 October) 2013.

All invertebrates were transferred to vials of 70% ethanol in the field for storage. In the laboratory, all were sorted to order, class (Collembola, Diplopoda) or superorder (Acari) using keys in CSIRO Division of Entomology (1991). Samples were sorted using a Nikon stereomicroscope. All Coleoptera, Diptera, Hemiptera and Hymenoptera were sorted to Family using Lawrence et al. (2000) for Coleoptera, Hackston (2015), McAlpine (1958) and Cutter (2004) for the Diptera and CSIRO (2012) for the Hymenoptera. We chose to focus on insect taxa that were abundant in the samples, are highly diverse taxonomically and are more likely to have arrived on the roofs independently, rather than through passive transport (e.g. wind or in soil, which is likely for, for example, the Collembola). All bee and hoverfly specimens were then sorted to species where possible, and otherwise to morphospecies. Larvae were excluded from analyses because they were not the target of the trapping methods. Representative bee and hoverfly specimens from each species were air dried and pinned for taxonomic verification by Dr Ken Walker, an expert entomologist from Museums Victoria, Melbourne, and a reference collection assembled. Families identified were verified by staff from the Australian Museum. We pooled data from each trapping method and used the average per site across the two seasons sampled. With these data we calculated (i) total invertebrate abundance, (ii) the number of orders found per site (ordinal richness), (iii) Coleoptera family richness, (iv) Diptera family richness, (v) Hemiptera (Heteroptera, Auchenorrhyncha and Sternorrhyncha) family richness, (vi) Hymenoptera family richness; and (vii) combined richness and abundance of bee (Hymenoptera: Apidae) and hoverfly (Diptera: Syrphidae) (hereafter pollinator) species.

Analysis

To assess the richness and abundance of invertebrate orders, richness of Coleoptera, Diptera, Hemiptera and Hymenoptera families and species richness and abundance of pollinators we used generalised linear mixed models (GLMMs) using the lme4 package (Bates et al. 2015) in R (R Core Team 2017), where 'site'

was considered a random effect (to indicate which of the six site clusters each sample was associated with) and 'type' a fixed effect. Type included green roofs, companion and nearby site types. All response variables were continuous. Order abundance and beetle family richness data were log transformed to improve the normality of the data. Models were run using a Gaussian distribution for total abundance (log), Coleoptera, Diptera, Hemiptera and Hymenoptera family richness, pollinator abundance (log) and a Poisson distribution for ordinal richness. Due to large abundance of Collembola in most sites, we re-ran these analyses with and without Collembola to assess their impact on the results. To test the effect of grassland versus succulent vegetation types on roofs specifically, we compared the mean of each of the invertebrate response variables using a t-test. We then performed a series of one way analysis of similarity (ANOSIM) using PRIMER (Clarke and Gorley 2006), to determine any differences in invertebrate assemblage composition between green roofs and companion sites, green roofs and nearby sites, and between grassland and succulent green roofs. Analyses were based on log +1 abundances, used to calculate a Bray–Curtis similarity with 999 permutations for each test. To evaluate the effect of site characteristics on the invertebrate assemblage, we again ran GLMMs. Due to small sample size, each site characteristic was entered into a univariate model only, where variables were standardised to have a mean of zero and standard deviation of one. GLMMs were chosen here to allow for the inclusion of 'Site' as a random effect. The fit of each model was assessed using the Akaike Information Criterion corrected for small sample size (AICc), using package 'AICcmodavg' (Mazerolle 2016). The model with the lowest AICc and highest model weight (wi) were considered as the top model. Models within two AICc points of the top model were also considered to have strong support and are reported (Burnham and Anderson 2002).

Results

In total 32 489 organisms were collected, but most of these were from the Collembola (25 457 samples). The majority of the specimens were collected from companion sites (20 980 individuals), followed by nearby sites (9360 individuals) and green roofs (2149 individuals). There was no significant difference in the abundance of individuals caught between seasons ($P=0.9$). Thirteen orders of invertebrates were collected: Amphipoda, Araneae, Coleoptera, Collembola, Dermaptera, Diptera, Hemiptera, Hymenoptera, Isopoda, Lepidoptera, Orthoptera, Thysanoptera and Trichoptera (Table 1).

Effects of site type and vegetation type

Total invertebrate abundance and composition

Companion sites and nearby sites both had a significantly greater abundance of individuals than green roofs ($P < 0.001$ and $P = 0.01$, respectively). No significant differences were found between sites types for ordinal richness. There was no difference between abundance and ordinal richness on grassland or succulent roofs ($P > 0.05$). The composition of ordinal assemblages did not significantly differ across site types or roof types, $P > 0.05$ for all tests. Results for abundance and richness did not change after removing Collembola from the dataset.

Families: Coleoptera, Diptera, Hemiptera and Hymenoptera

We found Diptera to be the most family rich group in this study, followed by Hymenoptera and Coleoptera (Table 2). Nearby sites

had a significantly greater richness of Diptera (flies) than green roofs ($P=0.02$), however companion and green roof sites had comparable Diptera richness ($P > 0.05$). No other significant differences were found between site types for Coleoptera (beetles), Hymenoptera (bees, wasps, ants) or Hemiptera (true bugs) family richness ($P > 0.05$ in all cases). There was a marginally greater Coleoptera family richness on grassland roofs than succulent roofs ($P=0.047$), where only one of the three succulent roofs were found to contain beetles. No differences between family richness on grassland or succulent roofs were found. ANOSIM analysis revealed the composition of Diptera families differed significantly between ground-level habitats and green roofs, where green roofs had a significantly different Diptera composition to companion sites ($P=0.028$) and nearby sites ($P=0.006$). Ground-level sites had more individuals within the Sciaridae, Cecidomyiidae, Ephydriidae and Phoridae families while the Chironomidae were abundant on green roofs. The composition of Coleoptera, Hemiptera and Hymenoptera families did not significantly differ across site types or roof types.

Pollinators: bees and hoverflies

We collected 137 individual bees (Hymenoptera: Apidae) and hoverflies (Diptera: Syrphidae) across all site types. Seven bee species (or morphospecies) were collected. These included the introduced European Honeybee *Apis mellifera* (Linnaeus 1758), and six native Australian bee species all within the Halictidae: *Homalictus (Homalictus) sphecodoides* (Smith 1853), *Lasioglossum (Chilalictus) brunnesetum* Walker, 1995, *Lasioglossum (Parasphcodes) hilactum* (Smith 1853), *Lasioglossum (Chilalictus) cognatum* (Smith 1853) and two morphospecies of *Homalictus* Cockerell 1919 (hereafter *Homalictus* sp. 1 and 2). *Homalictus (Homalictus) sphecodoides*, *Homalictus* sp. 1 and 2 and *Apis mellifera* all occurred on at least one green roof. One native hoverfly species *Melangyna (Austrosyrphus) viridiceps* (Macquart 1847) was detected on three of the green roofs. Due to the low number of species detected, we analysed total abundance and species richness only. There was no difference in the abundance or richness of pollinator species between green roof and both types of ground-level sites. There was no difference in pollinator abundance or richness between grassland and succulent roofs or grassland and succulent ground-level sites.

Effect of habitat variables

The per cent cover of green space surrounding each site was the most consistent predictor of many of the invertebrate communities' biodiversity indices. Increases in green space cover surrounding each site were positively correlated with the richness and abundance of all aspects of the invertebrate community sampled (Table 3). Higher green roofs were associated with a decrease in total invertebrate abundance and fly family richness (Table 3). Increasing site size positively influenced beetle family richness, and increasing site age positively influenced fly family richness (Table 3).

Discussion

We found that green roofs in Melbourne have lower invertebrate abundance than their respective nearby and companion sites. Other studies have also established that green roofs have a lower abundance of invertebrates when compared with ground-level sites (e.g. Colla, Willis, and Packer 2009; MacIvor and Lundholm 2011; Tonietto et al. 2011). Only Kadas (2006) has found that green roofs had equivalent or greater abundance of

Table 1: Taxa sorted by total abundance

Site type	Green roof		Companion		Nearby		Total
	Median	(Min, max)	Median	(Min, max)	Median	(Min, max)	
Collembola	31.5	(0, 690)	1580.5	(5, 11 496)	1134.5	(2, 2553)	25 457
Diptera	49.5	(19, 87)	90.5	(20, 153)	258	(69, 772)	2734
Hymenoptera	63	(7, 153)	89	(62, 498)	81.5	(21, 191)	1961
Hemiptera	37	(4, 185)	38.5	(5, 824)	25.5	(13, 45)	1485
Trichoptera	5.5	(0, 149)	0.5	(0, 11)	9.5	(0, 36)	285
Acari	2.5	(0, 14)	9	(3, 37)	7.5	(1, 44)	219
Coleoptera	1.5	(0, 7)	5.5	(0, 25)	5	(0, 21)	115
Araneae	3	(0, 6)	6	(1, 17)	7.5	(2, 12)	105
Diplopoda	0	(0, 2)	3.5	(1, 14)	1	(0, 7)	48
Lepidoptera	1	(0, 5)	0.5	(0, 4)	0.5	(0, 4)	25
Isopoda	0	(0, 0)	0.5	(0, 11)	0	(0, 2)	18
Dermaptera	0	(0, 0)	0	(0, 14)	0	(0, 0)	15
Orthoptera	0	(0, 1)	0	(0, 5)	0	(0, 2)	13
Gastropoda	0	(0, 1)	0	(0, 1)	0	(0, 1)	4
Thysanoptera	0	(0, 1)	0	(0, 0)	0	(0, 3)	4
Amphipoda	0	(0, 0)	0	(0, 1)	0	(0, 0)	1
Total	2149		20 980		9360		32 489

Table 2: Family richness of the most abundant invertebrate orders

Taxa	Green roof			Companion			Nearby			Total
	Median	Min	Max	Median	Min	Max	Median	Min	Max	
Coleoptera richness	1	0	2	3	0	5	3	0	9	12
Hymenoptera richness	3.5	2	5	5	1	5	4	1	6	19
Hemiptera richness	3.5	2	4	3	1	5	2	1	4	6
Diptera richness	11	10	16	15	7	21	18	12	24	35

invertebrates and she notes this was probably due to a high number of snails introduced on the pre-grown sedum mats.

We found no difference in the order-level composition of green roof invertebrate communities when compared to adjacent ground-level vegetation and similar ground-level vegetation nearby. Of the four orders identified to family level, only the composition of fly families differed significantly between green roofs and both nearby and companion ground-level habitats. This may be driven by a greater abundance of individuals in certain Diptera families at ground level, but three of four green roofs also had a greater proportion of individuals in the family Chironomidae than the ground-level sites. Most species in this family have aquatic larvae and short-lived adults that eat nectar and pollen. Their relatively high abundances on roofs may be due to colonisation of, and probable breeding in, standing water stored in the green roof drainage layers by individuals originating from nearby ground-level water bodies, which included ornamental ponds at two sites and ephemeral wetlands at a neighbouring brownfield development site. This highlights the influence of the surrounding landscape on the composition of green roof invertebrate communities.

Low numbers of native bees and hoverflies were found to use both succulent and grassland green roofs. However, no significant difference in pollinator richness, abundance or composition between green roof and ground-level sites and succulent and grassland sites was detected. This result differs from other studies, such as [Ksiazek, Fant, and Skogen \(2012\)](#) who found that green roofs supported a lower abundance of bees than nearby urban green space but may reflect the low abundance

and richness of native bees found in central Melbourne ([Mata et al. 2015](#)) where most of the green roofs sampled were located.

Few studies have compared the biodiversity of green roofs with different types of vegetation with the exceptions being [Madre et al. \(2013\)](#), who investigated arthropods and [Joimel et al. \(2018\)](#), who sampled Collembola. Apart from marginally greater beetle family richness, the invertebrate communities on the native grassland green roofs, that theoretically should favour Melbourne's endemic invertebrate biodiversity, were not substantially different from the succulent green roofs dominated by introduced sedum species and other exotic succulents. We expected to see a greater influence of green roof vegetation type across all invertebrate groups sampled because increased vegetation complexity and high proportion of native plant species is associated with higher biodiversity at ground level ([Tews et al. 2004](#); [Threlfall et al. 2017](#)) and on green roofs ([Madre et al. 2013](#)). The relatively young age of all the green roofs sampled could mean that differences are yet to develop and long-term monitoring may produce different results. Alternatively, because there are very few areas of native grassland remaining in Melbourne, and most of these are on the city's outskirts distant from the green roofs ([Williams, McDonnell, and Seager 2005](#)), specialist native grassland invertebrates may be unavailable to colonise the green roofs or the small patches of grassland we sampled. In addition, the ground-storey vegetation in most urban parks is dominated by exotic grasses and herbs meaning that native invertebrates, particularly specialist species that utilise native ground-storey plants, may be absent from large parts of the urban landscape. Confirming this is difficult due to the

Table 3: Strongest competing univariate linear mixed models of invertebrate communities on green roof, companion sites and nearby sites^a

Habitat characteristic	Invertebrate metric	AICc	AICc weight
% Green space (+)	Ordinal richness	13.1255	0.5357
	Ordinal abundance	69.878	0.2761
	Pollinator richness	13.5341	0.835
	Pollinator abundance	46.1642	0.913
	Beetle family richness	86.2208	0.4454
	Hymenoptera family richness	68.6346	0.8832
	Bug family richness	63.6905	0.7732
	Fly family richness	106.6914	0.4397
Height (-)	Ordinal abundance	68.743	0.4869
	Fly family richness	108.1221	0.215
Size (+)	Beetle family richness	87.6881	0.2139
Age (+)	Fly family richness	108.1915	0.2077

^aOnly models within two AICc points of the top model are listed. The sign within parentheses indicate the direction of the relationship for each model parameter.

lack of research on the distribution and composition of Melbourne's invertebrate communities and the effects of urbanisation on them, which are hampered by a lack of taxonomic expertise (Yen 2011).

We found that roof height negatively influenced ordinal abundance and fly family richness. This result is similar to other studies, such as MacIvor (2016), who found a decline in the number of solitary bees and wasps utilising artificial nests with increasing green roof height, and Madre et al. (2013), who found height negatively affected the richness and abundance of spiders and taxonomic composition of true bugs and beetles on French green roofs. As green roof height increases, invertebrates, particularly those with limited dispersal ability, may find them increasingly difficult to colonise, leading to less diverse invertebrate communities compared to ground level. Our results also suggest that the insects that are able to disperse to green roofs do not increase in abundance to numbers similar to ground-level sites. This could be because the sampled green roofs could not provide sufficient resources for a large abundance of individuals due to low primary productivity caused by the low nutrient inorganic substrates used. Similar processes have been shown to influence invertebrate communities in ground-level urban habitats (Shochat et al. 2004; Threlfall, Law, and Banks 2012).

The per cent cover of surrounding green space was the most frequent habitat variable affecting green roof invertebrate communities in our study. It was positively correlated with all measures of the assemblage—total abundance, ordinal richness, richness of all families, and richness and abundance of pollinator species (bees and hoverflies). This finding is consistent with other studies which have found the abundance and/or richness of bees (Tonietto et al. 2011; Braaker et al. 2014; MacIvor 2016; Ksiazek-Mikenas 2018), beetles (Braaker et al. 2014; Ksiazek-Mikenas 2018; Kyrö et al. 2018) and weevils (Braaker et al. 2014) increased with greater proportions of green space in the surrounding landscape.

The age and size of the green roofs did not consistently affect the green roof invertebrate communities. Site age was expected to increase habitat complexity leading to an enhancement of invertebrate taxa richness, similar to findings from ground-level urban green space (Smith et al. 2000; Sattler et al. 2010), and a recent study in Melbourne that demonstrated that older green spaces with more native vegetation cover provided habitat for a range of native bee species (Threlfall et al. 2015). However, green roofs are relatively new in Australian cities

(Williams, Rayner, and Raynor 2010) and the oldest green roof in this study was only 7 years old, and the others were between 1 and 3 years. Consequently, age effects may become more apparent in the future. The abundance and richness of invertebrates collected from green roofs was expected to increase with green roof area, but area was not found to consistently influence invertebrate communities, only affecting beetle family richness. Similar to the roof age, the range of roof areas was quite narrow due to the still limited number of extensive green roofs in Melbourne. Braaker et al. (2014) also found that green roof area was not an important determinant of their invertebrate communities but we acknowledge that different invertebrate taxa may respond to area differently.

This study sampled invertebrates on all available extensive green roofs in Melbourne in spring and autumn, using standard methods (pitfall and pan traps) over adequate time periods [7 days (once per season) and 24 h (three times each season), respectively]. We recognise that replication is low due to the relatively few extensive green roofs in Melbourne and that using additional sampling methods over longer time periods would probably have collected a greater abundance and richness of taxa. There is also potential sampling bias in our results as pan traps may not effectively sample larger bees (Roulston, Smith, and Brewster 2007) although it should be noted there are no *Bombus* species on mainland Australia (Buttermore 1997). Notwithstanding these limitations, our results are sufficient for an initial comparison with ground-level sites sampled in the same way. We found relatively few invertebrates utilising the green roofs and the resulting small dataset has restricted the explanatory power of some of our analyses. For example, we have not been able to run multivariate mixed models that could better elucidate the drivers of invertebrate responses. Similarly, the level of taxonomic resolution we have been able to achieve limits interpretation of our results. Rather than focussing on a specific taxa inhabiting green roofs and identifying samples to species level, because there have been no prior published studies of invertebrates on Australian green roofs, we chose to examine the broad range of invertebrate orders occurring on them. The most abundant orders were further identified to family level but resources did not allow species-level identification which is problematic for many Australian insects due to the large number of undescribed species (Yeates, Harvey, and Austin 2003). However, higher levels of taxonomic resolution can reflect patterns at lower taxonomic resolution in terrestrial invertebrate communities (Timms et al. 2013).

Despite our small sample size and coarse taxonomic resolution, our results suggest that although invertebrate abundance on green roofs in Melbourne is lower than nearby ground-level sites their community composition is similar, and they are significantly influenced by the amount of green space in the surrounding landscape. We did not find that green roofs specifically designed to support local biodiversity had greater invertebrate habitat value than succulent green roofs, although all of the green roofs sampled were comparatively young and differences may become more apparent as the roofs age.

Our results suggest that green roofs in Melbourne largely rely on the surrounding environment for their invertebrate biodiversity and that their effectiveness as invertebrate habitat is highly dependent on location and their horizontal and vertical connection to other habitats. This is consistent with studies in Zurich, Toronto, Chicago and Helsinki (Tonietto et al. 2011; Braaker et al. 2014; MacIvor 2016; Kyrö et al. 2018). When in close proximity to green space and natural areas, and at low height, green roofs have considerable habitat potential and could act as stepping stones, linking otherwise isolated habitat pockets as part of an urban habitat corridor for mobile invertebrates (Braaker et al. 2014; Mayrand and Clergeau 2018). Long term and species-specific studies will be necessary to further refine our understanding of the biodiversity value of green roofs, verify the long-term effect of vegetation type and the surrounding landscape and to establish if green roofs are able to support the colonisation of target species.

Supplementary data

Supplementary data are available at JUECOL online.

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Data are available from the City of Melbourne Open Data Portal <https://data.melbourne.vic.gov.au/> and on request from the authors.

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