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Bee-friendly community gardens: Impact of environmental variables on the richness and abundance of exotic and native bees

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Abstract With their abundant floral resources, urban community gardens have the potential to play an important role in pollinator conservation. At the same time, the gardens themselves are dependent upon the pollination services provided by insects. Thus, understanding the variables that can increase bee richness or abundance in community gardens can contribute to both urban agriculture and pollinator conservation. Here we examine the impact of several environmental variables on bee abundance and diversity in urban community gardens in Sydney, Australia. We used hand netting and trap nests to sample bees in 27 community gardens ranging from inner city gardens with limited surrounding green space, to suburban gardens located next to national parks. We did not find strong support for an impact of any of our variables on bee species richness, abundance or diversity. We found high abundance of a recently introduced non-native bee: the African carder bee, *Afranthidium repetitum* (Schulz 1906). The abundance of African carder bees was negatively correlated with the amount of surrounding green space and positively correlated with native bee abundance/ species richness. Our results highlight the seemingly rapid increase in African carder bee populations in inner city Sydney, and we call for more research into this bee's potential environmental impacts. Our results also suggest that hard-to-change environmental factors such as garden size and distance to remnant forests may not have a strong influence on native bee diversity and abundance in highly urbanized area.

Keywords Community gardens · Hymenoptera · Urban conservation · Pollinators · Exotic species

Introduction

Small scale, community-based agricultural systems such as community gardens and urban food forests are becoming increasingly common in major cities around the world (Guitart et al. 2012). Although they provide a variety of mental and physical health benefits, the main reason people join community gardens is for food production (Guitart et al. 2012). In order to be productive, community gardens, like other horticultural systems, require adequate pollination services from a range of insects, the most important of which are likely to be bees and flies (Hennig and Ghazoul 2012, Orford et al. 2015, Larson et al. 2001). Insect pollination increases productivity in 68 % of the world's leading 57 crops (Klein et al. 2007) and is essential for fruit set in crops including watermelon, vanilla, kiwi, passionfruit, squashes and pumpkins. Even self-fertile plants, which are capable of fruiting in the absence of pollinators, can benefit from insect pollination; these benefits include better taste, increased fruit set, longer shelf life, and more symmetrical shape (Klatt et al. 2014). While most studies on pollination services have focused on rural horticultural systems there is no reason to believe that pollinators will be less important in urban systems. Indeed, Lowenstein et al. (2015) found that fruit and seed set in cucumbers and eggplants in small urban gardens in Chicago (USA) was positively influenced by the diversity and abundance of wild bees. Most community gardens are characterized by low or no pesticide

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use and high floral diversity and may therefore provide important habitat for urban bee populations. Understanding the factors that increase or decrease pollinator abundance and diversity in community gardens could improve garden yields, while simultaneously aiding in the conservation of wild bee species.

Studies on the factors which influence urban pollinator diversity have had varied results. In a study of urban community gardens in New York City, Matteson and Langellotto 2010 found that the best predictors of bee and butterfly richness were floral area and sunlight availability. A study of domestic gardens in the UK found that the richness of native plant species, the number of surrounding houses and the extent of low canopy vegetation had a strong influence on the species richness of solitary bees, while habitat diversity and intensity of management drove diversity patterns of bumble bees (Smith et al. 2006). Threlfall et al. (2015) found that the density of non-native honey bees (*A. meliferra*) was positively influenced by increases in flowering native plants, while members of the native, ground dwelling genus *Homalictus* were most common in areas with low flower diversity and less surrounding impervious surfaces (Threlfall et al. 2015). While these studies highlight the potential importance of factors such as floral diversity on bee diversity and richness, few have focused exclusively on small, food-producing systems such as community gardens.

Here we address this knowledge gap by investigating the factors that drive the abundance and diversity of bees in community gardens in Sydney, Australia. Our two overarching goals were 1) to determine which bee species were common in Sydney community gardens and 2) to determine which factors, if any, influenced bee diversity and abundance. While community gardens differ in factors like size, degree of surrounding green space and distance from unaltered habitat, they are all similar in terms of plant composition, being dominated primarily by non-native crop plants such as tomatoes, capsicums, chili peppers, eggplants, squashes and brassicas (T Latty, pers comm). This similarity in vegetation allows for an examination of how environmental factors such as garden size drive bee abundance and diversity independent of local vegetation characteristics.

Methods

Bees in the Sydney region

Australia is home to at least 1600 species of bee, of which ~300 are found within the Sydney basin (Dollin et al. 2000). Two exotic species, the western honey bee (*Apis mellifera*) and the African carder bee (*Afranthidium repetitum*) are also present. The Australian bee fauna is dominated by bees in the family Colletidae which make up 50 % of the bee fauna (Dollin et al. 2000). The remainder are distributed amongst

the Halictidae (20 %), Megachillidae (10 %) and Apidae (10 %)(Dollin et al. 2000). The vast majority of bees in Sydney are solitary, with only two eusocial species, *Apis mellifera* and *Tetragonula carbonaria*.

Study area

We focused on the heavily urbanized Sydney metropolitan area (population ~ 4 million) located in coastal south eastern Australia. The metropolitan area is bordered by two large national parks and contains several smaller national parks and reserves. The original vegetation of our study area is a mix of sandstone woodland and heath and blue glum high forest (Benson and Howell 1994), although this has been highly fragmented since the arrival of Europeans in 1778.

We sampled 26 community gardens within the Sydney area (Fig. 1, Appendix Table 4). We focused on gardens in the relatively urbanized councils of Leichardt, Balmain, City of Sydney, Marrickville, Chatswood, Ryde, Ku-ring-gai, Willoughby, Ashfield and Waverley. Inclusion in our study was based primarily upon our obtaining the permission of garden steering committees and of local councils. For comparison, we also selected sampling sites in three National parks within our study area: Sydney Harbour National Park, Lane Cove National Park and Beowra Valley National Park (Fig. 1).

Sampling techniques

We sampled bees between November 2013 and March 2014, which is within the main flight period for the majority of Australian native bees (Dollin et al. 2000). Each site was sampled twice during the study period. The first sampling period was 10 October to 18 December 2013. The second sampling period was from 28 January to 25 February 2014. The community garden steering committees were strongly opposed to lethal sampling of bees. We therefore employed two non-lethal sampling techniques: visual surveying and trap nests. Visual surveys were further divided into patch based sampling and time-standardized sampling methodologies.

Visual surveys

Sampling was always conducted on bright, sunny days between 11:00–15:00, where the temperature was over 22 C. We first identified and numbered all flowering patches within each garden, where flowering patches were at least 85 cm in diameter as determined using a hula hoop to delimit the borders of our sampling area. We then randomly selected 4 flowering patches from the set of flowering patches. Each selected patch was observed for 15 min, during which time we used an insect net to catch every bee that landed on the flowering patch. Bees were immediately placed into

Fig. 1 The location of the 27 community gardens (*red circles*) and 3 national park (*green diamonds*) assessed in this study. All study sites lie withing the greater Sydney region
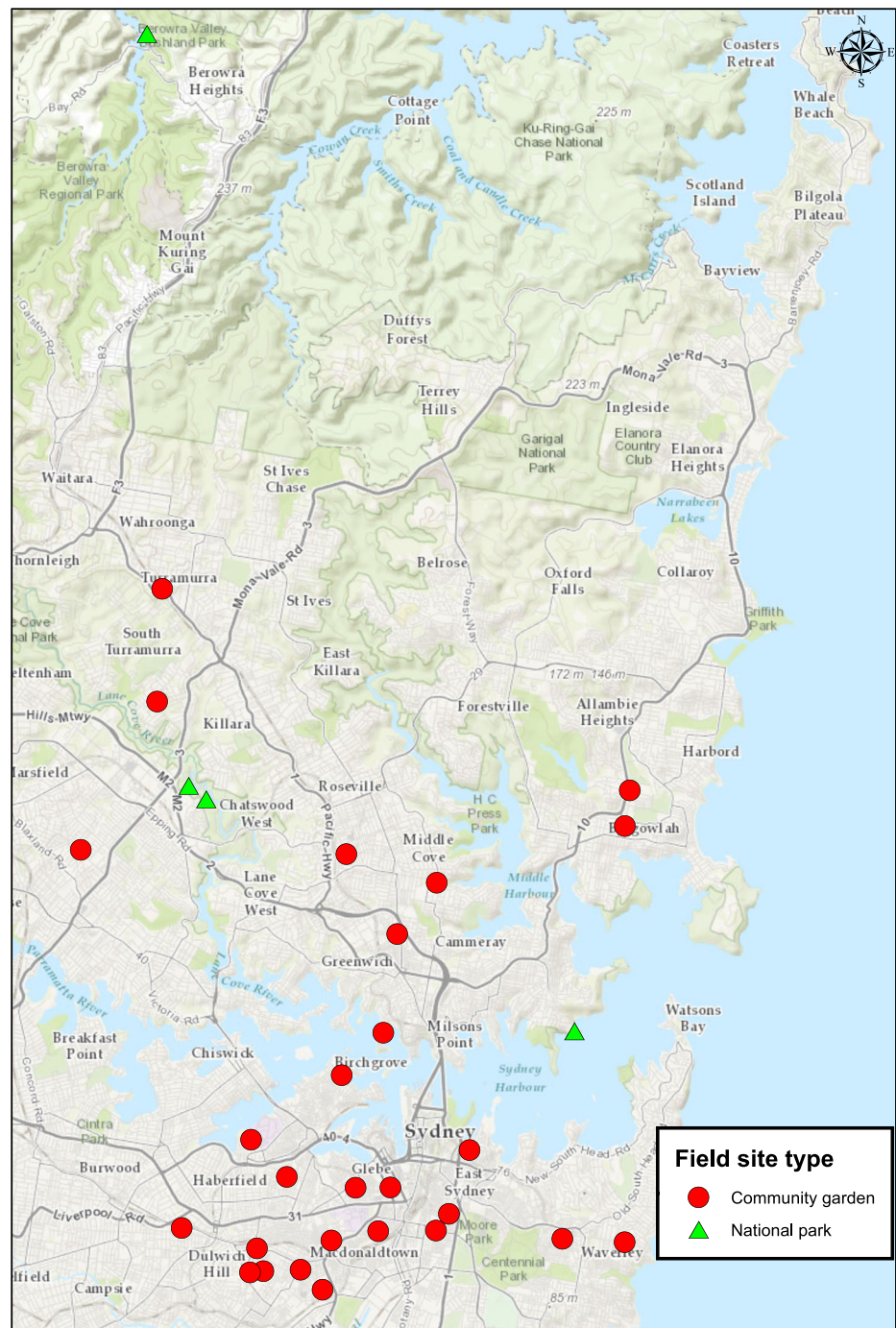
individual tubes. The tubes were placed into an insulated cooler with ice packs. Once the observation period ended we photographed every bee using a Canon DSLR 350D with a 100 mm Tamron macro lens. We took care to photograph each bee from multiple angles including at least a dorsal, ventral, side and face shot. Whenever possible, we also tried to get photographs of wing venation. All bees were released after being photographed. We identified each bee to the lowest possible taxonomic level with the help of a bee expert from the Australian Museum (Michael Batley).

For each garden, we estimated the density of flower patches by dividing the total number off flower patches (as defined above) in the garden by garden size. We visually estimated the proportion of the whole site that received full sun at the time of sampling.

Given the large size of national parks, we modified the above methods slightly. We counted the number of flower patches within a 100 m radius of a focal location within each National park. Focal locations were selected based on accessibility and distance from built-up regions, and the same

locations were used throughout the sampling period. As with gardens, we visually estimated the proportion of the whole focal location that received full sun at the time of sampling.

Herein, we use the term 'site' to refer to both gardens and national parks.

Time-standardized sampling

During pilot trials, we noticed that our patch-based sampling method was not effective at catching bees that were not actively foraging on our patches. Since the four sampled patches were selected randomly, we sometimes missed highly attractive flower patches. Thus, we decided to use a complementary time-standardised sampling technique in addition to our patch based survey technique. Time-standardised surveys were 20 min long, during which time the observer walked methodically around the site capturing every bee that they encountered. As above, bees were photographed in the field and released. Data obtained from time-standardised surveys were only used to assess species presence, and were not used to calculate abundance.

Trap nests

Trap nests are artificial nesting substrates used to sample insect populations. Several studies have successfully used trap nests to sample hymenopteran populations (Alves-dos-Santos 2003; Gathmann et al. 1994; Steffan-Dewenter 2002; Tscharntke et al. 1998). We used two types of trap nests: drilled wood blocks and reed bundles. Drilled wood blocks consisted of 190 × 190 × 59 mm blocks drilled with 3,6, and 8 mm holes (4 of each size) and covered with a metal roof. Reed bundles consisted of between 20 and 30, 30 cm long lantana canes. Lantana canes are one of the preferred nesting materials of reed bees, *Exoneura* spp. (Dollin et al. 2000). The bundles were held together with gardening wire.

Two trap nests of each type were placed within each site. Nests were generally hung from branches 2 m above the ground, although in some cases fences or other man made structures were used when branches were unavailable. We placed nests in the sunniest location possible, with the entrances oriented to the north. We checked trap nests fortnightly, and removed them if there were at least 3 sealed holes; all used nesting material was replaced with fresh material. Colonized nesting materials were returned to the social insect rearing facility (University of Sydney) where they were stored at ambient temperature. Each nest entrance was covered by a plastic falcon tube affixed with blue tac. Tubes were checked daily and any emerged bees were collected, photographed and released (except for those retained for our reference collections). This setup allowed us to monitor bee emergence, and to link individual bees back to their nest (and site) of origin. Monitoring of the nests continued for 9 months after the visual

sampling ended at which point bee emergence had largely ceased. We judged 9 months to be a reasonable observation period as the vast majority of insects had emerged, with only a few sealed nests remaining.

Statistical analysis

Quantifying diversity and abundance

We calculated species richness (the total number of species) in each site by combining species presence data from our trap nests and visual surveys (patch based + time-standardised). We calculated abundance by tallying the total number of individual bees observed during our patch-based surveys. We did not use trap nest data to calculate abundance because female bees regularly fill multiple holes, making the 'number of filled holes' an inaccurate representation of the number of bees actually present (personal observation, T Latty and J Makinson). We quantified bee diversity using Shannon's H index.

Variable selection and model fitting

We were interested in determining which variables influence the abundance and diversity of native bees within community gardens in urban landscapes. To achieve this goal, we first selected a set of potentially important variables that could drive differences in bee diversity and abundance. Variable selection was based on a review of the literature combined with our own knowledge of bee ecology. The set of factors included site size, the density of flowering patches (flower patches/m^2), % sun exposure, and distance from closest forest remnant. Using Arc Map (ESRI, Redlands, California, USA, version 10.1), we also measured the percent cover of mapped native vegetation, percentage of surrounding green space and dwelling density, all within a 500 m buffer of each site. We chose 500 m because this is the likely predicted flight range for most of our bee species. We used the Index of Relative Advantage and Disadvantage (IRSAD) to quantify the socio-economic status of each site. IRSAD scores and dwelling density (housing density per ha) for each of the districts within which our sites resided was obtained from the Australian Bureau of Statistics (ABS) 2011 Census.

We constructed separate sets of candidate models to study the influence of site characteristics on species richness, bee abundance, and bee diversity. Our candidate models contained the above eight variables (garden size, flower density, distance to forests remnants, socioeconomic index, % green space within 500 m, % sun cover, housing density and % cover of native vegetation), but were restricted to combinations of up to three variables per model to reduce over-fitting. To avoid collinearity in the models, we only included variables that were not strongly collinear (Spearman's Rho <0.7), and inspected variance inflation factors for all highly ranked

models (which were all <2). This led to the construction of 54 candidate models for each of the three analyses.

We fitted generalized linear models using a normal distribution for bee abundance and diversity. We ensured that the assumptions of homoscedacity and normality were met by examining residuals after model fitting. We used a square root transformation on the dependent variable 'bee abundance' to normalize the data. We fitted models using a poisson distribution for species richness, where dispersion was examined after model fitting. We calculated R^2 for models where appropriate, or used a pseudo-R^2 (in the case of poisson and binomial models) to evaluate percent variation explained. Pseudo R^2 was calculated as 1-(residual deviance/null deviance).

Model selection

We chose to compare all model subsets because our study was fundamentally explorative in nature. We used AICc (AIC corrected for small sample sizes; (Burnham and Anderson 2004)) to rank the candidate models such that models with lower AICc values were ranked more highly than those with high AICc values. We also calculated Akaike weights for each model. The Akaike weight ranges from 0 to 1 and indicates the probability that a given model is the best approximating model (Symonds and Moussalli 2011). We used ΔAICc to differentiate between models; if ΔAICc was <2, we considered the models to be equally well supported (Symonds and Moussalli 2011). For each bee response variable, a 95 % confidence set of models was constructed listing all models that have a summed Akaike weights (w_i) >0.95 (Burnham and Anderson 2002). The "relative importance" of each variable group was then calculated by summing the weight for all of the models incorporating that predictor (Burnham and Anderson 2002). We selected our final model based on the lowest AICc and the highest model w_i (Burnham and Anderson 2002).

African carder bee analysis

During our study we were surprised to discover that a recently introduced non-native bee species, the African carder bee, was present in high numbers throughout many of our sites. We conducted additional analyses on the African carder bee data to determine how the density of flowering patches (flower patches/m^2) and % green space within 500 m influenced the number of African carder bees. Since our African carder bee dataset contained many zeroes, we used a zero inflated Poisson regression with a log link function. We also examined the relationship between African carder bee abundance and native bee abundance, richness or diversity using the zero inflated Poisson regression described above. Analyses were conducted in JMP Pro 11 (SAS) and 'R', version 3.0.1 (http://www.r-project.org/).

General results

Our attempt to sample native bees in national parks was unsuccessful. None of our trap nests were colonized, and we did not observe any bees during our visual surveys. Our sites were dominated by tall Eucalyptus sp. and Corymbia sp. trees which we were unable to sample adequately due to their height. The lack of bees during visual surveys may have been due to the dearth of flowering plants; very few flowers (at ground level or in the canopy) were observed during our sampling period. Thus, we have omitted the national parks from subsequent analyses.

Trap nests

Overall, 492 hymenopterans emerged from our wood block trap nests, of which 165 were bees and the remainder were wasps. No bees colonized our reed bundle nests. Wood block trap nests were colonized by at least 1 bee in 44 % of our sites. Nine native bee species emerged from the wood block trap nests, the most common of which were *Hylaeus honestus* (106 individuals) followed by *Megachile erythropyga* (17 individuals). *Hylaeus honestus, H. concinna* and *M. erythropyga* were present in the greatest number of sites (Table 1).

Visual surveys

Overall, we captured and identified 454 individual bees, representing 12 species (Table 2). We observed bees at every site. The three most abundant bee species were *Apis mellifera*, *Amegilla sp.*, and *Afranthidium repetitum*. The most commonly observed bees (in terms of presence in sites) were Amegilla sp. (22 sites), followed by *M. serricauda* (20 sites) and *A. meliferra* (18 sites).

Influence of site characteristics on bee abundance and diversity

Species richness

We calculated the total number of bee species present in each garden by combining species presence data from visual surveys and trap nests. The mean number of species per garden was 2.75 with a maximum of 7 and a minimum of 1.

Our best model, as identified by AICc, was one containing '% green space within a 500 m radius' and the index of socio-economic status (Table 3), both of which had a negative relationship with increasing bee species richness. This model had a pseudo-R^2 of 0.15. Flower density was also ranked as a good predictor of species richness, as it featured in several of the highly ranked models (Table 3), where it had a negative relationship with bee species richness. The large number of

Table 1 Results from trap nests

Species	Family	Number of individuals emerging from trap nests	# of sites (of 27)	Pollen specificity
Afranthidium repetitum	Megachillidae	5	1	P
Hylaeus alcyoneus	Colletidae	8	1	O
Hylaeus euxanthus	Colletidae	1	1	P
Hylaeus honestus	Colletidae	106	6	P
Hylaeus nubilosus	Colletidae	6	1	P
Hyleoides concinna	Colletidae	8	3	P
Megachile aurifrons	Megachillidae	11	2	P
Megachile erythropyga	Megachillidae	17	3	Few records
Megachile lucidiventris	Megachillidae	3	1	P

For pollen specificity, *O* oligolectic (collects pollen from a single family or genus), *P* polylectic (collects pollen from many plants)

models included in the 95 % confidence set for species richness (Table 3) indicated that there was a very high degree of uncertainty in predicting bee species richness. Full details of candidate models can be found in Appendix Table 5.

Bee abundance

We calculated bee abundance by counting the total number of individual bees observed in each garden (visual surveys only). The index of socio-economic status (negative relationship), % sun exposure (positive relationship) and to a lesser extent, flower density (negative relationship), were the best predictors of bee abundance (Table 3). The model with the lowest AICc only contained the socio-economic variable. The IRSAD had a negative but non-significant relationship with bee abundance ($p = 0.55$) and the R^2 of this model was low (0.026). Again, the large number of models included in the 95 %

confidence set for abundance (Appendix Table 6) indicated that there was a very high degree of uncertainty in predicting bee abundance. Full details of candidate models can be found in Appendix Table 6.

Diversity

Garden size (positive relationship), % green space within a 500 m radius (negative relationship) and flower density (negative relationship) were the best predictors of bee diversity (Table 3). The model with the lowest AICc contained only garden size, which had a non-significant but positive relationship with bee abundance ($p = 0.07$). This model also had a low R^2 (0.09). The large number of models in the 95 % confidence set, and for which had an AICc < 2 indicates bee diversity is also hard to predict. Full details of candidate models can be found in Appendix Table 7.

Table 2 Species observed during visual surveys

Species	Family	Number of individuals captured	# of sites (of 27)	Pollen specificity	Nesting substrate
Afranthidium repetitum	Megachillidae	62	16	P	W
Amegilla bombiformis	Apidae	2	2	P	G
Amegilla sp	Apidae	100	22	P	G
Apis mellifera	Apidae	200	18	P	C
Exoneura sp	Apidae	3	3	P	T
Homalictus sp	Halictidae	30	12	P	G
Lasioglossum lanarium	Halictidae	1	1	P	G
Lipotriches flavoviridis	Halictidae	5	4	P	G
Lipotriches phanerura	Halictidae	24	14	P	G
Megachile serricauda	Megachillidae	27	20	P	C
Megachile sp.	Megachillidae	1	1	P	
Tetragonula carbonaria	Apidae	1	1	P	H
Thyreus caeruleopunctatus	Apidae	1	1	NA	K

For pollen specificity, *O* oligolectic (collects pollen from a single family or genus), *P* polylectic (collects pollen from many plants)

For nesting substrate, *g* ground nesting, *C* cavity, *T* twig/reed nesting, *W* wood nesting, *K* kleptoparasite

Table 3 Relative importance indices, following Burnham and Anderson (2002), calculated from AICc values of the 95 % confidence set of the GLMs for bee responses

Response	Species Richness	Abundance	Diversity
% Sun	0.1107	**0.2663**	0.1142
Garden size	0.1689	0.1548	**0.4123**
% green space in 500 m	**0.4684**	0.1338	**0.342**
Flower density	**0.2578**	**0.222**	**0.3236**
Distance to forest	0.1077	0.1488	0.2638
% cover native vegetation in 500 m	0.1154	0.0855	0.0889
Dwelling density in 500 m	0.0743	0.0958	0.1055
IRSAD	**0.4977**	**0.3187**	0.1167

The highest values for each bee response is highlighted in bold

IRSAD Index of socio-economic advantage and disadvantage, as described in the Methods

The African carder bee

African carder bees were present in 14 of our 27 sites (52 %) and their abundance in community gardens ranged between 0 and 16 individual bees. The % of green space had a negative effect on African carder bee abundance (Wald $\chi 2 = 9.83$, estimate = -3.27 ± 1.04, $P = 0.0017$). The density of flower patches within the site did not have a significant effect on African carder bee abundance (Wald $\chi 2 = 0.05$ estimate = -0.27 ± 1.20, $P = 0.819$). African carder bees were positively correlated with the abundance and species richness (abundance: Wald $\chi 2 = 107.36$, estimate =1.7 ± 1.66, $P < 0.00$; species richness: Wald $\chi 2 = 4.88$, estimate =0.9 ± 0.41, $P = 0.027$). African carder bee abundance did not have a significant effect on species diversity (Wald $\chi 2 = 0.90$, estimate =0.88 ± 0.92, $P = 0.34$).

Discussion

The aim of our study was to explore variables that might have an impact on the diversity and abundance of native bees in urban community gardens. None of the models we investigated (including the models identified as being the best) were a particularly good fit to our data (R^2 values below 0.20). Further, for each biodiversity measure (species richness, bee abundance and diversity index) we experienced a high degree of model uncertainty, with several univariate models having ∆AICc's of <2 indicating that they were as 'good' as our best fit model (Symonds and Moussalli 2011). Finally, in none of our analyses did any of our leading models have a high Akaike weight, suggesting that none of them were strongly supported. Our results suggest that rather than a system in which a few variables have strong effects, we are dealing with a system where many variables have small, complex effects ('many tapering effects', sensu (Burnham and Anderson 2004). Alternatively, we may not have measured some key, driving variable. For example, Threlfall et al. (2015) found that the amount of native vegetation on a site had a positive influence on honey bees, *Lasioglossum* spp. and bees within the family Colletidae. We did not measure the amount of native vegetation within each garden; since our sites were community gardens, they rarely, if ever, contained any native flowers. Native plants such as those within the family Myrtaceae are commonly used as landscaping elements within Sydney and their presence in the landscape surrounding the gardens could have had an impact on bee diversity and abundance. We did however measure the amount of native vegetation within 500 m of each garden, but found this variable had no strong effect on any of our bee response variables. It should be noted that our study was relatively short term, and a longer sampling period with greater sampling effort might have allowed us to detect more subtle impacts on bee diversity and abundance.

We found that the density of flowers in each garden had a consistent (albeit weak) negative impact on several measures of the bee community. This may be due to the large number of exotic plant species located in community gardens, many of which may not provide good quality forage for bees. Tomatoes and eggplants, for example were common plantings in community gardens, but neither species provides nectar. We suspect that much of the variation in bee diversity and abundance may be linked to the presence/absence of specific flowering plant species, rather than density of all flowers. Although all community gardens planted similar crops (tomatoes, eggplants, beans, cucumber, for example), several had also planted highly bee-attractive flowers such as the African blue basil (*Ocimum kilimandscharicum* × *basilicum* 'Dark Opal') and borage (*Borago officinalis*) (Latty and Makinson, pers comm). Other gardens had highly bee-attractive flowering herbs such as coriander *(Coriandrum sativum)*, mint (*Mentha* spp) and Lavender (*Lavandula* sp.) In their study of bumblebee diversity in urban gardens Gunnarsson and Federsel (2014) similarly suspect that bee diversity can largely be attributable to the presence/absence of specific attractive flower species. Research is underway to determine the overall impact of bee-attractive flowers on bee abundance and diversity.

In addition to flower density, the percent cover of surrounding green spaces, socio-economic status of the surrounding area and the percent sun exposure all had some influence on the bee community. These results are consistent with many other

studies, where it has been found that heavily maintained lawns such as those found in urban green spaces do not provide adequate foraging or nesting resources for bees (Threlfall et al. 2015; Tonietto et al. 2011). It may also be expected that areas of lower socio-economic status contain more unkempt or abandoned lots, providing resources for some bee species; indeed, vacant lands are often associated with increased insect biodiversity (reviewed in Gardiner et al. 2013). Socio-economics have been shown to be a strong driver of urban vegetation cover and diversity in other cities (Hope et al. 2003), and hence it may be possible that changes to vegetation in areas surrounding each of our gardens could also be influenced by socio-economics. In agreement with Matteson and Langellotto (2010) we also found a greater abundance of bees in gardens with greater sun exposure, although all three of these variables were found to be relatively weak drivers of the bee community.

We intended to compare bee abundance and diversity in urban areas to that found in relatively unaltered natural habitats. However both our sampling techniques failed to catch any bees within the three national parks we sampled. The national parks were dominated by very tall trees, with very little flowering understory. Although we did not explicitly quantify tree flowering at our field sites (many of the trees were too tall for accurate sampling), we did not observe any trees to be in flower at the time of the study. Thus, there may have been a dearth in floral resources, leading to a lack of bees. Even if trees were in flower, the bees would have been foraging high in the canopy, well beyond the reach of our nets. It is also possible that our results accurately reflect a real lack of bees in forested landscapes. Several studies have found that bees are more common in open lands with low canopy coverage rather than in heavily forested areas with dense canopy coverage (For example, see: Grundel et al. 2010, Hoehn et al. 2010).

Similar explanations (lack of floral resources, bees avoiding forest) could also explain the lack of uptake for our trap nests. It is also possible that our nesting materials were unattractive when compared to abundant natural nesting sites. Our bee nests were located relatively low in the canopy; if bees are flying at a higher level in order to take advantage of flowering trees, then they may not have encountered our nests boxes. In a Brazilian study, twice as many bees inhabited trap nests mounted at 8 or 15 m than those mounted at 1.5 m (Morato 2001),

suggesting that nest height is a key factor influencing nest uptake in forested environments. Alternatively, the group of bees we found in the trap nests may represent 'urban adapted' species which have proliferated in urban environments, but which occur at much lower numbers (or are virtually absent) in unaltered habitats. In that case, our trap nests may have been a true indication of bee diversity. Future studies will need to compare the efficacy of different sampling regimes (trap nests, hand sampling, pan traps and intercept traps) in order to find suitable techniques for sampling bees in forested Australian ecosystems.

By design, wooden block trap nests only attract species that will nest in drilled wooden blocks. Since 60 % of Australian bee taxa are ground nesting (Dollin et al. 2000), we expected that that the composition of bee fauna in our trap nests would differ from those found in visual surveys. Indeed, of the bee species found in trap nests, only one, the invasive African carder bee, was also observed foraging in the community gardens. This suggests that the bee species found in trap nests either do not forage on crop plants (for example, they might prefer native plants) or they forage at a time of day outside our sampling period. Although we cannot rule out the possibility that trip nesting bees were nocturnal or crepuscular, we believe this is unlikely as night-active bee species are rare (Somanathan et al. 2008). Our results highlight the importance of using both trap nests and visual surveys in order to get a broader representation of the bee fauna.

Five of the nine species we collected in trap nests were members of the family Colletidae. This included our most abundant trap nesting species, *Hylaeus honestus* which was present in 6 sites. The Colletidae are short tongued bees and as such are thought to be restricted to feeding on shallow flowers; it is therefore not surprising that they were rarely seen in the community gardens which are dominated by deeper, more complex flowers. Instead, we suggest that the colletids were feeding on the abundant native trees planted as landscape elements around the gardens. Our results reinforce the utility of using trap nests to sample otherwise underrepresented colletid bees.

Perhaps the most surprising result in our study was the high abundance of African carder bees. The African carder bee was first observed in Australia in southeastern Queensland in 2000 (Baumann et al. 2016). It has since undergone a massive and rapid range expansion and has been discovered as far south as Melbourne (Baumann et al. 2016). By the time of our study in

2013/2014, the African carder bee had become one of the most commonly observed bees in the Sydney area. We found that African carder bees were more abundant in sites that had less surrounding green space, potentially suggesting that they thrive in disturbed habitats. Our study also suggested a positive relationship between African carder bee abundance and native bee abundance/species richness. We suggest that our finding reflects the fact that sites that were suitable for native bees were likely also suitable for African carder bees rather than any sort of facilitation effect. It should be noted that our study was not explicitly setup to examine competition between African carder bees and native bees. The fact that at least one African carder bee moved into our trap nests suggests that there might be some overlap in nesting requirements with native bees and wasps. Worryingly, to date there are no peer reviewed papers on this introduced bee's behaviour or ecology, either in Australia or in its native South Africa (although see Baumann et al. 2016 for a discussion of the African carder bees range in Australia). Exotic bees can compete with native bees for food and nesting sites, can introduce novel pathogens, and can increase weed abundance (Stout and Morales 2009). The complete dearth of research on the African carder bee makes it difficult to speculate on the potential impacts it may have on Australian ecosystems.

Our study suggests that difficult-to-alter environmental features such as garden size, socio-economics and distance from surrounding green space have, at most, a weak influence on the abundance and diversity of bees in urban community gardens. This is encouraging news, as it implies that inner city community gardens will have access to pollination services and can provide food resources to bees, even if they are small and located in the heart of the city. Since we were unable to sample national parks, we cannot make a strong statement about the conservation value of community gardens relative to unaltered landscapes. However, the fact that we found a large number of bees foraging within community gardens suggests that they can provide important food resources for urban bee populations. We suggest that future research should be targeted at developing techniques for increasing bee diversity and abundance within community gardens. Based on our findings, we suggest that planting flowering species known to be attractive to bees will be a good first step. Well designed, bee-friendly community gardens have the potential to increase urban food yield while simultaneously providing habitat refuges for bee populations.

Appendix

Table 4 Community gardens used in our study

Community Garden Name	Longitude (East)	Latitude (South)	Garden size m^2
241 Bondi Rd	151.2669	33.89482	271
Addison Rd	151.1603	33.90264	1022
All Saints	151.1582	33.89543	87.3
Arthur St	151.2142	33.88715	19.8
Balgowlah	151.2671	33.79196	675
Bongalong St	151.198	33.81835	121
Camdenville Paddock	151.175	33.90671	1033
Charlie's Garden	151.1945	33.89192	112
Chatswood	151.1845	33.79894	56
Coal Loader	151.1942	33.84341	332
Cottage in the Graveyard	151.1803	33.894	186
Eora Garden	151.1358	33.89127	140
Francis St	151.171	33.90164	70.8
Glebe	151.1874	33.88126	476
Glover's Garden	151.1563	33.86708	568
James St	151.2112	33.89168	171
Manly Vale	151.2685	33.78325	761
Mort Bay	151.1834	33.85281	1720
Ryde	151.1059	33.7979	252
Turramurra	151.1301	33.73356	1448
Ultimo	151.1975	33.88103	416
Waverley	151.2459	33.89389	471
West Pymble	151.1279	33.76059	6.1
White St	151.1674	33.87868	756
Wilkins School	151.1538	33.90128	602
Willoughby	151.2106	33.80607	57
Woolloomooloo	151.2212	33.87157	387

Table 5 Full set of candidate
models examining the influence
of variables on bee species
richness

Model	AICc	Delta_AICc	AICcWt	Cum. Wt	Log Likelihood
PcGS + IRSAD	112.7321	0	0.136	0.136	-52.8206
IRSAD	113.5778	0.8457	0.0891	0.2252	-54.5281
PcGS	114.0321	1.3	0.071	0.2962	-54.7661
Flowers.m2 + IRSAD	114.6595	1.9274	0.0519	0.3481	-53.7843
PcGS + Flowers.m2 + IRSAD	115.0809	2.3488	0.042	0.3901	-52.5881
Flowers.m2	115.332	2.5999	0.0371	0.4272	-55.416
PcGS + Flowers.m2	115.4969	2.7648	0.0341	0.4614	-54.2267
LogSize + PcGS + IRSAD	115.5444	2.8123	0.0333	0.4947	-52.8198
LogPc.veg + IRSAD	115.7094	2.9773	0.0307	0.5254	-54.3092
Dist.forest + IRSAD	115.955	3.2229	0.0272	0.5526	-54.432
LogSize + IRSAD	116.0312	3.299	0.0261	0.5787	-54.4701
houses.ha + IRSAD	116.0803	3.3481	0.0255	0.6042	-54.4947
Pc.sun + IRSAD	116.1093	3.3771	0.0251	0.6294	-54.5092
PcGS + houses.ha	116.2998	3.5677	0.0229	0.6522	-54.6282
PcGS + Dist.forest	116.3569	3.6248	0.0222	0.6744	-54.6567
PcGS + LogPc.veg	116.4813	3.7491	0.0209	0.6953	-54.7189
Pc.sun + PcGS	116.5349	3.8028	0.0203	0.7156	-54.7457
LogSize + PcGS	116.5591	3.8269	0.0201	0.7357	-54.7578
LogPc.veg	116.9124	4.1803	0.0168	0.7525	-56.2062
Dist.forest	117.3326	4.6004	0.0136	0.7662	-56.4163
LogSize	117.3329	4.6008	0.0136	0.7798	-56.4165
Flowers.m2 + Dist.forest + IRSAD	117.4331	4.7009	0.013	0.7928	-53.7642
LogSize + Flowers.m2	117.609	4.8769	0.0119	0.8046	-55.2828
houses.ha	117.6918	4.9596	0.0114	0.816	-56.5959
Pc.sun	117.6942	4.9621	0.0114	0.8274	-56.5971
Flowers.m2 + Dist.forest	117.7863	5.0542	0.0109	0.8383	-55.3714
Flowers.m2 + LogPc.veg	117.7877	5.0556	0.0109	0.8491	-55.3721
Flowers.m2 + houses.ha	117.8747	5.1426	0.0104	0.8595	-55.4156
Pc.sun + Flowers.m2	117.8753	5.1432	0.0104	0.8699	-55.4159
LogSize + PcGS + Flowers.m2	117.9403	5.2081	0.0101	0.88	-54.061
PcGS + Flowers.m2 + houses.ha	118.0921	5.3599	0.0093	0.8893	-54.1369
LogPc.veg + houses.ha + IRSAD	118.0995	5.3674	0.0093	0.8986	-54.0974
PcGS + Flowers.m2 + Dist.forest	118.184	5.4518	0.0089	0.9075	-54.1829
PcGS + Flowers.m2 + LogPc.veg	118.2706	5.5385	0.0085	0.9161	-54.2262
Dist.forest + LogPc.veg + IRSAD	118.5128	5.7807	0.0076	0.9236	-54.304
Pc.sun + LogSize + IRSAD	118.841	6.1089	0.0064	0.93	-54.4681
LogSize + PcGS + houses.ha	119.0728	6.3406	0.0057	0.9358	-54.6273
LogSize + PcGS + Dist.forest	119.1121	6.38	0.0056	0.9414	-54.647
LogSize + LogPc.veg	119.1923	6.4602	0.0054	0.9467	-56.0744
LogPc.veg + houses.ha	119.2246	6.4924	0.0053	0.952	-56.0906
LogSize + PcGS + LogPc.veg	119.2418	6.5097	0.0052	0.9573	-54.7118
Pc.sun + LogSize + PcGS	119.2692	6.5371	0.0052	0.9625	-54.7255
Pc.sun + LogPc.veg	119.3693	6.6371	0.0049	0.9674	-56.1629
LogSize + Dist.forest	119.5186	6.7865	0.0046	0.972	-56.2376
Pc.sun + Dist.forest	119.8183	7.0862	0.0039	0.9759	-56.3874
Pc.sun + LogSize	119.8347	7.1026	0.0039	0.9798	-56.3956
Dist.forest + houses.ha	119.8355	7.1034	0.0039	0.9837	-56.396
LogSize + houses.ha	119.8666	7.1345	0.0038	0.9875	-56.4116
Pc.sun + houses.ha	120.2348	7.5026	0.0032	0.9907	-56.5956

Table 5 (continued)

Model	AICc	Delta_AICc	AICcWt	Cum. Wt	Log Likelihood
Pc.sun + LogSize + Flowers.m2	120.3124	7.5803	0.0031	0.9938	-55.2471
Flowers.m2 + Dist.forest + houses.ha	120.5526	7.8205	0.0027	0.9965	-55.3672
Pc.sun + LogSize + LogPc.veg	121.9555	9.2234	0.0014	0.9979	-56.0687
Pc.sun + LogSize + Dist.forest	122.2925	9.5604	0.0011	0.999	-56.2372
Pc.sun + LogSize + houses.ha	122.6049	9.8728	0.001	1	-56.3934

ΔAICc values are ΔAICc <2 and are considered 'as good' as the best model (Symonds and Moussalli 2011)

PcGS percent green space, *IRSAD* Index of socio-economic advantage and disadvantage, *LogSize* the size of each garden (logx + 1), *LogPc.veg* percent cover of native vegetation (logx + 1), *Houses.ha* Housing density per ha, *Dist.forest* distance to nearest forest area, *Pc.sun* percent cover of sun in the garden

Table 6 Full set of candidate models examining the influence of variables on bee abundance

Model	AICc	Delta_AICc	AICcWt	Cum. Wt	Log Likelihood
IRSAD	79.7496	0	0.0719	0.0719	-36.3294
Pc.sun + IRSAD	79.8295	0.0799	0.0691	0.1409	-34.9624
Pc.sun	79.9807	0.2311	0.064	0.205	-36.4686
Flowers.m2	80.4513	0.7016	0.0506	0.2556	-36.7039
Pc.sun + Flowers.m2	81.0158	1.2662	0.0382	0.2938	-35.5988
LogSize	81.0433	1.2937	0.0376	0.3314	-36.9999
Dist.forest	81.2715	1.5219	0.0336	0.365	-37.114
Dist.forest + IRSAD	81.4026	1.653	0.0315	0.3964	-35.7489
Flowers.m2 + IRSAD	81.4874	1.7378	0.0301	0.4266	-35.7913
Flowers.m2 + Dist.forest	81.5212	1.7716	0.0296	0.4562	-35.8515
PcGS	81.5846	1.835	0.0287	0.4849	-37.2706
houses.ha	81.7799	2.0302	0.026	0.511	-37.3682
LogSize + IRSAD	81.7909	2.0413	0.0259	0.5369	-35.9431
houses.ha + IRSAD	81.8214	2.0718	0.0255	0.5624	-35.9583
PcGS + IRSAD	82.0307	2.2811	0.023	0.5854	-36.063
LogPc.veg	82.0449	2.2953	0.0228	0.6082	-37.5007
Flowers.m2 + LogPc.veg	82.1537	2.4041	0.0216	0.6298	-36.1678
LogPc.veg + IRSAD	82.2999	2.5503	0.0201	0.6499	-36.1976
Pc.sun + PcGS	82.359	2.6093	0.0195	0.6694	-36.2704
Pc.sun + LogSize	82.3764	2.6268	0.0193	0.6887	-36.2791
Pc.sun + Dist.forest	82.5364	2.7868	0.0178	0.7065	-36.3591
Pc.sun + houses.ha	82.613	2.8634	0.0172	0.7237	-36.3974
Flowers.m2 + houses.ha	82.7191	2.9695	0.0163	0.74	-36.4504
Flowers.m2 + Dist.forest + IRSAD	82.7341	2.9845	0.0162	0.7562	-34.8671
Pc.sun + LogPc.veg	82.7432	2.9936	0.0161	0.7723	-36.4625
LogSize + Dist.forest	82.852	3.1024	0.0152	0.7875	-36.5169
Pc.sun + LogSize + IRSAD	82.8563	3.1067	0.0152	0.8027	-34.9282
PcGS + Flowers.m2	83.1085	3.3588	0.0134	0.8161	-36.6451
LogSize + Flowers.m2	83.1654	3.4158	0.013	0.8291	-36.6736
PcGS + Dist.forest	83.2637	3.5141	0.0124	0.8415	-36.7228
LogSize + houses.ha	83.513	3.7634	0.0109	0.8525	-36.8474
LogSize + PcGS	83.5411	3.7915	0.0108	0.8633	-36.8615
PcGS + houses.ha	83.5534	3.8037	0.0107	0.874	-36.8676
LogSize + LogPc.veg	83.5857	3.836	0.0106	0.8846	-36.8837
Pc.sun + LogSize + Flowers.m2	83.8537	4.104	0.0092	0.8938	-35.4983
PcGS + LogPc.veg	83.919	4.1694	0.0089	0.9027	-37.0504
Dist.forest + houses.ha	83.9765	4.2269	0.0087	0.9114	-37.0792

Table 6 (continued)

Model	AICc	Delta_AICc	AICcWt	Cum. Wt	Log Likelihood
Dist.forest + LogPc.veg + IRSAD	84.2676	4.5179	0.0075	0.9189	-35.6338
PcGS + Flowers.m2 + Dist.forest	84.3903	4.6407	0.0071	0.926	-35.7666
PcGS + Flowers.m2 + IRSAD	84.4384	4.6888	0.0069	0.9329	-35.7192
Flowers.m2 + Dist.forest + houses.ha	84.5101	4.7605	0.0067	0.9395	-35.8265
LogPc.veg + houses.ha	84.5545	4.8049	0.0065	0.946	-37.3682
LogSize + PcGS + IRSAD	84.5757	4.8261	0.0064	0.9525	-35.7879
PcGS + Flowers.m2 + LogPc.veg	84.7765	5.0268	0.0058	0.9583	-35.9597
LogPc.veg + houses.ha + IRSAD	84.9036	5.154	0.0055	0.9638	-35.9518
Pc.sun + LogSize + Dist.forest	85.0806	5.3309	0.005	0.9688	-36.1117
Pc.sun + LogSize + PcGS	85.1559	5.4062	0.0048	0.9736	-36.1494
Pc.sun + LogSize + houses.ha	85.2757	5.526	0.0045	0.9781	-36.2093
PcGS + Flowers.m2 + houses.ha	85.4059	5.6563	0.0042	0.9824	-36.2744
Pc.sun + LogSize + LogPc.veg	85.4129	5.6633	0.0042	0.9866	-36.2779
LogSize + PcGS + Dist.forest	85.4639	5.7142	0.0041	0.9907	-36.3034
LogSize + PcGS + houses.ha	85.9754	6.2258	0.0032	0.9939	-36.5591
LogSize + PcGS + LogPc.veg	86.0658	6.3162	0.0031	0.997	-36.6043
LogSize + PcGS + Flowers.m2	86.0892	6.3396	0.003	1	-36.616

ΔAICc values are ΔAICc <2 and are considered 'as good' as the best model(Symonds and Moussalli 2011). Variable abbreviations follow Appendix Table 5

Table 7 Full set of candidate models examining the influence of variables on bee diversity (Shannon H index)

Model	AICc	Delta_AICc	AICcWt	Cum. Wt	Log Likelihood
LogSize	39.6641	0	0.0813	0.0813	-16.3103
Flowers.m2	39.8657	0.2016	0.0735	0.1549	-16.4111
LogSize + Dist.forest	40.2573	0.5932	0.0605	0.2154	-15.2195
PcGS	40.5945	0.9304	0.0511	0.2664	-16.7755
LogSize + PcGS	40.7584	1.0943	0.0471	0.3135	-15.4701
Dist.forest	41.2847	1.6206	0.0362	0.3497	-17.1206
PcGS + Flowers.m2	41.3506	1.6865	0.035	0.3847	-15.7662
Flowers.m2 + Dist.forest	41.4274	1.7633	0.0337	0.4184	-15.8046
PcGS + Dist.forest	41.6845	2.0204	0.0296	0.448	-15.9331
LogSize + Flowers.m2	41.7954	2.1313	0.028	0.476	-15.9886
LogSize + PcGS + Dist.forest	41.9023	2.2382	0.0266	0.5026	-14.5226
Pc.sun + LogSize	41.9879	2.3238	0.0255	0.528	-16.0849
LogSize + IRSAD	42.228	2.5639	0.0226	0.5506	-16.1616
Flowers.m2 + IRSAD	42.2411	2.577	0.0224	0.573	-16.1682
LogSize + LogPc.veg	42.2546	2.5905	0.0223	0.5953	-16.2182
LogSize + houses.ha	42.4122	2.7481	0.0206	0.6159	-16.297
Flowers.m2 + LogPc.veg	42.6098	2.9457	0.0186	0.6345	-16.3958
Flowers.m2 + houses.ha	42.6331	2.969	0.0184	0.653	-16.4075
Pc.sun + Flowers.m2	42.6403	2.9762	0.0184	0.6713	-16.411
IRSAD	42.8646	3.2005	0.0164	0.6878	-17.8869
LogPc.veg	42.8853	3.2212	0.0162	0.704	-17.9209
PcGS + IRSAD	43.0219	3.3578	0.0152	0.7192	-16.5586
PcGS + houses.ha	43.1045	3.4404	0.0146	0.7338	-16.6432
PcGS + Flowers.m2 + Dist.forest	43.2275	3.5634	0.0137	0.7475	-15.1852
Pc.sun + LogSize + PcGS	43.2687	3.6046	0.0134	0.7609	-15.2058
Pc.sun + LogSize + Dist.forest	43.2803	3.6162	0.0133	0.7742	-15.2116

Table 7 (continued)

Model	AICc	Delta_AICc	AICcWt	Cum. Wt	Log Likelihood
Pc.sun	43.2944	3.6304	0.0132	0.7875	-18.1255
houses.ha	43.295	3.631	0.0132	0.8007	-18.1258
Pc.sun + PcGS	43.3494	3.6853	0.0129	0.8136	-16.7656
PcGS + LogPc.veg	43.3664	3.7023	0.0128	0.8264	-16.7741
LogSize + PcGS + Flowers.m2	43.5361	3.872	0.0117	0.8381	-15.3395
LogSize + PcGS + IRSAD	43.5569	3.8928	0.0116	0.8497	-15.2785
Dist.forest + IRSAD	43.6617	3.9976	0.011	0.8607	-16.8785
Dist.forest + houses.ha	43.6967	4.0326	0.0108	0.8716	-16.9393
Pc.sun + Dist.forest	43.7123	4.0482	0.0107	0.8823	-16.9471
LogSize + PcGS + houses.ha	43.7151	4.0511	0.0107	0.893	-15.429
LogSize + PcGS + LogPc.veg	43.797	4.1329	0.0103	0.9033	-15.4699
Flowers.m2 + Dist.forest + IRSAD	44.0674	4.4033	0.009	0.9123	-15.5337
Flowers.m2 + Dist.forest + houses.ha	44.132	4.468	0.0087	0.921	-15.6375
PcGS + Flowers.m2 + LogPc.veg	44.1441	4.48	0.0087	0.9297	-15.6435
PcGS + Flowers.m2 + houses.ha	44.1777	4.5136	0.0085	0.9382	-15.6603
PcGS + Flowers.m2 + IRSAD	44.1839	4.5198	0.0085	0.9467	-15.592
Pc.sun + LogSize + Flowers.m2	44.6201	4.956	0.0068	0.9535	-15.8815
Pc.sun + LogSize + IRSAD	44.7917	5.1276	0.0063	0.9598	-15.8959
Pc.sun + LogSize + LogPc.veg	44.9877	5.3236	0.0057	0.9655	-16.0653
Pc.sun + LogSize + houses.ha	45.0238	5.3597	0.0056	0.9711	-16.0833
LogPc.veg + IRSAD	45.3319	5.6678	0.0048	0.9758	-17.7136
LogPc.veg + houses.ha	45.5159	5.8518	0.0044	0.9802	-17.8489
Pc.sun + LogPc.veg	45.6031	5.939	0.0042	0.9844	-17.8924
Pc.sun + IRSAD	45.6689	6.0048	0.004	0.9884	-17.8821
houses.ha + IRSAD	45.6775	6.0134	0.004	0.9924	-17.8864
Pc.sun + houses.ha	46.0681	6.404	0.0033	0.9957	-18.125
Dist.forest + LogPc.veg + IRSAD	46.1575	6.4934	0.0032	0.9989	-16.5788
LogPc.veg + houses.ha + IRSAD	48.2957	8.6316	0.0011	1	-17.6478

ΔAICc values are ΔAICc < 2 and are considered 'as good' as the best model(Symonds and Moussalli 2011). Variable abbreviations follow Appendix Table 5

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