Gardening in the desert changes bee communities and pollination network characteristics

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Abstract

Conversion of natural habitats to human settlements creates an alternative habitat with different bio-physical characteristics such as micro-climatic conditions and resource availability. Deserts are especially sensitive to such effects due to generally low nutrient levels and water availability. Gardens in human settlements in the desert are often a main source of alien plant species that provide ample foraging resources year round. These changes in floral composition and availability may alter pollinator community composition and foraging behaviour, as well as pollination network characteristics. We investigated the effects of desert gardening on pollinator communities and pollination networks in the Jordan Rift Valley (Israel), an arid agro-natural landscape south of the Dead Sea. We studied seasonal diversity patterns of plants and wild bees in natural habitats and in gardens in settlements. Wild bees and blooming plants were sampled from February to July in 2007 and in 2008. We constructed plant–pollinator networks and compared bee communities between the two habitat types along the season. We found that bee abundance was greater in the gardens, and that rarefied bee species richness was greater in the natural habitat. Bee species richness and abundance exhibited contrasting seasonal patterns between habitats. Bee community composition also differed greatly between habitats, and species in the gardens had a generally wider geographical range in comparison to species in the natural habitat. We also found a higher level of generalisation of the pollination network in the gardens compared to the natural habitat, which may indicate a response to a disturbed and unstable environment. We conclude that gardening in deserts although promoting overall bee abundance, negatively affects species richness and changes community composition and network characteristics, with possible implications on the composition of native flora in the natural habitat surrounding the gardens.

Zusammenfassung


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Bees are one of the most important groups of pollinators (Kearns & Inouye 1997), and in some places may constitute up to 94% of the pollinator community (Potts, Dafni, & Ne’eman 2001). They contribute to the pollination of both wild plants and agricultural crops (Klein et al. 2007), and fulfill a fundamental role in the function of ecosystems all over the world (Buchmann & Nabhan 1996). Recent declines in managed honey bee (Apis mellifera L.) populations led researchers to turn to wild bees as alternative pollinators and a form of insurance against possible pollination shortages (Klein, Steffan-Dewenter, Buchori, & Tscharntke 2002; Klein 2009; Winfree, Williams, Dushoff, & Kremen 2007; Winfree, Williams, Gaines, Ascher, & Kremen 2008), yet recent declines in wild bee populations have been reported as well (Aizen & Feinsinger 1994; Klein et al. 2002; Biesmeijer et al. 2006).

Habitat loss, degradation and fragmentation are among the main threats to bees today (Kears, Inouye, & Waser 1998; National Research Council 2006; McFrederick & LeBuhn 2006; Brittain et al. 2011), mainly due to agricultural intensification and urbanization (Klein et al. 2002; Cane, Minckley, Kervin, Roulston, & Williams 2006; Williams & Kremen 2007; Kearns & Oliveras 2009). Global human population has doubled in the past 50 years, which led to a sharp increase in the rate at which natural landscapes are being converted to intensive agricultural and human settlements (Reid et al. 2005).

Human settlements create an alternative habitat that differs in its physical characteristics, its micro-climatic conditions and its resource availability from the surrounding environment (Czech, Krausman, & Devers 2000). In order to survive in a habitat, bees require suitable floral resources, nesting sites and nesting material within flight range of each other. Many of the man-induced habitat changes affect the availability of these resources and may therefore alter a habitat’s suitability to different bee species (Cane 2001).

Most human settlements include gardens which may vary in size, shape and function as possible bee habitats (McIntyre & Hostetler 2001; Cane et al. 2006; McFrederick & LeBuhn 2006). Gardens may supply attractive floral resources for bees in an environment that does not offer any (Comba, Corbet, Hunt, & Warren 1999; McFrederick & LeBuhn 2006), and provide suitable nesting sites that may attract bees from afar (Cane et al. 2006). However, depending on their floral composition and phenology, gardens might fail to provide bees with the resources necessary for their survival (Cane et al. 2006; Ahné, Bengtsson, & Elmqvist 2009). Some gardens may resemble the natural surrounding landscape in the structure and composition of their plant communities, whereas others may greatly vary from their surrounding landscape, depending on the planting regime employed. Plants in particular have a major role in shaping the structure of bee communities, attracting and providing resources to some species while deterring others (Potts, Vulliamy, Dafni, Ne’eman, & Willmer 2003; Morales & Aizen 2006; Totland, Nielsen, Bjerknes, & Ohlson 2006; Bartomeus, Vila, & Santamaria 2008). Alien plants in gardens may differ in their blooming morphology and phenology, and thus affect the composition and/or seasonality of the local bee community (Kato, Shibata, Yasui, & Nagamasu 1999; Totland et al. 2006). Ultimately, these changes in plant and bee communities caused by the introduction of alien plants in gardens, may affect the local plant–pollinator network (Aizen, Morales, & Morales 2008; Bartomeus et al. 2008; Padrón et al. 2009). Alien plants will usually attract the more generalist bee species, increasing their number of partners (Lopezaraiza-Mikel, Hayes, Whalley, & Memmott 2007; Bartomeus et al. 2008; Padrón et al. 2009), thereby increasing the generalisation of the entire network.

In deserts, the contrast in biophysical conditions between gardens and the surrounding landscape is at its most extreme. Desert ecosystems are very poor in nutrients and water, and their plant communities are especially adapted to these conditions. Vegetation within the gardens is usually irrigated year-round, in contrast to the limited and highly seasonal rainfall sustaining the natural vegetation. This may greatly affect plant community composition, plant cover, blooming duration, and quality and quantity of both pollen and nectar foraged by bees. All these may greatly affect local bee communities and their interactions with the plant community.
The present study was conducted in the extremely arid Jordan Rift Valley (Israel), where communal settlements with gardens were established over the last 50 years. These gardens introduce new, often exotic, plant species, and provide a bounty of floral resources and constant water supply throughout the year. We investigate the effects of these biophysical differences on wild bee communities, and on plant–pollinator network structure.

Materials and methods

Study region

The Central Jordan Rift Valley is located in the southeastern part of Israel (Fig. 1). It is an extreme desert environment with average temperatures ranging between 27 and 40 °C in summer and 8 and 22 °C in winter and an average annual rainfall of less than 40 mm. The extreme climate limits natural vegetation growth mostly to wadis, with a rather short blooming season (ca. 16 weeks) that peaks around March and April for most local plant species. During the rest of the year, floral resources in the natural area are scarce and highly patchy.

Data collection

During February to June of 2007 and 2008 we sampled bees and plants in natural habitats and in ornamental gardens. The natural plots are located in wadis and contain only native plant species. The gardens are located within the small settlements, up to 1 km from the settlement boundary. They are rather small in size (~3000 m² on average) and are comprised of planted annual and perennial exotic species. In each habitat type we marked four 2500-m² plots that were at least 1.7 km apart. The natural plots were at least 10 km away from the settlements and agricultural areas, but maintained environmental conditions and vegetation type similar to the ones found in the natural habitat bordering the gardens. Each year we conducted 6 sampling rounds at ca. 3 weeks intervals. Bee sampling was conducted only if temperatures were above 18 °C and wind speed below 2.5 m/s. In each sampling round in each plot, bees were net-collected for 40 min between 08:00 and 12:00, and for 40 min between 12:00 and 16:00. During sampling, collectors walked the entire plot and examined all flowering plants encountered. Bees were collected directly from plants, and plants were recorded to establish a bee-plant interaction. Handling time was not included in search time. Together with bee collection, we recorded blooming plants (plant species and number of fresh flowers, i.e. petals and anthers intact and not dried) in 50 1-m² quadrats spread in a regular grid across the entire plot. We counted inflorescences rather than single flowers in plants from several families: Asteraceae, Lamiaceae, Resedaceae and Verbenaceae, as well as in Lobularia maritima.

We identified plant species in the field and in the lab with the help of Feinbrun-Dothan and Danin (1998) for wild species and Meltzer (2007) for garden plant species. Bee specimens were pinned and sent to taxonomists for identification: Prof. Holger Dathe, Hylaeus; Dr. Terry Griswold, Megachilidae; Dr. Sebastian Patiny, Panurgines; Prof. Alain Pauly, Halictidae; Dr. Stephan Risch, Euceridae; Dr. Christophe Praz, Megachile; Dr. Erwin Scheuchl, Andrenidae; Dr. Maximilian Schwarz, parasitic bees; and Dr. Michael Terzo, Xylocopidae. Since sampling was conducted similarly in 2007 and 2008, we were able to clump together catches from both years.

Data analysis

All tests were calculated using Excel and SPSS 17 unless stated otherwise. We used EstimateS 8.2 (Colwell 2009) to construct cumulative species richness curves and standardise differences in overall abundance between habitats.

Bee abundance and species richness

We compared bee abundance and species richness between habitats using a Mann–Whitney U test. To compare abundance-standardised species richness, we calculated the
expected species richness for equal abundances in the two habitats using 1000 randomizations in MATLAB R2008a.

The effects of season and habitat on wild bee species richness and abundance were tested using a two-way ANOVA with repeated measures. All variables were normally distributed (Kolmogorov–Smirnov test, all p-values non-significant). In the abundance analysis, Mauchly’s test indicated that the assumption of sphericity had been violated, $\chi^2 = 28.847, \text{df} = 14, p = 0.022$, therefore degrees of freedom were corrected using Greenhouse–Geisser estimates of sphericity ($\varepsilon = 0.315$).

We performed a linear regression analysis to examine the relationship between bee species richness and abundance and plant species richness and floral abundance in both habitats.

**Bee community composition**

We calculated a Yates-corrected $\chi^2$ test to investigate the habitat preference of bee species that were collected in both habitats. To assess the flight seasons of bee species, we compared the average number of samplings in which each bee species was collected throughout the season in both habitats using a Mann–Whitney $U$ test.

To explore differences in geographical distribution of bee species in the two habitats, we used the Discover Life (database at: http://www.discoverlife.org) and Atlas Hymenoptera (database at: http://www.atlashymenoptera.net) databases to give all possible bee species a global distribution categorical value (Appendix A: Table 1). Species with the narrowest distribution, i.e. the ones known only from the Middle East were assigned a value of 1. Species known also from North Africa were assigned a value of 2. Species known from around the Mediterranean or from countries adjacent to the Middle East were assigned a value of 3. Species with wider distributions – known from other areas (Europe, Africa and Asia) – were assigned a value of 4. We then compared the bees’ frequency distribution in the four categories between habitats using a $\chi^2$ goodness-of-fit test.

Using Michener (2007) and Müller (2010), we determined the nesting guilds (ground-nesting, plant-nesting, cleptoparasites) for all known bee species. We then compared bee species’ frequency distribution between habitats for ground- and plant-nesting species using a $\chi^2$ goodness-of-fit test.

**Network analysis**

We used the frequency-based index $H_2'$ to calculate network-level specialisation (Blüthgen, Menzel, & Blüthgen 2006). $H_2'$ ranges from 0 to 1.0 for extreme generalisation and specialisation, respectively. Since the number of plant species in the gardens is almost three times higher than in the natural plots due to gardening choices, we decided to compare specialisation separately for the pollinator level using the frequency-based community index $\langle d'_i \rangle$ (Blüthgen et al. 2006). This index also ranges from 0 to 1.0 for extreme generalisation and specialisation, respectively. We used these frequency-based indices since they are more robust against differences in abundance and are often more meaningful than metrics that only assign links as present or absent (Blüthgen, Fründ, Vázquez, & Menzel 2008). The values of both habitats were compared using the Mann–Whitney $U$ test. In order to examine whether the same species exhibit different levels of specialisation between habitats, we calculated species-level specialisation index $d'_i$ (Blüthgen et al. 2006) for the 13 wild bee species present in both habitats observed with two or more individuals. The values were compared between habitats using a paired $t$-test. The Bipartite package (Dormann, Fründ, Blüthgen, & Gruber 2009) in R 2.9.1 was used for calculating all indices and network diagrams.

**Results**

**Bee abundance and species richness**

In total, 1778 wild bees were collected, representing 81 bee species. Four hundred and seventy individuals and 55 species were collected in natural plots, compared to 1308 individuals and 50 species collected in gardens. Bee abundance in the gardens was significantly higher than in the natural plots ($U_{26,10} = 0, n_1 = 4, n_2 = 4, p = 0.029$; Mean ± S.E: Natural = 117.5 ± 8.1, Gardens = 327.0 ± 38.9). Species richness

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<th>Habitat</th>
<th>Independent variable</th>
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<th>$t$</th>
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<td>2.69</td>
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* Significant values.
** Marginally significant values.
was not significantly different between the two habitats ($U_{19,17} = 0$, $n_1 = 4$, $n_2 = 4$, $p = 0.886$; Mean ± S.E: Natural = 26.0 ± 2.7, Gardens = 27.8 ± 2.8). However, rarefaction curves showed a higher species richness in the natural habitat throughout the range inspected (Fig. 2). Abundance-standardised species richness in the gardens for 470 individuals drawn randomly out of the total 1308 collected individuals was 38.02 only; (95% CI 33–43 species, 1000 randomizations) this was much lower than the total of 55 species collected in the natural plots for the same abundance.

Bee abundance was significantly affected by season ($F_{1.57,30} = 5.4$, $p = 0.033$), by habitat ($F_{1.6} = 20.3$, $p = 0.004$), and by the interaction between season and habitat ($F_{1.57,30} = 8.2$, $p = 0.011$). The two habitats exhibited a somewhat inverse seasonal pattern in bee abundance (Fig. 3); abundance in the natural habitat peaked early in the season and then declined, while abundance in the gardens was initially similar to the natural habitats but increased later in the season.

Wild bee species richness was significantly affected by season ($F_{5.30} = 9.0$, $p < 0.001$), by habitat ($F_{1.6} = 7.6$, $p = 0.033$) and by the interaction between season and habitat ($F_{5.30} = 16.0$, $p < 0.001$). Wild bee species richness was higher in the natural habitat than in gardens early in the season (Fig. 4). In mid-season, bee species richness did not differ between habitats, and towards the end of the season species richness was higher in the gardens than in the natural habitat.

In the natural habitat, bee abundance significantly increased with plant species richness and flower abundance, and bee species richness was marginally positively related to plant species richness and flower abundance (Table 1). However, in the gardens neither bee species richness nor abundance was significantly related to plant species richness and floral abundance.

**Bee community composition**

The 10 most abundant wild bee species in each habitat comprised 79% of the total bee abundance in gardens and 66% in the natural plots (Table 2). Only one bee species, *Nomioides rotundiceps* Handlirsch, appears in the 10 most abundant species list of both habitats. Thirty one species were represented by only a single individual. Twenty one of them were collected in the natural plots and 10 in the gardens.

Twenty four out of a total of 81 wild bee species sampled were found in both habitats. We calculated habitat preferences for 16 of these 24 species (for the other eight species abundances were too low for appropriate analysis) (Appendix A: Table 2). Only four of the 16 shared species did not exhibit any habitat preference. Nine species preferred gardens, while three preferred the natural habitat.

Bee species in gardens had a longer activity season compared to species in natural plots ($U_{26,10} = 0$, $n_1 = 4$, $n_2 = 4$, $p = 0.029$; Mean no. of samplings ± S.E: Natural = 1.8 ± 0.1, Gardens = 2.8 ± 0.2).

The geographical distribution of bees in the gardens was generally wider than that of bees in the natural habitat ($\chi^2 = 1298$, df = 3, $p < 0.001$). The majority of bees in the natural habitat (56%) were known only from the Middle East and North Africa, while most of the bees in the gardens (90%)
were more widely distributed and were generally known from Europe, Asia, and other parts of Africa.

We found appropriate data to the nesting habits for 71 of the 81 bee species. Seven of them are cleptoparasites, 14 are plant-nesting, 45 are ground-nesting and five are known to use both plants and soil for nesting. We found no differences in the proportion of plant- and ground-nesting bee species between both habitats ($\chi^2 = 1.1, df = 1, p = 0.293$).

Network analysis

The pollination network of wild bees and plants in the gardens was comprised of 50 wild bee species and 58 plant species displaying 284 different links (Fig. 5A). The network in the natural plots was comprised of 55 wild bee species and 21 plant species displaying 106 links (Fig. 5B). The garden network was almost three times as large as the natural network, 1308 interactions compared to 470 respectively. The garden network was dominated by one wild bee species, *Ceylalictus variegatus* Olivier, which accounted for 34% of total interactions in the entire network. This bee species mainly interacted with the most dominant plant species in the network, *Sesuvium portulacastrum* L., which was involved in 16% of the interactions in the network. The natural habitat network was dominated by a different bee species, *Nomioides rotundiceps*, but it accounted for 16% only of the total interactions in the network. This species was connected almost exclusively to the most dominant plant species in the network, *Ochradenus baccatus* Delile, which was involved in 37% of the interactions of the entire network.

Comparing network-level generalisation $H_2'$, we found the networks in the gardens more generalised than the ones in the natural plots ($U_{26,10} = 0$, $n_1 = 4$, $n_2 = 4$, $p = 0.029$; Mean ± S.E: Natural = 0.767 ± 0.139, Gardens = 0.450 ± 0.075). Comparing bee community-level generalisation ($d'_i$) between habitats we found no differences in specialisation of the whole bee community ($U_{21,15} = 5$, $n_1 = 4$, $n_2 = 4$, $p = 0.486$; Mean ± S.E: Natural = 0.440 ± 0.132, Gardens = 0.372 ± 0.119). Comparing species-level generalisation $d'_i$, we found no differences in the specialisation exhibited by single bee species collected in both habitats ($df = 12, t = -0.071, p = 0.944$).

Discussion

Wild bee communities in gardens compared to natural habitat were found to be different in a desert ecosystem; overall abundance was higher in gardens, but rarefied species richness was higher in the natural habitat and community composition, seasonal community patterns, and plant–pollinator network characteristics were significantly different in desert gardens compared to the surrounding natural habitat.

Bee abundance and species richness

Most studies addressing the effects of anthropogenic disturbance found a negative effect on bee abundance and/or species richness (McFrederick & LeBuhn 2006; Kwaiser & Hendrix 2008; Winfree, Aguilar, Vázquez, LeBuhn, & Aizen 2009), whereas we found a positive effect on bee abundance. Our results differ from what was found in temperate, Mediterranean and tropical ecosystems and emphasize the difference in effects of anthropogenic disturbance on a desert environment in comparison to other non-arid ecosystems. The arid environment is very poor in water and nutrients, and vegetation and flower resources are therefore very scarce. The habitat change caused by gardening in the desert adds food resources to the system rather than depletes it, and thus increases its carrying capacity for wild bees. This is a major functional difference between desert habitats and other systems where water and/or nutrients are not that strongly limited.

McIntyre and Hostetler (2001), working in the arid Phoenix, Arizona, metropolitan area, found a decline in both richness and abundance of wild bees in gardens in comparison to natural desert environment. The contrasting effect on abun-
dance we found may be explained by the difference in capture methods used. The Phoenix study was based on pan-trap catches rather than net-collecting. Pan-trap catches may correspond very poorly to net-collecting, and yield fewer species and specimens, especially in areas where floral resources are abundant, as is the case in gardens (Cane, Minckley, & Kervin 2000; H.Y. and G.A., unpublished data collected in the study area, but see Westphal et al. 2008). Another possible reason for the difference could be the modest size of the settlements in our study (less than 2 km²), limiting the distance between gardens and the nearby desert area to a few hundreds of metres. Medium sized bee species (e.g. _A. mellifera_) have an average home-range radius of 2.8 km with a median of 1.5 km, and even small species such as _Lasioglossum umbripenne_ ellis are known to travel 200 m (reviewed in Cane 2001; Greanleaf, Williams, Winfree, & Kremen 2007). Hence, at least some of the bees sampled in the gardens could have used the desert habitat for nesting, while foraging in the gardens. The plots sampled in the Phoenix area were in some cases more than 10 km away from natural habitat and suit-
able nesting sites. It is known that the availability of nesting resources plays a key role in determining the structure of a bee community (Potts et al. 2005). Lack of nesting habitats was the major limiting factor for ground nesting bees in Salamanca, Spain (Gayubo & Torres, 1989), and a rise in the availability of cavities in the Tucson Basin of Arizona increased the abundance of cavity-nesting bee species (Cane et al. 2006). The fact that no differences were found in the distribution of ground- and plant-nesting bees between habitats in our study supports the hypothesis that nesting sites were not a limiting factor in our study.

Wild bees in our study differed in their seasonal trends between habitats. Bee species richness and abundance vary along the season and show an inverse trend between habitats. Abundance in the natural habitat peaks early in the season then decreases, while abundance in the gardens is low early in the season and peaks towards its end. In the natural habitats, bee species richness was highest in the beginning of the season, while in the gardens species richness was higher later in the season. Floral communities shape bee communities (Potts, Vulliamy, Dafni, Ne’eman, & Willmer 2003; Potts et al. 2006; Ebeling et al. 2008). We found that bee abundance and species richness in the natural habitat were positively correlated with plant species richness and flower abundance. The harsh desert climate dictates the blooming season in the natural plots, and thus the seasonal dynamics of bees. The gardens however, are disconnected from the arid conditions due to irrigation and planting. Bloom may no longer be the limiting factor in gardens, thus some bee species may elongate their activity season.

**Bee community composition**

We found bee community composition to differ greatly between habitats. Weiner et al. (2011), who compared intensively and extensively used grasslands, found a 42% overlap in flower visitors between habitats, which indicates a general high impact of changing environmental conditions on pollinators. However, in our study the difference in bee community composition was even greater; only 24 of the total 81 species could be found in both habitats, and only four of these showed no habitat preference. The others were either found only in one habitat or showed a significant habitat preference. We found that nine of the 10 most abundant species in each habitat were different, and that the two species groups seem to differ fundamentally; the bee community in the natural plots was dominated by species with a relatively narrow biogeographical range, mostly restricted to desert, whereas the dominant species in gardens had a generally broader biogeographical range, and included also non-arid ecosystems of Mediterranean, temperate and partly even tropical type. This may also explain the seasonality pattern of bees in the gardens, which was similar to that found in the Mediterranean region of Israel (Potts, Vulliamy, Dafni, Ne’eman, O’Toole, et al. 2003).

Bee species introduction to a new region is in general more limited than in other invertebrate groups (Invasive Species Specialist Group Database at http://www.issg.org/database). Many species of insects such as ants, whiteflies, aphids and beetles may arrive accidentally in potted plants and agricultural shipments (Invasive Species Specialist Group Database, http://www.issg.org/database), but most invasions of bees are of managed *Apis* and *Bombus* species gone feral (Kato et al. 1999; Ings, Ward, & Chittka 2006; Dohzono, Kunitake, Yokoyama, & Goka 2008). The bee species of broader biogeographical range found in our gardens are not invasive exotic species. They have been recorded in desert ecosystems in the past. However, they do differ from the dominant natural habitat species in their ability to colonize a wider spectrum of habitats, and can therefore take advantage of the bounty of foreign floral resources provided in the gardens and increase their abundance in that habitat. It is important to note that many of the species in both habitats were collected with very few individual numbers, and therefore could be easily overlooked in any of the plots. It is likely that the bee species found in the gardens exist in the adjacent desert habitat, but in a much lower abundance and were therefore not collected.

**Network analysis**

The pollination network of the gardens was dominated by a single bee species, much more so than the natural habitat network, and that one species interacted with many different plant species. This was in contrast to the dominant bee species in the natural habitat network, which interacted almost exclusively with just one plant species. This difference also applied to the network level; the garden network showed a higher generalisation level in comparison to the network of the natural habitat. One reason for this is simply the higher diversity of flower resources available to bees in gardens, and it is supported by the finding that generalisation levels displayed by the entire bee community level and by single shared bee species did not differ between habitats. Another reason for the higher network generalisation level in the gardens is the fact that nearly all plant species in the gardens in our study are alien to the region, did not evolve with the local bee fauna, and therefore mostly lack plant–pollinator specialisation, which takes place on an evolutionary time scale (Gomez & Zamora 2006; Minckley & Roulston 2006). A higher level of generalisation contributes to network stability, as it is less prone to extinction, and can therefore usually be found in more disturbed habitats (Gomez & Zamora 2006; Taki & Kevan 2007; Petanidou, Kallimanis, Tzanopoulos, Sgardelis, & Pantis 2008). When a species depends only on one or very few partners as food source or pollinator, it is more sensitive to the partner’s disappearance (Gomez & Zamora 2006; Pauw 2007). Each plant species in the gardens plays a smaller part in the network compared to the natural habitat, and will thus have limited effect on bee communities if eliminated. In
the undisturbed natural network, the two most visited plant species are very important to most of the bees in the network and constitute ca. 66% of all recorded visits. In the desert habitat, patches of floral resources change location and size between years. However, on the ecosystem level this limited diversity repeats itself, unlike many of the plant species in the gardens, which are taken in and out of the gardens as part of the local gardening regime. In that sense, the stability and predictability of plant species in the natural habitat, as well as their limited diversity, promote the strong and specialised local plant–pollinator relationships.

We conclude that gardening in deserts greatly affects bee communities and pollination networks. The differences found between natural habitat and gardens in the phenology and composition of flower resources are expected to be a major driving force in shaping bee communities in this ecosystem. In this study, the gardens serve as relatively rich bee reservoirs, accommodating many local species, probably due to the overwhelming abundance and diversity of flower resources they provide. However, the relatively small size of the settlements and distance from natural landscape may play an important role here. Larger settlements and greater distances from natural landscape might result in a poorer and less diverse bee community (McIntyre & Hostetler 2001; McFrederick & LeBuhn 2006). In addition, composition of bee communities differed significantly between habitats. Therefore, from a conservation perspective, while the current gardening in the study region may provide some source populations to nearby natural habitats in case of local extinctions, it may also alter the composition of natural communities with possible functional implications on pollination services to wild plants in the ecosystem (Blüthgen & Klein 2011, this feature). Future research should look into the effects of gardening regime on the local bee community. Gardening with local plant species will allow examining the effect of irrigation and consequent increase in flower resources vs. the effect of exotic plant species and increasing in floral diversity on local bee communities and local pollination network structure.

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Appendix A. Supplementary data


References


