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# Rainwater harvesting and arthropod biodiversity within an arid agro-ecosystem

Olivia Norfolk[a,*], Mahmoud Abdel-Dayem[b,c], Francis Gilbert[a]

[a] School of Biology, University of Nottingham, University Park, Nottingham NG7 2RD, United Kingdom
[b] Department of Entomology, Faculty of Science, Cairo University, Giza 12613, Egypt
[c] Plant Protection Department, College of Food and Agricultural Sciences, King Saud University, P.O. Box 2460, Riyadh 11451, Saudi Arabia

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ABSTRACT

The Bedouin of South Sinai, Egypt, use a technique known as runoff agroforestry to capture rainwater and increase the agricultural potential of the arid landscape. Utilising water that would otherwise be lost from the vicinity allows the creation of multi-strata orchards with higher plant densities than the surrounding environment. We used pitfall traps to compare ground arthropod communities within 15 agricultural gardens to those in the external habitat. Total arthropod abundance and species richness were significantly higher inside the gardens, with no loss of alpha-diversity. Species level analysis of ants revealed that six out of seven recorded species were more abundance inside the gardens. There were significant differences in the responses of Coleopteran functional groups, with scavengers occurring in higher numbers inside, predators higher outside, and herbivores showing no difference. There was a high presence of the ant *Monomorium venustum* and the scavenger beetle *Lagria* sp. inside the gardens, but no negative impacts on species accumulation rates or alpha-diversity. In conclusion, runoff agroforestry seems to enhance local arthropod abundance, without a strong negative affect on biodiversity.

## 1. Introduction

Agriculture can have dramatic effects on biodiversity, species composition and ecosystem functioning (Altieria, 1999; Hooper et al., 2005). Though intensive agriculture tends to have negative impacts on biodiversity (Le Féon et al., 2010; Prober and Smith, 2009; Reidsma et al., 2006; Robinson and Sutherland, 2002; Tscharntke et al., 2005), diverse agroforestry systems can limit these impacts and can provide important habitat for numerous species of insects (Hemp, 2005; Klein et al., 2002; Perfecto et al., 1997), birds (Lozada et al., 2007; Selmi and Boulinier, 2003) and mammals (Nyhus and Tilson, 2004). Creating diverse agro-ecosystems can also maintain natural processes on which farmers rely, such as ecosystem services that maintain soil fertility (Munyanziza et al., 1997), water retention (Roose and Ndayizigiye, 1997), temperature control (Lin, 2007), pollination (Jha and Vandermeer, 2010; Klein et al., 2003) and pest control (Shah et al., 2003; Trujillo-Arriaga and Altieri, 1990).

In the mountains of South Sinai, Egypt, the Bedouin people utilise a traditional agricultural strategy known as runoff agroforestry. Rainfall is scarce in the region, but when it falls, it often comes as heavy flash floods. Most of this water is lost as surface runoff, flowing from the mountainous wadis before it has time to penetrate the rocky soil. Using an ancient system of dams and walled gardens, the Bedouin direct this surface water into agricultural plots, giving it time to soak into the soil and thus increasing the potential for plant growth (Zalat and Gilbert, 2008). The gardens were traditionally positioned where permeable dykes of rock ran across the wadi, allowing water accumulation (and hence wells) if the water had time to seep into the bedrock.

Within these agricultural gardens the Bedouin grow a wide diversity of intercropped cultivated plants, with vegetables and other crops growing beneath fruit trees. Wild plants are generally tolerated within the gardens, with farmers weeding only the immediate vicinity of their crops. The Bedouin do not use any form of agro-chemicals, but do apply goat manure as fertiliser. The higher availability of water combined with a low-intensity approach to farming mean that these gardens provide a refuge for a variety of native flora: surveys conducted in 71 gardens by Zalat et al. (2001) recorded a total of 116 wild species within the gardens.

This study looks at the effects of run-off agroforestry by comparing the biodiversity of ground arthropods within the gardens to that of the surrounding natural habitat. Ground-dwelling insects are particularly suitable for making this kind of environmental impact assessment, because of their high biomass, ubiquitous presence (Rosenberg et al., 1986) (even in a relatively sparse desert environment such as Sinai), ease of capture (Kromp, 1999) and

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* Corresponding author. Tel.: +44 7725112071.
*E-mail addresses:* plxon1@nottingham.ac.uk, olivia_norfolk@hotmail.com (O. Norfolk), msabdeldayem@sci.cu.edu.eg (M. Abdel-Dayem), francis.gilbert@nottingham.ac.uk (F. Gilbert).

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multifaceted ecological significance (Losey and Vaughan, 2006). Particular attention is paid to Formicidae and Coleoptera, groups that are often described as good bio-indicators of landscape quality (Abdel-Dayem et al., 2007; Majer, 1983; Orabi et al., 2011; Pearce and Venier, 2006; Rainio and Niemelä, 2003; Underwood and Fisher, 2006).

## 2. Materials and methods

### 2.1. Study site

This study was conducted throughout May to June 2010 in the St Katherine Protectorate, South Sinai, Egypt. It is an arid, mountainous region with altitudes of 1500–2624 m. The climate is extremely dry, with hot and rainless summers and cool winters, receiving an average of 57 mm of rainfall a year (Ayyad et al., 2000).The landscape is dominated by rugged mountains, interspersed with steep-sided valleys (known as wadis); along the bottom of these wadis run riverbeds that remain dry for most of the year, only temporarily returning to rivers during the intermittent flash floods.

The geology of the high mountains of South Sinai is extremely complex, fundamentally consisting of basement granitic rocks with volcanic intrusions but with a long and complex geological history (Greenwood, 1997). Very little is known about the soils of South Sinai, making it difficult to make comparisons with elsewhere: however, from growing maize seedlings in soils from a number of sites, Abd El-Wahab et al. (2006) suggested that water was the dominating factor limiting plant growth, rather than any particular or combination of mineral nutrients.

The mountains are geologically typified by black volcanic rock and red granite. The black rock easily crumbles and is permeable to water, absorbing it rapidly and leaving it unavailable for agricultural use. The impermeable red granite is the key to the success of the agricultural gardens; it combines large, flat impenetrable surfaces with deep cracks and crevices. These underground cracks and dykes collect the run-off rainwater, providing a semi-permanent water source that can facilitate agriculture (Perevoltosky, 1981).

Bedouin gardens are strategically positioned above these underground water sources, with wells dug to provide irrigation for crops. Consequently the gardens tend to be clustered along the base of the wadis. Some of these traditional gardens can be traced back to 4th–6th century AD (Zalat and Gilbert, 2008), so do not represent a recent anthropogenic change to the landscape. The Bedouin are traditionally nomadic people and the gardens are not associated with the permanent human settlements that now occur in St Katherines town, but occur in the mountains, generally containing a small house used as a summer family residence.

### 2.2. Sampling techniques

We used a paired design, with 25 pitfall traps inside each garden and 25 outside. Fifteen gardens were selected from four valleys surrounding the town of St Katherine (Shraij, Tuffaha, Itlah and Abu Fraish). Garden choice was based on permission from garden owners and whether suitable trapping sites were available in the surrounding vicinity. External traps were placed a minimum of 15 m from their associated gardens, with approximately half placed along the base of the wadi and half placed higher up, upon the wadi slopes. It was not possible to place traps on inaccessible steep slopes or busy paths where they were frequently disturbed.

Traps were laid out in 5 × 5 grids, with 2 m between each trap. The simple trap design consisted of a plastic cup flush with the ground and filled to one-third with a solution of water and detergent. Traps were set between 9 am and 11 am in the mornings and emptied 24 h later. The contents of the traps were collected and stored in alcohol. Insects were grouped initially by family and then sorted into morpho-species based on visual characteristics. Representative individuals of beetle, spider and ant morpho-species were stored for identification by Dr Mahmoud Abdel-Dayem of Cairo University (beetles), Hisham El-Hennawy of Cairo (spiders) and Dr Brian Taylor of Nottingham (ants).

Pitfall traps do have limitations and cannot provide accurate estimates of true abundance, nor the relative abundance of the different functional groups, because certain groups tend to be over-represented (Lang, 2000). However, they can still provide relative numbers for a comparison between the two habitat types. Other methods of sampling were ruled out due to impracticality of the landscape–a high proportion of plants inside and outside of the gardens have thorns and spines making sweep-netting unfeasible. Although vegetation within the gardens is denser than outside, there are still large areas of exposed sandy soil with a similar composition to that found outside. Traps were all set in these open areas and not within areas of higher leaf litter (such as flower-beds), allowing for a comparable trapping effort inside and outside.

### 2.3. Statistical analyses

Statistical analyses were performed using R ver. 2.11.1 (R Development Core Team, 2010). Abundance of each insect taxon was typically low per trap, so data were pooled from the 25 traps associated with each grid. Generalized mixed effects models with Poisson errors (*lmer* function) included the following response variables: abundance of (1) Arachnea; (2) Coleoptera; (3) Collembola; (4) Formicidae; (5) Hemiptera; and (6) Orthoptera. *Habitat* (inside or outside of the garden) was included as a fixed factor and *wadi* ($N = 4$) and *garden* ($N = 15$) were included as random factors to account for potential spatial non-independence (Crawley, 2007). Model simplifications were based on AIC values. Fixed effects were dropped from the minimum sufficient model and tested for significance using $\chi^2$. This method was used in all subsequent models. Total species richness and alpha diversity per site were also included as response variables, with Poisson and normal error structures, respectively. Alpha diversity was calculated per garden using Simpson's Diversity Index, taking alpha as $1 - D$, where $D = \sum (n_i/N)^2$, $n$ = number of individuals of species $i$, and $N$ = the total number of individuals of all species.

Ant species differences were also analysed using generalized mixed effects models that included a *species * habitat* interaction. Beetles displayed high species richness, but many species occurred in low numbers making species-level analysis unfeasible. Similarly spiders did not occur in sufficient numbers for species level analysis. Beetles and spiders were sorted into functional groups and then analysed using generalized mixed effect models with a *functional group * habitat* interaction. Beetles were separated into three categories; herbivores, scavengers and predators. Spiders were separated into two categories, web-weavers and hunters.

EstimateS Version 8.2 (Colwell, 2009) was used to calculate the number of observed species for each sampled site (Sobs MauTau) and the mean diversity index (Shannon and Simpson) for each level of sampling, with samples added to the pool at random. Species accumulation curves were created for Formicidae and Coleoptera.

The community data were analysed for Formicidae and Coleoptera using the Non-Metric Multidimensional (NMS) scaling function in Community Analysis Package Version 4.1.3 (Seaby and Henderson, 2007). The analyses used PCA to determine the start position, Bray Curtis similarity measures and 200 iterations. The sites were grouped into *gardens* and *outside*, and the calculated Axis-1 coordinates were compared using a paired Wilcoxon signed rank test.

## 3. Results

Total arthropod abundance was significantly higher within the agricultural gardens than outside (Table 1). Collembola, Formicidae and Hemiptera showed a significant difference between the two habitat types. All other groups followed the same pattern, thus contributing towards the overall trend. Total species richness across groups was significantly higher within the gardens, but there was no difference in Simpson alpha-diversity.

### *3.1. Ant abundance and diversity*

Ants were the dominant group in both habitats. A total of 1511 individuals were caught, comprising 74.0% of the catch inside the gardens and 55.5% outside. Seven species were recorded, all of which occurred both in the gardens and outside. Ant species richness was higher within the gardens at an average of 4.5 (±0.43) species per site and 3.4 (±0.43) outside, but the difference was not significant (*lmer*: $\chi^2_1 = 2.44$, $P = 0.119$). All species occurred in higher numbers within the gardens, apart from *Crematogaster inermis* Mayr, 1862, which occurred in higher numbers in outside (Fig. 1). Ant abundance differed significantly between species ($\chi^2_6 = 800.07$, $P = 0.001^{***}$), with a significant species–habitat interaction ($\chi^2_6 = 173.95$, $P = 0.001^{***}$). Fig. 2a and c shows that rates of species accumulation and diversity were higher inside the gardens.

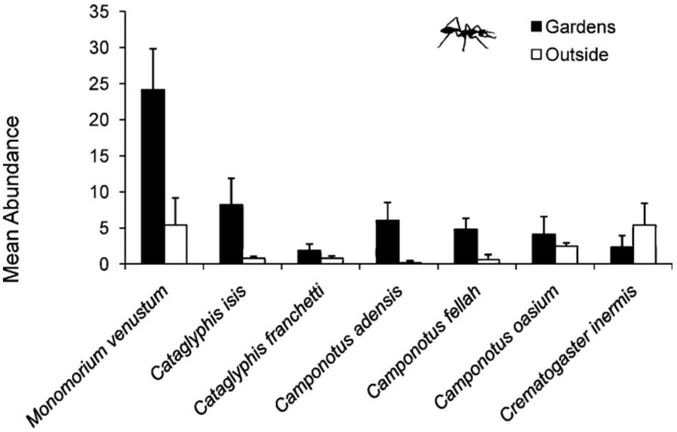

**Fig. 1.** Mean abundance of ant species per habitat type.

Non-metric multidimensional scaling showed that there was considerable overlap between the species composition of the communities within the gardens and outside, though the gardens typically had lower values along Axis 1 (Fig. 3a). Comparison of the axis-1 coordinates (gardens versus outside) showed that this difference was not significant (Wilcoxon: $V = 29$, $P = 0.0833$).

### *3.2. Beetle abundance and diversity*

Beetles were the most diverse group of arthropods in the survey. A total of 139 individuals were caught; 18 species from 11 families

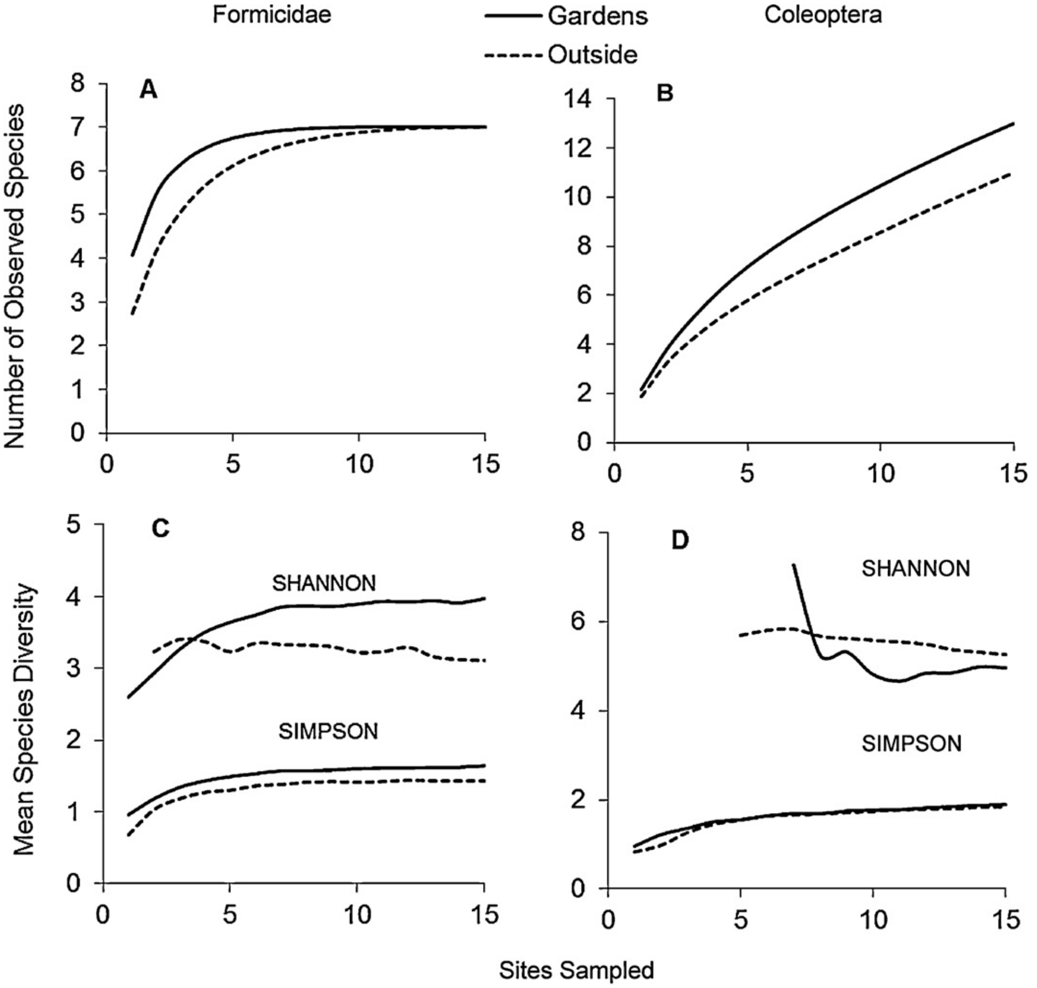

**Fig. 2.** Species accumulation curves and diversity indices for (a) and (c), Formicidae; and (b) and (d), Coleoptera.

**Table 1**
Average arthropod abundance, species richness and diversity.

| | Average ± S.E. (per site) | | Test of the difference | | |
|---|---|---|---|---|---|
| | Gardens | Outside | $\chi^2$ | *df* | *P* |
| *Abundance* | | | | | |
| All Species | 122.07 ± 21.39 | 40.73 ± 5.15 | 475.3 | 1 | 0.001 |
| Araneae | 2.50 ± 0.31 | 2.33 ± 0.39 | 0.024 | 1 | 0.876 |
| Coleoptera | 5.50 ± 1.91 | 4.13 ± 1.01 | 1.622 | 1 | 0.203 |
| Collembola | 10.50 ± 2.50 | 4.50 ± 1.28 | 45.56 | 1 | 0.001 |
| Formicidae | 78.20 ± 22.33 | 22.60 ± 6.08 | 486.76 | 1 | 0.001 |
| Hemiptera | 5.00 ± 1.81 | 3.92 ± 0.86 | 4.60 | 1 | 0.032 |
| Orthoptera | 1.70 ± 0.40 | 1.50 ± 0.14 | 2.50 | 1 | 0.114 |
| SPECIES RICHNESS (All) | 20.27 ± 1.92 | 14.53 ± 1.06 | 14.23 | 1 | 0.001 |
| SIMPSON DIVERSITY (All) | 0.741 ± 0.05 | 0.82 ± 0.02 | 1.63 | 1 | 0.202 |

Generalized mixed effect models with habitat (in/out of garden) as the fixed effect and garden and wadi as random effects. Abundance and richness models fitted with Poisson errors and Laplace approximations. Alpha diversity model fit with normal errors and REML approximations.

(species names listed in Appendix A). Species richness was higher inside the garden at an average of 4.2 (±1.7) species per site and 3.6 (±1.06) outside, but the difference was not significant (*lmer*, $\chi_1^2 = 0.69$, $P = 0.405$).

There were three identifiable functional groups; scavengers, predators and herbivores (Table 2). Fig. 4 shows that scavenger beetles occurred in higher numbers within the gardens and predatory beetles occurred in higher numbers outside. There was little difference between the abundances of herbivorous beetles. Beetle abundance differed significantly between the three functional groups (*lmer*, $\chi_2^2 = 12.61$, $P = 0.002^{**}$), with a significant interaction between group and habitat type ($\chi_2^2 = 16.58$, $P < 0.001^{***}$).

Fig. 2b and d shows that species accumulation rates and Simpson alpha-diversity were higher within the gardens than outside. Shannon alpha-diversity showed a different pattern; diversity was initially higher within the gardens but dropped with the number of sites sampled, finally ending up with the lower diversity.

Non-metric multidimensional scaling showed that the communities within the gardens and outside overlapped along their two main axes (Fig. 3b) and there was no significant difference between the axis-1 coordinates of the two groups (Wilcoxon: $V = 38$, $P = 0.968$).

### 3.3. Spider abundances

Spiders were less abundant than the ants and beetles, with a total catch of just 28 individuals. We identified seven families of spiders within the gardens and six outside (details specified in Appendix B). The abundance of web-weaving spiders was higher inside the gardens, with an average abundance (per site) of 0.40 (±0.06) inside and 0.06 (±0.02) outside. Hunting spiders were more abundant outside of the gardens with an average abundance of 0.533 (±0.05) inside and 0.733 (±0.06) outside. There was a significant difference between the abundances of the two functional groups (*lmer*, $\chi_1^2 = 7.10$, $P = 0.008^{**}$) and a significant habitat–group interaction ($\chi_1^2 = 8.43$, $P = 0.015^{*}$).

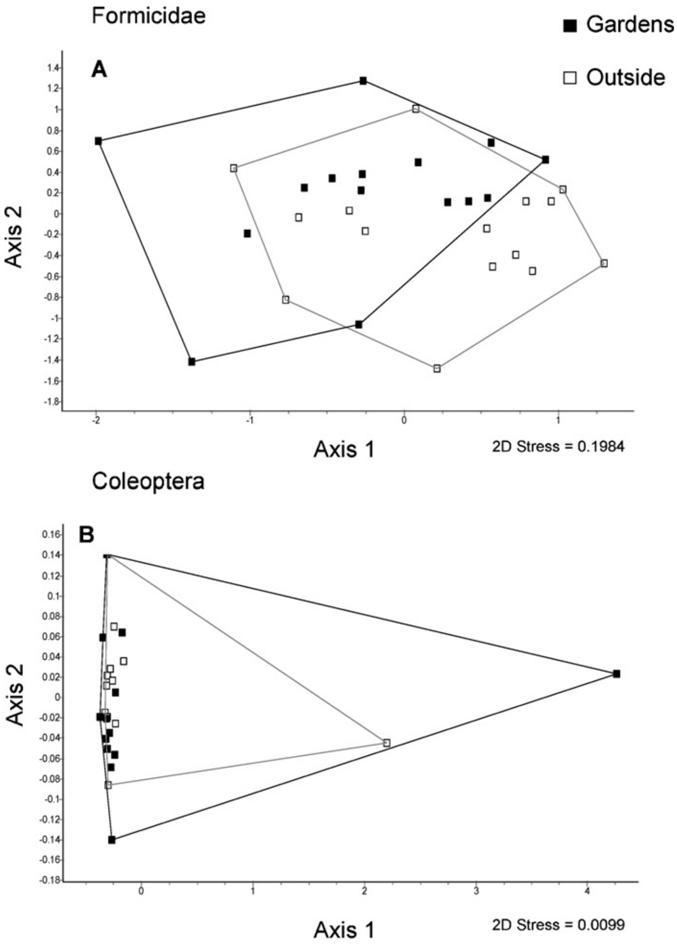

**Fig. 3.** Non-metric multidimensional scaling ordination plots of (a) Formicidae; and (b) Coleoptera.

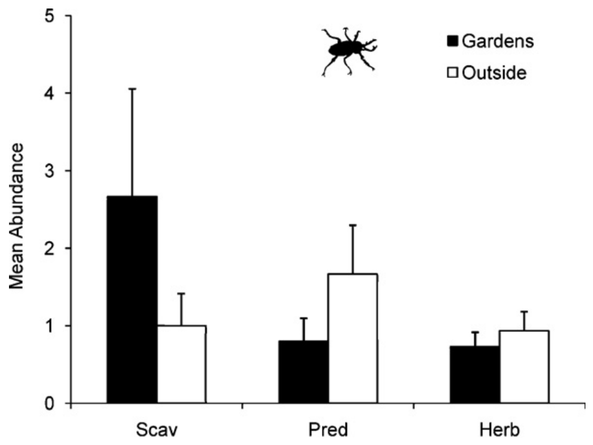

**Fig. 4.** Mean abundance of beetle functional groups per habitat type.

**Table 2**
Beetle functional groups and abundances.

| Family | Total abundance | | Functional group[a] |
|---|---|---|---|
| | Gardens | Outside | |
| Chrysomelidae | 0 | 1 | Herbivore |
| Curculionidae | 1 | 0 | Herbivore |
| Elateridae | 1 | 0 | Herbivore |
| Buprestidae | 9 | 13 | Herbivore/live wood |
| Carabidae | 2 | 2 | Predator |
| Histeridae | 0 | 1 | Predator |
| Meloidae | 0 | 5 | Predator |
| Cantharidae | 10 | 17 | Predator |
| Anthicidae | 1 | 1 | Scavenger |
| Scarabaeidae | 2 | 0 | Scavenger |
| Tenebrionidae | 37 | 14 | Scavenger |

[a] As determined by Lassau et al. (2005) and Susilo et al. (2009).

## 4. Discussion

The agricultural gardens farmed by the Bedouin of South Sinai contain higher ground-arthropod abundance than the surrounding natural habitat. There was no detectable loss of biodiversity, with overall species richness, species accumulations and alpha diversity remaining higher within the gardens. However, we have documented differential responses among arthropod species and functional groups.

Ant abundance was significantly higher within the gardens, with no decrease in species richness or diversity. This is consistent with previous work on the ants of South Sinai (Orabi et al., 2011), which has shown that altitudinal changes in abundance are not associated with a change in species richness, Shannon's diversity or evenness. However, the results contrast with most studies of ant communities in agro-ecosystems, which tend to demonstrate a negative impact of land conversion. For example, in the Solomon islands, forest clearance for coconut plantations leads to a decrease in ant species richness, with richness correlating with the biomass of vegetation present (Greenslade and Greenslade, 1977). In Costa Rica, coffee, banana and cacao agro-ecosystems all display significantly lower diversity of ground-dwelling ants than nearby forest (Roth et al., 1994; Perfecto et al., 1997). These losses are minimized in diverse planting systems, with ground ant diversity increasing with the complexity of vegetation (Perfecto and Snelling, 1995).

The arid conditions in Sinai are dramatically different from those found in the wet-tropics, a contrast that is likely to explain the differences in the responses observed. In the tropics, agricultural conversion typically involves deforestation and a loss of the biomass and complexity of vegetation. In Sinai the presence of agriculture, and the associated water harvesting techniques, actively increase plant densities, so it seems consistent that arthropod populations respond accordingly.

In accordance with the findings of Roth et al. (1994) and Samways (1983), who observed an increase of dominant species in agricultural plots, we observed a rise in the dominance of one ant species, the native *Monomorium venustum* (Smith, 1858). Despite the high abundance of this species, the rate of species accumulation was still higher within the gardens and unlike both of the cited studies there was no overall loss of diversity.

Beetle abundance and species richness did not differ between the internal and external habitats. In previous studies on beetles in the region (Abdel-Dayem et al., 2007; Semida et al., 2001), abundance and species richness (but not diversity) have been shown to differ amongst localities and altitudes, which authors attributed to habitat heterogeneity. Despite the obvious differences between the micro-habitat within the gardens and outside, we did not detect any changes in abundance. Our findings are dissimilar to European studies, where densities of beetles and other arthropods are higher in semi-natural grasslands than on farms themselves (Pfiffner and Luka, 2000; Purtauf et al., 2005); and tropical studies, where beetle diversity was lower in coffee agroforests than in natural forest (Perfecto et al., 1997).

Non-metric multidimensional scaling did not reveal any differences in species community structure, however there were significant differences between the abundances of beetle functional groups. Scavenger beetles occurred in higher numbers inside the gardens, though this was mainly due to the dominance the Tenebrionid, *Lagria* sp. (Appendix A). In Fig. 3d there is a sharp drop in the Shannon diversity within the gardens, which may be caused by the dominance of this species. Predatory beetles occurred in lower numbers within the gardens, which may reflect a lower abundance of suitable soft-bodied prey. Though many wild plants are tolerated inside the gardens, it is possible that gardeners consciously exclude certain plant species that support high numbers of aphids; for example the toxic *Gomphocarpus sinaicus* was frequently observed covered in aphids, but was rare within the gardens.

Hunting spiders did not show any differences between the two habitats. Web-weaving spiders occurred in significantly higher numbers inside the gardens. This may be related to the higher availability of suitable habitats (such as dry stone walls and old trees) or to the higher abundance of flying insects within the gardens (Personal Observation).

## 5. Conclusions

Home gardens and agroforestry systems in the tropics have frequently been linked with the conservation of valuable species (Bhagwat et al., 2008; Hemp, 2005) and the maintenance of ecosystem services (Altieria, 1999; Henry et al., 2009; Jose, 2009; Torquebiau, 1992). This study suggests that home gardens in arid environments may have similar conservation potential; rainwater harvesting has created agricultural gardens that enhance the abundance of ground arthropods (in particular the bio-indicator group Formicidae), whilst maintaining external levels of species richness and biodiversity.

## Acknowledgements

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## Appendix A. Total abundance of beetle species caught within the gardens and outside

| Family (genus and species) | | Total abundance | |
|---|---|---|---|
| | | Gardens | Outside |
| Anthicidae | *Pseudoleptaleus unifasciatus* (Desbrochers des Loges, 1875) | 1 | 1 |
| Buprestidae | *Acmaeoderella squamosa* (Thery, 1912) | 6 | 10 |
| | *Acmaeoderella pharao* (Obenberger, 1923) | 1 | |
| | *Anthaxia angustipennis* (Klug, 1829) | 2 | 3 |
| Cantharidae | *Chauliognathus* sp. | 10 | 17 |
| Carabidae | *Calosoma olivieri* (Dejean, 1831) | 2 | 1 |
| | *Laemostenus quadricollis* (L. Redtenbacher, 1843) | 0 | 1 |
| Chrysomelidae | *Oulema* sp. | 0 | 1 |
| Curculionidae | Unknown sp. | 1 | 0 |
| Elateridae | *Heteroderes abyssinus* (Candeze, 1859) | 1 | 0 |
| Histeridae | *Saprinus rubber gemmingeri* (Marseul, 1864) | 0 | 1 |
| Meloidae | *Mylabris* sp. | 0 | 5 |
| Scarabaeidae | *Pentodon algerinus algerinus* (Fuessly, 1778) | 1 | 0 |
| | *Rhyssemodes orientalis* (Mulsant and Godart, 1875) | 1 | 0 |
| Tenebrionidae | *Alphitobius diaperinus* (Panzer, 1796) | 12 | 1 |
| | *Lagria* sp. | 23 | 13 |
| | *Micipsa philistina* (Reiche and Saulcy, 1857) | 1 | 0 |
| | *Trachyderma genei* (Solier, 1836) | 1 | 0 |

## Appendix B. Spider identifications and abundances

| Family (genus and species) | | Total abundance | | | |
|---|---|---|---|---|---|
| | | Gardens | Outside | Ecological info. | Group |
| Gnaphosidae | *Pterotricha lesserti* (Comte de Dalmas, 1921) | 2 | 6 | Ground spiders – hunter, lacks prey capture web | H |
| | *Zelotes* sp. | 0 | 2 | | |
| Lycosidae | Unknown sp. | 3 | 0 | Wolf spider – hunter with keen eyesight, lacks web | H |
| Oecobiidae | *Uroctea* sp. | 4 | 0 | Small web weaver | W |
| Palpimanidae | *Palpimanus* sp. | 1 | 1 | Palp-footed spider | H |
| Pholciade | Unknown sp. | 2 | 0 | Long-legged web weaver | W |
| Sicariidae | *Loxosceles* sp. | 0 | 1 | Recluse spider – venomous, web builder | W |
| Thomisidae | *Xysticus tristami* (Kulczynski, 1908) | 1 | 0 | Crab spider – ambush hunter, lacks web | H |
| Zoldarilidae | *Zodarion* sp. | 3 | 2 | Ant hunting spider – often ant mimics | H |

H, hunter; W, web-weaver.

## Appendix C. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.agee.2012.08.007.

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