Traditional agricultural gardens conserve wild plants and functional richness in arid South Sinai

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Abstract

Maintaining agricultural diversity is important for the conservation of rare species and for preserving underlying ecosystem processes on which smallholder farmers rely. The positive effects of crop diversity are well documented in tropical systems, but the conservation potential of arid agricultural systems is less clear. This study assesses the impact of three arid agroforestry systems on plant diversity and functional richness in South Sinai, Egypt: (1) mountain orchard gardens, (2) modern town gardens and (3) low desert date-palm gardens. We surveyed plants (cultivated and wild) within gardens and control plots of natural habitat and allocated each plant eight biological traits that are recognised as being linked with major ecosystem processes. Species diversity was quantified using three measures (Hill's numbers) and total species diversity was significantly higher within gardens than in the surrounding habitat at all three levels of diversity and across the three agroforestry systems. Species similarity was high between gardens and the surrounding habitat, and there was a strong overlap in the functional traits of wild plants and cultivated non-tree species. Despite the clear presence of trees within the gardens, the community weighted trait means (CWMs) showed that chamaephyte perennials were the dominant life-forms in both the gardens and the natural habitat. Functional richness differed between the three agroforestry systems, but was significantly higher within the gardens. Functional richness has been linked to increased productivity and CWMs showed that plants within the gardens were considerably taller than outside, suggesting higher biomass accumulation. These findings suggest that Bedouin agricultural practices are not having a negative effect on the flora of the region and that the continuation of these indigenous farming practices can actively benefit rare wild plants in the region. On a wider scale, this study supports the view that smallholder farms and homegardens can be valuable tools in conservation, preserving local species and maintaining ecosystem functioning.

Zusammenfassung

Landwirtschaftliche Diversität zu erhalten ist wichtig für den Schutz seltener Arten und für den Erhalt grundlegender Ökosystemprozesse, auf die Kleinbauern angewiesen sind. Die positiven Effekte der Kulturpflanzendiversität sind für tropische Systeme gut dokumentiert, aber das Naturschutzpotential von landwirtschaftlichen Systemen in ariden Gebieten ist weniger klar. Die vorliegende Studie untersucht den Einfluss von drei ariden Agroforstsystemen auf die Pflanzendiversität und funktionale Vielfalt auf dem südlichen Sinai (Ägypten): (1) Obstgärten im Gebirge, (2) moderne Stadtgärten und (3) Dattelpalmgärten in der tiefer gelegenen Wüste. Wir erhoben Daten zu Wild- und Kulturpflanzen in Gärten und auf Kontrollflächen mit natürlichem Bewuchs und wiesen jeder Pflanze acht biologische Merkmale zu, die mit wichtigen Ökosystemprozessen in Zusammenhang stehen. Die Artenvielfalt wurde mit drei Maßzahlen quantifiziert (Hills effektive Artenzahlen). Für die drei Agroforstsysteme war die Gesamtartendiversität in den Gärten signifikant höher als in der Umgebung (alle Hill-Zahlen). Die Artenähnlichkeit zwischen Gärten und Umgebung war groß, und es gab eine hohe Übereinstimmung bei den funktionalen Merkmalen der Wildpflanzen

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und der Kulturarten (ohne Bäume). Trotz der offensichtlichen Anwesenheit von Bäumen in den Gärten zeigten die gewichteten Merkmalsmittel der Gemeinschaften (CWM), dass ausdauernde Chamaephyten sowohl in den Gärten als auch in den natürlichen Habitaten dominierten. Die funktionelle Vielfalt der Agroforstsysteme war unterschiedlich, aber immer in den Gärten signifikant höher. Die funktionale Vielfalt soll eine höhere Produktivität bedingen, und die CWM-Werte zeigten, dass die Pflanzen innerhalb der Gärten beträchtlich größer waren als außerhalb, was eine höhere Biomassebildung nahelegt. Diese Befunde legen nahe, dass die landwirtschaftliche Praxis der Beduinen keine negativen Folgen für die regionale Flora hat. Vielmehr kann die regionaltypische Bewirtschaftung aktiv seltene Wildpflanzen in der Region begünstigen. Generell unterstützt diese Untersuchung die Ansicht, dass kleinbäuerliche Betriebe und Hausgärten wertvolle Hilfsmittel des Naturschutzes sein können, indem sie lokale Arten bewahren und Ökosystemfunktionen aufrechterhalten.

Introduction

Intensive agriculture is notorious for its negative impacts on diversity with the simplification of complex ecosystems into monocultures leading to inevitable species erosion (Robinson & Sutherland 2002; Tscharntke, Klein, Kruess, Steffan-Dewenter, & Thies 2005; Prober & Smith 2009; Le Féon et al. 2010). Traditional homegardens across the world tend to maintain higher levels of crop diversity, because cultivating a range of sequentially ripening crops can provide year-round food security, whilst buffering against unpredictable environmental events such as droughts and pest outbreaks (Fernandes & Nair 1986; Jose & Shanmugaratnam 1993). As well as having practical benefits from the farmers' perspective, diverse agro-ecosystems can provide numerous social and environmental benefits (Jackson, Pascual, & Hodgkin 2007; Sandhu, Wratten, & Cullen 2010; Calvet-Mir, Gómez-Baggethun, & Reyes-García 2012), such as the provision of ecosystem services that maintain pest control (Trujillo-Arriaga & Altieri 1990), soil fertility (Munyanziza, Kehri, & Bagyaraj 1997), water retention (Roose & Ndayizigiye 1997) and pollination (Klein, Steffan-Dewenter, & Tscharntke 2003; Jha & Vandermeer 2010).

In the St Katherine's Protectorate of South Sinai, Egypt, the local Bedouin tribe have a long tradition of agriculture and goat herding, and have been cultivating mountain gardens for over one thousand years (Zalat & Gilbert 2008). These walled mountain gardens are arid-land equivalents of agroforests that depend on runoff rainwater for the growth of a variety of orchard products as well as vegetables and herbs (Norfolk, Abdel-Dayem, & Gilbert 2012). Runoff rainwater is captured using a system of walls and dams, giving it time to seep into the bedrock where it accumulates and can be accessed with wells and used for year-round irrigation (Perevolotsky 1981). Due to these rainwater harvesting techniques, the gardens have a higher potential for plant growth than the external environment and appear as 'oases' of greenery in the arid mountains. This unusual distribution of resources makes this a novel location in which to study the diversity effects of agro-ecosystems.

The impacts of agricultural diversity have been well-documented in tropical agro-ecosystems, with diverse systems providing important habitat for insects (Hemp 2005; Jha & Vandermeer 2010), forest birds (Waltert, Bobo, Sainge, Fermon, & Muhlenberg 2005; Beukema 2007; Clough, Dwi Putra, Pitopang, & Tscharntke 2009) and mammals (Nyhus & Tilson 2004). The conservation potential of arid agro-ecosystems has received less attention, perhaps because arid lands do not hold the same conservation significance as tropical forests. However, they are home to one third of the human population (MEA 2005) who may suffer with increasing pressures on food security in the face of predicted climate change. Restoring agricultural diversity could potentially help buffer against future climatic instability and minimise the risk to farmers in arid lands. In this study we consider three different agroforestry systems in South Sinai: (1) the traditional high-mountain orchard gardens, (2) their modern equivalents in the town of St Katherine, and (3) low-altitude desert gardens, which are dominated by date palms.

The aim of this study was to assess the impact that gardens (from all three agroforestry systems) have upon natural plant communities, using both species-based and functional-trait-based analyses. Southern Sinai contains a high diversity of wild plants, but approximately one third of the 600 plant species present are classified as very rare within Egypt (Ayyad, Fakhry, & Moustafa 2000). We are interested in whether these gardens are (a) helping to protect the rare wild plants within this protected area, and (b) whether they are supporting plants with similar ecosystem function as those in the surrounding natural habitat. We surveyed plants (cultivated and wild) within gardens and control plots of natural habitat and allocated each plant biological traits that are recognised as being linked with major ecosystem processes. Functional traits are often regarded as more informative indicators of ecosystem functioning than species measures alone (Dı′az & Cabido 2001; McGill, Enquist, Weiher, & Westoby 2006) and a trait-based approach allows us to assess whether these gardens are supporting plants with a similar ecological niche as those in the surrounding habitat, whilst giving insight into the underlying ecosystem processes within the gardens.

Materials and methods

Study sites and sampling methods

Our study was conducted in and around the St Katherine Protectorate, South Sinai, Egypt (Fig. 1), during April and May 2012 at the peak of the flowering season (Danin 2006). The Sinai Peninsula lies in the arid belt of North Africa, in the low rain belt of Egypt. The study area has a true desert climate, experiencing extremely dry, hot summers and cold winters. Average rainfall is 57 mm a year, with maximum temperatures of 35 °C and lows of 5 °C (St Katherine Protectorate Management Unit, pers. comm.). The high mountains (1600–2460 m asl) surrounding the town of St Katherine receive higher levels of precipitation, of up to 100 mm per 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 becoming rivers during the intermittent flash floods that occur every 10–20 years.

A total of 30 gardens from seven areas were randomly selected for sampling (subject to permission from garden owners); ten from the high mountains (Wadi Gebel ~1800 m asl and Wadi Itlah ~1500 m asl), ten from within the boundary of the town of St Katherine (St Katherine village and Raha, ~1500 m asl) and ten from much lower altitudes just outside the high Ring Dyke region (Sheikh Awad, ~1100 m asl), and westwards (Wadi Feiran ~700 m asl) and eastwards (Ein Hodra Oasis ~700 m asl) towards the base of the mountain massif (collectively called here the 'low-desert gardens'). From satellite imaging we have estimated that there are between 500 and 600 gardens in the St Katherine Protectorate. In the mountains and towns they form a dense network of walled gardens that run along the base of mountain valleys, but in the low desert they are much sparser, reflecting the lower availability of natural water sources. Photographs of typical gardens from the three agroforestry systems are shown in Fig. 2. The co-ordinates of the specific gardens sampled are included as supplementary material (Appendix A: Table 1).

Plant communities have been shown to display high levels of spatial variation within the region and can differ significantly between individual wadis (Ayyad et al. 2000), so one control plot was allocated to each of the seven wadis. These control plots were a minimum of 200 m away from all gardens and were selected to typify the microhabitat found in the gardens (along the base of the wadi, with sandy soil equivalent to that found within the gardens). Ideally one control plot would have been sampled per garden, but because gardens are highly clustered along the wadi bottoms (see Appendix A: Fig. 1) it was simply not feasible to find enough remaining natural habitat. In preliminary work we randomly positioned one control plot in the vicinity of each garden, but they tended to fall along the steep rocky sides and not along the wadi base, so consisted of very different rocky habitats from those within

the gardens and contained much lower densities of plants. In this study, plant abundance was not significantly different between gardens and control plots (lmer: $\chi^2 = 1.65$, df = 1, $P = 0.198$) and we believe it is an improvement on previous methods.

We recorded all plants along two 50 m transects that reached diagonally across the garden (or control plot) from one corner to the other. All plants within 1 m either side of the transects were counted. We choose to diagonally position the transects in order to prevent oversampling of flowerbeds and crops that tended to run linearly with respect to the garden walls. Plants were identified in the field where possible or collected for identification in the laboratory using Boulos (1999–2005). Plants were classified as either wild or cultivated, with cultivated defined as any species actively tended for food, household, medicinal or ornamental purposes. Trait data were compiled for each species, with eight traits coded as either quantitative or ordinal variables (Table 1). Data were gathered from Boulos (1999–2005) and the Flora of Israel Online (Danin 2006) and traits were collated in accordance to the LEDA Traitbase (Kleyer, Bekker, Knevel, Bakker, & Thompson 2008). See LEDA (2012) for more information.

Statistical analyses

Hill's numbers (species richness [^{0}D], the exponential of Shannon entropy [^{1}D] and the inverse Simpson index [^{2}D]) (Hill 1973) were used as diversity measures in accordance with current consensus (Jost 2006; Tuomisto 2010; Leinster & Cobbold 2011; Chao, Chiu, & Hsieh 2012). Hill's numbers are defined to the order of q (^{q}D), whereby parameter q indicates the weight given towards rare or common species. ^{0}D (species richness) is insensitive to relative frequencies, and is therefore weighted towards rare species. ^{1}D (exponential of Shannon entropy) is weighted towards common species, and ^{2}D (inverse Simpson concentration) is weighted towards abundant species. Diversity measures were calculated in SPADE (Chao & Shen 2010). Species richness (^{0}D) was estimated using *Chao1-bc*, a bias-corrected form of Chao1 (Chao 2005). ^{1}D and ^{2}D were estimated using a maximum likelihood estimator (Magurran 1988). Plant abundance was quantified as the total number of counted individuals along both transects and diversity indices were calculated from the summed data.

Plant abundance and the three measures of diversity (^{0}D, ^{1}D, ^{2}D) were compared between gardens and their control plots and across agroforestry systems using linear mixed-effect models using package *lme4* (Bates, Maechler, & Bolker 2011) in R.15.1 (R Core Team 2012). *Plant abundance/diversity* was included as the response variable, *garden/control* and *agroforestry system* (mountain/town/low desert) as explanatory variables and *Wadi* as a random factor to account for spatial variation within the three sites. Model simplifications followed Zuur, Ieno, Walker, Savelieve, and
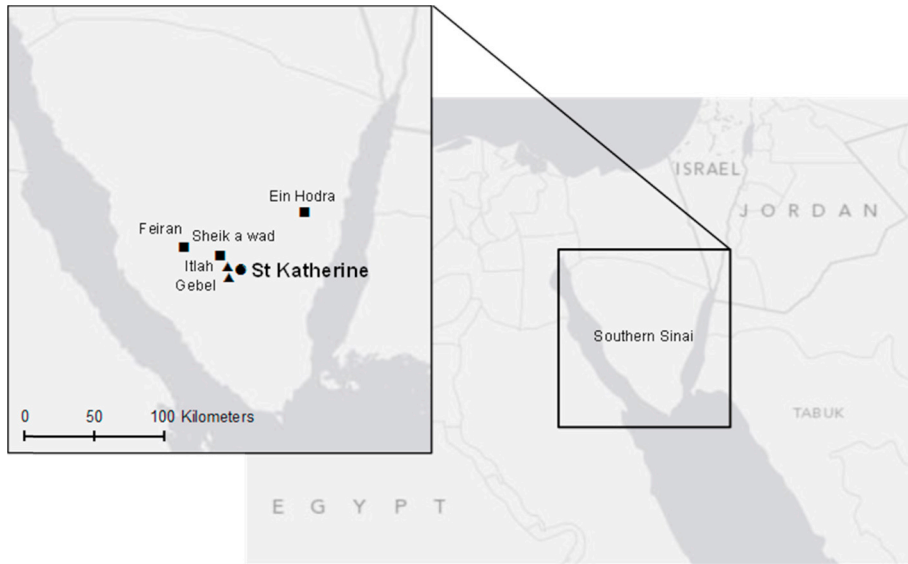
Fig. 1. Map of study sites in South Sinai, Egypt, showing St Katherine town (circle), mountain sites (triangles) and low desert sites (squares).

Smith (2009). We considered the size (m^2) of each garden as a potential confounding variable, but linear mixed-effect models showed that there were no significant relationships between area and plant abundance ($\chi^2_1 = 0.15$, $P = 0.697$), ^{0}D ($\chi^2_1 = 0.51$, $P = 0.477$), ^{1}D ($\chi^2_1 = 0.01$, $P = 0.917$) or ^{2}D ($\chi^2_1 = 0.04$, $P = 0.849$) so area was not included in our main analyses.

A standardised principal components analysis (PCA) was performed to explore the distribution of cultivated and wild species within the trait space. Cultivated plants were separated into two groups (trees and other plants) and their distribution along the first principal component axis was

compared to that of wild plants using Tukey's HSD test. In order to compare overall functional differences between sites we examined functional richness as the amount of functional niche space filled by species in the community (Mason, Mouillot, Lee, & Wilson 2005). Functional richness was calculated using the *dbFD* function in the *FD* package (Laliberté & Legendre 2010), which uses a distance-based approach to compute multidimensional functional diversity indices from a species-by-traits matrix. The functional traits were of various statistical types (quantitative and ordinal) so a Gower dissimilarity matrix was used to calculate the functional differences amongst species. Distances were not Euclidean so a
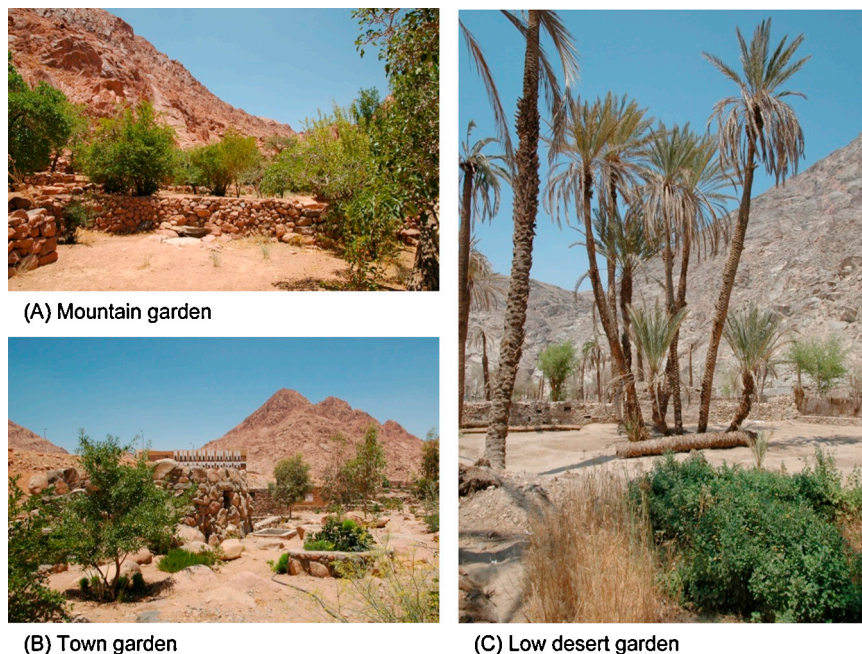

(A) Mountain garden

(B) Town garden

(C) Low desert garden

Fig. 2. Photographs depicting typical gardens from the three agroforestry systems.

Table 1. List of species traits used for the classification of functional groups; data types are either quantitative (Q) or ordinal (O).

Data type	Trait	Attributes	Category/unit(s) of measurement
	Whole plant traits		
O	Plant growth form	Phanerophyte	1
		Chamaephyte	2
		Hemicryptophyte	3
		Cryptophyte	4
		Therophyte	5
Q	Canopy height		m
O	Plant life span	Annuals	1
		Short to medium lived perennials (<10 yrs)	2
		Long-lived perennials (>10 yrs)	3
	Leaf traits		
Q	Leaf length		cm
Q	Leaf width		cm
	Stem traits		
O	Woodiness	Woody	1
		Semi-woody	2
		Herbaceous (non-woody)	3
O	Shoot growth form	Lianas, climbers and scramblers	1
		Stem erect	2
		Stem ascending to prostrate	3
		Stem prostrate	4
O	Leaf distribution along the stem	Rosette/tufted plant	1
		Semi-rosette	2
		Leaves distributed regularly along the stem	3
		Shoot scarcely foliated	4
		Tufts and crowns at the top of shoot or stem	5
		Other	6

Cailliez correction was applied (Cailliez 1983). Community-level weighted means (CWMs) of trait values were calculated using the *functcomp* function in *FD*, whereby the CWM for quantitative traits is the mean trait value of all species present in the community weighted by their relative abundances, and the CWM for ordinal traits is the most dominant trait in the community.

Results

Plant abundance and diversity

In total 5112 plants were recorded, belonging to 96 species from 36 families (for species descriptions see Appendix A: Table 2). There was no significant difference between the total abundance of plants within the gardens and their associated control plots (lmer: $\chi^2 = 1.65$, df = 1, $P = 0.198$), nor across the three agroforestry systems ($\chi^2 = 5.12$, df = 2, $P = 0.077$), although town gardens had the highest overall abundance (Fig. 3A).

All three Hill-number measures of diversity were significantly higher within the gardens than in control plots (^{0}D: $\chi^2 = 30.83$, df = 1, $P < 0.001$; ^{1}D: $\chi^2 = 7.13$, df = 1, $P = 0.008$; ^{2}D: $\chi^2 = 27.95$, df = 1, $P < 0.001$, Fig. 3B–D). Species richness (^{0}D) differed significantly among the three agroforestry systems ($\chi^2 = 11.38$, df = 2, $P = 0.004$) and was highest within the high mountains, but ^{1}D ($\chi^2 = 4.88$, df = 2, $P = 0.087$) and ^{2}D ($\chi^2 = 1.63$, df = 2, $P = 0.443$) showed no significant differences among systems.

Separating plants into cultivated and wild species showed that the high-mountain gardens had the highest proportion of wild plants (60% of all plants) and town and low-desert gardens the lowest (~30% respectively). When cultivated plants were removed from the analyses the species richness of wild plants was still significantly higher in the gardens ($\chi^2_1 = 9.14$, $P = 0.003$), with twice the species richness found in the control plots. ^{1}D of wild plants was 70% higher within the gardens than the control plots ($\chi^2_1 = 8.45$, $P = 0.004$), but ^{2}D was not significantly different ($\chi^2_1 = 1.93$, $P = 0.164$). This implies a greater number of scarce and moderately common species but a similar number of dominants.

86% of wild plants within the gardens were native to the Middle East, 27% of which were endemic or near-endemic. In total, eight such taxa were recorded in the gardens (assessed according to Boulos 1999–2005): *Origanum syriacum sinaicum*, *Phlomis aurea*, *Plantago sinaica*, *Silene schimperiana* (all Sinai only), *Crataegus sinaica* (Sinai, Syria and Saudi Arabia), *Tanacetum sinaicum* (Sinai, Palestine and Saudi Arabia), *Fagonia mollis* and *Paronychia sinaica* (both Egypt and Palestine only). The average Sørenson's
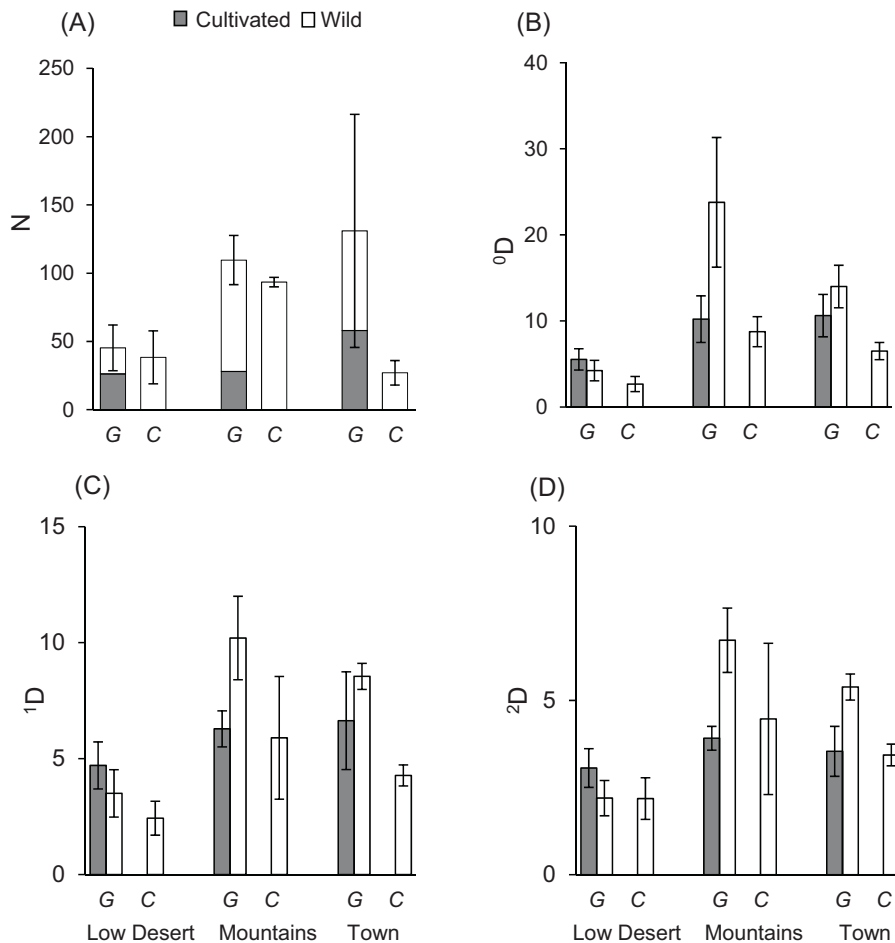
Fig. 3. Mean plant (A) abundance, (B) ^0D, species richness, (C) ^1D, exponential of Shannon entropy, and (D) ^2D, inverse Simpson per garden or control plot (200 m^2). Error bars represent the standard errors of the mean and in (A) this is for all plants (cultivated and wild combined). G = gardens, C = control plots.

similarity index of wild plants, when compared to their associated control plots, was 0.785 ($\pm$0.039) per garden.

Functional trait analyses

Principal components analysis of the eight functional traits explained 66% of the variance between plant species along its two main axes (Fig. 4). The first principal component accounted for 46% of the species variation and had high positive loadings with canopy height and plant life span, and negative loadings with plant growth form and stem woodiness. Cultivated tree species were clearly separated from other cultivated and wild species along axis 1 (Tukey's HSD test: $P < 0.001$), but there was considerable overlap between all other cultivated and wild species (Tukey's HSD test: $P = 0.828$). The second principal component accounted for 21% of the variation, with high positive loadings with leaf length and leaf width, but this was primarily due to one outlying species, *Phoenix dactylifera*, which had considerably larger leaves than all other species (top right corner of plot).
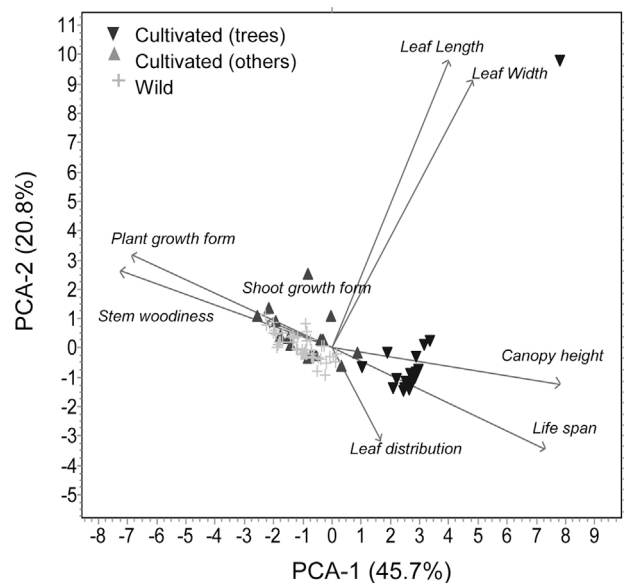

Fig. 4. Standardised principal components analysis (PCA; first vs. second axes) of cultivated and wild species characterised by eight plant traits. For list of trait variables see Table 1.

Table 2. Community weighted trait means across the three agroforestry systems; quantitative traits have mean with SEM and ordinal traits have modal values.

	Low desert		Mountain		Town	
	Garden	Control	Garden	Control	Garden	Control
Plant growth form	Chamaephyte	Chamaephyte	Chamaephyte	Chamaephyte	Chamaephyte	Chamaephyte
Canopy height (m)	9.5 ± 1.6	0.7 ± 0.3	1.6 ± 0.2	0.6 ± 0.01	1.4 ± 0.2	0.4 ± 0.01
Plant life span	Perennials <10 yrs	Perennials <10 yrs	Perennials <10 yrs	Perennials <10 yrs	Perennials <10 yrs	Perennials <10 yrs
Leaf length (cm)	149.0 ± 36.4	2.7 ± 0.3	10.0 ± 3.5	5.8 ± 0.4	8.4 ± 1.4	1.9 ± 0.4
Leaf width (cm)	30.5 ± 7.2	0.5 ± 0.2	2.7 ± 0.6	1.4 ± 0.01	2.3 ± 0.4	0.6 ± 0.2
Woodiness	Semi-woody	Semi-woody	Herbaceous (non-woody)	Herbaceous (non-woody)	Herbaceous (non-woody)	Herbaceous (non-woody)
Shoot growth form	Stem ascending	Stem ascending	Stem ascending	Stem ascending	Stem ascending	Stem ascending
Leaf distribution	Regularly along stem	Regularly along stem	Regularly along stem	Regularly along stem	Regularly along stem	Regularly along stem

The community weighted means (CWMs) of trait values (Table 2) show that all three sites were dominated by perennial chaemaphyte sub-shrubs in both the gardens and the control plots. Although trees were not the dominant growth form within the gardens, they had a strong influence on the CWM for canopy height, which was considerably higher within the gardens than in the control plots. Though many of the CWM trait values were the same in gardens and their control plots, the overall functional richness (the number of unique trait combinations in the community) was significantly higher within the gardens than the control plots (Fig. 5; lmer: $\chi^2 = 15.12$, df = 1, $P < 0.001$), as was the functional richness of wild species alone ($\chi^2 = 4.73$, df = 1, $P = 0.029$). Total functional richness also differed significantly between the three agroforestry systems ($\chi^2 = 9.04$, df = 2, $P = 0.012$) and was lowest in the low desert gardens.

Many of the common cultivated species could be paired to one or more wild species with high taxonomic relatedness and overlapping functional traits. For example: cultivated rocket (*Eruca sativa*) with wild wall-rocket (*Diplotaxis harra*); cultivated fennel (*Foeniculum vulgare*) with umbellifer *Deverra triradiata*; cultivated rosemary (*Rosmarinus officinialis*) and oregano (*Origanum sinaicum*) with wild labiates *Stachys aegyptica* and *Ballota undulata*; and cultivated beans (*Phaseolus vulgaris*) and alfafa (*Medicago sativa*) with wild nitrogen-fixing legumes such as *Raetama raetam* and *Astralagus* species.

Discussion

This study shows that traditional agricultural gardens in the arid region of South Sinai maintained high levels of native plant diversity, with higher plant functional richness than that found in the surrounding environment. In the tropics, smallholder agricultural systems have been shown to combine high yields with high biodiversity (Clough, Barkmann, Juhrbandt, Kessler, & Wanger 2011; Tscharntke et al. 2012) and this study suggests that the same land-sharing approach to agriculture can be applied to arid landscapes.

Total plant diversity was significantly higher within the gardens at all three levels of diversity (^0D, ^1D, ^2D), suggesting that rare, common and dominant species are all enhanced within the gardens. When cultivated plants were excluded from analyses the gardens still had a positive effect on the diversity of wild plants, the majority of which were native to the Middle East and one quarter endemic to the region. The biggest diversity increase was seen in wild plant species richness, suggesting that scarce wild plants were the most positively influenced and demonstrating the role of the gardens in conserving rare species.

There was a high species similarity between the wild plants found growing inside and outside of the gardens, suggesting that the gardens support plants with a similar ecological niche to those in the natural habitat. This was confirmed by the functional trait analyses, which showed that community weighted
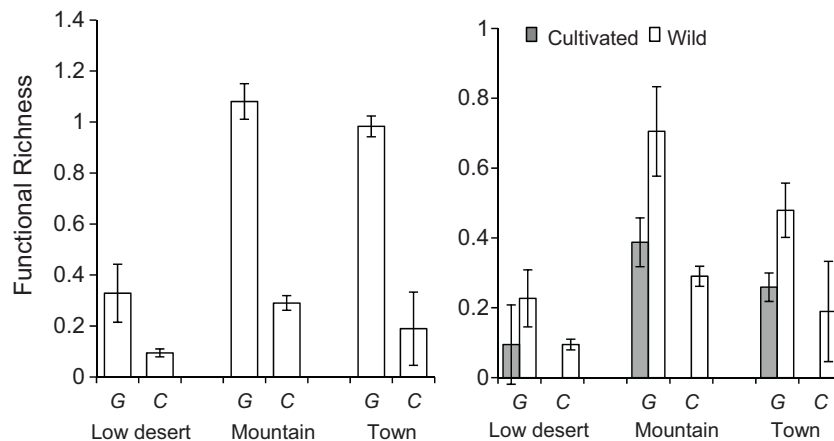
Fig. 5. Mean functional richness per garden or control plot (200 m²) for (A) all plants, and (B) cultivated and wild plants separately. G = gardens, C = control plots.

trait means overlapped considerably between species inside and outside of the gardens. Despite the obvious presence of trees within these orchard gardens, it was chaemaphyte perennial sub-shrubs that formed the dominant trait combination in all three agroforestry systems (low desert, mountain and town), as they did in the natural habitat. Modern low-altitude forest plantations elsewhere in Sinai have also been shown to increase the diversity of wild plants above those in the surrounding environment (Farahat & Linderholm 2012). However, unlike the Bedouin traditional gardens over half of the new species within the plantations were agricultural weeds. The forests were described as dense plantations of tall exotic trees (Indian rosewood, eucalyptus, cypress), which blocked out light and prevented the growth of desert shrubs. The Bedouin gardens are run on the principles of agroforestry so the smaller orchard trees are widely spaced to allow light to reach the cultivated vegetables and herbs growing beneath them, allowing the growth of native desert shrubs with higher ecological value than agricultural weeds.

Principal component analysis showed that the functional traits of cultivated tree species were clearly separated from other cultivated and wild species along the primary axis, but revealed considerable overlap between all other cultivated and wild species. The convergence of traits in wild and cultivated non-tree species further suggests that the gardens support plants with a similar ecological niche to those in the natural habitat; firstly by providing habitat for wild species, and secondly through the cultivation of plants with similar traits and ecosystem functioning.

Functional richness was significantly higher within the gardens than in the surrounding environment, representing a higher number of unique trait combinations. The cultivation of the wide variety of vegetables, vines and trees brings additional functional richness, above and beyond that seen in the naturally occurring desert shrubs. Many important ecological processes such as biomass accumulation (Tilman et al. 1997; Hector, Schmid, Beierkuhnlein, Caldeira, & Diemer 1999; Reich et al. 2004) and decomposition (Scherer-Lorenzen 2008) have been positively linked with plant functional richness. Though plant abundance was not significantly higher within the gardens, the community weighted trait means showed that plants tended to be considerably taller (1–8 m) than plants in the control plots (0.6 m) suggesting that overall productivity and biomass accumulation is higher within the gardens.

Conservation implications

Our study highlights the promising conservation potential of agroforestry within South Sinai, by showing that agricultural gardens support a more diverse plant community than control plots of natural habitat. In both temperate and tropical environments agricultural conversion often involves deforestation and a decrease in the biomass and complexity of vegetation. The loss of dependent wildlife can be minimised

by diverse planting systems (Perfecto & Snelling 1995; Perfecto, Rice, Greenburg, & van der Voort 1996; Bhagwat, Willis, Birks, & Whittaker 2008), but even the most diverse agroforests will still represent greatly depauperate versions of pristine forests. The situation is very different in an arid environment like Sinai, where the presence of agriculture and the associated rainwater-harvesting techniques are shown here to actively increase plant diversity and average canopy height above those found in unmodified habitat.

The mountain and low-desert gardens have been a fixture of the Sinai landscape for up to one thousand years (Zalat & Gilbert 2008), whereas the gardens in the town of St Katherine represent a recent anthropogenic change to the landscape (∼50 years) (Gilbert 2011). The modern town gardens had just as high abundance and species richness of wild plants as those found in the mountain gardens, showing that the benefits of arid agroforestry can be created within a relatively short timescale.

Creating new gardens has the potential to provide conservation benefits, particularly in the town and low desert where abundances and diversity of wild plants in the surrounding environment are lowest. However, there are several caveats to this. Firstly, the region is extremely water-limited so large scale expansion could put excessive demands on the limited water supply and endanger surrounding plant communities. Secondly, this study is limited in size and more extensive sampling could detect rare species with specialised niches that are not suited to the microhabitat found within the gardens (such as arid specialists). On a similar note, it is inevitable that gardeners do not tolerate all wild species equally and toxic plants such as *P. aurea* and *Gymnocarpos decandrus* are more likely to be excluded than harmless or useful species. Habitat specialists and 'undesirable' species could suffer from the further conversion of natural habitat, so the priority should be in optimising current gardens and preventing the loss of this valuable cultural practice.

The low-desert gardens had the lowest plant species richness, which is consistent with other studies in the region that have shown that plant coverage and species richness increase with altitude (Guenther, Gilbert, Zalat, Salem, & the volunteers of Operation Wallacea in Egypt 2005). The extremely high temperatures in the low desert put a limit on which species can be cultivated, and can explain the high dominance of the date palm, a heat-resilient species that can tolerate temperatures up to 50–60 °C (FAO 1993). Temperatures in the town of St Katherine and the surrounding mountains can be up to 10 °C cooler than those on the coast (Ayyad et al. 2000) and these lower temperatures, associated with increased rainfall, make the mountain and town gardens of St Katherine ideal for cultivating a wider diversity of orchard and vegetable produce.

Livestock were observed grazing on wild plants inside several of the low desert gardens and this is likely to have contributed to the lower abundances and diversity. We recommend the halting of grazing within low desert gardens

and would predict an associated rise in their conservation potential. Traditional practices of Bedouin in the town and mountain differ on the issue of grazing, and while goat and sheep are grazed in the desert and mountains, they are never allowed into the gardens. Some believe that grazing has a negative effect on wild plants (Moustafa, Zaghloul, 'Abd-el Wahab, & Shaker 2001), but a recent study re-evaluating the grazing pressures in the region found that wild plants formed just 2–3% of flocks' diets and that the average grazing time per km^2 was just 33 min per month (Rashad, Abd el Basset, Hemeed, Alqamy, & Wacher 2002; Gilbert 2013). A much higher component of flocks' diets was sourced from plants cultivated within the gardens (*M. sativa*) or from imported supplementary fodder. The fact that grazing is banned inside the walled gardens may have some influence on the higher plant abundances, but at such low overall levels it is unlikely to be the sole explanation and other environmental factors, such as higher water availability, are more likely to explain the trends that we have observed.

Gilbert (2013) discusses a lack of evidence-based management practice within the St Katherines Protectorate and suggests that institutionalised prejudice against the Bedouin people has led to a dismissal of traditional ways of life. This study shows that Bedouin agricultural practices do not have a negative effect on the flora of the region and that the continuation of these indigenous farming practices can actively benefit rare wild plants in the region. On a wider scale, this study supports the view that smallholder farms and home-gardens can be valuable tools in conservation, helping to maintain species diversity and protect underlying ecosystem processes (Altieri, Anderson, & Merrick 1987; Altieri 2004; Kumar & Nair 2004), whilst playing a vital role in food production for the world's poorest people (Tscharntke et al. 2012).

Acknowledgements

Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at http://dx.doi.org/10.1016/j.baae.2013.10.004.

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