The impact of Bedouin agricultural gardens on biodiversity in South Sinai, Egypt

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“Hard as the mountains, soft as the sand, fast as the wind, forever free”

Dedicated to the Bedouin of St Katherine

ABSTRACT

This project assesses the impact of traditional Bedouin agricultural gardens on biodiversity within the St Katherine Protectorate, South Sinai, Egypt. The Bedouin harvest rainwater from intermittent flash floods, allowing them to cultivate a wide range of trees and crops throughout the year. Rainwater harvesting techniques such as these can improve crop yields and enhance food security in arid regions, but this is one of the first studies to address the impact upon dependent wildlife. The results showed that the irrigated gardens support a more diverse plant community than the surrounding unmanaged habitat, providing an abundance of floral resources which in-turn enhance pollinator abundance and species richness. The inclusion of a diversity of culturally important minority crops had a dramatic effect upon the structure of plant-pollinator visitation networks, with cultivated plants supplementing the resources provided by wild flowers. The presence of simultaneously flowering crops also had a positive effect upon pollination services to the primary crop (almond), by attracting higher densities of wild pollinators into the gardens and facilitating enhanced fruit set. The higher abundance of resources within the gardens also had a positive impact upon birds in the region, with gardens supporting higher densities and species richness than the unmanaged habitat. Gardens were particularly important for migratory species, providing an important stop-over for numerous small passerines. In conclusion this study provides evidence that irrigated agriculture in arid environments has the potential to increase biodiversity above that found in the unmanaged environment. The implications on a local scale are that traditional Bedouin practices can have a positive influence on wildlife within the Protectorate, thus initiatives to fund and support gardeners should be encouraged. On a wider scale the results suggest that rainwater harvesting may provide a sustainable mechanism for increasing food security in arid regions, offering a low-cost strategy for increasing agricultural productivity that does not undermine the biodiversity on which it depends.

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Publications

A list of peer-reviewed publications based on the work presented in this thesis:

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Norfolk, O., Eichhorn, M., & Gilbert, F. (2014). Contrasting patterns of turnover between plants, pollinators and their interactions. *Diversity and Distributions.* In press.

Norfolk, O., Eichhorn, M. & Gilbert, F. (2014). Flowering ground vegetation increases wild pollinator densities and enhances fruit set of an orchard crop. *Agriculture, Ecosystems and Environment*, In review.

Norfolk, O., Power, A., Eichhorn, M. & Gilbert, F. (2014) Migratory bird species benefit from traditional agricultural gardens in arid South Sinai. *Journal of Arid Environments.* In revision.

Contents

Chapter 1.

Introduction: *An overview of the study system*

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1.1 St Katherine Protectorate

This research took place in the St Katherine Protectorate (StKP) in South Sinai, Egypt. The StKP was established in 1996 by the Egyptian Environmental Affairs Agency with the goal of ensuring the conservation and sustainable development of the natural and cultural resources of the region (Grainger 2003). Covering an area of 4350 km², it is one of Egypt's largest protected areas, encompassing the entire mountainous region known as the Ring-Dyke Massif (Fig 1.1). The region supports an unusual high-mountain ecosystem and harbours a diverse community of species that are distinct from those in the surrounding lowland deserts.

South Sinai falls along the arid belt of North Africa so is characterised by a Saharo-Mediterranean climate. Within the StKP the high altitude moderates temperatures, which can be 10 °C cooler than those experienced on the coast, but summers are still relatively hot, reaching a mean maximum temperature of 36 °C (August) (Grainger 2003). Winters are cool and the mountains frequently experience snowfall (Grainger & Gilbert 2008). The region is classified as hyper-arid, with mean annual rainfall ranging between 10 mm/year on the coast up to 60 mm/year in the high mountains. The topography of the high mountains mean they receive additional orographic precipitation (generated by a forced upward movement of air upon encountering the physiographic upland). This orographic precipitation is often in the form of snow, and at times this can amount to 300 mm annually (Grainger 2003). Rainfall is sporadic, but usually occurs between October and May. When is does rain, the entire annual rainfall can often fall within the space of a few days and tends to result in heavy flash floods (Cools et al. 2012).

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 floods. The geology of the Ring-Dyke Massif is unusual and complex, with the mountains fundamentally consisting of basement red granitic rocks with intrusions of volcanic rock (Greenwood 1997). The impermeable red granite combines large, flat impenetrable surfaces with deep cracks and crevices. The black volcanic rock crumbles easily and is permeable to water, absorbing it rapidly and allowing it to seep into the crack and dykes within the red granite. In combination this unusual geology facilitates the collection of runoff rainwater in underground pools, providing a semi-permanent water source. This water can be accessed year-round via wells and gives these apparently arid mountains the potential to support permanent agriculture (Perevolotsky 1981).

Flora and fauna

The Sinai Peninsula forms a land-bridge between Africa and Asia and consequentially it supports an unusual combination of flora and fauna from both continental masses. The plant community is particularly diverse and is considered an important centre of plant diversity for the Saharo-Sindian (Irano-Turanian) region of the Middle East (Grainger 2003). The Sinai Peninsula as a whole supports approximately 1285 plant species, 800 of which have been recorded in South Sinai. The high mountains are of particular conservation interest because they have been shown to support a disproportionately high number of rare and endemic species (Ayyad et al. 2000).

The natural fauna of the region is less diverse, with only 25 species of mammal occurring in South Sinai (Hoath 2009). Several species of large carnivores are present within the protectorate, but with the exception of the highly abundant red fox (*Vulpes vulpes*), their conservation status is unknown. The Sinai leopard (*Panthera pardus* *nimr*) is now presumed extinct, but recent camera trapping studies have confirmed that striped hyena (*Hyaena hyaena*) and wolf (*Canis lupus arabs*) are still present in the region (Gecchele, pers. comm.). Nubian ibex (*Capra nubiana*) occur in low numbers throughout the mountains and relict populations of Dorcas gazelles (*Gazella dorcas*) can also be found in isolated populations in the lower deserts (Hoath 2009). The region does support a diverse bird community, with Sinai forming an important migratory corridor for numerous species of passerine, birds of prey and storks (Frumkin et al. 1995). The high mountains also support an interesting and diverse insect community and a higher number of butterfly species than the rest of Egypt (Gilbert & Zalat 2008), including two endemic species, the Sinai Hairstreak (*Satyrium jebelia*) and the Sinai Baton Blue (*Pseudophilotes sinaicus*) (James 2006).

Cultural heritage

The region has an extremely rich and diverse religious history and the St Katherine area was designated a UNESCO World Heritage site in 2004 (UNESCO, 2014). Of particular cultural significance is the St Katherine Monastery, which was founded in the 6th Century BCE making it the oldest monastery still used for its initial function (Forsyth 1968). The mountains surrounding the monastery also have strong religious ties and Jebel Musa (Mount Moses) is considered to be the biblical Mount Sinai where Moses encountered the burning bush and received the Ten Commandments from God (Forsyth 1968). The mountains have religious significance for Christianity, Judaism and Islam, and Jebel Musa regularly receives pilgrims from multiple faiths. In combination the mountain and monastery provide an important influx of tourists and income to the region.

There are four main tribes of Bedouin people that inhabit the region in and around the protectorate (Fig 1.1), the Jebeliya, Awlaad Sa’iid, Gararsha and Muzayna tribes (Hobbs 1996). The most populous tribe, the Jebeliya inhabit the area directly surrounding St Katherine and have strong historical links with the monastery. Some believe they are the direct decedents of Macedonian guards, who were sent to Sinai from Eastern Europe by Emperor Justinian in 527 A.D with the specific role of protecting the monastery (Grainger & Gilbert 2008). Now converted to Islam, the Jebeliya maintain a close relationship with the monks, and the historical connections have led to Jebeliya being granted exclusive rights to guide pilgrims and tourists around the mountains surrounding Jebel Musa and to work as labourers within the monastery gardens.

In addition to their favourable relationship with the monastery, the Jebeliya also inhabit the most resource-rich territory of South Sinai, making them the most prosperous tribe. The topologically diverse region provides higher water availability than the surrounding desert, which allows them to practice pastoralism and orchard agriculture in addition to working for wages in the monastery and with tourists (Grainger 2003).

In contrast, the neighbouring Awlaad Sa’iid tribe inhabits a more extensive, but resource-poor territory in the west. They are primarily pastoralists and are one of the poorest tribes in the region. The Gararsha live to the north of the Protectorate, in and around the oasis of Wadi Feiran. The oasis offers some opportunity for agriculture and is famous for its date palms, but the tribe receive very little of the tourist revenue enjoyed by the Jebeliya, which can lead to tensions. The Muzayna inhabit the coastal regions along southeastern Sinai and enjoy the benefit of tourists visiting coastal resorts in addition to traditional practices of fishing and pastoralism (Hobbs 1996).

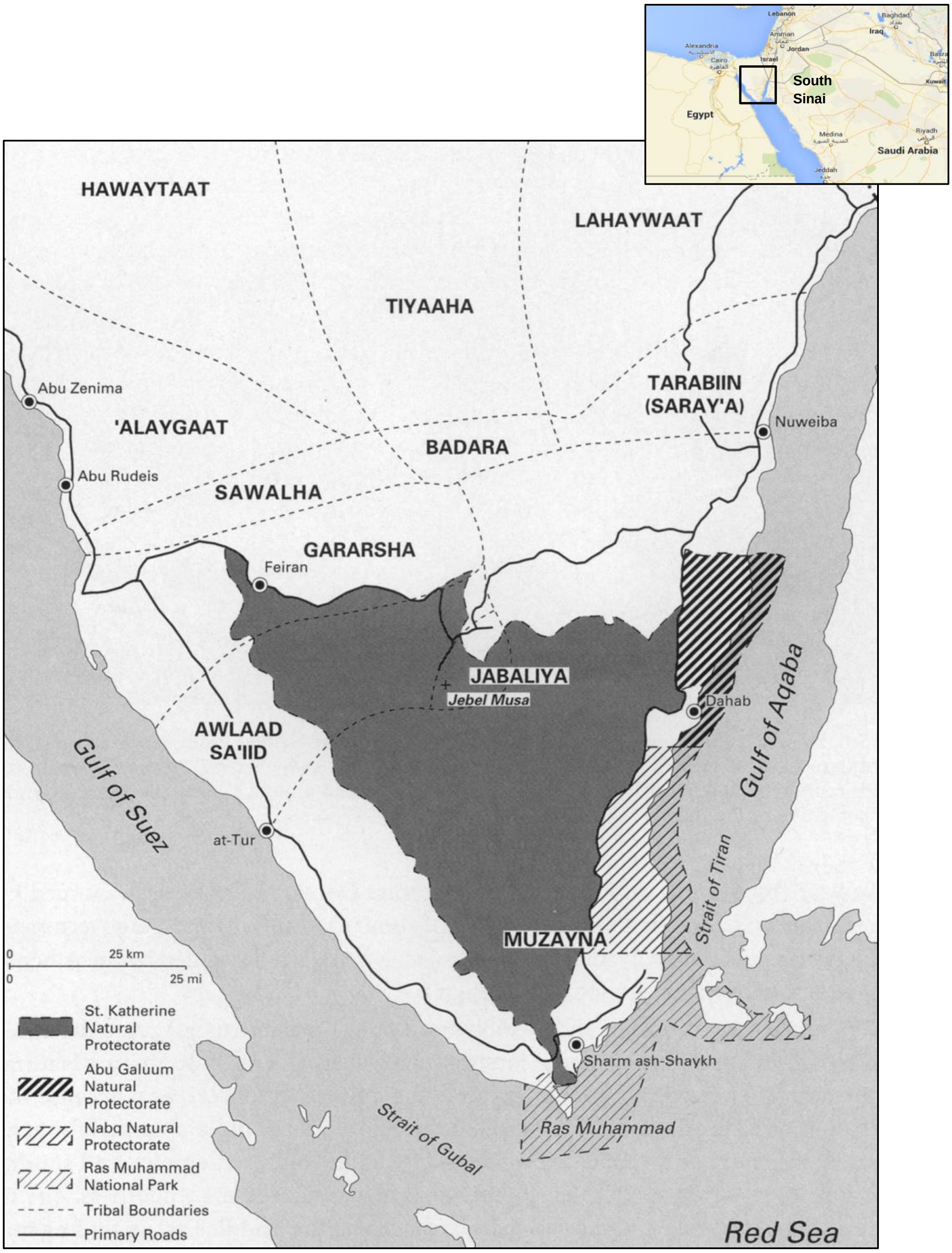

Figure 1.1. The boundaries of Natural Protectorates and Ras Mohammed National Park, with associated tribal territories. St Katherine Protectorate is shown in dark grey. Cartography by Adrianne Nold, Missouri Geographic Resouces Centre. (taken from: Hobbs 1996).

1.2 Jebeliya gardens

The Jebilaya are the only tribe that extensively practice orchard agriculture. This is partly due to their well-placed territory within the cooler mountains, but also from their mutualistic relationship with the monks of St Katherine. The mountain gardens were probably introduced to the region by hermits settling in the early Byzantine era, and the tradition was invigorated by the monks of the Monastery, who brought Greek fruit and olive trees along with knowledge of grafting techniques (Zalat et al. 2001; Grainger & Gilbert 2008).

The Jebeliya traditionally practised a semi-nomadic system of transhumance; in winter they would be pastoralists in the lower mountains, while in the early summer they would move their households into the higher, cooler mountains to tend to and harvest their mountain gardens. This lifestyle was established over one thousand years ago, but is still very much apparent in the modern-day culture. Though most people have settled in the ever-expanding town, hundreds of gardens are still maintained within the protectorate and many people still keep small herds of goats and sheep. Paid work and proximity to the school tie people to the town for the majority of the year, but many families continue the practice of transhumance and move to gardens in the high mountains for several months during the school summer holidays.

The walled gardens are a distinctive feature of the high mountains (Fig 1.2). From satellite photographs I have estimated that there are approximately 500 gardens within the StKP (Fig 1.3). Some of the larger gardens are still owned by the monastery and are tended by Bedouin in return for half the produce, but the majority are Bedouin owned.

Figure 1.2. Walled gardens form a distinctive feature of the landscape within the St Katherine Protectorate.

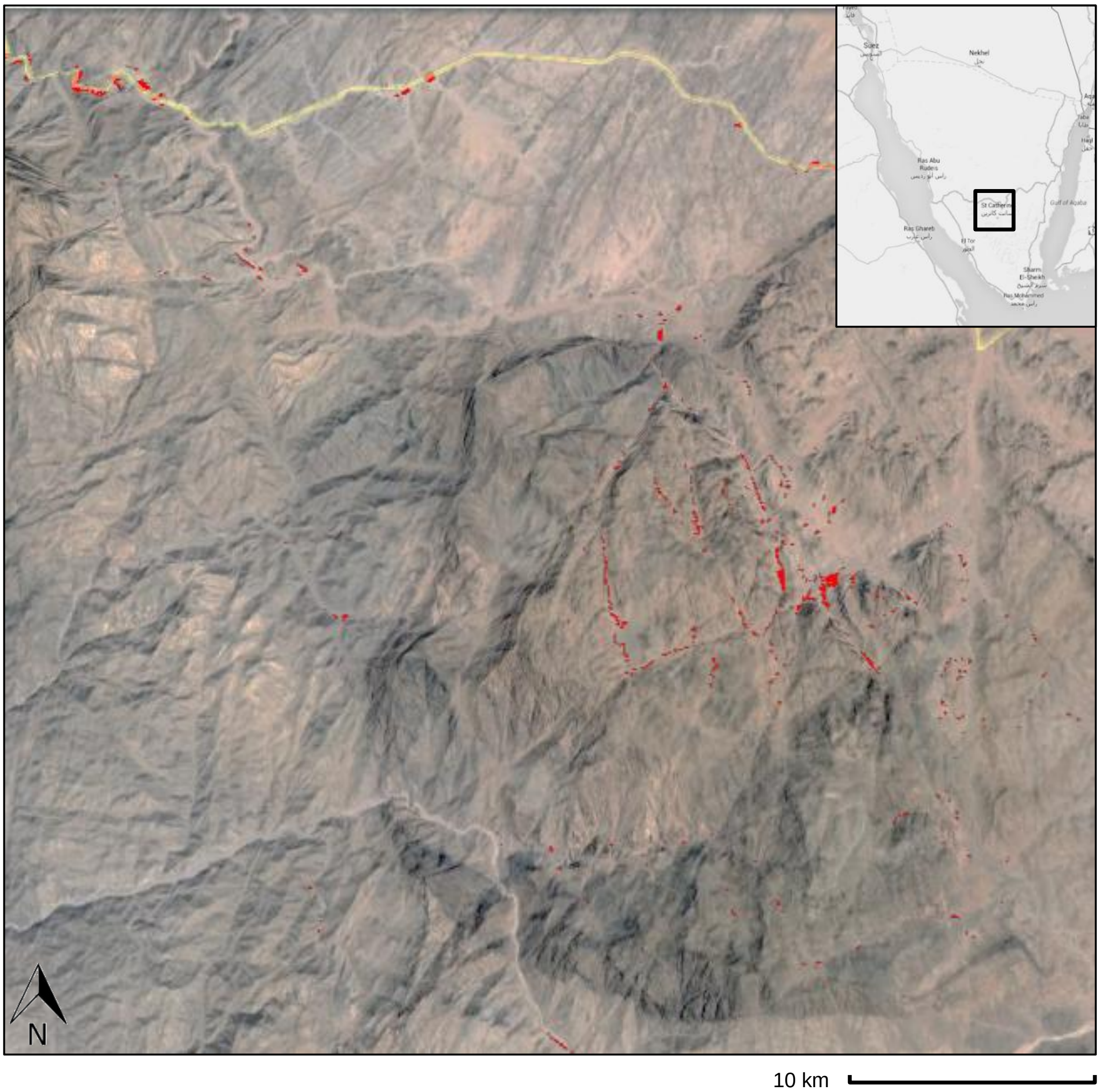

Figure 1.3. Satellite image of gardens in and around the St Katherine Protectorate. Gardens are highlighted in red.

The garden walls keep out grazing animals, help to retain soil and act as dams to capture rainwater from the flash floods. By making the most of water harvested from these floods, the Jebilaya are able to cultivate a wide variety of crops throughout the year. Fruit trees form the bulk of garden produce, but orchard trees are under-cropped with a high variety of crops, vegetables and herbs (Zalat & Gilbert 2008). Previous surveys have recorded a total of 39 species of cultivated tree and plant growing within the gardens (Zalat et al. 2001), the most common of which are listed in Table 1.1. The gardens are organic since the Bedouin avoid all agro-chemicals and only apply goat manure as fertiliser. Wild plants are generally tolerated within the gardens, with farmers weeding only the immediate vicinity of their crops. Thus the gardens can provide habitat for a wide variety of native flora, with 104 species of wild plants recorded growing alongside the cultivated crops (Zalat et al. 2001).

Table 1.1. Most abundant crop species in the gardens. Data taken from Zalat et al. (2001) from surveys of 71 gardens in the StKP.

Crop	Proportion of gardens containing crop (%)
Almond	94
Grape	93
Fig	77
Apricot	77
Pomegranate	74
Apple	66
Olive	66
Jujube	36
Quince	30
Tomato	30
Peach	27
Walnut	26
Bean	26
Plum	24
Pear	24
Carob	21
Aubergine	20

Motivations for keeping the gardens may differ between families, but in recent interviews with thirty garden-owners (Dunne 2012), tradition was the most popularly cited motivation for keeping the gardens (50%). Despite many Bedouin citing cultural rather than economic motivations, the gardens do provide a substantial contribution to many families food needs (Fig 1.4).

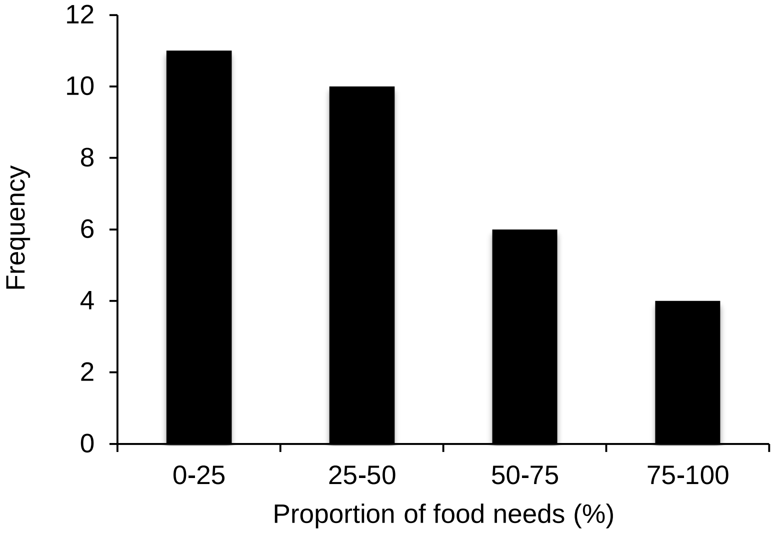

Figure 1.4 Proportion of families whose total food needs are met by garden produce. Data taken from Dunne (2012).

Anthropologist Emanuel Marx (1999) believes that there is a more complex strategy behind the maintenance of the gardens, and that both the gardens and goat herds are retained as a back-up economic strategy by the Jebilaya. When paid work is abundant Bedouin continue to cultivate the gardens and raise flocks, which can become alternative sources of livelihood if times get tough. He observed that when the going is good, Bedouin invest limited amounts of time into garden and keep small numbers of animals, maintaining their alternative economy at idling speed. When things become difficult, herd sizes can be increased rapidly and cultivation of the gardens can be intensified. My own personal observations after the Egyptian Revolution of 2011 are consistent with his theory. The political troubles led to a crash in the tourist industry, which created mass unemployment in St Katherine. There has been a dramatic increase in the size of goat herds and many men have begun to invest more time on their gardens. When times became difficult, it became much clearer how essential these traditional methods of subsistence are for the Jebilaya. In times of plenty the gardens can appear as recreational and maintained purely for tradition, but observations over the last six years suggest that they continue to provide an essential fall-back for a community that faces an extremely unstable and insecure economy.

1.3 Rainwater harvesting

Successful agriculture in arid regions depends on careful management of the scarce water resources. One way to maximise productivity in harsh arid environments is to utilise the runoff water by rainwater harvesting. Runoff agriculture and rainwater harvesting can be used as synonymous terms and are defined broadly as “farming in dry regions by means of runoff rainwater from whatever type of catchment or ephemeral stream” (Bruins et al. 1986). This can encompass a wide form of techniques, such as terracing, ditches, micro-catchments and dams, and though broad in method these techniques all share some key characteristics that are distinctive to runoff agriculture (Boers & Ben-Asher 1982):

1) They occur in arid regions where surface runoff occurs as a discrete event often followed by long dry periods. This ephemerality of water availability means that water storage is an integral part of all rainwater-harvesting practices.

2) They depend upon local water such as surface runoff, creek flow or springs, which does not include the storage of river water in large reservoirs or the mining of groundwater.

3) As a consequence of the first two features, they tend to be small-scale operations in terms of catchment area and volume of storage.

The geology of the Sinai facilitates natural rainwater harvesting, with the wadis funnelling water into underground pools and dykes. The Bedouin take advantage of the beneficial geology by strategically positioning gardens at points where permeable black volcanic rock intersects with the red granite; this tends to coincide with underground water sources that can be accessed via shallow wells. The design of the gardens further enhances the natural runoff, with many gardens built in terraces (Fig 1.5a), with stone walls acting as dams to slow the water and give it time to soak into the underground pools. Additionally, within the walled gardens, many Bedouin build low sand dams around trees and flowerbeds (Fig 1.5b) to retain further runoff from the floods and to facilitate effective irrigation.

A recent charity-funded initiative has also seen an increase in the number of dams that span the width of the wadis (Fig 1.5c and d). These dams can capture large quantities of water, which with time will seep into the underground pools and re-fill the wells of all downstream gardens. A comprehensive understanding of how to manage water is evident in the design of the gardens, wells and piping, and this inherited knowledge seems to be essential for maintaining productive agriculture in this region.

Figure 1.5**.** Examples of rainwater harvesting techniques used in South Sinai; a) terraces, b) sand ditches around crops, and dams c) immediately after heavy rain and d) several months after heavy rain.

Social and ecological benefits of rainwater harvesting

Rainwater harvesting can bring many social benefits to people living in harsh environments and many believe it provides an important and underexplored option for increasing food production in arid environments (Helmreich & Horn 2009; Vohland & Barry 2009; Barrow 2014). Though it requires a detailed understanding of the environmental conditions of a region (topology, soil type, rainfall patterns), the techniques themselves are simple and easily learnt and tend to make use of cheap and locally available materials such as stone walls and sand ditches (Barrow 2014). Once the knowledge is in place, rainwater harvesting techniques can spread with little risk of dependency or need for outside assistance, so it seems to offer an exciting opportunity for tackling food security in arid regions.

By maximising limited water resources, rainwater harvesting techniques can increase security of harvest and provide new opportunities for crop diversification. In arid regions of India, ridges and furrows around trees lead to a significant increase in biomass accumulation and root mass of three cultivated tree species (Gupta 1995), and lined micro-catchments have also been shown to enhance seedling recruitment in Jujube plantations (Ojasvi et al. 1999). In arid regions of China, ridges and furrows are associated with increased recruitment and yields of potatoes and corn (Tian et al. 2003; Xiaolong et al. 2008), and artificial irrigation experiments show that they are most effective at enhancing yields in low rainfall conditions (Xiaolong et al. 2008). A meta-analysis of rainwater harvesting practices in the drylands of Africa also found a positive impact upon yield and food security, with the majority of studies focussing on cereal crops such as sorghum and maize (Vohland & Barry 2009).

Rainwater harvesting practices can also enhance soil quality and help to reduce soil erosion and degradation. In the highlands of Saudi Arabia, terraces significantly reduced the amount of runoff and soil loss, leading to an increase in the density and recruitment of cultivated Juniper trees (Atta & Aref 2010). In southeastern Tunisia, micro-catchments experience a build-up of sediment after rain events which increases the quality of the soil (Schiettecatte et al. 2005) and this sediment trapping combined with increased soil moisture can increase the rate of organic matter degradation and nutrient release (Zougmoré et al. 2003; Fatondji et al. 2009). The stone walls and terraces of the Bedouin gardens in Sinai are likely to have similar positive effects upon soil quality, and soil within the gardens has been shown to contain significantly higher concentrations of nitrogen, phosphorus and organic carbon than that in the surrounding mountains (Norfolk 2010).

The positive impacts of rainwater harvesting has been well established regarding soil quality and biomass accumulation, but there is a poor understanding of its impact upon dependent wildlife (Vohland & Barry 2009). Initial work in Sinai suggested that the irrigated gardens can actively increase the abundance and diversity of ground arthropods (Norfolk et al. 2012) and flower visitors (Norfolk & Gilbert 2014), suggesting that the benefits of rainwater harvesting may extend from increased yields into biodiversity conservation. The aim of this project is to establish a fuller understanding of how Bedouin gardens impact upon patterns of diversity within the St Katherine Protectorate, and to increase our understanding of how rainwater harvesting affects dependent wildlife.

1.4 Thesis outline

The overall aim of this body of work is to understand how the traditional Bedouin gardens affect biodiversity within the St Katherine Protectorate and to investigate how these patterns diversity vary temporally, spatially and between groups of organisms. The following gives an outline of the thesis structure, with a brief survey of the objectives covered in each chapter:

- Chapter 2 assesses the impact of gardens on plant communities in the region and compares three styles of agroforestry systems across the whole of South Sinai: those practised by the Jebeliya in the (a) high mountains and (b) town of St Katherine; and (c) those practised by the Muzayna and Garasha tribes in the low desert regions.
- Chapter 3 investigates how the modification of the plant community within the gardens impacts interacting pollinators, and specifically addresses how the presence of the gardens impacts upon the availability of floral resources and the abundance and diversity of pollinators.
- Chapter 4 focusses on disentangling the patterns of diversity exhibited by two interacting groups of organisms, plants and pollinators. In this chapter, plant-pollinator interactions are used to test two conceptual models for explaining patterns of beta diversity across the landscape. The aim is to determine whether the local enhancement of habitat within the gardens (a) leads to an increase in habitat heterogeneity (by supporting novel species); or (b) increases densities of the same species found in the unmanaged habitat. It also asks whether plants and pollinators exhibit contrasting responses.
- Chapter 5 investigates temporal changes in the relative importance of cultivated and wild flora for pollinators within the gardens. This chapter utilises a visitation network approach to assess how plant-pollinator

interactions differ across the season and between years, and assesses whether cultivated flowers are widely utilised by pollinators.

- Chapter 6 builds on the previous chapter to assess whether enhancement of the pollinator communities within the gardens has a positive effect on pollination services to crops. This chapter focusses on the primary crop almond, and asks whether the diversity of flower visitors within the gardens is positively linked to visitation rates and fruit set of the trees.
- Chapter 7 investigates whether the higher availability of plant and insect resources within the gardens influences the distribution of birds. It compares the diversity and functional richness of bird communities within gardens and unmanaged habitat, and assesses the importance of the gardens for migratory bird species.
- Finally, Chapter 8 brings together all of the research described in the previous chapters, discussing general trends and conclusions in order to evaluate the overall impact that the Bedouin gardens have upon biodiversity within the St Katherine Protectorate.

Chapter 2.

Plant diversity and functional richness[*]:

How do the gardens impact upon plant communities in the region?

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[*] A modified version of this chapter was published in *Basic and Applied Ecology:*
Norfolk, O., Eichhorn, M. & Gilbert, F. (2013). Traditional agricultural gardens conserve wild plants and functional richness in arid South Sinai. *Basic and Applied Ecology*, 14, 659-669.

Chapter 2. Plant diversity and functional richness: *How do the gardens impact upon plant communities in the region?*

Abstract

Maintaining agricultural diversity is important for the conservation of rare species and for preserving underlying ecosystem processes upon 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 chapter assesses the impact of three Bedouin agroforestry systems on plant diversity and functional richness: 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 unmanaged 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 unmanaged 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, desert shrubs were frequently found growing between the trees and community weighted trait means (CWM) 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 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.

2.1 Introduction

Intensive agriculture is notorious for its negative impacts on diversity with the simplification of complex ecosystems into monocultures leading to inevitable species erosion (Le Féon et al. 2010; Prober & Smith 2009; Robinson & Sutherland 2002; Tscharntke et al. 2005). 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 (Calvet-Mir et al. 2012; Jackson et al. 2007; Sandhu et al. 2010), such as the provision of ecosystem services that maintain pest control (Trujillo-Arriaga & Altieri 1990), soil fertility (Munyanziza et al. 1997), water retention (Roose & Ndayizigiye 1997) and pollination (Klein et al. 2003; Jha & Vandermeer 2010).

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 et al. 2005; Beukema 2007; Clough et al. 2009) and mammals (Nyhus & Tilson 2004). The conservation potential of arid agro-ecoystems 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.

The rainwater-harvesting techniques utilised by the Bedouin of South Sinai means that the gardens have a higher potential for plant growth than the external environment, with gardens appearing 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. In this chapter I consider three different agroforestry systems in South Sinai: 1) the traditional Jebeliya 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 chapter 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 et al. 2000). In this chapter I assess whether the gardens (a) help to protect the rare wild plants within the StKP and (b) whether they support plants with similar ecosystem function as those in the surrounding unmanaged habitat. I 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 (Diaz & Cabido 2001; McGill et al. 2006) and a trait-based approach allowed me to assess whether these gardens are supporting plants with a similar ecological niche as those in the surrounding habitat, whilst giving insight into the underlining ecosystem processes within the gardens.

2.2 Methods

Study sites and sampling methods

Plant surveys were conducted during April and May 2012 at the peak of the flowering season (Danin 2012). 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 ca.1800 m asl and Wadi Itlah ca. 1500 m asl), ten from within the boundary of the town of St Katherine (St Katherine village and Raha, ca. 1500 m asl) and ten from much lower altitudes just outside the high Ring Dyke region (Sheikh el-wad, ca. 1100 m asl), and westwards (Wadi Feiran ca. 700 m asl) and eastwards (Ein Hodra Oasis, ca. 700 m asl) towards the base of the mountain massif (collectively called here the ‘low-desert gardens’) (Fig 2.1). From satellite imaging we have estimated that there are between 500 - 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 Figure 2.2. The co-ordinates of the specific gardens sampled is included as supplementary material (Appendix 2.1: Table 2.1).

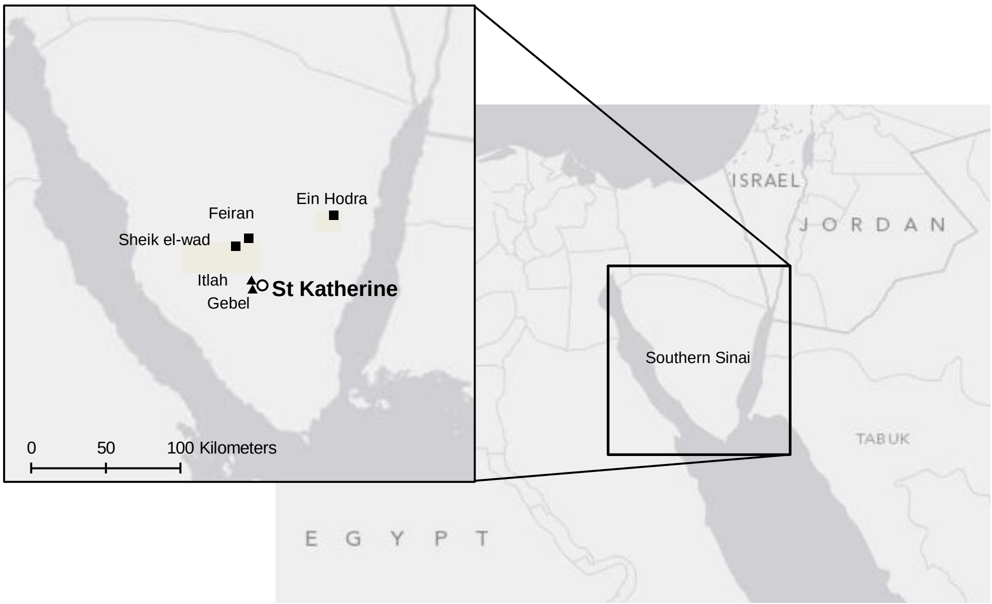

Figure 2.1. Map of study sites in South Sinai, Egypt, showing St Katherine town (circles), mountain sites (triangles) and low desert sites (squares).

A) Mountain garden

B) Town garden

C) Low desert garden

Figure 2.2. Photographs depicting typical gardens from the three agroforestry systems.

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 2.2: Fig 2.1) it was simply not feasible to find enough remaining natural habitat. In previous work control plots were randomly placed in the vicinity of each garden (Norfolk et al. 2013), but I noted that 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 the present study, plant abundance was not significantly different between gardens and control plots (lmer: $\chi^2 = 1.65$, df=1, $P = 0.198$) so I believe it is an improvement on previous methods.

Plants were recorded 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. Transects were positioned diagonally in order to prevent over sampling 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 that was actively tended be it 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 2.1). 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 et al. 2008) which descibes plant traits that relate to key feautures of plant dynamics, such as persistence and regeneration. See LEDA (2012) for more information.

Statistical analyses

Hill’s numbers (species richness [^0D], the exponential of Shannon entropy [^1D] and the inverse Simpson concentration [^2D]) (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 (^qD), whereby parameter q indicates the weight given towards rare or common species. ^0D (species richness) is insensitive to relative frequencies, and is therefore weighted towards rare species. ^1D (exponential of Shannon entropy) is weighted towards common species, and ^2D (inverse Simpson concentration) is weighted towards abundant species. Diversity measures were calculated in SPADE (Chao & Shen 2010). Species richness (^0D) was estimated using *Chao1-bc*, a bias-corrected form of Chao1 (Chao 2005). ^1D and ^2D 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 (^0D,^1D, ^2D) 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, 2013). *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 among the three sites. Model simplifications followed Zuur et al. (2009). The size (m^2) of each garden was considered as a potential confounding variable, but linear mixed-effect models showed that there were no significant relationships between area and plant abundance (χ^2_1= 0.15, P = 0.697), ^0D (χ^2_1= 0.51, P = 0.477), ^1D (χ^2_1= 0.01, P = 0.917) or ^2D (χ^2_1= 0.04, P = 0.849) so area was not included in the 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 Tukeys 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 et al. 2005). Functional richness was calculated using the *dbFD* function in the *FD* package (Laliberté and 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 Cailliez correction was applied (Cailliez 1983). Community-level weighted means of trait values (CWM) were calculated using the *functcomp* function in *FD*, where-by 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.

Table 2.1. 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

2.3. Results

Plant abundance and diversity

In total 5112 plants were recorded, belonging to 96 species from 36 families (for species descriptions see Appendix 2.3: Table 2.2). There was no significant difference between the total abundance of plants within the gardens and their associated control plots (lmer: χ^2= 1.65, df=1, $P = 0.198$), nor across the three agroforestry systems (χ^2= 5.12, df=2, $P = 0.077$), although town gardens had the highest overall abundance (Fig 2.3a).

All three Hill-number measures of diversity were significantly higher within the gardens than in control plots (^{0}D: χ^2= 30.83, df=1, $P < 0.001$; ^{1}D: χ^2= 7.13, df=1, $P = 0.008$; ^{2}D: χ^2= 27.95, df=1, $P < 0.001$, Fig. 3B-D). Species richness (^{0}D) differed significantly among the three agroforestry systems (χ^2= 11.38, df=2, $P = 0.004$) and was highest within the high mountains, but ^{1}D (χ^2= 4.88, df=2, $P = 0.087$) and ^{2}D (χ^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 (χ^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 (χ^2_1= 8.45, $P = 0.004$), but ^{2}D was not significantly different (χ^2_1= 1.93, $P = 0.164$). This implies a greater number of scarce and moderately common species but a similar number of dominants.

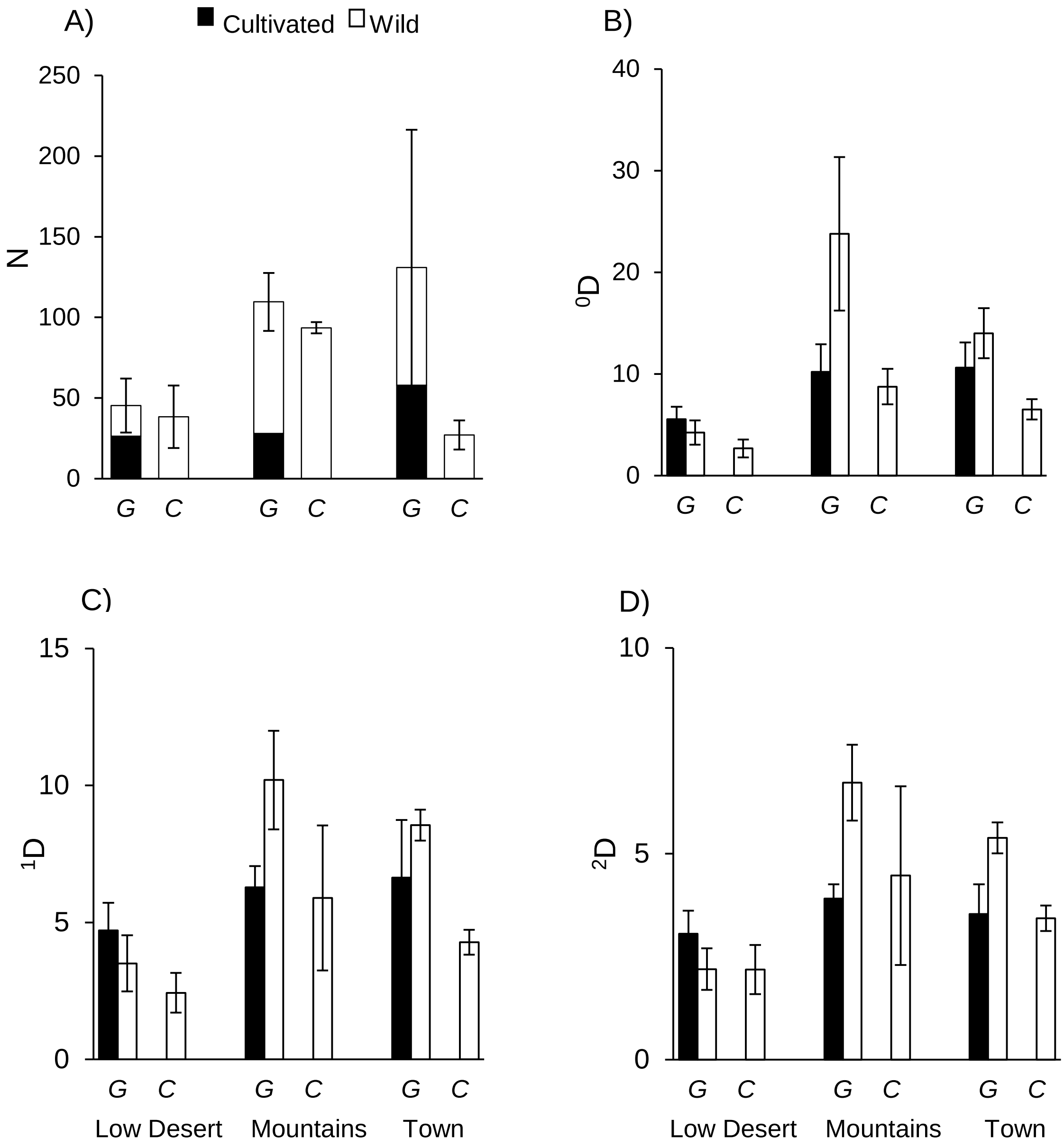

Figure 2.3. Mean plant a) abundance, b) ^{0}D, species richness, c) ^{1}D, exponential of Shannon entropy, and (D) ^{2}D, inverse Simpson per garden or control plot (200 m²). Error bars represent the standard errors of the mean and in (A) this is for all plants (cultivated and wild combined). G= gardens, C=control plots.

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

Functional trait analyses

Principal components analysis of the eight functional traits explained 66% of the variance among plant species along its two main axes (Fig 2.4). The first principal component accounted for 46% of the 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 axis1 (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).

The community weighted means (CWM) of trait values (Table 2.2) show that all three sites were dominated by perennial chaemaphyte sub-shrubs in both the gardens and the control plots. Although numerically trees were not the dominant growth form within the gardens, they did have 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 2.5; *lmer:* χ^2= 15.12, df=1, *P* <0.001), as was the functional richness of wild species alone (χ^2= 4.73, df=1, *P* = 0.029). Total functional richness also differed significantly between the three agroforestry systems (χ^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.

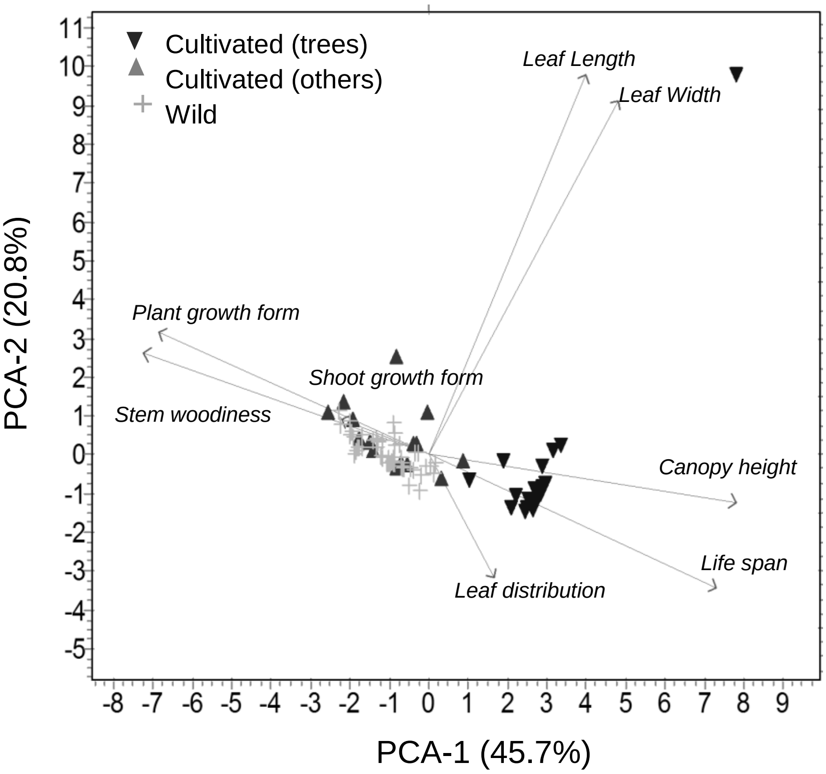

Figure 2.4. Standardized Principal Components Analysis (PCA; first vs. second axes) of cultivated and wild species characterized by eight plant traits. For list of trait variables see Table 1.

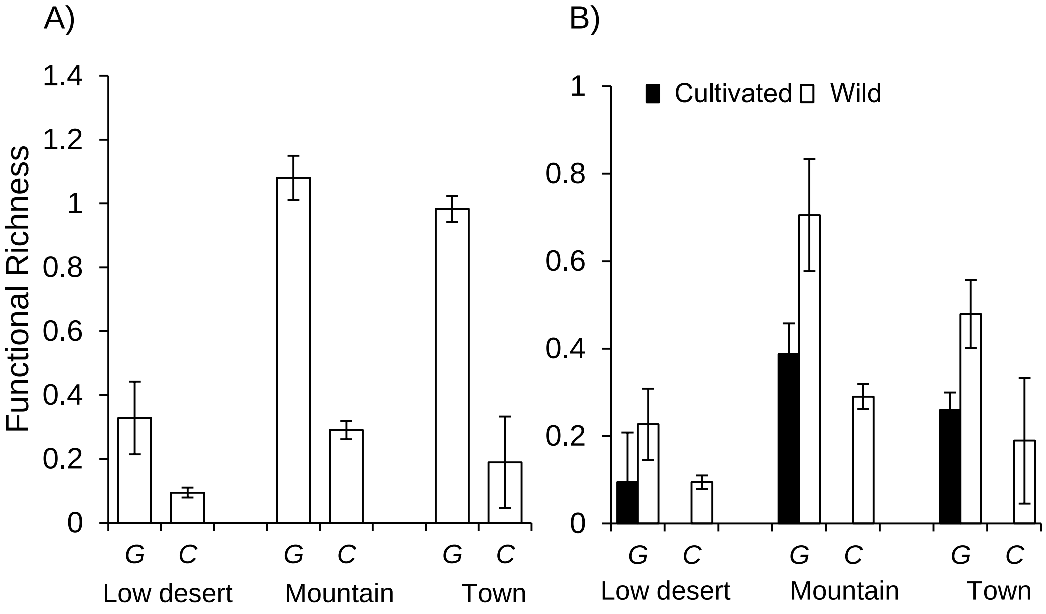

Figure 2.5. Mean functional richness per garden or control plot (200m^2) for a) all plants, and b) cultivated and wild plants separately. G= gardens, C=control plots.

Table 2.2. 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 <10yrs	Perennials <10yrs	Perennials <10yrs	Perennials <10yrs	Perennials <10yrs	Perennials <10yrs
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

2.4. Discussion

This chapter shows that traditional agricultural gardens in South Sinai maintain 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 et al. 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 (${}^{0}D$,${}^{1}D$, ${}^{2}D$), suggesting that effective numbers of 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 largest diversity increase was seen in wild plant species richness, suggesting that scarce wild plants were the most positively influenced and demonstrating the gardens' role 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 are supporting 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 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 are supporting 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 functions.

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 et al. 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

These results highlight 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 et al. 1996; Bhagwat et al. 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 *Phlomis 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 et al. 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 50-60 ºC (FAO 1993). Temperatures in the town of St Katherine and the surrounding mountains can be up to 10 ºC 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 the several of the low desert gardens and this is likely to have contributed to the lower abundances and diversity. We would 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, 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 minutes per month (Rashad et al. 2002; Gilbert 2013). A much higher component of flocks’ diets was sourced from plants cultivated within the gardens (*Medicago 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 chapter 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 work supports the view that smallholder farms and homegardens can be valuable tools in conservation, helping to maintain species diversity and protect underlying ecosystem processes (Altieri et al. 1987; Altieri 2004; Kumar & Nair 2004), whilst playing a vital role in food production for the world’s poorest people (Tscharntke et al. 2012).

Chapter 3.

Characterising the pollinator community:

What effect do gardens have upon pollinator abundance and species richness?

Chapter 3. Characterising the pollinator community: *What effect do gardens have upon pollinator abundance and species richness?*

Abstract

In tropical and temperate environments, agriculture and anthropogenic land use change typically have negative impacts upon pollinator communities. There is much less research on how pollinators respond to land use change in arid regions, but the irrigation that is typically associated with arid agriculture means that farms can provide higher levels of floral resources than unmanaged desert habitat, which may actively benefit pollinators. In this chapter I use flower visitor data collected from eight months of field work in 2012 and 2013 to test how the presence of the Bedouin gardens affects flower visitors. The results contain an up-to-date species list of the pollinators found within the St Katherine Protectorate, including five bee species new for Egypt and one bee species previously undescribed. Pollinator abundance and species richness were significantly higher within the gardens than in plots of unmanaged habitat, directly attributable to the higher levels of floral resources within the gardens. The impact of the gardens differed across an altitudinal gradient; at low altitudes gardens contained much higher abundance and species richness of pollinators than the unmanaged habitat, but a higher altitudes (>1800 m) levels were equal in the two habitat types. The results show that St Katherine Protectorate supports a diverse pollinator community (that includes several rare and endemic species of bee) and that the irrigated Bedouin gardens have a positive influence on the abundance and diversity of these pollinators. Gardens were particularly beneficial at lower altitudes where natural floral resources were scarcer. On a wider scale, these results demonstrate that irrigated agriculture has the potential actively to benefit pollinators and that conclusions gathered from temperate and tropical regions cannot be directly extrapolated to arid regions.

3.1 Introduction

Pollinators provide an essential ecosystem service, helping to maintain the yields of approximately 75% of crop species and 94% of wild flowering plants (Klein et al. 2007). Worldwide these services have been valued at US$215 billion (Gallai et al. 2009), but despite their extremely high ecological and monetary value, pollinators have been experiencing widespread declines. There are thought to be multiple drivers behind these declines (Potts et al. 2010); land-use change and loss of natural habitat tend to lead to a reduction in pollinator abundance and species richness (Winfree et al. 2009) perhaps due to lower availability of floral and nesting resources, and these issues may be further compounded by additional pressures such as disease (Cox-Foster et al. 2007), improper use of pesticides (Brittain et al. 2010) and climate change (Hickling et al. 2006; Dormann et al. 2008).

Agricultural conversion of natural habitat is typically associated with a decrease in pollinator diversity (Ferreira et al. 2013), with crop visitation rates and pollinator richness declining exponentially with distance from natural habitat (Ricketts et al. 2008). These negative impacts of agriculture are minimised in diversified or organic systems and in landscapes that contain higher proportions of semi-natural habitat (Kennedy et al. 2013), but the majority of pollinator work to date comes from temperate and tropical environments (Mayer et al. 2011), with a distinct lack of research in arid regions such as North Africa and the Middle East (Archer et al. 2014). In resource-poor regions, irrigated agriculture has the potential to boost floral resources above those found in the natural habitat, so trends and conclusions drawn from tropical and temperate regions should be extrapolated with care. Indeed initial impressions suggest that irrigation in arid region can actively benefit desert pollinators, with previous work in South Sinai showing that wild plants within agricultural gardens receive elevated rates of floral visitation (Norfolk & Gilbert

2014) and work in Israel showing that irrigated ornamental gardens contain higher abundances of wild bees than the natural desert habitat (Gotlieb et al. 2011).

This chapter outlines the methods and protocols used for collecting the plant-pollinator data that form the bulk of this thesis. It then addresses the question of how the irrigated Bedouin gardens influence the abundance and diversity of local pollinator communities. Without human interference the rainwater from flash floods is quickly lost from the system, as the impenetrable red granite is unable to retain water. However by digging wells the Bedouin are able to access the water in underground pools and bring it back to the surface, giving these gardens a higher potential for plant growth than the unmanaged desert habitat (Norfolk et al. 2013). This chapter addresses two main hypothesises: as a result of rainwater-fed irrigation, gardens will (1) contain higher levels of floral resources than the unmanaged habitat; and (2) be able to support a higher abundance and species richness of pollinators. Pollinator communities were compared between the gardens and unmanaged habitat using a species-based approach, with mixed-effect models used to assess which environmental variables had the strongest effect upon pollinator abundance and species richness. The results showed that the gardens had a positive impact upon pollinator communities due to the increased availability of floral resources.

3.2 Methods

Monthly flower visitor surveys were carried out from April to July in 2012 and April to July 2013. Average monthly daytime temperatures in the region range from 22°C in April, 28°C in May, up to 32°C in June and July (RP5 2013). No rain was recorded during either field season, but there were heavy floods in February 2013 meaning water availability was higher in the second year (personal obs.), leading us to classify 2012 as a pre-flood year and 2013 as a post-flood year. Prior to these floods there had been no rain since spring 2011.

Flower visitor sampling protocol

Four monthly flower visitor surveys were carried out in each garden and unmanaged plot from April - July in 2012 and April – July 2013. In each garden and control plot five 10 x 10 m² quadrats were measured out for repeat surveys across the season. Quadrats were placed contiguously, with the first quadrat randomly placed at a point along the garden wall and others towards the centre of the garden, giving a total survey area of 500 m^2 per garden. Surveys were always carried out during sunny, non-windy days between 9am and 4pm. During sampling, a single collector walked at a steady pace around each 10 x 10 m^2 quadrat, searching each flowering plant in turn and recording all flower visiting insects. If there were no flower visitors they continued the walk and moved on to the next plant. All observed flower visiting insects were net-collected directly from the plants, unless confident identification was possible in the field (honeybees and distinctive butterflies). A visit was determined as any insect observed in contact with the stamen or stigma of a flower; resting upon the petal was not sufficient. The identity of each visited plant species was also recorded in order to establish the flower-insect interaction. When multiple visitors were observed simultaneously on one plant, no more than five minutes (excluding handling time) was spent catching insects from that particular plant.

Plants were identified in the field where possible or collected for identification using Boulos (199-2005). All plants were classified as either wild or cultivated, with cultivated defined as any plant actively tended for consumption, household use or ornamental purposes. All captured insects were pinned and identified to species level for orders Hymenoptera and Lepidoptera and family Syrphidae by taxonomists. Coleoptera and non-syrphid Diptera were identified to family level and have been grouped into morphospecies based on visual characteristics for analyses. Capture rates were 92 % of observed insects; visitors that evaded capture were excluded from further analyses since species-level identification was not possible. Ants and pollen beetles are not generally considered pollinators so were excluded from analyses. Throughout this thesis I refer to the remaining flower-visiting insects as pollinators, in reference to their functional role as pollinators. However it should be noted that flower visitation cannot be considered a quantitative proxy for pollination, because species and individuals can differ considerably in their pollination efficiency.

In 2013 floral abundance and floral species richness were recorded in the gardens and unmanaged plots. Floral abundance per garden or unmanaged plot was calculated as the total number of fresh flowers (i.e. petals and anthers intact and not dried) in the five quadrats. For plants with clustered, umbelled or spiked flower arrangements we counted the number of inflorescences rather than the number of single flowers; the average number of flowers per inflorescence was then calculated from three flower heads in the field, with floral abundance equal to the total number of inflorescences multiplied by the average number of flowers per inflorescence.

Site selection

In 2012 monthly flower visitors surveys were conducted in the same 30 gardens described in the previous chapter, with one control plot of unmanaged habitat per wadi (N=7). The precise location of these gardens and unmanaged plots is shown in Figure 3.1. Security issues meant it was not possible to repeat the surveys in the low desert (Wadi Feiran, Sheik El-wad, Oasis Ein Hodra) in the second year, but monthly flower visitors surveys were repeated in the twenty gardens in the mountain and town in 2013. To compensate for the loss of the low desert gardens, 14 additional mountain gardens were surveyed in 2013, with gardens randomly selected from Wadi Itlah (1300 m a.s.l), Wadi Gebel (1800 m a.s.l) and the previously unsampled Wadi Tinya (1850 m a.s.l). In 2013 the number of unmanaged plots was also increased to provide a more robust comparison of pollinator numbers within the gardens and unmanaged habitats. A total of 20 unmanaged plots were chosen to typify the habitat of the wadi, with sandy soil and low-growing desert shrubs. See Figure 3.2 for the locations of the 34 gardens and unmanaged plots that were surveyed in 2013.

Gardens tended to occur in tight clusters along the base of the wadis so the choice of unmanaged plots was highly constrained, but within wadi all gardens and unmanaged plots were within 1 km of each other, with a mean distance of 461 m ± 73 between gardens and the nearest controls. The maximum foraging range of many solitary bees is 600 m (Osborne et al. 1999; Gathmann & Tscharntke 2002) and wild pollinators generally respond to landscape factors within a 1 km radius (Steffan-Dewenter et al. 2002). As such, it can be assumed that within wadi all samples were taken from the same available pollinator community, with habitat type (garden versus unmanaged) being the main varying factor.

Garden size varied significantly between the low desert and the mountains and town (glm: $F_{60,62}=10.58$, $P < 0.001$), with average garden sizes in the low desert of 14,800 m^2 (± 6500), compared to an average of just 2100 m^2 (± 240) in the mountains and town. However there was no significant correlation between garden size and the abundance (lmer: $\chi^2 = 0.009$, df =1, $P = 0.921$) or species richness ($\chi^2 = 1.55$, df=1, $P = 0.213$) of pollinators, so garden size has not been included in further analyses.

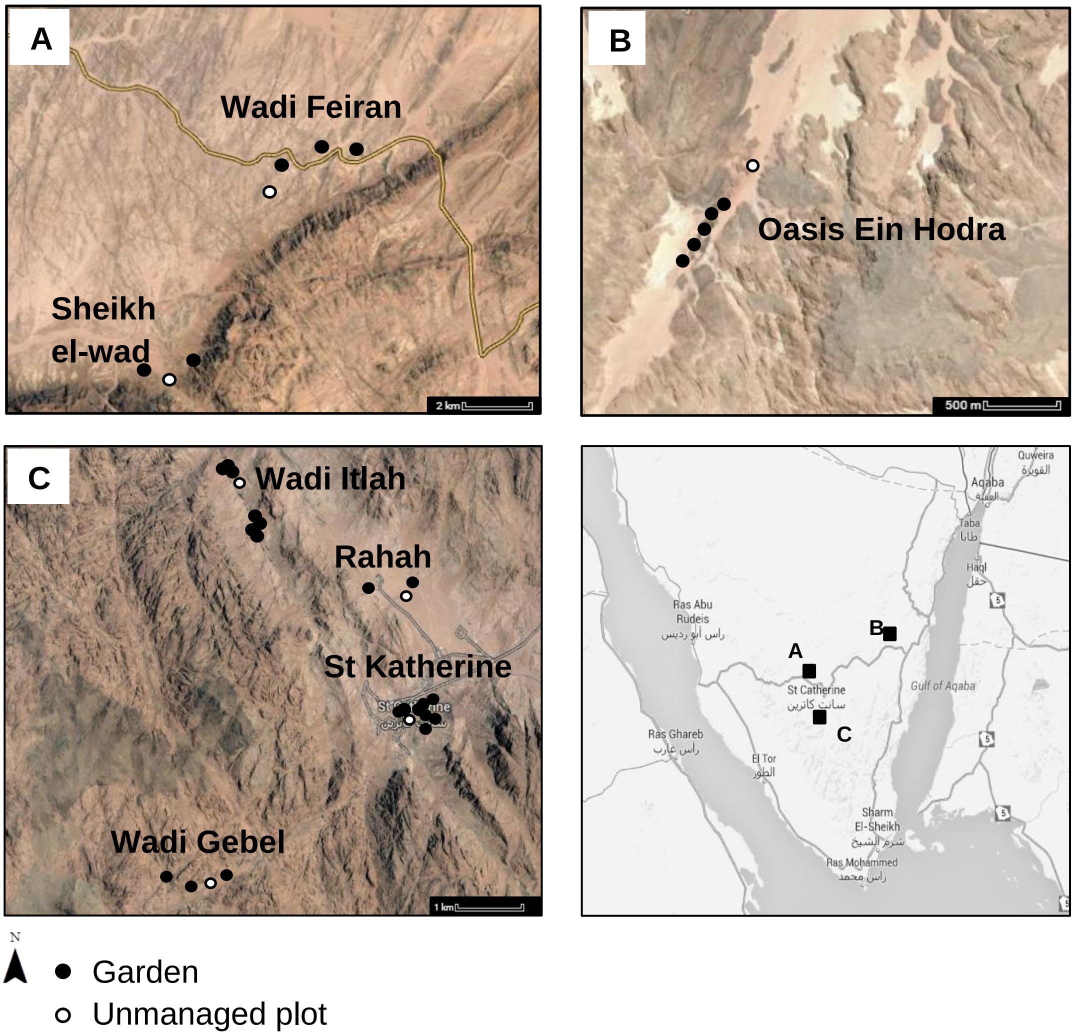

Figure 3.1. The locations of surveyed gardens (N= 30) and unmanaged plots (N=7) in 2012.

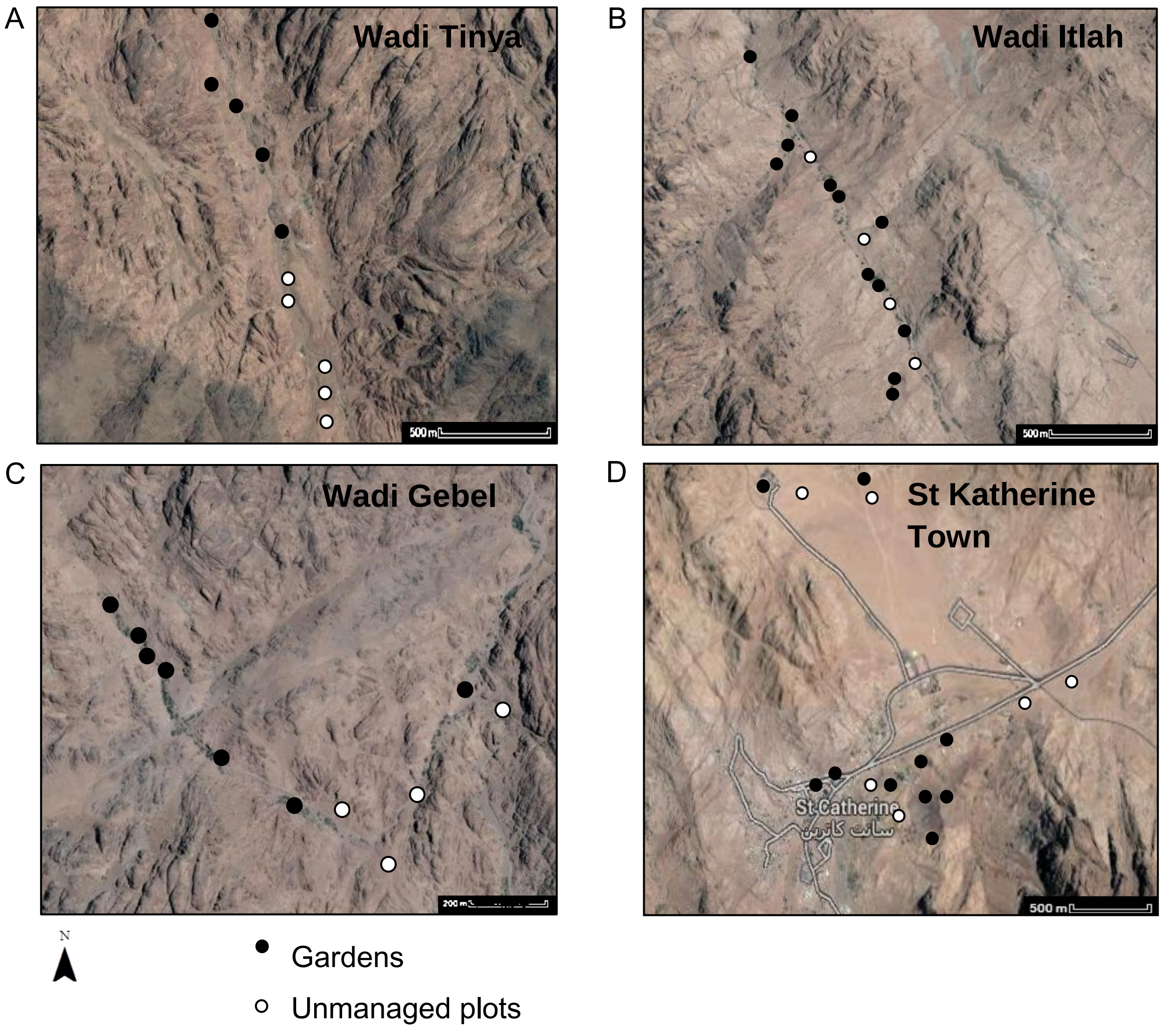

Figure 3.2. The locations of surveyed gardens (N= 34) and unmanaged plots (N=20) in 2013.

Statistical analyses

Data were pooled for each garden/plot across the four sampling rounds, with data from 2012 and 2013 included in the analyses. Firstly, a generalised linear model was used to compare pollinator a) abundance and b) species richness, between gardens and unmanaged plots. The interaction between *garden/unmanaged* and *year* was included as a fixed effect to see whether the same pattern was observed in both years.

The data were then modelled more extensively using linear mixed-effect models (*lme4* package) (Bolker et al. 2009), which included numerous environmental variables as fixed effects. Pollinator a) abundance and b) species richness were response variables, with *floral abundance*, *floral species richness*, *garden/unmanaged* and *altitude* included as fixed effects. *Wadi* was included as a random effect to account for any spatial differences in flower visitor abundance or richness. Model fit was based upon AIC and followed Zuur et al. (2009), with the significance of fixed effects and their interactions tested by comparing models with a likelihood ratio test (distributed as Chi-squared). R^2 values were obtained for linear mixed-effect models using the MuMIn package (Barton 2014), with marginal R^2_{GLMM} values representing the variance explained by each fixed effect (Nakagawa & Schielzeth 2013).

3.3 Results

Characterising the pollinator community

In total 5243 pollinators were recorded; 1928 in 2012, with insects belonging to 137 species from 37 families, and 3315 in 2013 with insects belonging to 185 species from 29 families (Table 1). For a full species list see Appendix 3.1. Hymenoptera were the most abundant order, making up of 53% of the total pollinator community. Of these the majority were solitary bees (61%), followed by honeybees (25 %) and wasps (14%).

Hymenoptera were also the most diverse order of pollinators, with over 96 species recorded from 14 families. Five of these species were new records for Egypt*;* *Lasioglossum erraticum* (previously only recorded in Greece, Cyrprus, Turkey and Armenia), *Megachile montenegrensis* (previously in Greece, Turkey and Iran), *Megachile insignis*, (previously in Greece, Syria, Turkey and Israel), *Colletes tuberculatus* (previously in Jordan and Israel) and *Bembecinus hebraeus* which was previously thought to be endemic to Israel. One species of hoverfly was also a new record for Egypt, *Melanostoma scalare* (previously in Europe).

Two rare and range-restricted species of colletid bee were locally abundant; the endemic *Hylaeus sinaiticus* and *Hylaeus* Sinai sp1, which has now been confirmed as a new species by Holger Dathe (personal comm.) and is pending description. There were several unusual species of Anthophorine bee (*Amegilla* Sinai sp1, *Anthophora* Sinai sp1, *Anthophora* Sinai sp2 and *Anthophora (Heliophila)* Sinai sp1), which are awaiting description by Chris O'Toole pending access to type material to confirm that they are new. The taxonomy of the Anthophorini is somewhat confused, but these are likely to be new, undescribed species. Two *Megachile* species from the complex subgenus *Eutrichaea* may also be previously undescribed species and are under further investigation by Christophe Praz.

Gardens versus unmanaged plots

Pollinators occurred in significantly higher numbers within the gardens than the unmanaged plots (glm: $F_{86,88} = 8.15$, $P < 0.001$) and had a significantly higher species richness ($F_{86,88} = 21.73$, $P < 0.001$). These patterns were observed in 2012 (Fig 3.3a; gardens: N=30, unmanaged: N =7) and in 2013 when the number of unmanaged plots were increased (Fig 3.3b; gardens: N=34, unmanaged: N=20), with no significant interaction between *year* and *garden /unmanaged* (abundance: $F_{86,87} = 0.001$, $P=0.997$; species richness: $F_{86,87} = 0.14$, $P=0.713$). On average pollinator abundance was significantly higher in 2013 than 2012 ($F_{87,88} = 5.11$, $P=0.026$), as was species richness ($F_{87,88} = 23.51$, $P < 0.001$). Within the gardens, average abundance increased by one third in 2013 (as compared to 2012) and in the unmanaged habitat abundances increased ten-fold. Grouping the polliantors by order showed that Hymenoptera were the most abundant flower visitor in 2012 and 2013, followed by Diptera (Fig 3.4). All five orders showed similar patterns in relation to the presence of the gardens, and all occurred in higher abundances within the gardens than in the unmanaged plots.

Environmental predictors

Linear mixed-effect modelling showed that floral abundance was the strongest predictor of pollinator abundance and species richness (Fig 3.5a; Table 3.2). The species richness of pollinators was also strongly correlated with the species richness of flowers found in each plot (Fig 3.5b). Whether plots were in gardens or unmanaged habitat did have a significant effect upon pollinator abundance and species richness, but the model revealed that there was a significant interaction with altitude (Table 3.2). At lower altitudes, gardens had a strong positive effect upon pollinator abundance and species richness, containing much higher numbers than the unmanaged plots (Fig 3.6a & b). However at altitudes greater than 1800 m a.s.l. the gardens supported an equal abundance and species richness of pollinators as the unmanaged plots.

These altitudinal differences appeared to relate directly to the floral availability along the altitudinal gradient, with floral abundance (Fig 3.7a) and floral species richness (Fig 3.7b) exhibiting very similar patterns in response to altitude. At lower altitudes, gardens supported a more abundant and diverse floral community than the unmanaged habitat, but at altitudes above 1800 m a.s.l. the floral abundance and species richness increased in the unmanaged habitat and approached the levels found within the gardens.

Table 3.1. Pollinator families recorded in the gardens and unmanaged habitat.

	2012				2013			
	Abundance		**Species richness**		**Abundance**		**Species richness**	
	Garden	Unmanaged	Garden	Unmanaged	Garden	Unmanaged	Garden	Unmanaged
COLEOPTERA								
Burpestidae	7	1	1	1	28	8	2	2
Coccinellidae	23	0	1	0	110	0	1	0
Chrysomelidae	54	0	3	0	0	0	0	0
Dermestidae	73	2	6	2	121	32	3	3
Pyrochroidae	17	0	1	0	106	0	1	0
Scarabidae	5	0	2	0	33	1	3	1
DIPTERA								
Anthomyiidae	26	0	4	0	0	0	0	0
Acroceridae	1	0	1	0	0	0	0	0
Bombyliidae	4	2	2	1	23	9	10	6
Caliphoridae	57	0	5	0	14	6	2	2
Chloropidae	1	0	1	0	0	0	0	0
Drosophilidae	2	0	1	0	0	0	0	0
Muscidae	20	0	4	0	0	0	0	0
Platypezidae	4	0	1	0	0	0	0	0
Sarcophagidae	37	0	9	0	26	20	9	8
Tephritidae	8	1	5	1	16	22	5	2
Syrphidae	293	17	16	3	577	50	12	8

HEMIPTERA								
Acanthosomatidae	26	0	2	0	0	0	0	0
Lygaeidae	1	0	1	0	1	1	1	1
HYMENOTPERA								
Andrenidae	10	0	2	0	39	0	2	0
Apidae	500	4	17	4	523	61	21	8
Colletidae	102	0	5	0	193	80	13	2
Halictidae	106	3	25	2	245	70	15	5
Megachillidae	136	3	13	3	240	77	21	11
Chalcidoidea	0	0	0	0	5	4	2	1
Chrysididae	4	0	3	0	3	7	2	3
Crabonidae	49	0	16	0	68	120	13	17
Evaniidae	3	0	3	0	0	0	0	0
Ichneumonidae	3	0	2	0	2	1	1	1
Scoliidae	0	0	0	0	30	8	1	1
Sphecidae	7	0	3	0	9	0	6	0
Tiphiidae	0	0	0	0	3	0	1	0
Vespidae	11	1	6	1	42	15	9	4
LEPIDOPTERA								
Hesperiidae	1	0	1	0	4	0	1	0
Lycaenidae	292	0	5	0	56	146	3	3
Nymphalidae	4	1	2	1	1	3	1	1
Pieridae	6	**0**	5	0	40	10	5	3
Sphingidae	1	0	1	0	6	0	1	0
TOTAL	**1894**	**35**	**174**	**19**	**2564**	**751**	**167**	**93**

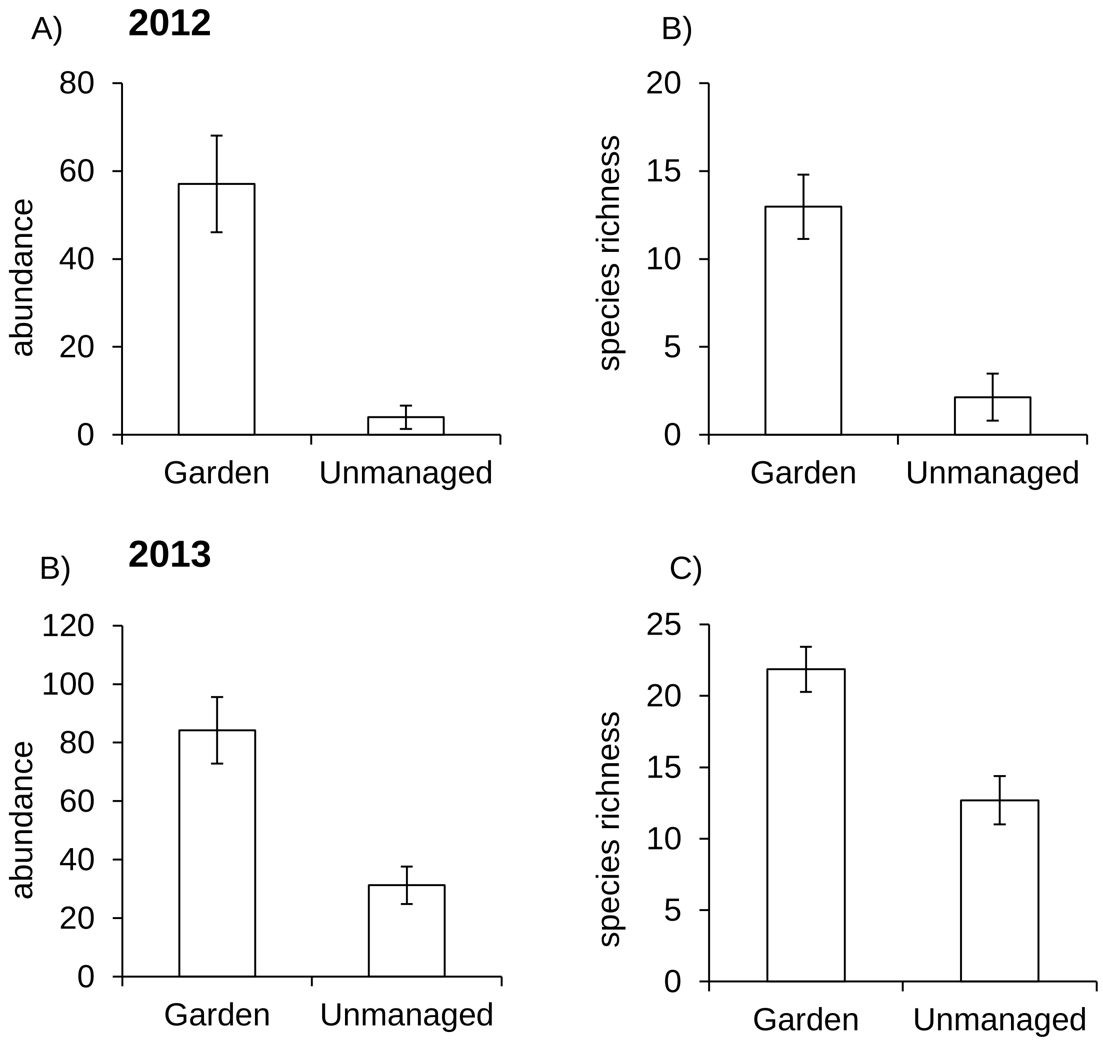

Figure 3.3. Mean pollinator abundance and species richness in gardens and unmanaged plots. Error bars represent the SEM.

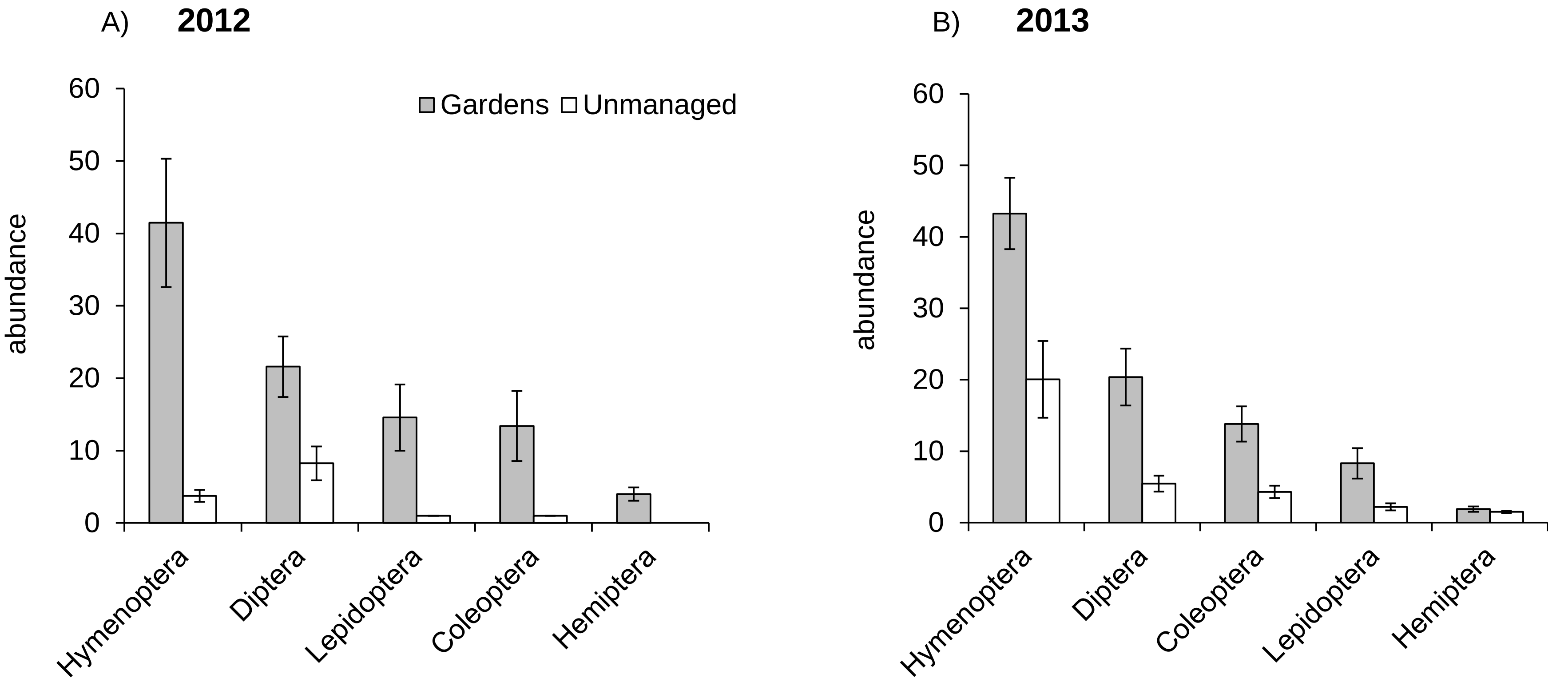

Figure 3.4. Mean abundance (± SEM) of the five pollinator orders per garden and unmanaged plot.

Table 3.2.Environmental predictors of pollinator abundance and species richness; results from linear mixed effect models that included wadi as a random effect. R^2_{GLMM} is the marginal R^2 value and represents the variance explained by the fixed factors. X^2 tests the difference between models after the associated fixed effect has been dropped.

Response variable	Fixed effect	R^2_{GLLM}	χ^2	df	*P*
Pollinator abundance	floral abundance	0.23	438.56	1	**0.001**
	garden/unmanaged	0.13	7.35	1	**0.007**
	altitude*garden/unmanaged	0.16	4.51	1	**0.034**
	floral richness	0.17	0.91	1	0.338
	altitude	0.02	0.047	1	0.829
	full model	*0.35*			
Pollinator species richness	floral abundance	0.15	24.10	1	**0.001**
	floral richness	0.36	22.01	1	**0.001**
	garden/unmanaged	0.10	8.59	1	**0.003**
	altitude *garden/unmanaged	0.21	3.99	1	**0.046**
	altitude	0.09	0.67	1	0.412
	full model	*0.43*			

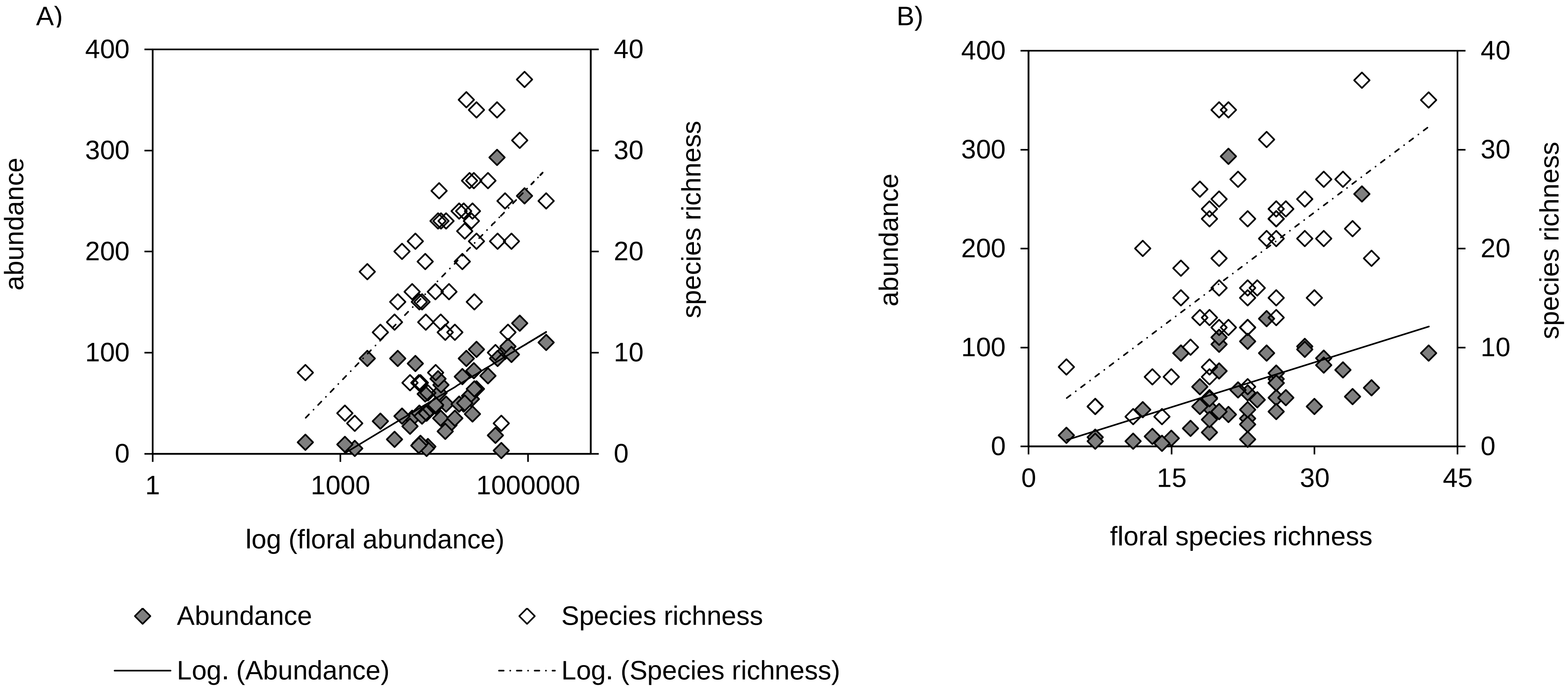

Figure 3.5. Pollinator abundance and species richness in relation to a) floral abundance and b) floral species richness. For associated R^2_{GLMM} values see Table 3.2.

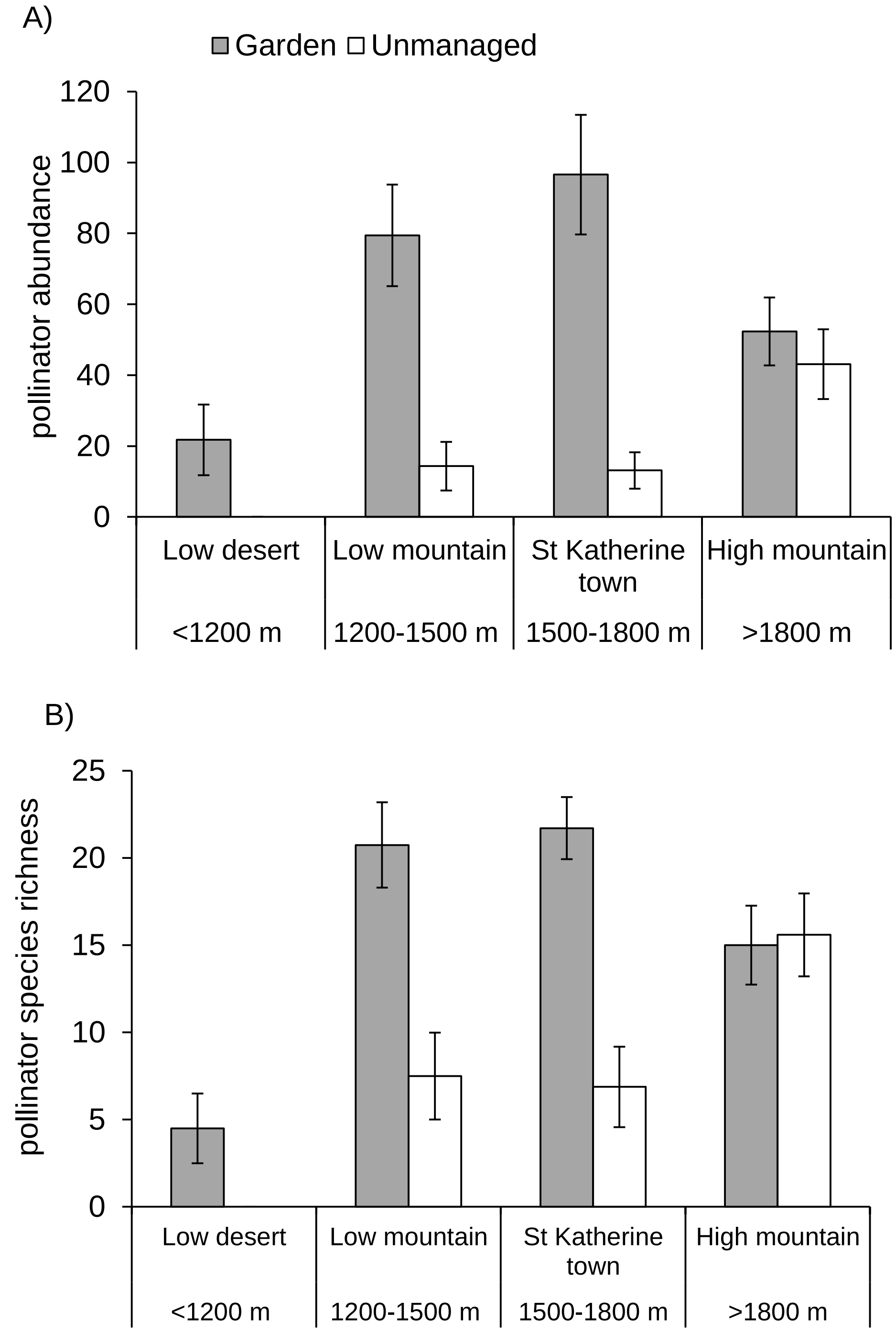

Figure 3.6. Mean pollinator a) abundance and b) species richness per garden and unmanaged plot along an altitudinal gradient. Error bars represent SEM.

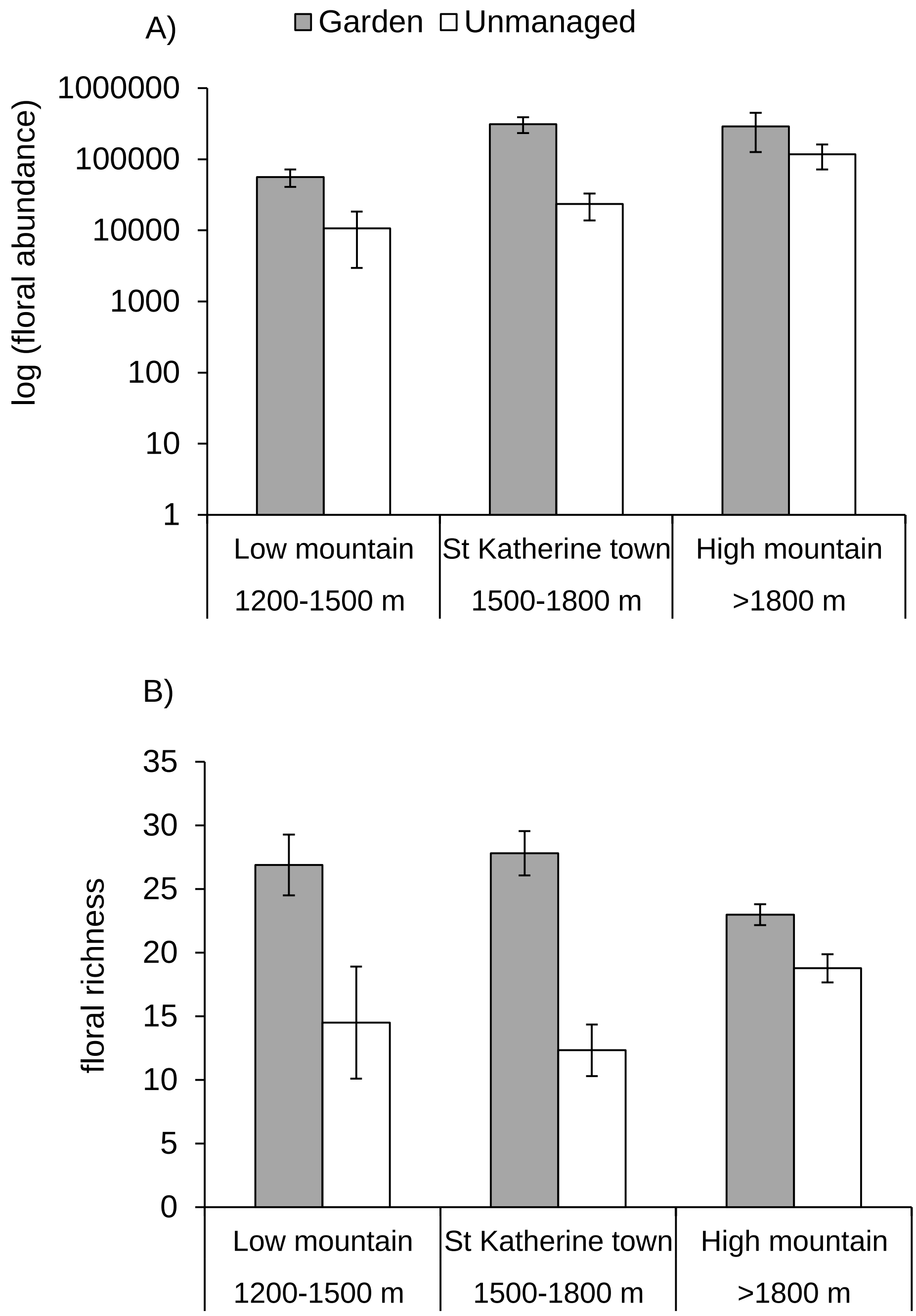

Figure 3.7. Mean a) floral abundance and b) floral species richness in gardens and unmanaged plots along an altitudinal gradient in 2013. Error bars represent SEM.

3.4 Discussion

The rainwater irrigated gardens in this system actively enhanced pollinator abundance and species richness, highlighting the positive potential of arid land agriculture for pollinator conservation. The majority of work on pollinators comes from temperate and tropical regions (Archer et al. 2014), where agricultural conversion typically results in a reduction in bee abundance and diversity (Ferreira et al. 2013). We demonstrate that the situation is very different in understudied arid environments like South Sinai, where irrigated agricultural gardens can actively increase pollinator abundances.

Composition of the pollinator community and the impact of the gardens

This study provides one of the first comprehensive species lists of pollinators in the St Katherine Protectorate. For orders Coleoptera and non-syrphid Diptera we were unable to establish species level identifications, but the vast majority of bees and wasps have been identified to the species level. Hymenoptera were the most abundant and diverse order of pollinators in South Sinai, with a total of 96 species recorded, five of which were new records for Egypt, and several of which may be previously undescribed endemics.

Gardens contained a significantly higher abundance and species richness of pollinators than the unmanaged habitat, and this was true of all five insect orders. In temperate and tropical environments agricultural conversion of natural habitat often reduces landscape complexity and leads to a loss of floral resources and nesting sites for pollinators (Ferreira et al. 2013). However in arid environments, the limited availability of water and nutrients means that natural floral resources are scarce and agricultural interventions can increase the availability of floral resources. Here, the irrigated agricultural gardens actively increased floral abundance and species richness, resulting in a dramatic increase in the abundance and diversity of dependent pollinators. Though these results contrast with the trends observed in other regions (Winfree et al. 2009; Ferreira et al. 2013), they are consistent with patterns observed in neighbouring Israel, where irrigated ornamental gardens have been shown to dramatically increase the abundances of wild bees (Gotlieb et al. 2011).

The higher abundance of pollinators was strongly correlated with the floral abundances within the gardens. As discussed in the previous chapter, livestock are not allowed in the Jebeliya gardens, but they do graze on wild vegetation in the unmanaged habitat (Gilbert 2013). The fact that grazing is banned within the gardens may contribute towards the elevated levels of floral availability, but water is more likely to be the key factor in determining plant productivity in this hyper-arid region (Fischer & Turner 1978). Indeed, the fact that pollinator abundance and species richness increased significantly following the heavy rains in early 2013 is consistent with the hypothesis that water is the primary limiting factor in this system. Within the gardens, the floods led to a marginal increase in pollinator numbers, but in the unmanaged habitat they resulted in a ten-fold increase in pollinator abundances. The muted response within the gardens suggests that the water-fed irrigation from wells buffers against the environmental fluctuations of water availability experienced in the unmanaged habitat, allowing gardens to support elevated numbers of pollinators.

Pollinator species richness was strongly linked to both the abundance and diversity of floral resources, which tended to be higher within the gardens. The positive link between pollinator diversity and floral diversity has been observed in other studies (Potts et al. 2003; Gotlieb et al. 2011) and farms that contain a higher diversity of flowering plants tend to support more diverse bee communities (Holzschuh et al. 2007; Kennedy et al. 2013). The presence of water within the gardens may be the key factor influencing plant and pollinator abundances, but the maintenance of agricultural diversity (both crops and wild plants) seems to be important in determining the diversity of the pollinator communities.

The impact of the gardens differed across an altitudinal gradient. At low altitudes gardens contained much higher abundance and species richness of pollinators than the unmanaged habitat, but a higher altitudes (>1800m) levels were equal in the two habitat types. These altitudinal differences related directly to the floral availability along the altitudinal gradient, which increased in the unmanaged habitat at higher elevations. The high mountains have a cooler climate with higher levels of precipitation (often as snow) and as a result they tend to support a more abundant and diverse plant community than the low mountains and desert (Ayyad et al. 2000). It seems that at higher altitudes, when water resources are naturally more abundant, the irrigation within the gardens had less of a pronounced effect.

Conservation implications

The Bedouin have received somewhat unwarranted negative attention regarding the damaging effects of their grazing and hunting (Gilbert 2013), but these results suggest that their primary source of subsistence actively benefits plants and pollinators within the St Katherine Protectorate. Over the past decade there have been contrasting views on how best to conserve biodiversity and many of these ideas can be summed up via the ‘parks versus people’ debate (Miller et al. 2011). Some take the ‘parks’ view, arguing that they only way to conserve wildlife is to exclude people completely from parks. Others take a more social view of conservation taking the side of the ‘people’, arguing that parks should work with local people, encouraging sustainable use of resources, with an emphasis on poverty alleviation. Win-win solutions that simultaneously conserve biodiversity and promote human well-being are hard to find (McShane et al. 2011), but the results of this study suggest that Bedouin gardens can benefit biodiversity within the StKP whilst providing a sustainable food source for local people.

Chapter 4.

Contrasting patterns of turnover between plants, pollinators[*]: *How do the gardens influence patterns of beta diversity across the landscape?*

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[*] A modified version of this chapter is in press at *Diversity and Distrubtions*:
Norfolk, O., Eichhorn, M., & Gilbert, F. (2014). Contrasting patterns of turnover between plants, pollinators and their interactions. *Diversity and Distributions*. In press.

Chapter 4. Contrasting patterns of turnover between plants, pollinators and their interactions: *How do the gardens influence patterns of beta diversity across the landscape?*

Abstract

Biogeographers typically assess patterns of diversity across landscapes. Since interacting groups often exhibit contrasting trends, this leads to variation in the structure of interaction networks and thereby influences ecosystem processes. Here we aim to disentangle how patterns of diversity differ between species (plants, pollinators) and their interactions across this agricultural landscape. Previous chapters have demonstrated that the irrigated gardens enhanced plant and pollinator diversity and appear as high-diversity islands within the arid mountainous habitat. This chapter assesses whether this local enhancement: (a) increases landscape heterogeneity by supporting novel species and interactions; (b) increases local diversity by supporting higher densities of species that also occur in the unmanaged habitat; (c) whether these patterns differ between plants, pollinators and their interactions. The results showed that the impact of the gardens differ with respect to the landscape context; in the low mountains, gardens enhance the abundance and diversity of plants, pollinators and interactions, but in the high mountains they had no effect. Plants exhibited high levels of species turnover, with gardens increasing landscape heterogeneity by supporting novel species. In contrast, pollinators exhibited low levels of turnover, with gardens in the low mountains increasing local abundance and diversity by supporting species that were shared with the matrix species pool. The diversity of interactions was strongly influenced by the composition of the plant community and showed extremely high levels of turnover across the landscape. These results show that interacting species can display highly contrasting patterns of turnover across a shared landscape. Though local habitat enhancement had the potential to benefit pollinators, landscape

heterogeneity was also required in order to maintain the diversity of plant-pollinator interactions.

4.1 Introduction

Understanding landscape-scale patterns of diversity is an important challenge in conservation biogeography because it can help inform which strategies will be most effective at maximising diversity. Beta diversity is maintained across a landscape by two processes, nestedness and spatial turnover (Wright & Reeves, 1992; Baselga, 2010). Nestedness occurs when less diverse assemblages of species form a nested subset of those present in the entire species pool and usually reflects the non-random process of species exclusion from less diverse sites (Ulrich & Gotelli, 2007). In contrast, spatial turnover occurs when certain species are actively replaced by others, creating distinct assemblages that each support novel species (Leprieur et al., 2009). It is useful to understand these patterns of beta diversity because communities exhibiting high nestedness versus high spatial turnover require contrasting conservation strategies; in nested communities the targeted conservation of the most diverse habitat patches can benefit the majority of species, but in those with high spatial turnover it is essential to maintain a number of patches with high habitat heterogeneity in order to conserve all the species in the community (Wright & Reeves, 1992; Baselga, 2010).

Deciding on the target organism also has a strong influence on the most appropriate conservation strategy because different taxa can display contrasting patterns of beta diversity across a shared landscape (Fleishman et al. 2002; Soininen et al. 2007). Species with higher dispersal abilities tend to show lower levels of turnover (Soininen et al., 2007) and herbivorous insects show much lower levels of spatial turnover than plants due to their ability to exhibit generalised foraging behaviour (Novotny et al. 2007). Since pollinators are more mobile than plants and known to exhibit generalised foraging behaviour (Bjerknes et al. 2007; Graves & Shapiro 2003; Williams et al.

2011) they are likely to exhibit much lower levels of turnover than the plants on which they forage.

In reality groups of organisms cannot be considered in isolation, with communities consisting of complex networks of interacting species from different trophic levels (Tylianakis et al. 2010). Conservation is traditionally aimed at rare and threatened species, so often fails to take into account the networks of interactions that are responsible for maintaining ecosystem services such as pollination and pest control (Memmott et al. 2004; Fontaine et al. 2005,Tylianakis et al. 2007; Macfadyen et al. 2009). We are currently moving towards a more holistic approach to conservation that focusses on preserving ecosystem functioning (MEA 2005; Balvanera et al. 2006), and so must consider how to best conserve the networks of interactions between species. In the context of plants and pollinators, we must decide whether we want to prioritise the conservation of a) pollinator species or b) pollination services. If it is the latter, then it may be more useful to focus on conserving plant-pollinator interactions rather than pollinator species per se. This chapter compares patterns of alpha and beta diversity between plants, pollinators and their interactions, in order to disentangle how community composition changes between species and their interactions across a shared landscape.

The unusual distribution of resources associated with the study site in South Sinai makes it an ideal location to compare patterns of landscape-scale diversity. It is an arid mountainous region, but the presence of rainwater harvesting allows the cultivation of agricultural gardens with a higher potential for plant growth than unmanaged habitat (Norfolk et al. 2013). These gardens appear as resource-rich islands in an arid landscape and have been shown to support a higher diversity of wild plants (Norfolk et al. 2013) which receive higher rates of floral visitation (Norfolk & Gilbert 2014) than the surrounding habitat. This chapter extends early work, in order to assess how the gardens affect the alpha diversity of plants, pollinators and their interactions.

Landscape context is known to have a strong influence on the composition of pollinator communities (Holzschuh et al. 2007), with the species richness of crop pollinators declining with distance from natural or semi-natural habitat (Ricketts et al. 2008) and increasing with the quality of the surrounding habitat (Kennedy et al. 2013). Results of the previous chapter suggested the impact of the gardens differs with altitude in accordance to the quality of the surrounding habitat. Here, gardens were selected from two altitudinal categories with distinct environmental properties: (a) the high mountains (isolated, cooler temperatures, higher water availability); and (b) the low mountains (close proximity to villages, more disturbed, lower water availability). Specific predictions were that the irrigated gardens would increase the abundance and alpha diversity of plants, pollinators and interactions above those found in the unmanaged habitat, with a greater effect in the low mountains due to a higher contrast with the quality of the matrix.

This chapter utilises new techniques in similarity analyses (Gotelli & Chao 2013) to test two models for explaining how beta diversity is maintained across this agricultural landscape. The first model (A) predicts high levels of spatial turnover, with gardens increasing landscape heterogeneity by supporting novel species not present in the unmanaged habitat (Fig 4.1a). The second model (B) predicts that diversity is nested, with low levels of turnover across the landscape (Fig 4.1b); in this model, gardens create local enhancement by increasing the densities of species already present in the unmanaged species pool.

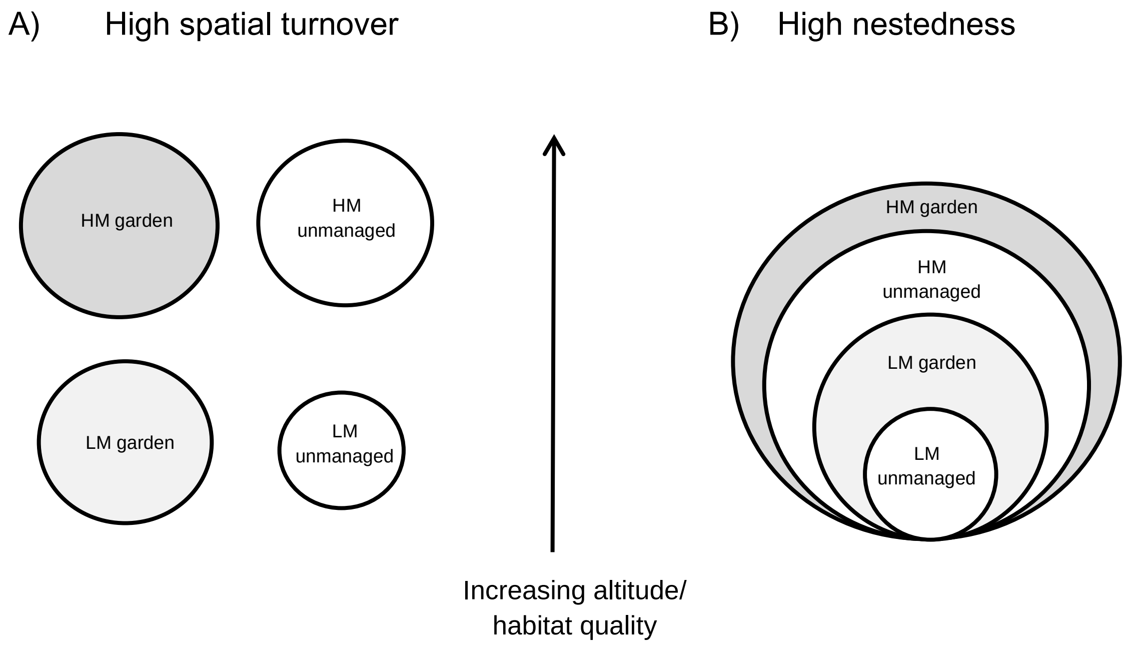


Figure 4.1 The two conceptual models describing patterns of diversity between gardens and unmanaged habitat in the high mountains (HM) and low mountains (LM): (a) a high spatial turnover predicts that gardens and unmanaged habitat will support distinct assemblages of novel species; and (b) a high nestedness predicts that gardens will increase diversity by supporting higher numbers of species already present in the unmanaged species pool.

Specifically the predictions were that plants and pollinators would show contrasting levels of turnover and that:

1) Plants would follow model A, exhibiting high levels of spatial turnover with gardens increasing overall landscape heterogeneity.

2) Pollinators would follow model B, showing much lower levels of spatial turnover (due to their greater mobility and generalised foraging behaviour) with gardens creating local enhancement.

3) Plant-pollinator interactions would be influenced by the distribution of both plants and pollinators, and so would display higher levels of spatial turnover across the landscape, following model A.

The results revealed highly contrasting patterns of turnover between plants, pollinators and their interactions, and showed that patterns of alpha diversity are strongly influenced by the landscape context.

4.2 Methods

Data collection

In order to investigate patterns of diversity in plants and pollinators, monthly plant-pollinator surveys were conducted in gardens and unmanaged plots throughout April to July 2013, using the flower-visitor survey protocol described in the previous chapter (see section 3.2). For the similarity analyses used in this chapter I felt it imperative to have an equal number of garden and unmanaged plots, because I was concerned that unequal sampling effort in the gardens versus the unmanaged plots could be artificially inflating the number of species observed within the gardens. To make sure this was no the case I only used the data from 2013 (gardens, N=34; unmanaged, N=19) and filtered these data by randomly selecting 19 gardens from the available 34 to equal the number of unmanaged plots. Spatial matching of unmanaged plots and gardens was maintained by randomly selecting the appropriate number of gardens within each wadi. After establishing the importance of altitude in Chapter 3, I then categorised the plots into two altitudinal zones, (a) high mountains, 1800-1850 m a.s.l. (N = 9), and (b) low mountains, 1300-1550 m a.s.l. (N = 10). The locations of the selected gardens and unmanaged plots are shown in Figure 4.2.

Data analyses

Spatial patterns in alpha diversity were explored using Hill's numbers (species richness [${}^0D$], the exponential of Shannon entropy [${}^1D$] and the inverse Simpson index [${}^2D$]) (Hill 1973) in accordance with current consensus (Chao et al. 2012; Jost 2006; Leinster & Cobbold 2011). Hill's numbers are defined to the order of q (${}^qD$), whereby parameter q indicates the weighting given to rare or common species. ${}^0D$ is insensitive to relative frequencies, and is therefore weighted towards rare species. ^{1}D is weighted towards common species, and ^{2}D is weighted towards abundant species. The same concept was also applied to the interactions, with ^{0}D defined as the number of unique links between plant and pollinator species, ^{1}D as the Shannon diversity of these interactions and ^{2}D as the inverse Simpson diversity of interactions. Diversity measures were calculated in package *vegan* in R version 3.0.2 (R Core Team 2013; Oksanen et al. 2013). Data from the four-month sampling period were pooled for each garden and unmanaged plot. Pollinator abundance was defined as the total number of insects recorded visiting flowers in each plot, thus is equivalent to the abundance of interactions.

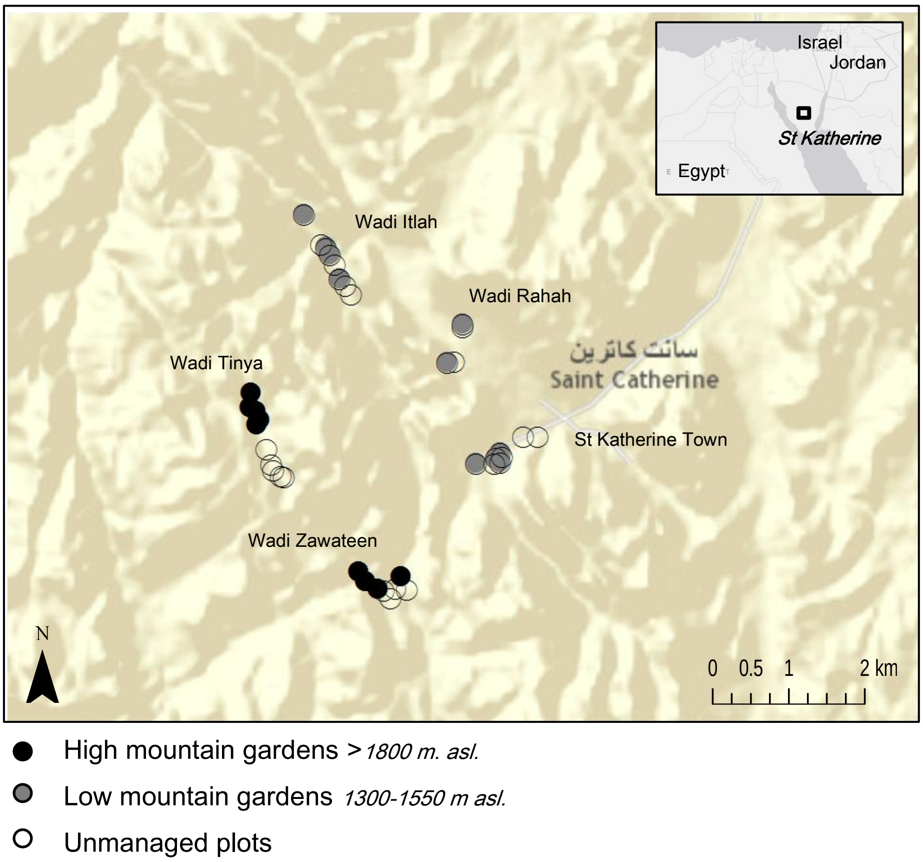


Figure 4.2. Map of study site in St Katherine Protectorate, South Sinai, with locations of gardens and unmanaged plots.

The abundance and diversity (^0D, ^1D, ^2D) of plants, pollinators and their interactions were analysed using linear-mixed effect models (*lme4* package) (Bolker et al. 2009). Models included an interaction between altitudinal category (high mountains vs. low mountains) and habitat (garden and unmanaged habitat) as predictors, and wadi and garden area as random factors; wadi was included to account for spatial variation amongst plots, and garden area to account for the variation in the sizes of gardens. Model fit was based upon AIC and simplification followed Zuur et al. (2009), with the significance of fixed factors tested by comparing models with a likelihood ratio test (distributed as Chi-squared). A Tukey's post-hoc test was used to compare the effect of habitat (garden vs. unmanaged) in the high mountains and low mountains.

To visualise the interactions between plants and pollinators at a community level cumulative visitation networks were created for gardens and unmanaged plots in the high and low mountains using the R package *bipartite* (Dormann et al., 2009). These visitation networks were derived from quantitative interaction matrices with n rows (representing plant species) and m columns (representing insect species), with the value at the intersect representing the number of interactions observed between flower and insect.

Similarity analyses

In order to evaluate whether the gardens increased species turnover or lead to local enhancement we compared the similarity of species and interactions in the gardens and unmanaged habitats using three measures of beta diversity derived from the C_{qN} measure (Gotelli & Chao 2013). As with the previously described Hill's numbers, q is a parameter that determines the measures' sensitivity to species' relative abundances and N is the number of assemblages (in this case N = 2 for the high and low mountains respectively). For q=0, C_{0N} is the Sørenson similarity index; for q=1, C_{1N} is the Horn overlap index; and for q=2, C_{2N} is the Morisita-Horn similarity index. These three similarity indices were calculated for flower-visitors, flowers and their interactions in using SPADE with 200 iterations (Chao & Shen 2010). C_{qN} ranges between unity (when communities are identical) and zero (when communities are completely different). Higher similarity means more species are shared between gardens and unmanaged plots, indicating the potential to increase local diversity. Lower similarity means fewer shared species, indicating that the gardens support a distinct community of species and thus increase landscape heterogeneity.

The Sørenson similarity index was also used (with 200 iterations) to estimate the total relative abundance of the shared species and interactions in (a) the garden assemblage and (b) the unmanaged assemblage (ie. the proportion of species within the garden that were shared with the unmanaged habitat, and vice versa). This provided additional insight into whether any dissimilarity was due to the two habitats supporting a completely different suite of species, or whether dissimilarity was due to the presence of additional species within the gardens.

4.3 Results

Plant-pollinator interactions in the gardens and unmanaged habitat

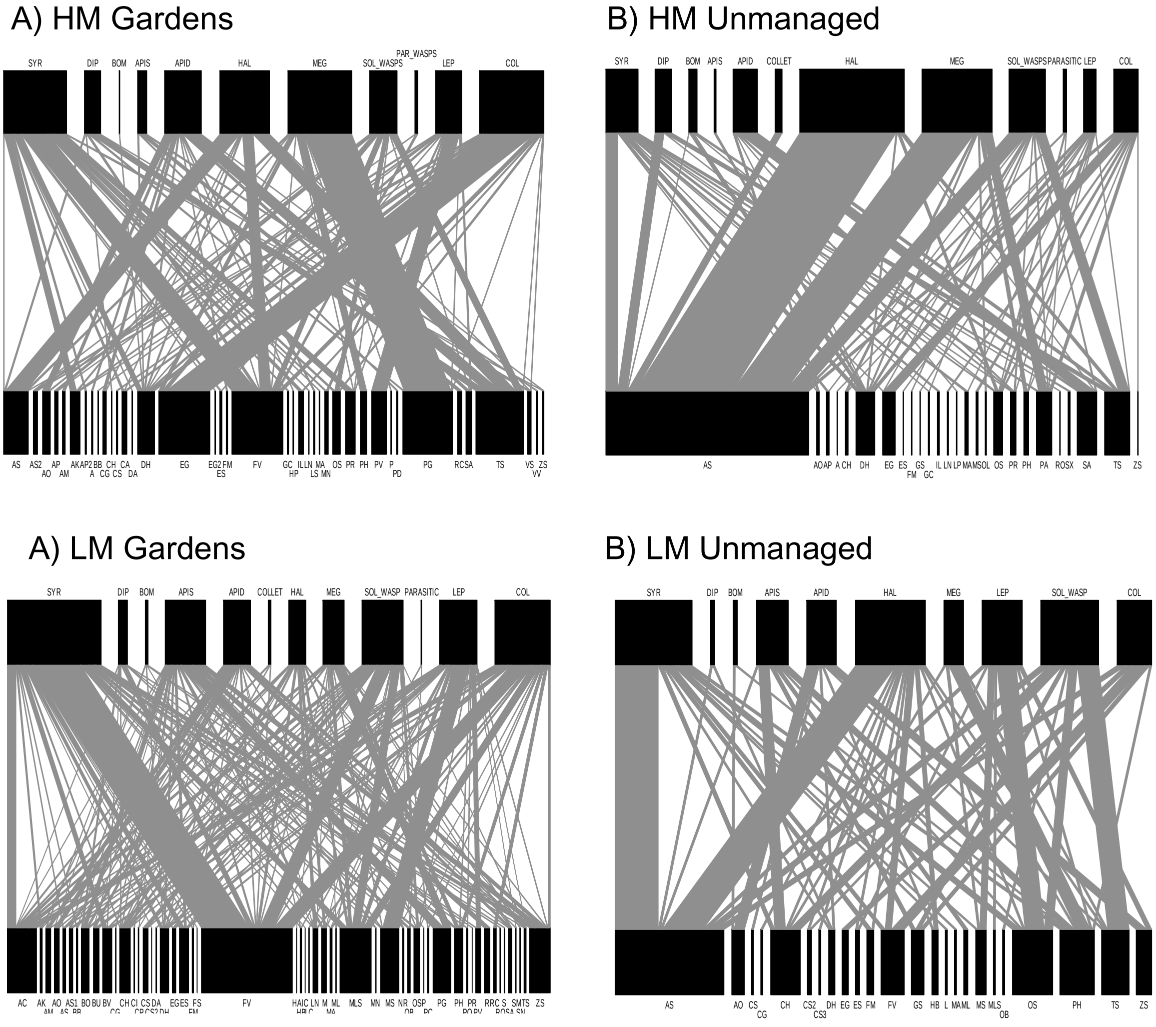
In total we recorded 2410 interactions between 159 pollinator species and 81 plant species. The average number of observed interactions was 88 ± 13 in the gardens and 37 ± 9 in the unmanaged habitat. Visitation networks are shown in Figure 4.3. Plants, pollinators and their interactions displayed significant statistical interactions between habitat type (garden/ unmanaged) and altitude (Table 4.1), with gardens having a much stronger positive effect upon abundance and diversity in the low mountains.

In the high mountains, habitat type had little impact upon plant abundance (Mean ± S.E. garden: 68.67 ± 5.39; unmanaged: 70.33 ± 6.48) or pollinator abundance (garden: 56.22 ± 9.80; unmanaged: 45.89 ± 10.47) and garden and unmanaged plots supported similar levels of plant and pollinator diversity (${}^{0}D$, ${}^{1}D$ and ${}^{2}D$) (Fig 4.4a & b). The diversity of plant-pollinator interactions (^{0}D, ^{1}D and ^{2}D) did not differ between gardens and unmanaged habitat (Fig 4.4c) which can be visualised by the similar complexities of the visitation networks (Fig 4.3a). In the low mountains, habitat type had a much stronger effect, with gardens supporting a more abundant and diverse community of plants and pollinators than the unmanaged habitat. Plant abundance was twice as high within the low mountain gardens (garden: 98.20 ± 10.14; unmanaged habitat: 47.40 ± 7.37) and pollinator abundance increased by seven-fold (garden: 117 ± 21.09, unmanaged: 18.10 ± 13.16). Plant diversity (^{0}D, ^{1}D and ^{2}D) and pollinator diversity (^{0}D and ^{1}D) were also significantly higher within the gardens than the unmanaged habitat, with plant and pollinator species richness doubling within the gardens (Fig 4.4a & b). The diversity of their interactions was higher still (0D, 1D and 2D), with the richness of interactions increasing four-fold within the gardens (Fig 4.4c).

The ten most abundant pollinator species for each habitat are given in Appendix 4.2. In the high mountains, seven of these ten species occurred in both gardens and unmanaged habitats, with *Megachile walkeri* the dominant species in both. In the low mountains, six of the ten species occurred in both gardens and unmanaged habitats, with *Syritta fasciata* and *Apis mellifera* the dominant species in both. In accordance with the Hill's diversity estimates (Table 4.1), there tended to be one or two highly abundant pollinator species in each habitat, followed by many rarer species.

Species similarity of plants, pollinators and their interactions

Plants exhibited low levels of similarity between gardens and unmanaged plots in both the high and low mountains, with the similarity of interactions lower still (Fig. 4.5). In contrast, pollinators exhibited much higher levels of similarity between gardens and unmanaged plots in both the high and low mountains (Fig. 4.5). The similarity of plants and interactions decreased steeply with the order of *q*, suggesting that there was high similarity between the presence/absence of species in the gardens and unmanaged plots, but that there were important differences in the relative frequency of dominant species and that when these differences were accounted for the similarity between the two communities decreased.



SYR= Syrphidae, DP = true flies, BOM = Bombylidae, APIS= *Apis mellifera*, APID= other Apidae, HAL= Halictidae, MEG= Megachillidae, CRAB= Crabonidae, VESP= Vespidae, LEP= Lepidoptera, COL= Coleoptera

Figure 4.4. Quantitative bipartite networks of interactions between flowers and insect-visitors in gardens and unmanaged habitats (based upon pooled data) in the high mountains (HM) and low mountains (LM). In each network the rectangles represent plants (bottom row) and pollinators (top row) and the connecting lines represent links between species. The width of the rectangle represents the total number of interactions, and the widths of the connecting lines represent the number of interactions observed for that link. The insects in the top row are grouped by taxonomic groups for simplicity, though interaction analyses within the text were performed on a species level and were based upon individual networks. Plants in the bottom rows represent species, with species names listed in Appendix 4.1.

Table 1. Results of linear mixed-effect models comparing the two habitats (garden versus unmanaged) and the interaction between altitudinal category (high mountain vs. low mountain). Models contained abundance or diversity as the response variable, habitat and altitudinal category as fixed effects and wadi as a random effect. *df* =1 in all models.

	Results of linear mixed-effect models					
	Habitat * Altitude		Habitat		Altitude	
	χ^2	*P*	χ^2	*P*	χ^2	*P*
Plants						
N	13.15	< 0.001	2.58	0.108	0.03	0.860
^{0}D	8.31	0.004	7.39	0.007	0.18	0.671
^{1}D	7.32	0.007	10.83	< 0.001	1.23	0.257
^{2}D	4.99	0.025	9.41	0.002	0.16	0.692
Pollinators						
N	19.54	< 0.001	1.13	0.285	1.68	0.195
^{0}D	12.54	< 0.001	1.18	0.276	0.54	0.460
^{1}D	5.45	0.020	3.18	0.074	0.01	0.988
^{2}D	1.366	0.243	1.73	0.188	0.01	0.959
Interactions						
^{0}D	19.89	< 0.001	1.06	0.304	1.46	0.226
^{1}D	10.78	0.001	3.85	0.049	0.07	0.788
^{2}D	10.14	0.001	1.26	0.262	0.58	0.447

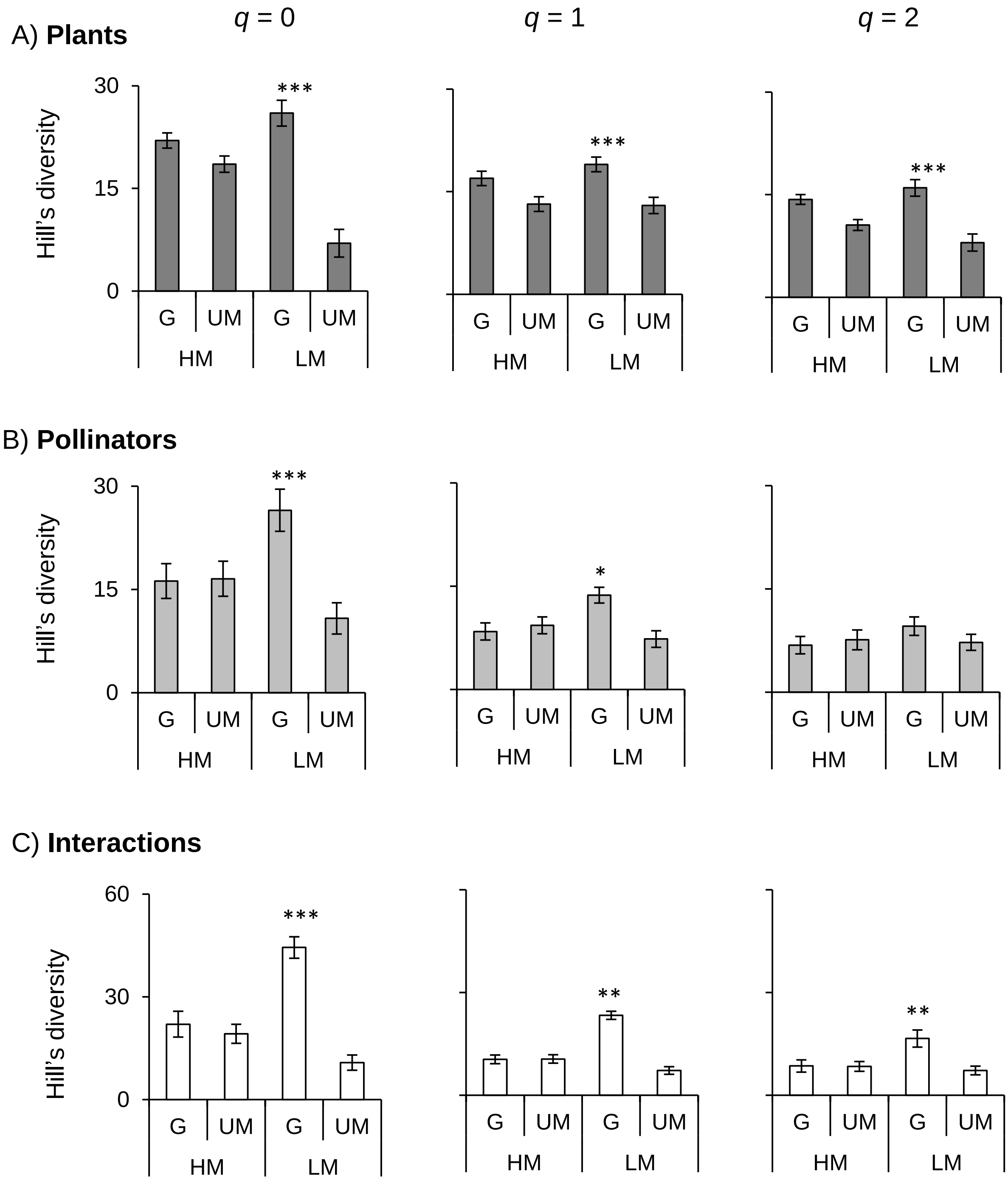
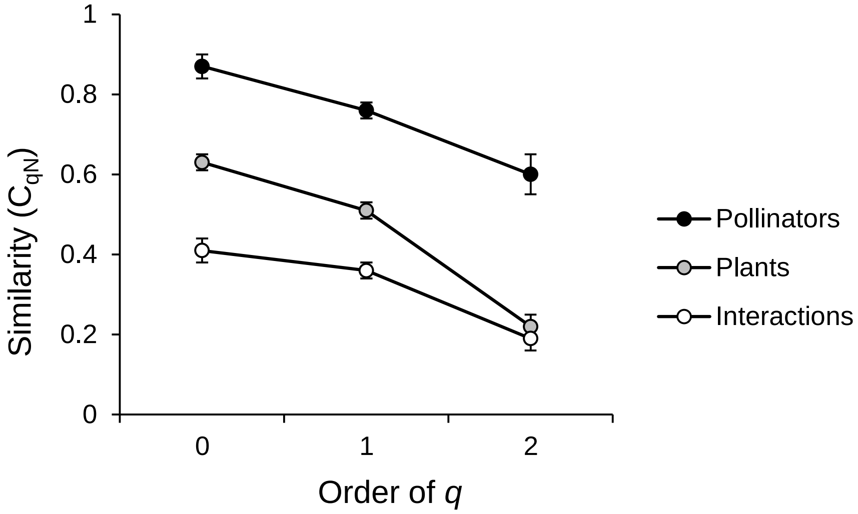


Figure 4.4. Mean Hill's diversity of (a) plants, (b) pollinators, and (c) their interactions, for *q*= 0, 1, 2. Each bar compares the mean diversity (± S.E.) between gardens (G) and unmanaged plots (UM) in the two altitudinal categories, high mountains (HM) and low mountains (LM). Asterisks represent a significant difference between gardens and unmanaged habitat within altitudinal categories as determined by Tukey post-hoc tests.


A) High mountains

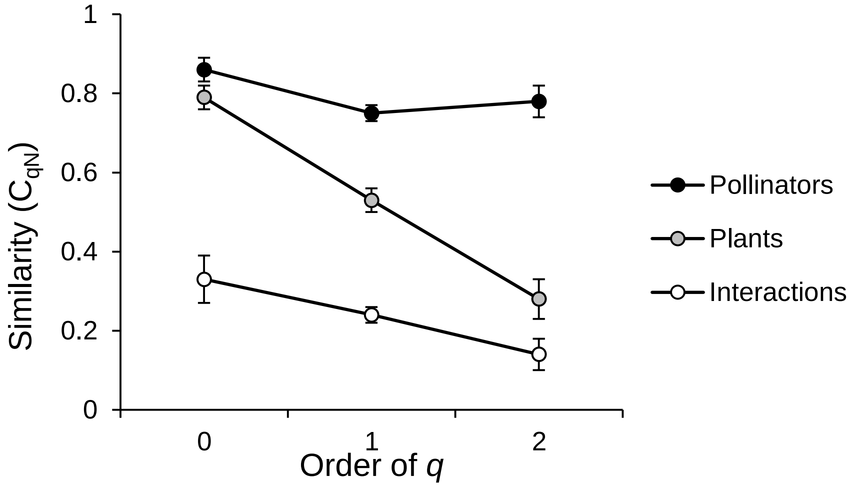

B) Low mountains

Figure 4.5. The similarity profile C_{qN} of species and interactions in gardens and unmanaged plots for a) high mountains and b) low mountains, for q =0, 1, 2. C_{qN} ranges between unity (when communities are identical) and zero (when communities are completely different). Error bars represent standard errors estimated from 200 iterations.

The vast majority of plants and pollinators observed within the unmanaged plots were shared with the gardens, with approximately 90% of the species and interactions from the natural habitat also found within the gardens (Fig. 4.6). Within the gardens, the majority of pollinators were shared with the natural habitat, but the proportion of shared plants and interactions was considerably lower with approximately half of all plants and interactions unique to the gardens. This suggests that the dissimilarities in community structure are primarily due to the presence of novel plant species and interactions within the gardens, and not due to a loss of species or interactions in either habitat.

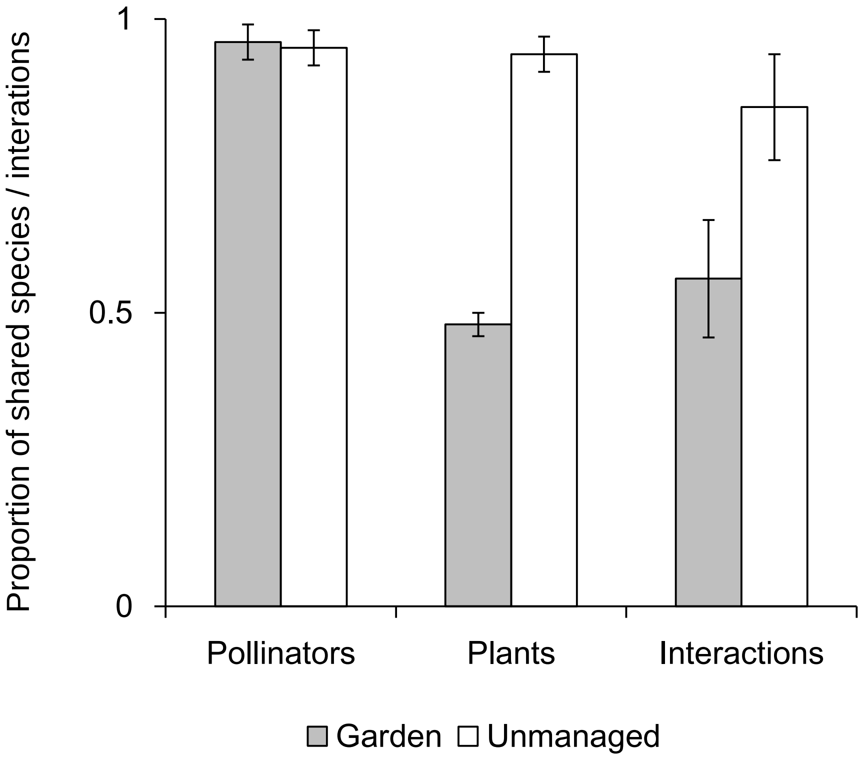


Figure 4.6. Mean relative abundance of shared species and interactions in all gardens and natural plots, estimated using an adjusted Sørenson's similarity index with 200 iterations. Error bars represent standard errors.

4.4 Discussion

Plants and pollinators showed highly contrasting patterns of landscape scale diversity. As predicted, plants followed model A (Fig. 4.1a), with gardens increasing overall landscape heterogeneity by supporting a distinct assemblage of species highly dissimilar to that found in the unmanaged habitat. Pollinators exhibited extremely low levels of turnover across the landscape, with gardens increasing local diversity (in the low mountains) by supporting higher densities of species also present in the unmanaged species pool (Fig. 4.1b). The identity of the plant-pollinator interactions was strongly affected by the composition of the plant communities, with pollinators showing the ability to modify their foraging behaviour. Thus interactions showed even higher levels of turnover than the plants, with gardens and unmanaged habitats containing extremely dissimilar networks of interactions despite supporting the same pollinator species.

The impact of the gardens and the importance of landscape context

The quality of the surrounding habitat affected how the pollinator community responded to the presence of the agricultural gardens. At higher altitudes the natural habitat is relatively undisturbed, with a higher availability of water and containing a high abundance and diversity of wild flora (Norfolk et al. 2013; Ayyad et al. 2000). In this high-quality habitat, gardens supported an equally abundant and diverse plant community as the unmanaged habitat and had no impact upon the abundance or diversity of pollinators or interactions. Conversely, in the low mountains where natural floral resources were scarce, the gardens actively increased the abundance and diversity of pollinators and interactions. Both ornamental and agricultural gardens have been known to boost pollinator abundances in other resource-limited habitats, such as desert environments (Gotlieb et al. 2011), heavily developed cities (Matteson et al. 2008) and intensively managed farmlands (Samnegård et al. 2011). The Bedouin agricultural gardens seem to have a similar positive effect upon pollinator abundances in the low mountains where the surrounding environment is particularly sparse.

Gardens in the poorer-quality landscape received twice as many pollinators as those in the high mountain gardens, despite gardens supporting an equal abundance and species richness of flora. These inflated abundances could be indicative of a crowding effect in the low mountains, with gardens acting as florally-rich islands that collect species from the surrounding sparse habitat. The crowding effect has been documented for arthropods in highly fragmented habitats (Collinge & Forman 1998; Debinski 2000; Zhao et al. 2011), and predicts that when habitat is removed from a landscape, surviving individuals in the disturbed matrix move into the remaining habitat fragments, leading to elevated densities (Grez et al. 2004). In a reversal of typical habitat fragmentation, the human-modified gardens may be acting as resource-rich islands in the low-quality desert habitat, resulting in elevated densities of pollinators within the gardens. In recently fragmented habitats, crowding effects tend to be transient, with inflated densities adjusting to a lower equilibrium within a matter of months (Debinski 2000; Grez et al. 2004), although abundances can be maintained through sustained immigration from neighbouring populations (Bowman et al. 2002). The gardens in the low mountains all date back 50 years or more (Gilbert 2011), so the high abundances of pollinators are unlikely to be transient, but it is possible they are being maintained through sustained immigration from the high mountains.

Contrasting turnover between plants, pollinators and their interactions

Plants exhibited high levels of spatial turnover across the landscape, with distinct communities of species in the gardens and the unmanaged habitat. This was primarily due to the additional presence of cultivated species within the gardens rather than a reflection of a loss of wild plant species: gardens supported the vast majority of the wild flowers (95%) and interactions (85%) present in the unmanaged habitat. These results are consistent with the patterns observed in Chapter 2, and further suggest that the gardens have a positive role in the conservation of native flora in this region.

The presence of cultivated flora led a major restructuring of the plant-pollinator interaction networks, with changes in interaction diversity directly reflecting the modified plant community within the gardens. Pollinators were able to adapt to the novel floral resources within the gardens, with interactions with cultivated flora augmenting those with wild species. Such generalised foraging behaviour has been observed in other systems, with many alien flowers receiving substantial levels of visitation from native pollinators (Bjerknes et al. 2007; Graves & Shapiro 2003; Williams et al. 2011). Alien flora can become well integrated in visitation networks (Memmott & Waser 2002; Vilà et al. 2009) to such an extent that the simulated removal of alien plants can lead to species extinctions if flower visitors fail to reorganise their interactions (Valdovinos et al. 2009). In accordance with other studies, cultivated flora (some of which were alien to the region) were deeply integrated into visitation networks within the gardens and provided important resources for native pollinators.

This is one of the first studies to use the C_{qN} similarity analyses described by Gotelli and Chao (2013), and it seems clear that utilising three measures of both alpha and beta diversity provided substantial insight into the relative abundances of rare, common and abundant species within the two habitats. Pollinators showed a highly uneven distribution of alpha diversity in both habitats, with a high number of rare species accompanied by several highly abundant, dominant species. This phenomenon of widespread rarity appears to be pervasive in bee communities (Williams et al. 2001), and high numbers of singleton species accompanied by several dominant species have been noted in communities of desert bees elsewhere in the Middle East (Potts et al. 2003; Gotlieb et al. 2011) and in North America (Hostetler & McIntyre 2001; Minckley 2014).

Levels of beta diversity also decreased sharply to the order of q for plants, pollinators and their interactions, suggesting that the relative frequency of dominant species (and interactions) differed between the gardens and unmanaged habitats. For plants, this pattern likely reflects the fact that actively cultivated flora tended to be more abundant, thus dominant within the gardens, with the less abundant wild species shared with the unmanaged habitat. Although the vast majority of pollinator species occurred in both habitats (high similarity based upon presence/absence), the modification of the floral community within the gardens seems to have influenced the relative abundances of these species resulting in different dominant species in each habitat.

Conclusions

These results show that interacting species can display highly contrasting patterns of turnover across a shared landscape and provide a clear conceptual framework for explaining the patterns of turnover exhibited by plants, pollinators and their interactions. In this system the enhancement of local habitat had the potential to boost pollinator numbers, but habitat heterogeneity was also required in order to maintain the diversity of plant-pollinator interactions. In terms of management, the results suggests that improvement of local habitat and habitat heterogeneity are both important tools in conservation, but that a combined approach may be necessary in order to conserve the diversity of interactions between species.

Chapter 5.

Seasonal trends in the importance of cultivated and wild flora[*]: *How important are flowering crops for pollinators within the gardens?*

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[*] A modified version of this chapter was published in *Biodiversity and Conservation*: Norfolk, O., Eichhorn, M., & Gilbert, F. (2014). Culturally valuable minority crops provide successional resources for flower visitors in orchard gardens. *Biodiversity and Conservation*, 23, 3199-3217.

Chapter 5. Seasonal trends in the importance of cultivated and wild flora: *How important are flowering crops for pollinators within the gardens?*

Abstract

Agricultural intensification typically has detrimental effects on pollinator communities, but diverse cropping systems that contain sequentially-flowering crops have the potential to benefit pollinators through the provision of additional floral resources. In this chapter I look specifically at the importance of cultivated flora for pollinators in the ten town gardens, which were surveyed consecutively in 2012 and 2013. Plant-pollinator interactions in gardens and unmanaged plots were surveyed across a four-month period in two years with distinct environmental properties (pre-flood and post-flood). Despite containing an equal abundance and diversity of wild plants as unmanaged habitat, gardens supported a higher abundance and diversity of pollinators due to the additional presence of cultivated flora. Visitation networks exhibited dramatic intra-annual changes in composition, with cultivated plants becoming increasingly important in later months. Trends were conserved across two years despite highly contrasting rainfall. Several key crop species were important in shaping the structure of the networks, the majority of which were herbs with strong cultural significance (fennel, rosemary, mint) grown incidentally alongside the primary orchard crops. Minority crops are frequently overlooked in agricultural systems due to their low economic value, but these results show that they can have a dramatic influence upon the structure of visitation networks, increasing both pollinator abundance and diversity, and emphasising the link between cultural practices and biodiversity conservation.

5.1 Introduction

Many agricultural pollination studies focus on the intensive agricultural systems that dominate temperate regions (Holzschuh et al. 2013; Le Féon et al. 2010; Steffan-Dewenter & Westphal 2008), but on a global scale 90% of all farms are less than two hectares in size (Tscharntke et al. 2012), with smallholder farms and homegardens making an essential contribution to food security in poorer regions (Horlings & Marsden 2011). These small-scale agricultural systems typically serve just one household and are used primarily for subsistence crops, with cash crops sometimes grown to supplement household incomes. They often employ the principles of diverse farming and habitually cultivate a range of crops that ripen in succession throughout the year (Fernandes & Nair 1986; Jose & Shanmugaratnam 1993). The presence of sequentially ripening and flowering crops is likely to influence and potentially increase the availability of floral resources for insect visitors across the entirety of their flight season.

In contrast to temperate systems, the actively irrigated Bedouin gardens have been shown to support a higher diversity of wild plants than the unmanaged desert habitat (Norfolk et al. 2013), with wild plants in the gardens receiving elevated levels of floral visitation (Norfolk & Gilbert 2014). Ornamental gardens have also been shown to enhance bee abundance in neighbouring Israel (Gotlieb et al. 2011) and in this chapter I build upon these previous studies in order to determine the specific importance of crops for pollinators in this arid environment.

Smallholder farms in Africa and the Middle East tend to be heavily reliant on the economic returns from pollinator-dependent crops such as fruits and vegetables, and this leaves them particularly vulnerable in the face of pollinator declines (Gallia et al. 2009; Kasina et al. 2009). Despite the region’s vulnerability, there is a marked geographical bias in the focus of pollination research, with the vast majority relating to temperate regions, in particular Europe and the USA (Archer et al. 2014; Mayer et al. 2011). The lack of research in poorer regions such as Northern Africa appears to be linked to a lack of funding opportunities and research infrastructure (Archer et al. 2014). Understanding the drivers of pollinator losses is important for tackling future food security and it is unfortunate that the most at-risk nations are those lacking the relevant research. This research is aimed at filling in some of the knowledge gaps relating to smallholder agriculture and pollinators in this under-studied hyper-arid environment.

This chapter uses a visitation network approach to quantify the plant-pollinator interactions within ten town gardens across two four-month periods. The specific aims were: (1) to evaluate the relative importance of cultivated and wild flora for pollinators; (2) to assess whether the sequential flowering of crops influences the structure of visitation networks across the year; and (3) to determine which plant species are most integral to the structure of the visitation networks.

Plant-pollinator interactions were also compared between the gardens and unmanaged plots to assess whether the additional cultivated flora of the gardens have a positive impact upon pollinator abundance and diversity in the area. The results demonstrate that the town gardens supplement wild floral resources through the provision of sequentially-flowering crops throughout the season.

5.2 Methods

Data collection

Monthly surveys were carried out from April - July in 2012 and 2013 in ten gardens within the town of St Katherine. Pollinator surveys followed the protocol described in section 3.2. In 2013 six control plots in areas of unmanaged land were also surveyed within the town to give an indication of the plants and insects that would be present without active cultivation of the gardens. The control plots of unmanaged land were chosen to typify the desert habitat of the area, with sandy soil and low-growing desert shrubs. The location of these control plots was determined by the availability of suitable sites within the town and was highly constrained by the density of gardens and buildings. See Figure 5.1 for a map of the study site.

Average monthly daytime temperatures ranged from 22°C in April, 28°C in May, up to 32°C in June and July (RP5, 2013). No rain was recorded during the study period, but there were heavy floods at the beginning of 2013; thus water availability was higher in the second year (personal obs), and I therefore classify 2012 as a pre-flood and 2013 as a post-flood year.

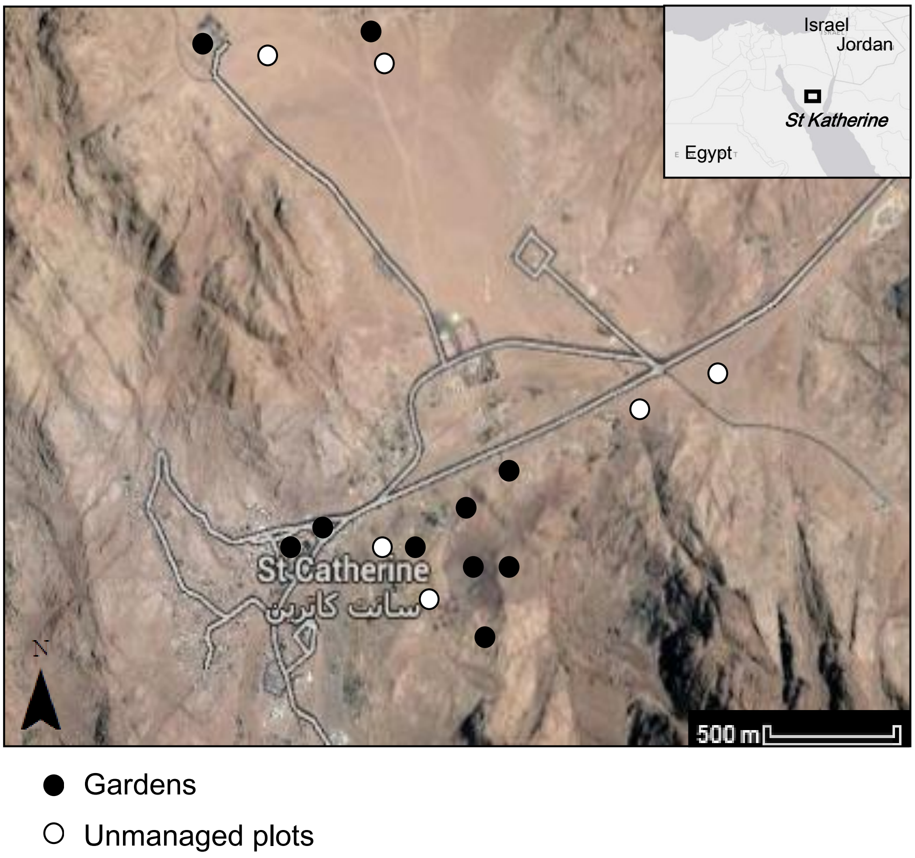

Figure 5.1. Map of study site in St Katherine Protectorate, South Sinai, with locations of gardens and unmanaged plots.

Statistical analyses

Visitation networks

In order to compute network statistics, visitation webs were created for each plot as quantitative interaction matrices with n rows (representing plant species) and m columns (representing insect species), with the value at the intersect representing the number of interactions observed between flower and insect. Monthly networks were constructed for each garden in both years (a total of 80 networks) and the control plots in 2013 (24 networks). Each garden network was then split into two networks, one containing only interactions with cultivated plants and the other containing only interactions with wild plants (a total of 160 networks).

Network-level statistics were computed in R package *bipartite* (Dormann et al. 2009). The number of interactions, number of links and interaction diversity were computed for each network. Interaction diversity was defined as the exponential of the Shannon diversity of interactions (Dyer et al. 2010). All statistical analyses were performed with R.3.02 software (R Team, 2013).

Linear mixed-effect models from the R package *lme4* (Bates et al. 2011) were used to test for: (1) seasonal patterns in the abundance and diversity of cultivated and wild flora; and (2) seasonal patterns in the number and diversity of pollinator interactions experienced by cultivated and wild plants. All models included *month* and *cultivated/wild* as the fixed effects and *garden* (N=10) as a random effect to account for spatial variation. Response variables tested were: (a) floral abundance and species richness; and (b) number of interactions, number of links and interaction diversity. The data from 2012 and 2013 were pooled for the cultivated/wild analyses, because although there were a higher number of interactions in the post-flood year (lmer: year; χ^2=77.1, df=1, P = 0.001), there were no significant differences between the seasonal patterns in the two years for the mean number of interactions (month*year; χ^2=2.88, df=3, $P = 0.411$), number of links (month*year; $\chi^2 = 3.11$, df=3, $P = 0.375$) or interaction diversity (month*year; $\chi^2 = 1.10$, df=3, $P = 0.778$). Model fit was based upon AIC and followed Zuur et al. (2009), with the significance of fixed effects and their interactions tested by comparing models with a likelihood ratio test (distributed as χ^2).

Species similarity indices

Species similarity of insects visiting wild and cultivated plants were compared using three complementary measures of beta diversity derived from C_{qN} which together provide insight into the degree of overlap in rare, common and abundant pollinators (Gotelli & Chao 2013). As with Hill's numbers, *q* is a parameter that determines sensitivity to relative abundance (Hill 1973) and N is the number of assemblages (in this case $N = 2$). C_{0N} (the Sørenson similarity index) is an incidence-based index weighted towards rare species; C_{1N} (the Horn overlap index) is an abundance-based similarity index weighted towards common species; and C_{2N} (the Morisita-Horn similarity index) is an abundance-based similarity index weighted towards abundant species. C_{qN} ranges between unity (when communities are identical) and zero (when communities are completely different). The three indices were calculated for cultivated and wild pollinators (pooled from 2012 and 2013) using SPADE with 200 iterations (Chao & Shen 2010).

Topological importance

Topological importance was used to determine the relative importance of cultivated and wild species and to assess their integration within the structure of the networks. One of the most direct measures of topological importance was used, unweighted degree, because despite its relative simplicity it performs well when compared to other topological centrality measures (Pocock et al. 2011). Topological importance (degree) is defined here as the total number of pollinator species that visited each plant species;

a well-linked plant will have a higher topological importance and is likely to be a key species within the network. Another index, partner diversity, was measured as the exponential Shannon diversity of the insect visitors. Degree and partner diversity were calculated for plants using *specieslevel* in the package *bipartite* (Dormann 2011) from cumulative networks of all ten gardens.

The average a) topological importance and b) partner diversity of pollinators were compared between wild and cultivated plants using linear-mixed-effects models with *cultivated/wild* as a fixed effect and *month* as a random effect. The model for topological importance was fitted with a Poisson error distribution and partner diversity with a normal error distribution. As with the previous models, data from 2012 and 2013 were pooled.

Gardens and control plots

Floral abundance, floral species richness and the three network statistics were compared between the 2013 gardens and unmanaged control plots. Plot type (garden or control) was included as a fixed effect, with the identity of each plot as a random effect. In order to determine whether wild plants received more visits within the gardens or the control plots, the above models were also run with just wild species for comparison.

5.3 Results

Characterisation of the insect-flower interactions within the gardens

A total of 2298 plant-pollinator interactions were observed between 114 pollinator species and 59 plant species within the gardens over the course of the two years. Approximately three quarters of these interactions were with cultivated plants (1579 interactions) and one quarter with wild plants (621 interactions). Pollinators interacted with 33 wild species and 26 cultivated plant species (Appendix 5.1), the most abundant of which were wild species *Achillea santolina* (17% of all garden visits; present in eight of the ten gardens) and *Chenopodium album* (16 %; eight gardens); and cultivated species *Beta vulgaris* (14 %; seven gardens) and *Foeniculum vulgare* (10 %; eight gardens).

The most abundant visitors to cultivated plants belonged to the order Hymenoptera (34%)(Table 5.1). Solitary bees were the most abundant group of Hymenoptera visiting cultivated plants (39%), followed by solitary wasps (34%) and managed honeybees (28%). The most abundant visitors to wild species also belonged to the order Hymenoptera (41%). Of these, solitary bees were the most abundant group (68%), followed by honeybees (17%) and solitary wasps (12%).

Many of the most abundant pollinator species were observed visiting both cultivated and wild plant species (see Appendix 5.2 for species list), with honeybees a common visitor to both. These shared visitors included generalist, cosmopolitan species with ranges spreading across Europe and Northern Africa such as *Apis mellifera*, *Lampides boeticus* (Long-tailed Blue butterfly) and hoverflies *Eupeodes corollae* and *Syritta fasciata*. Flowering crops also supported species with higher conservation importance, such as the leaf-cutter bee *Megachile walkeri* (endemic to the Middle East), the colletid bee *Hylaeus sinaitus* (endemic to Southern Sinai) and *Hylaeus sp A*, a previously undescribed species (Dathe, pers. comm.). Despite some differences in the top ten species, the overall species similarity values were high between pollinators utilising cultivated and wild species, with beta diversity values C_qN of 0.812 ± 0.03 (q=0), 0.674 ± 0.02 (q=1) and 0.726 ±0.05 (q=2). The incidence-based estimate (q=0) was higher than the abundance-based estimates (q= 1,2), with rarer species showing a higher degree of overlap than common or abundant species.

Table 5.1. Total number of interactions (N) and total number of species (S) observed from each arthropod order, with families included for important pollinating groups.

	Cultivated flora				Wild Flora			
	N		**S**		**N**		**S**	
	2012	2013	2012	2013	2012	2013	2012	2013
Hymenoptera	214	347	42	47	166	163	30	23
Apidae	77	106	10	5	95	93	8	7
Halictidae	21	33	11	5	26	14	4	3
Colletidae	67	60	2	3	11	8	3	2
Megachillidae	20	23	5	4	25	23	9	3
Crabonidae	24	123	10	28	8	25	6	8
Lepidoptera	103	103	6	8	24	21	8	6
Lycaenidae	100	96	3	4	19	15	5	4
Nymphalidae	1	0	1	0	0	0	0	0
Pieridae	2	7	2	4	3	6	3	2
Diptera	248	339	27	22	85	81	19	17
Bombylidae	0	0	0	0	0	8	0	5
Syrphidae	138	300	8	10	66	65	8	5
Coleoptera	45	172	8	9	24	52	11	8
Hemiptera	4	11	3	1	4	1	1	1
Total:	**614**	**965**	**86**	**83**	**303**	**318**	**69**	**55**

Seasonal changes in network structure and the importance of cultivated and wild flora

Gardens contained a significantly higher floral abundance of cultivated plants than wild plants (Fig 5.2a; χ^2=11.98, df=1, $P < 0.001$). Despite the higher abundances of cultivated flora, wild plants had the higher species richness (Fig 5.2b; χ^2 =32.27, df=1, $P < 0.001$). Cultivated plants showed some increase in floral abundance across the year, but there was no seasonal interaction between the floral abundance of cultivated or wild plants (month*cultivated/wild: χ^2 =6.08, df=3, P =0.108). There was a strong seasonal interaction with the species richness of cultivated and wild flora; cultivated species richness stayed relatively constant in all four months, but wild plant species richness showed a steep decrease in June and July, reducing to half its initial level.

There were strong seasonal trends in the importance of cultivated and wild species within the visitation networks, with the same pattern observed in both two years, pre-flood and post-flood (Fig 5.3). In April 2012 and 2013 approximately 50% of interactions within the gardens involved wild plant species, but the proportion of interactions with wild plants decreased dramatically throughout the season and by July over 85% of interactions involved cultivated plants. Analysis of the network properties confirmed that these trends were highly significant (Table 5.2), with number of interactions (Fig 5.4a), number of links (Fig 5.4b) and interaction diversity (Fig 5.4c) all increasing for cultivated plants and decreasing for wild plants in the later months.

Topological importance

Topological importance (total number of insect species that visited each plant species) was used to estimate the relative importance of wild and cultivated plants within the visitation networks. There was a clear turnover in the identity of the topologically important species across the season (Table 5.3) with several key cultivated species recurring in consecutive years; *Eruca sativa* (rocket) and *Rosmarinus offinalis* (rosemary) in April; *Foeniculum vulgare* (fennel) from May through to July; *Origanum syriacum* (oregano) and *Medicago sativa* (alfalfa) in June; and *Mentha longifolia schimperi* (habak mint) and *M. sativa* (alfalfa) in July.

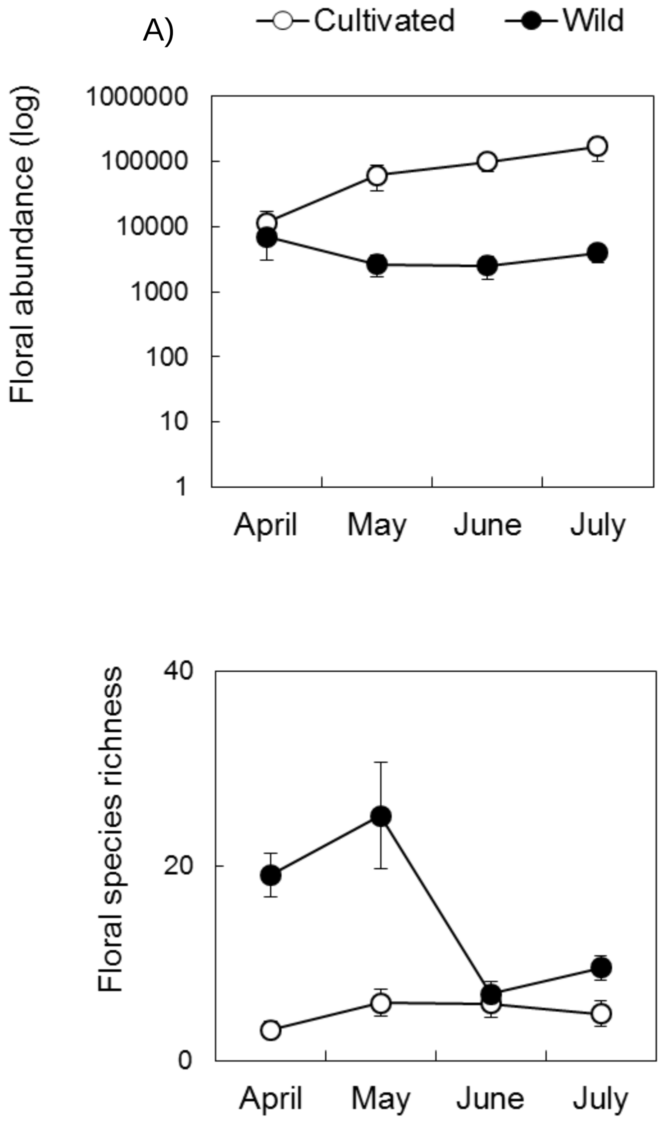

Figure 5.2. Mean a) floral abundance, and b) floral species richness, of cultivated and wild plants in the gardens across 2012 and 2013 (± S.E.M).

Table 5.2. Seasonal variations in cultivated and wild plant interactions. Output from linear mixed effects models containing cultivated (cultivated or wild) and month as fixed effects and garden as a random factor.

		lmer output		
Response variable	Fixed effects	X^2	*df*	*P*
Number of interactions	*month*cultivated*	8.39	3	0.039
	cultivated	14.18	1	0.001
Number of links	*month*cultivated*	75.41	3	0.001
	cultivated	69.54	1	0.001
Interaction diversity	*month*cultivated*	14.52	3	0.002
	cultivated	0.77	1	0.380

Cultivated plants tended to have higher topological importance than wild species, with an average of 4.9 (± 0.7) links to cultivated species and 3.2 (± 0.4) to wild species (χ^2=30.2, df=1, P <0.001). There was no significant difference between the diversity of insects visiting cultivated and wild species, with an average partner diversity of 0.89 (± 0.09) associated with cultivated species and 0.70 (± 0.08) with wild species (χ^2= 2.59, df=1, P = 0.108).

Gardens versus unmanaged plots

In 2013 the gardens contained significantly higher floral abundances (Fig 5.5a) and floral species richness (Fig 5.5b) than equal-sized plots of unmanaged land (abundance: χ^2= 13.80, df=1, P < 0.001, species richness: χ^2= 14.31, df=1, P <0.001), with a significantly higher average number of insect-flower interactions (χ^2= 19.68, df=1, P < 0.001). When cultivated plants were not considered, there was no difference between wild plant floral abundance (Fig 5.5a) or floral richness (Fig 5.5b) (abundance: χ^2= 0.57, df=1, P = 0.447, species richness: χ^2= 2.37, df=1, P = 0.123). Observed interactions with wild plants were still significantly more numerous within the gardens (Fig 5.5c: χ^2= 4.73, df=1, P = 0.030), with a higher average number of links with insect species (χ^2= 5.25, df=1, P = 0.022). There was no difference in the average wild plant interaction diversity in gardens and unmanaged plots (χ^2= 3.38, df=1, P = 0.066).

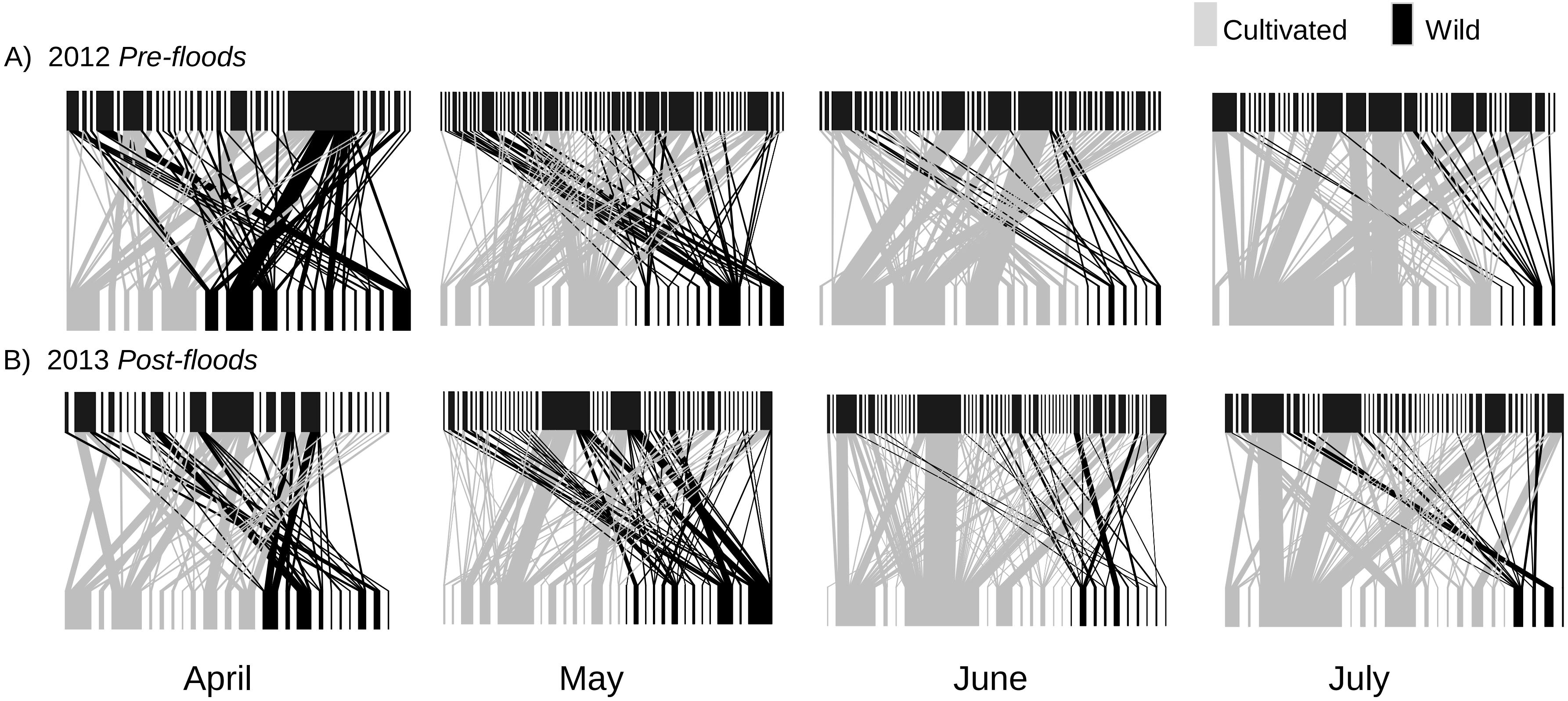

Figure 5.3. Quantitative plant-pollinator visitation networks for gardens across the sampling season, (A) pre-floods in 2012 and (B) post-floods in 2013. In each network the rectangles represent pollinator species (top row) and plant species (bottom row), and the connecting lines represent links between species. The width of the rectangle represents the total number of visits made, and the widths of the connecting lines represent the number of visits observed for that link. Links with cultivated plants are shown in grey and links with wild plants in black.

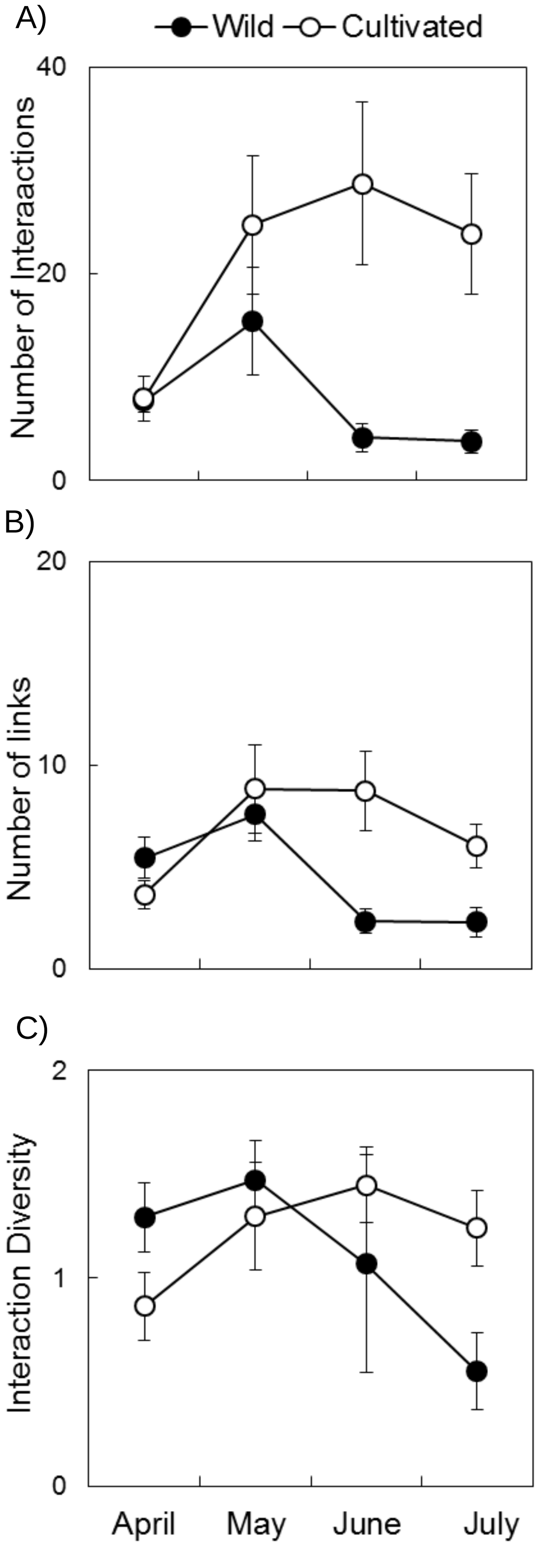

Figure 5.4. Network-level metrics for cultivated and wild plants within the gardens; mean number of a) interactions, b) links per network, and c) interaction diversity (± S.E.M).

Table 5.3. Seasonal trends in topologically important species, calculated from cumulative networks of all gardens. Plants with the highest topological importance are highlighted in bold. * indicates cultivated.

		2012				2013		
		Topological importance	Partner diversity	% of total links		Topological importance	Partner diversity	% of total links
April	***Eruca sativa***	**10**	**1.83**	**15**	***Eruca sativa***	**8**	**1.55**	**13**
	Caylusea hexagyna	**10**	**2.24**	**15**	**Salvia officinalis*	4	0.79	7
	**Rosmarinus officinalis*	7	1.82	10	**Rosmarinus officinalis*	4	1.39	7
	Arabidopsis kneuckeri	7	1.11	10	*Zilla spinosa*	4	1.24	7
	Zilla spinosa	4	0.79	6	*Alkanna orientalis*	4	1.08	7
May	***Foeniculum vulgare***	**19**	**2.46**	**18**	***Foeniculum vulgare***	**15**	**1.73**	**15**
	Petroselinum crispum	**19**	**2.47**	**18**	*Diplotaxis harra*	12	2.27	12
	Peganum harmala	18	2.76	17	*Peganum harmala*	10	1.66	10
	** Beta vulgaris*	13	2.41	13	*Zilla spinosa*	10	1.59	10
	**Allium cepa*	6	1.75	6	**Eruca sativa*	7	1.5	7
June	*** Foeniculum vulgare***	**22**	**2.76**	**31**	***Foeniculum vulgare***	**32**	**2.31**	**33**
	** Beta vulgaris*	11	1.7	16	**Allium cepa*	14	2.1	15
	** Medicago sativa*	5	0.56	7	**Origanum syriacum*	10	2.25	10
	Caylusea hexagyna	5	1.61	7	*Ballota undulata*	7	1.48	7
	** Origanum syriacum*	4	1.15	6	**Medicago sativa*	5	0.62	5
July	*** Foeniculum vulgare***	**15**	**2.21**	**25**	***Foeniculum vulgare***	**19**	**2.0**	**25**
	Achillea fragrantissima	14	2.27	23	**Mentha longifolia schimperi*	16	2.43	21
	Ochradenus baccatus	6	1.67	10	*Achillea santolina*	8	1.91	11
	** Medicago sativa*	5	1.02	8	**Beta vulgaris*	5	1.3	7
	** Mentha longifolia schimperi*	5	1.56	8	**Medicago sativa*	4	1.28	5

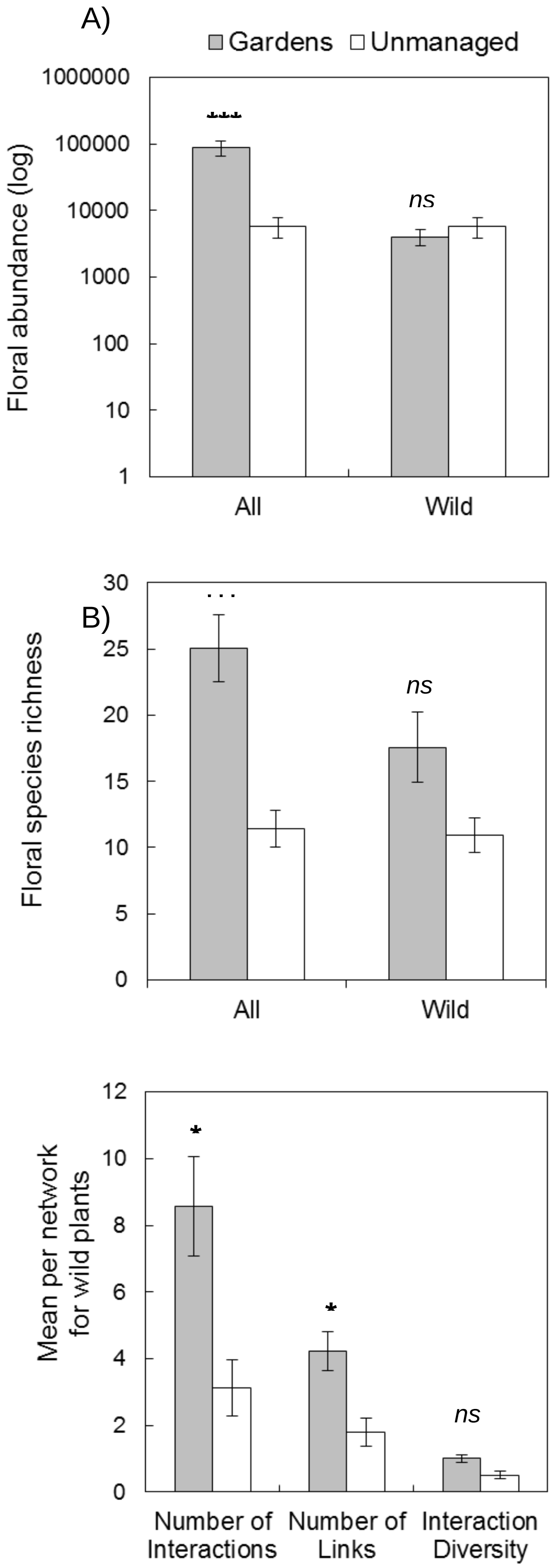

Figure 5.5. Comparison of (A) floral abundance, (B) floral species richness, and (C) wild plant network statistics, in gardens versus unmanaged plots in 2013. Values represent the mean per 500 m² plot (± SEM) across the year.

5.4 Discussion

Plant and pollinator communities within the gardens

The town gardens supported an abundant and diverse community of spontaneously occurring wild flora, with abundances matching those found in surrounding unmanaged habitat. Despite this, the majority of pollinators were found utilising the crops, which provided a more abundant (though less diverse) floral community than the wild species. Wild flora has previously been shown to provide an important resource for flower visitors in the ground cover of apple orchards in Europe (Rosa García & Miñarro 2014). In the Bedouin gardens flora in the ground cover between the trees also provided important resources for pollinators, but flowering vegetables and herbs were more significant for the pollinator community than wild flora.

As well as supporting many common pollinating species, such as honeybees and hoverflies, cultivated plants were also visited by a number of regionally endemic solitary bees, such as *M. walkeri* (Middle East) and *H. sinaitus* (Sinai). There was considerable overlap in the insect species visiting cultivated and wild flowers, with beta diversity estimates confirming high similarity between the two communities. The incidence-based diversity estimate was higher than the abundance-based estimates, with rarer species showing a higher degree of overlap than common or abundant species. This implies that cultivated flora are not just visited by dominant generalist species, but provide resources for many of the rarer visitors that also visit wild species.

Seasonal changes in the importance cultivated flora

Analysing the temporal changes in the insect-flower interactions revealed dramatic seasonal patterns in the importance of cultivated and wild flora within the visitation networks. In spring, wild plants played a large role within the networks, but in later months the majority of interactions were with cultivated flowers. This decline in wild plant interactions coincided with a decrease in wild flower species richness within the gardens. Pollinator abundance has been positively linked to floral species richness in other agro-ecosystems (Holzschuh et al. 2008; Kennedy et al. 2013) and it appears that cultivated plants provide an alternative source of nutrition for insects during the hotter and drier months of the year, when wild plant floral richness is low.

The distinct temporal trend in the importance of cultivated plants was highly conserved across both years. Such a low level of inter-annual variation is particularly striking because heavy floods at the beginning of 2013 meant that water availability was considerably higher in the second year. There was a clear succession of key cultivated species, which played an integral role in network structuring across the four-month period. The same topologically important species occurred in both years and this may help to explain why the visitation networks exhibited such similar patterns despite the extreme environmental variation.

Seasonal planting typically provides households with year-round food security, but none of the topologically important plant species were food staples and all formed relatively minor parts of local peoples’ diets in the form of salads and herbs. In fact many of the herbs that were deeply involved in the network structure (fennel, oregano, mint, rosemary) have a strong cultural significance, being widely consumed in Bedouin tea and used in traditional herbal medicines (Zalat & Gilbert 2008). The link between cultural practices, traditional ecological knowledge and biodiversity conservation has been widely noted (Barthel et al. 2010; Ormsby & Bhagwat 2010; Maffi 2005), and it is striking to think that a change in drinking preferences (from mint tea to instant coffee) could have serious consequences for pollination networks in this region. The inclusion of plants and flowers of cultural importance alongside food crops seems to have both social and ecological benefits that likely apply in other homegarden systems.

Conservation potential of agricultural gardens in arid regions

Agricultural gardens can boost flower-visitor abundances in heavily developed cities (Matteson et al. 2008), as can ornamental gardens in intensively managed farmlands (Samnegård et al. 2011); the Bedouin gardens seem to have a similar beneficial effect in a hyper-arid desert landscape, where particularly low nutrient levels and water availability limit floral abundances in the surrounding habitat. In our study, gardens contained more floral resources, with higher rates of pollinator visitation, than plots of unmanaged land within the town. Cultivated plants provided an important resource for these flower-visitors, but not at the expense of wild plants, which received more flower-visitors inside gardens than they did outside.

Cultivated flowers became increasingly important later in the season (June and July), when temperatures can exceed 30°C (RP5 2013) and water becomes more scarce. Similar seasonal patterns have been observed with bee abundances in ornamental gardens in Israel (Gotlieb et al. 2011); in early spring, gardens and natural habitat contained equal bee abundance, but by June and July, numbers in the natural environment had declined and there was a six-fold increase in bee abundance within the gardens. With global temperatures rising and rainfall becoming more erratic, we predict that such gardens will provide increasingly important habitat for desert species.

This chapter does not directly address the impact that the pollinators have upon the eventual pollination success of crops or wild flora; however increased visitation by wild insects has been linked to increased fruit set in 41 crop systems worldwide (Garibaldi et al. 2013) and wild bees are known to improve fruit set in several crops found within the gardens, such as tomatoes (Greenleaf & Kremen 2006a), alfalfa (Cane 2002) and almond (Kennedy et al. 2013). Increased visitation rates to crops seem likely to bring agricultural benefits, but the high floral abundances found within the irrigated gardens could pose a risk to native flora if pollinators are attracted away from wild species. Previous research in the region has shown that the seed set of two species of native plants is not affected by the presence of the gardens, and that native plants within the gardens tended to be larger in size than those in the surrounding natural habitat (Norfolk & Gilbert 2014). This suggests that the gardens do not have a negative effect on the pollination success of wild flora, although further research to rule out dilution effects would be helpful.

Conclusions

These results highlight the benefits of under-cropping within orchards and small-scale farms, demonstrating that cultivated flora can supplement wild floral resources, particularly at the end of the flowering season. These traditional agricultural gardens enhanced the abundance and diversity of pollinators above those in the unmanaged desert habitat, whilst maintaining the number of interactions with wild plant species. Minor crops with low economic but high cultural importance were the most utilised by flower visitors, and were strongly involved in shaping the structure of visitation networks, emphasising the positive link between cultural practices and biodiversity conservation.

Chapter 6.

Pollination services to an orchard crop[*]:

Do wild pollinators within the gardens improve the quality of pollination in almond?

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[*] A modified version of this chapter is under review at *Agriculture, Ecosystem and Environment:* Norfolk, O., Eichhorn, M. & Gilbert, F. (2014). Flowering ground vegetation increases wild pollinator densities and enhances fruit set of an orchard crop. *Agriculture, Ecosystems and Environment,* In review.

Chapter 6. Pollination services to an orchard crop: *Do wild pollinators within the gardens improve the quality of pollination in almond?*

Abstract

Agricultural conversion of natural habitat tends to have negative consequences for wild pollinators, undermining the provision of pollination services. Designing agricultural landscapes that can support stable pollinator populations could help slow declines and boost crop productivity. In South Sinai, almond is cultivated in traditional gardens that contain a mixture of fruit trees inter-planted with vegetables and herbs. This chapter investigated the relative contribution of honeybees and wild insects for the pollination of almond trees and assessed how flowering ground vegetation influenced pollinator densities and fruit set. Expectations were that the presence of simultaneously flowering plants would attract pollinators into the orchards and facilitate enhanced almond pollination. The results showed that almond was highly dependent on insect pollination, with bagged flowers producing less than 8% of the fruit set of insect and hand-pollinated flowers. Fruit set was correlated with wild pollinator visitation, but not honeybee visitation and the presence of honeybee hives had no effect upon fruit set. The abundance and species richness of flowering ground vegetation was positively related to pollinator densities within the gardens and was associated with enhanced fruit set. Over half of the flowering ground flora were minority crops grown alongside almond, suggesting that facilitation can occur between simultaneously flowering crops. If mutual facilitation can occur between other crop species, then diverse cropping systems might provide a mechanism for enhancing pollination services across agricultural landscapes.

6.1 Introduction

Agriculture is heavily dependent on the pollination services provided by both managed and wild insects, with an estimated 75% of world crops benefiting from insect pollination (Klein et al. 2007). Despite their ecological and economic importance, pollinators are experiencing widespread declines (Potts et al. 2010). Many commercial farms temporarily import honeybees in order to meet the pollination requirements of short-flowering crops, but the global stock of honeybees is growing at a slower pace than the increasing demand for agricultural pollination (Aizen & Harder 2009). Wild pollinators currently fill this pollination deficit, and even when honeybees are present the additional presence of wild pollinators can enhance crop fruit set (Garibaldi et al. 2013). Unfortunately agricultural conversion of natural habitat tends to have negative implications for the wild pollinators on which it depends (Ferreira et al. 2013); in order to secure the future productivity of pollinator dependent crops, it may be necessary to rethink the design of current agricultural systems and aim to create agricultural landscapes that can support stable populations of wild pollinators.

Almond is dependent on insect pollination, with commercial farmers relying heavily on the importation of rented honeybee hives. Approximately 70% of the world's almonds are produced in California (FAO, 2014) and during the flowering season this requires the services of approximately half of the honeybee hives across the entire United States (Sumner & Boriss, 2006). This industrial scale procedure would not be required if wild pollinators occurred in sufficient numbers to fulfil the pollination requirements of the crops; however intensive orchard landscapes do not provide the resources required by wild pollinators. Almond is a short-flowering crop, flowering for just two to three weeks in early February. During this time orchards are full of floral resources, but in order to support stable pollinator populations an alternative source of forage must be provided for the rest of the season.

One way in which to secure wild pollination services is to maintain areas of natural or semi-natural habitat in the vicinity of pollinator-dependent crops (Ricketts et al. 2008; Kennedy et al. 2013); Californian almond orchards situated in landscapes containing a higher proportion of natural habitat received increased levels of wild bee visitation (Klein et al. 2012) and the same was found in cherry orchards that were surrounded by bee-suitable habitats (Holzschuh et al. 2012). In both these examples, the increased visitation from wild pollinators actively increased pollination effectiveness and led to enhanced fruit set. In intensive landscapes where natural habitat is not available, wild pollinators can still benefit from the introduction of floral resources into the orchards themselves. Planting vegetation strips between trees can increase pollinator densities in almond orchards (Klein et al. 2012), and flowering ground vegetation can benefit pollinators in cherry (Holzschuh et al. 2012) and apple orchards (Rosa García & Miñarro 2014).

In South Sinai, almond is widely cultivated within the Bedouin gardens. The sparsely populated landscape contains a high proportion of semi-natural habitat, but unlike the previously studied systems in California, these irrigated gardens contain higher levels of floral resources than the uncultivated, semi-natural habitat (Norfolk et al. 2014). The low-intensity orchard gardens contain an abundance of flowering ground vegetation beneath the trees, which are utilised by an abundant and diverse community of wild pollinators (Norfolk et al. 2014). As shown in previous chapters, this flowering ground vegetation allows the gardens to support higher abundances of pollinators than the surrounding habitat. This chapter assesses how this affects the pollination services provided to orchard crops, testing the hypothesis that the presence of simultaneously flowering plants will draw additional pollinators into the gardens and facilitate increased visitation and fruit set in almond.

Thus the aim of this chapter is to evaluate how fruit set in almond is influenced by honeybee and wild pollinator visitation and to assess whether the presence of flowering ground vegetation has an effect upon pollinator densities and subsequent fruit set in almond. Honeybees (or any kind of social bee) are not native to South Sinai, and for centuries almond has been successfully cultivated there without their presence. The recent (1990s) introduction of hives for the production of honey allows us to compare sites with and without introduced honeybees and to assess their relative contribution to the pollination of almond. Specifically the following hypothesese were addressed:

(1) Insect pollination enhances fruit set in almond trees compared to fruit set after wind pollination alone.

(2) Honey bees and wild bees are similarly effective at providing pollination services and the introduction of managed honeybee hives will have a positive effect upon fruit set.

(3) Flowering ground vegetation within the orchards will have a positive effect upon fruit set, supporting a facilitation hypothesis.

The results showed that pollination effectiveness of wild pollinators surpassed that of honeybees and that the presence of flowering ground vegetation within gardens increased pollinator densities and enhanced fruit set.

6.2 Methods

Study site

The Bedouin gardens typically contain a variety of orchard produce (almond, apple, pear, apricot, fig, olive, pomegranate), inter-planted with vegetables and herbs for domestic use. Gardeners generally weed around the immediate vicinity of trees, but wild plants are tolerated within the orchards and frequently occur at higher numbers than they do in the surrounding habitat (Norfolk et al. 2013). Gardens are organic; no pesticides or herbicides are used on the trees and goat manure is the only fertiliser. Orchard produce is not produced commercially, but are primarily for local use.

Almond is the most abundant orchard fruit, grown in 96% of all gardens with an average of 20 trees per garden (Zalat et al. 2001). Almond trees tend to be planted in informal rows with an average spacing of 8 m (± 0.94) between individual trees, though some gardens contain single, isolated trees. The average size of gardens is 2000 m^2, with areas ranging from 500 m^2 to >6000 m^2. Almond is the first fruit tree to flower in the region, allowing us to study the impact of ground flora on almond pollination without any confounding competition from other flowering orchard crops.

Honeybees are not native to South Sinai, but were introduced in the 1990s for the production of honey (Semida & ElBanna 2006). Hives are common close to the towns of St Katherine and Abu Selah, but are absent in the high mountains. Thirty trees were selected at random from three wadis within the vicinity of St Katherine, two of which contained managed honeybee hives: ten trees were within the town gardens of St Katherine (~1500 m a.s.l.; hives present), ten within the low mountains gardens of wadi Itlah (~1350 m a.s.l.; hives present) and ten from the high mountain gardens of wadi Gebel (1800 m a.s.l.; hives absent).

Pollination treatments

Each tree was subjected to three pollination treatments in order to determine the relative impact of wind, insect and optimal pollination on the fruit set: (1) bagged flowers (wind pollination only), (2) open flowers (wind and insect pollination), (3) hand-pollinated open flowers (‘optimum’ pollination). Treatments were randomly assigned to three branches per each tree (totalling 90 treatments). For the ‘bagged flowers’, branches were covered with polyethylene tulle bags for the whole duration of flowering. Bags prevented access to pollinating insects, but allowed wind-mediated pollen grains to pass through. Bags were removed after petal abscission to avoid shading the developing leaves and fruits. For the ‘open’ treatment, branches were marked and left freely exposed to all insect pollinators. For the ‘hand pollinated’ treatment, flowers were freely exposed to insect pollinators and were additionally hand-pollinated with the pollen of three flowers from a neighbouring tree to test for pollen limitation. In July, before birds and small mammals started feeding on the developed fruits, the number of developed fruits was counted on the same branches and was used to calculate fruit set. Extremely small and deformed fruits were noted and excluded from analysis.

Pollinator visitation to almond

Flower-visiting pollinators (honey bees, wild bees, hoverflies and beetles) were recorded from the 10th February to 22nd February 2014 during the flowering period of the almond trees, with surveys beginning directly after bud opening. Two observation rounds (AM and PM) were carried out per experimental tree (N=30) with morning observations conducted between 10:00-12:30 and afternoon observations between 12:30-15:00. Surveys were only conducted when conditions were suitable for pollinator activity (temperature >18C, wind speed < 3 on the Beaufort scale, sunny weather). During the flowering period there were heavy rains and snow, which put surveying on hold for four days. Due to the small size of the trees and the relatively low abundances of flower-visitors it was possible to record all flower-visitors by walking slowly around the tree, with 10 minutes spent observing each tree. Pollinators were primarily identified in the field, with voucher specimens collected for identification in the lab. For calculation of visitation rates, the number of pollinators per tree was divided by the number of flowers in that tree. Visitation rates were averaged across the two rounds.

Local and landscape effects

The abundance and species richness of all flowering ground vegetation were recorded within a 10 m x 10 m quadrat centred at each focal tree. Pollinators utilising the ground flora were recorded within the quadrat in both the AM and PM survey sessions, with a single observer walking at a steady pace around the quadrat, examining each flower and recording all flower-visiting insects. Abundances were summed across the two rounds. For each tree the distance to the nearest neighbouring almond tree was also recorded, as was the number of flowers on conspecific trees within the quadrat.

The proportion of semi-natural habitat within the landscape was mapped using satellite imaging on Goggle Earth Pro version 7.0.3 within a circle of 1 km radius, centred on each of the surveyed trees. A 1 km radius was selected because solitary bees are known to be influenced by landscape factors at a relatively small scale, typically less than 1 km (Gathmann & Tscharntke 2002; Steffan-Dewenter et al. 2002).

Statistical analyses

All statistical analyses were performed with R.3.0.2 software (R Team 2013), with linear mixed-effect models conducted using the *lme4* package (Bolker et al. 2009). The fruit set resulting from the three pollination treatments was compared using a linear mixed-effect model with *pollination treatment* as a predictor and *tree*, *garden* and *wadi* as random factors (nesting *trees* within *garden* within *wadi*). A priori contrasts were used to compare the three treatments. Pollen limitation was calculated as fruit set after hand pollination divided by fruit set after open pollination. The relationship between fruit set of open-pollinated flowers and pollen limitation was assessed using a linear regression. The model included only pairs where pollen limitation values were higher or equal to 1 to exclude trees where the experimental supplement of pollen had probably failed (Larson & Barrett 2000).

The fruit set data were highly zero-inflated, but there was no evidence for pollen limitation being responsible for the low levels of fruit set (Fig 6.2); the high proportion of zeroes may have been a result of flowers failing to form fruit because of the extreme environmental conditions experienced during the surveying period. To account for the high occurrence of zeroes, the fruit set data were analysed using a hurdle-at-zero mixed-effect model in the *glmmADMB* package (Fournier et al. 2012; Skaug et al. 2014), which treated zero and non-zero outcomes as two separate categories; one model was fitted to the binary part of the data (zero vs. non-zero), with a binomial distribution, and then the non-zero values were analysed in a separate model with a truncated Poisson distribution (Bolker et al. 2012). Both models included garden nested within wadi as random factors.

Pollinator-mediated effects upon fruit set were assessed with the two hurdle-at-zero mixed-effect models with predictors of wild pollinator visitation and honeybee visitation. Local (non-pollinator mediated) effects on fruit set were assessed with models that included the predictors of distance to nearest conspecific tree, conspecific floral abundance and floral abundance and species richness of ground vegetation. Landscape-scale effects on fruit set were assessed in models containing the proportion of natural habitat as a predictor.

Wild pollinator and honeybee visitation in trees and wild pollinator and honeybee densities within the gardens were assessed in linear mixed-effect models with the predictors floral abundance and floral species richness. The impact of introduced honeybee hives was then assessed using linear mixed effect models with honeybee visitation, wild bee visitation and fruit set as response variables. All models contained garden nested with wadi as random factors to account for spatial variation. Model fit was based upon AIC and simplification followed Zuur et al. (2009), with the significance of fixed factors tested by comparing models with a likelihood ratio test (distributed as Chi-squared). R^2 values were obtained for linear mixed-effect models using the *MuMIn* package (Barton 2014), with marginal R^2_{GLMM} values representing the variance explained by each fixed effect (Nakagawa & Schielzeth 2013) (it is not currently possible to estimate R^2 for hurdle-from-zero models).

6.3 Results

Fruit set differed significantly between the three pollination treatments (Fig 6.1, *lmer*: $\chi^2 = 13.3$, df=2, $P = 0.001$) and was significantly lower for the bagged flowers (wind pollination only) which produced less than 8% of the fruits produced by open- and hand-pollinated flowers (A priori contrasts: open vs. hand: $P= 0.162$, hand vs. bagged: $P=0.007$, open vs. bagged: $P= 0.002$). The fruit set of open-pollinated flowers was unrelated to pollen limitation – calculated as the quotient of fruit set after hand pollination and fruit set after open pollination (Fig. 6.2, *lm*: $F_{1,15}= 0.008$, $P= 0.929$). Low level of fruit set in open flowers was not associated with high levels of pollen limitation, indicating that another factor (other than pollination limitation) was responsible for limiting fruit set.

In total 364 flower-visiting pollinators were recorded during the sampling period, 243 visiting the almond trees and 120 visiting flowering ground vegetation within the gardens. Approximately three-quarters of these flower-visitors were honeybees (77%). Of the wild pollinators, half were wild bees (51%; *Andrena* sp, *Anthophora* sp, *Xylocopa sulcatipes*), 20% were hoverflies (*Eupeodes corollae, Eristalinus aeneus*) and 23% were *Tropinota* sp. beetles. In total 14 species of ground flora were recorded growing beneath the trees (for full species list see Appendix 6.1); cultivated plants comprised 60% of the ground flora present, with wild plants representing 40%.

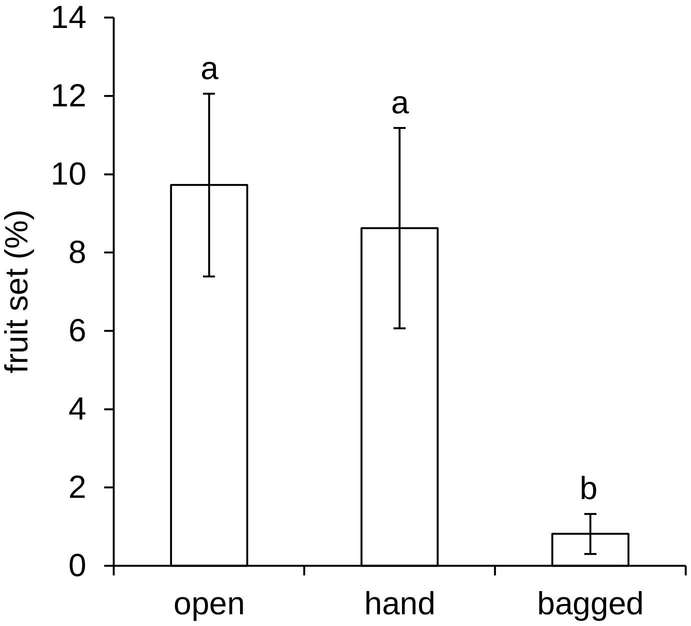

Figure 6.1. Mean fruit set (± Standard error) for the pollination treatments: open (insect + wind), hand (‘optimum’) and bagged (wind only).

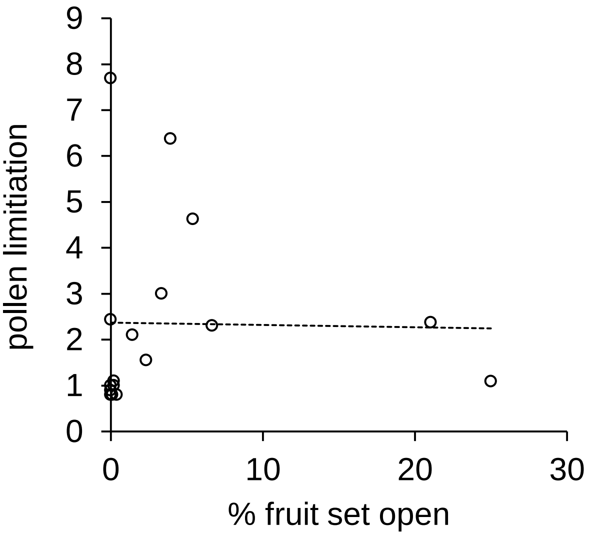

Figure 6.2. Relationship between pollen limitation (fruit set after hand pollination divided by fruit set after open pollination) and fruit set after open pollination. Data points represent pairs of hand-pollinated and open-pollinated trees where fruit set after hand-pollination was greater than fruit set after open pollination. Low levels of pollen limitation paired with low open fruit set indicate that another external factor is responsible for low levels of fruit set.

Fruit set was positively correlated with the visitation rate of wild pollinators (Table 6.1, Fig 6.3a), but not with honeybee visitation rate (Fig 6.3b). Wild pollinator and honeybee abundances within the gardens were positively associated with the abundance and species richness of ground flora (Fig 6.3c). Fruit set was not affected by the distance to the next almond tree or the floral abundance of almonds within the vicinity (all P >0.05), but was enhanced by the abundance and species richness of garden flora found within the vicinity of the tree (Table 6.1, Fig 6.3d). The proportion of semi-natural habitat in the landscape ranged from 70 – 90% and had no impact upon fruit set ($P > 0.05$).

The presence of honeybee hives had a highly significant effect upon honeybee visitation rates to almond (lmer: $\chi^2 = 7.36$, df = 1, $P = 0.007$). In sites where hives were present average visitation rates were 27 ± 4 per 1000 flowers, compared to 2 ± 0.9 in sites where hives were absent. The presence of the honeybee hives had no effect on wild pollinator visitation rates ($\chi^2 = 0.002$, df = 1, $P = 0.962$), nor on the subsequent fruit set of the almond trees ($\chi^2 = 1.11$, df = 1, $P = 0.290$).

Table 6.1. Results from generalized linear mixed-effect models assessing the impact of pollinator visitation, local effects and landscape effects on fruit set in almond, and the effect of floral abundance and species richness on pollinator densities within the orchards. Models included *garden* nested within *wadi* as random factors. χ^2 tests the difference between models after the associated fixed factor has been dropped. Fruit set was fitted with hurdle models that analysed both the binary data (zero vs. non zero) and values greater than zero separately. R^2_{GLMM} is given where available, and represents the variance explained by the associated fixed factor.

Response variable	**Fixed factor**		χ^2 (df=1)	***P***	R^2_{GLMM}
Fruit set (pollinator-mediated)	Wild pollinator visitation (+)	*binary*	4.39	0.036	
		greater than zero	4.63	0.031	
Fruit set (local effects)	Abundance of ground flora (+)	*binary*	10.36	0.002	
		greater than zero	3.70	0.054	
	Species richness of ground flora (+)	*binary*	16.23	< 0.001	
		greater than zero	5.18	0.022	
Wild pollinator orchard abundance	Floral abundance (+)		27.99	< 0.001	0.37
	Floral richness (+)		19.98	< 0.001	0.44
Honeybee orchard abundance	Floral abundance (+)		44.93	< 0.001	0.69
	Floral richness (+)		16.21	< 0.001	0.12

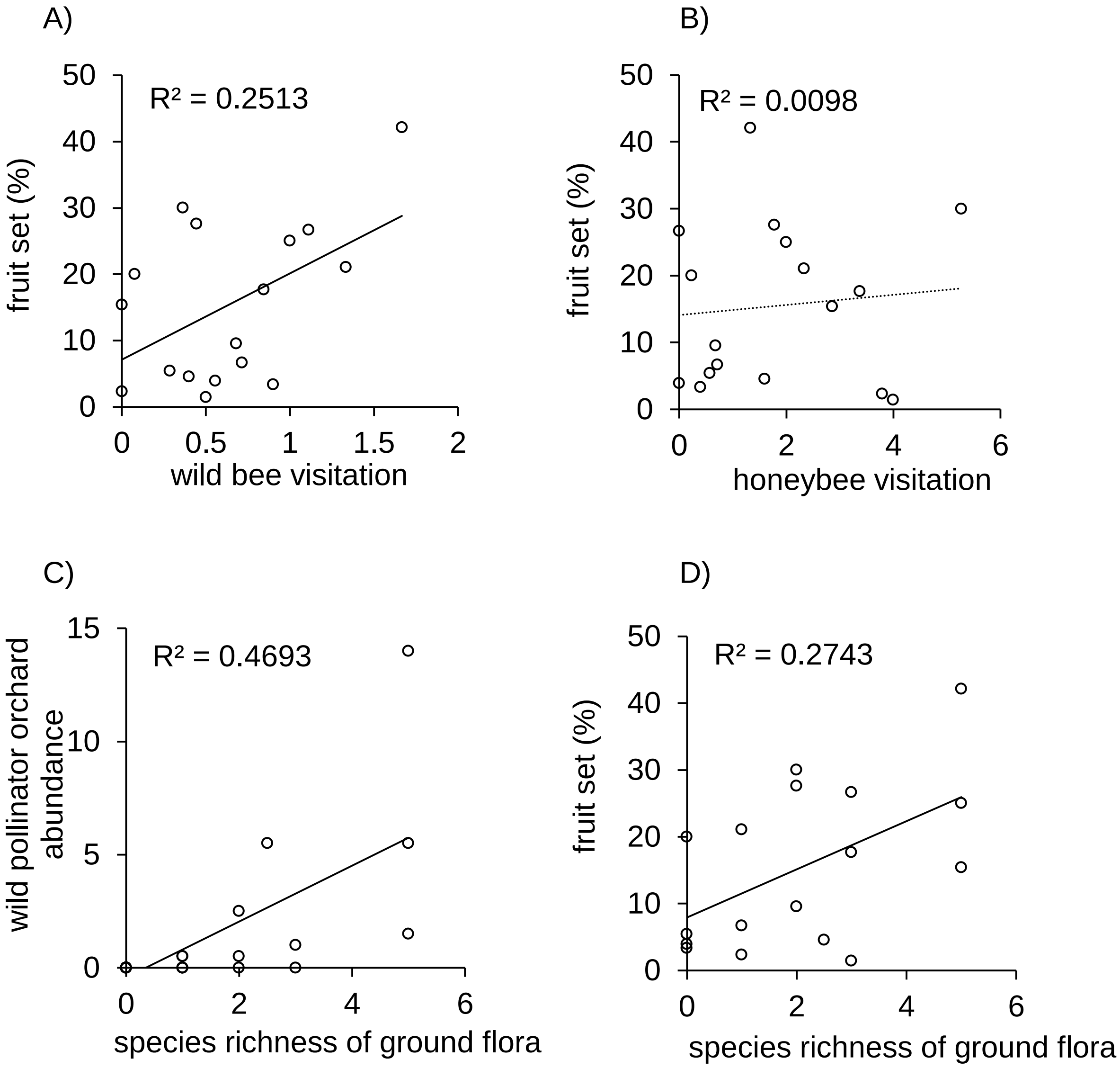

Figure 6.3. (a) Wild pollinator visitation vs. fruit set; (b) honeybee visitation vs. fruit set; (c) & (d) ground flora species richness vs. (c) wild pollinator abundance per 100 m² within the orchards, and (d) fruit set. Visitation rates are the number of individuals per 1000 flowers summed across the two sampling rounds. Solid lines indicate significant linear mixed-effect models (P< 0.05), dashed lines non-significant models (P> 0.05). Plots of fruit set (a & d) represent only trees with fruit set greater than zero.

6.4 Discussion

Pollination efficiency of wild pollinators versus honeybees

Almond in Sinai is highly dependent on insect pollination, with bagged flowers producing less than 8% of the fruits set by insect-pollinated flowers. Honeybees were the most abundant visitor to almond, outnumbering wild pollinators three to one, yet fruit set was positively correlated with the visitation rate of wild pollinators alone. This suggests that despite their lower abundance, wild insects were more effective pollinators than honeybees. Although the presence of the hives greatly increased levels of honeybee visitation to almond, it had no effect on wild pollinator visitation rates and did not lead to an increase in the fruit set. Since the additional presence of honeybees did not lead to enhanced fruit set, this further suggests that wild pollinators alone were sufficient to provide effective pollination services to almonds in this region.

Wild bees have been observed to provide higher levels of pollination efficiency than honeybees in other orchard crops through a variety of mechanisms. The solitary bee *Osmia cornuta* has been shown to provide higher levels of pollen deposition per visit than honeybees in almond (Bosch & Blas 1994), apple (Vicens & Bosch 2000) and pear (Monzón et al. 2004). Contact with the stigma depends on the collecting behaviour of the insect, and solitary bees often collect pollen and nectar simultaneously, providing higher levels of stigmal contact than honeybees (Bosch & Blas 1994). Higher levels of pollen deposition are only useful if the pollen is transferred from compatible cultivars, and again wild pollinators have been observed to move between cultivars more frequently than honeybees; in almond orchards *Osmia cornuta* is more likely to move between tree rows than honeybees, which rarely move between rows on a single foraging trip (Bosch & Blas 1994).

Interactions between species can also impact upon the quality of pollination services. Honeybees have been shown to modify their foraging behaviour in the presence of wild bees (Greenleaf & Kremen 2006b), and in Californian almond orchards honeybees moved between rows more frequently when they encountered wild bees, resulting in higher pollination effectiveness (Brittain et al. 2013b). Through this mechanism, wild pollinators have the potential indirectly to increase fruit set by modifying the behaviour of honeybees. If synergistic effects were responsible for the patterns observed in this study, then fruit set should increase in sites where both honeybees and wild pollinators were present. However, the presence or absence of hives had no effect upon fruit set, suggesting that the pollination contribution from wild pollinators exceeded that of honeybees.

Studies of Spanish almond found that hand-pollinated flowers achieved fruit sets ranging between 10 and 38 %, with 40 % considered the maximum achievable (Bosch & Blas 1994). In our study, mean fruit set was just under ten per cent for open flowers which is seems low, but we did not find any evidence of pollen limitation. It is difficult to directly compare the quality of pollination services between systems because interactive effects such as nutrient limitation or water deficiency can also limit the maximum achievable fruit set (Bommarco et al., 2013; Lundin et al., 2013). We suspect that fruit set may have been reduced by the extreme weather experienced during the flowering season rather than from a pollination deficit, though the single year design of the study makes this difficult to confirm. Cold weather and high winds are known to decrease honeybee activity (Brittain et al., 2013a) so pollination may have been disrupted during the storms, but it was also noted that the heavy rain and snow-fall resulted in high losses of flowers regardless of whether they had been pollinated.The importance of flowering ground vegetation

In conventional orchard systems, the proportion of natural habitat within the surrounding landscape can have a strong impact in determining the abundance of wild pollinators and the subsequent fruit set of crops (Holzschuh et al. 2012; Klein et al. 2012). In this low-intensity system, the gardens contain higher levels of floral resources than the expansive desert habitat (Norfolk et al. 2014) and the proportion of natural habitat in the landscape had no effect upon pollinator abundance. Local factors such as the floral availability within the orchards were better predictors of pollinator abundance and of subsequent fruit set. Setting aside semi-natural habitat can be a successful strategy for improving pollination services in some environments, but in arid regions where the natural habitat does not provide ideal conditions for bees, more effective results might be achieved by enhancing on-farm floral abundance and diversity.

Flowering ground vegetation is known to attract pollinators into orchards, and has been linked with increased densities of pollinators in almond (Klein et al. 2012), cherry (Holzschuh et al. 2012) and apple orchards (Rosa García & Miñarro 2014). There was also a positive association between ground flora abundance and pollinator abundance, and additionally a higher abundance and diversity of ground flora was associated with enhanced fruit set in the almond trees. Previous studies of orchard crops have not detected a positive relationship between ground vegetation and subsequent fruit set (Holzschuh et al. 2012), although facilitation effects have been observed in several other crop species such as sunflower and blueberry, where the presence of non-crop species attracted higher pollinator densities and increased yields (Carvalheiro et al. 2011; Blaauw & Isaacs 2014). These results suggest that ground vegetation within orchards can have a positive impact upon pollination services, and that simultaneously flowering plants are not a direct threat to tree yields.

Over half of the ground flora within the gardens were actively cultivated crops such as rocket, rosemary and strawberry, inter-planted amongst the trees. Previous studies have demonstrated the potential of wild and ruderal flowers for increasing crop productivity (Carvalheiro et al. 2011; Blaauw & Isaacs 2014) and these results additionally suggest that maintaining a diversity of co-flowering crop species can have the same beneficial effects. If mutual facilitation can occur between crop species, then diverse cropping systems may provide the opportunity to increase the effectiveness of pollination services in agricultural landscapes. Wild flower vegetation strips are typically used to supplement on-farm floral resources, and although they can be economically viable through the associated increases in crop yields (Blaauw & Isaacs 2014), farmers are often reluctant to sacrifice productive land. The introduction of additional flowering crops may provide equivalent benefits whilst creating an additional source of farm produce.

Conclusions

These results show that wild pollinators provided a higher level of pollinator service to almond than honeybees in these traditional Bedouin gardens. The introduction of honeybee hives had no impact upon fruit set, which was positively associated with visitation rates exclusively from wild pollinators. The presence of additional flowering vegetation, both cultivated and wild, had a beneficial effect upon pollinator abundance and was associated with enhanced fruit set. These results suggest that pollination services to almond in the region cannot be maximised through the introduction of more honeybees, but that increasing the abundance and diversity of ground flora within orchards could benefit wild pollinators and increase fruit set.

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Chapter 7.

Bird communities within the gardens[*]:

How do the gardens impact upon the functional composition of bird communities?

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[*] A modified version of this chapter is in revision for Journal of Arid Environments: Norfolk, O., Power, A., Eichhorn, M. & Gilbert, F. (2014) Migratory bird species benefit from traditional agricultural gardens in arid South Sinai. *Journal of Arid Environments*. In revision.

Chapter 7. Bird communities within the gardens: *How do the gardens impact upon the functional composition of bird communities?*

Abstract

In temperate and tropical regions, agricultural conversion of natural habitat typically has negative impacts upon the diversity and functional complexity of bird communities. In arid environments, however, the irrigation associated with agriculture can lead to an increase in local abundances of plant and insect resources, and so has the potential to benefit bird communities. Arid South Sinai is a key migratory corridor for many birds making the annual journey to and from wintering sites in Africa and breeding sites in Europe. This chapter assesses the importance of traditional Bedouin agricultural gardens for both resident and migratory species by comparing the density and functional composition of birds within the irrigated gardens to those in the unmanaged desert habitat. Estimated bird densities were significantly higher within the gardens than the unmanaged habitat, with a higher estimated species richness within the gardens. There were distinct differences in the functional composition of bird communities in the two habitats, with gardens supporting higher densities of insectivorous and migratory birds in addition to the resident desert species that were associated with the unmanaged habitat. The majority of resident species that were observed in the unmanaged habitat also occurred within the gardens, but overall the gardens increased landscape beta diversity and supported a distinct bird community due to the additional presence of migratory species. Migratory species were almost entirely absent from the unmanaged habitat, suggesting that this region would not be used as a migratory stop-off if not for the presence of the traditional agricultural gardens. Social and political changes have direct impacts upon the upkeep of gardens within the region, and this could have conservation implications for both resident and migratory birds.

7.1 Introduction

The impact of anthropogenic land-use on bird communities has received limited attention in arid regions. In temperate and tropical regions, land-use change such as agricultural conversion of natural habitat typically lead to the disruption of bird assemblages and a loss of functional diversity (Tscharntke et al. 2008; Flynn et al. 2009), but initial research suggests that the irrigation associated with agricultural systems and gardens in arid regions can actively increase the functional diversity of plants (Norfolk et al. 2013) and the abundance of pollinators (Gotlieb et al. 2011) and birds (Selmi & Boulinier 2003; Khoury & Al-Shamlih 2006). Human population growth and land-use change in arid environments are likely to have direct impact upon local bird communities, but may also have implications for the estimated 4 billion birds that make the bi-annual migration across the Saharan-Arabia desert belt, passing between wintering sites in Africa and breeding sites in Europe (Frumkin et al. 1995). In order conserve the migratory routes of these species, it is important to understand how birds and humans interact in the arid environments en route.

Crossing the inhospitable expanse of the Sahara desert poses major challenges for birds, including an exceptional energy demand (Zduniak et al. 2013). Strategies for crossing the desert differ between bird groups: raptors tend to use soaring flight, but small passerines use flapping flight which allows them to fly through the night and avoid high temperatures (Bruderer 1994; Chernetsov 2006). Many passerines make the entire 40-50 hour journey across the Sahara in one stretch, but others do it in stages, stopping along the way for refuelling either in the desert or in well-vegetated natural or man-made oases (Biebach 1990; Salewski et al. 2010). South Sinai forms a key migratory corridor of this migration and thousands of birds have been observed in oases and traditional gardens whilst on passage through the region (Bairlein 1992; White et al. 2007); most individuals rest for just a day, but some remain for lengthier refuelling periods (Lavee et al. 1991; Bairlein 1992). South Sinai is also known to support many species of over-wintering migrant that remain throughout the entire winter and make a shorter migration to Europe in the spring (Svensson et al. 2010).

The Bedouin gardens are known to support a highly modified plant community with higher functional complexity than the unmanaged habitat (Norfolk et al. 2013) and increased local abundances of insects (Norfolk et al. 2012). The higher availability of water, shelter and food resources within the gardens is likely to have a strong influence on the distribution and functional composition of bird communities in the area. This chapter compares bird communities within the irrigated gardens compared to those found outside in the unmanaged desert habitat.

Predictions were that the gardens would have a positive influence on bird densities, and that the modification of the vegetation and insect resources caused by the gardens would influence the functional composition of bird communities. Species-based and functional-trait based analyses were used to test these hypotheses, with specific attention paid to the relative importance of the gardens for resident versus migratory species. The results showed that active management of the gardens dramatically alters bird community composition, with gardens supporting higher densities of insectivorous and migratory birds in addition to the resident desert species also associated with the unmanaged habitat.

7.2 Methods

Data collection

The study was conducted between February and March 2014, allowing us to observe over-wintering migrants and the first wave of the passage migrants. The aim of this chapter was to compare the bird communities within the gardens to those that would be present in the absence of agriculture using a paired design, whereby each garden was matched with nearby associated unmanaged habitat. Each garden and associated unmanaged transect was repeatedly surveyed five times throughout the course of the study period. All surveys were conducted between 07:00 and 11:00 hours, with gardens and their matched unmanaged transect surveyed on the same day to ensure similar weather conditions.

Within the gardens, densities were estimated from a slow walk covering the whole area of the garden, recording all birds detected visually within the boundary. Due to high visibility in this sparse arid habitat, we are confident that this method produced high detection rates. The area of each garden was recorded using a Garmin eTrex GPS device: average garden size was 2700 (± 200) m^2. Bird densities in the unmanaged habitat were extremely low, meaning that it was not feasible to use an equivalent point count method. Point-counts were trialled, but nearly always resulted in no sightings. In order to make a more accurate density estimate in the unmanaged habitat we used line transects and a distance approach to estimate bird densities. All birds detected visually along the transect were recorded, as was their distance from the transect and the angle at which they were observed. A digital rangefinder was used to measure and estimate distances, with all observations beyond 50 m discarded. Out of 407 individuals recorded, just one remained unidentified (omitted from the analyses).

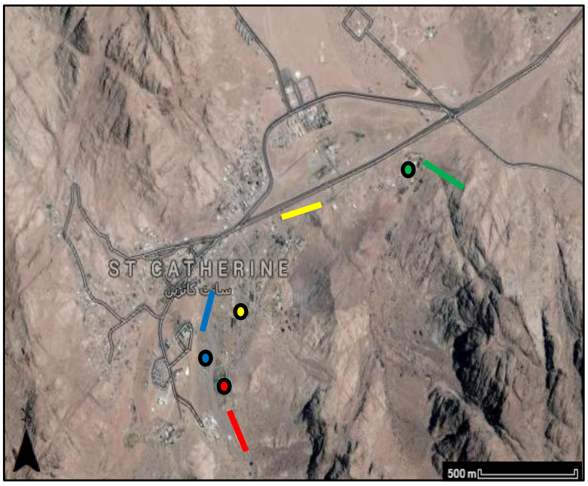
Site selection

In total 12 pairs of gardens and unmanaged transects were surveyed; four from St Katherine town, four from Wadi Itlah and four from Wadi Gebel. Gardens were selected at random from the gardens available in each wadi. Transects of unmanaged habitat were a minimum of 50 m from their associated garden, and were 200 m in length. In the mountains it was often only feasible to survey along the base of the steep-sided valleys, so transect locations were determined by geographic feasibility. Although complete randomisation of the transects would have been ideal, the gardens are also constrained to the wadi bases so our design allows for a fair comparison with the habitat that would be present in the absence of agriculture. Figure 7.1 shows the positions of the gardens and their associated transects.

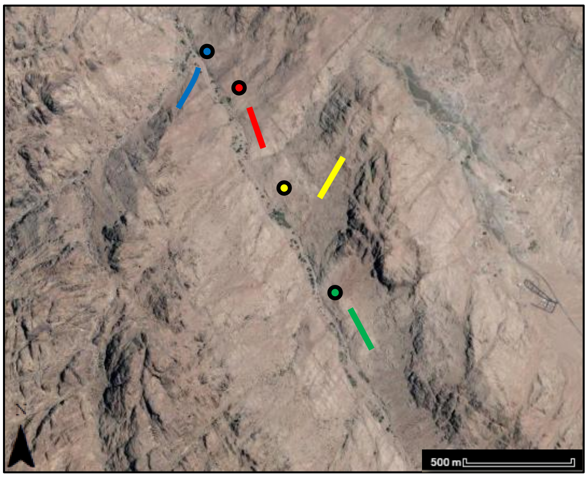
Functional groups

In order to establish how the gardens influence the functional composition of the bird communities, each species was classifed according feeding guild, habitat preference and migratory status (extracted from Svensson et al. 2010 and Hollom et al. 1988). Feeding guilds included insectivore, granivore, frugivore, nectarivore and carnivore, but for analyses we only compared the two most frequent guilds, insectivores and granivores. Habitat preference was divided into two categories; birds that preferred sparse habitats (such as rocky desert, mountains, wadis, low scrub), and those that preferred well-vegetated habitats (such as woodlands, gardens, parks and oases). Migratory status was also split into two categories, residents and migrants. The latter category included both passage migrants and over-wintering migrants, with feral species excluded from both categories.

A) Town



B) Wadi Itlah



C) Wadi Gebel

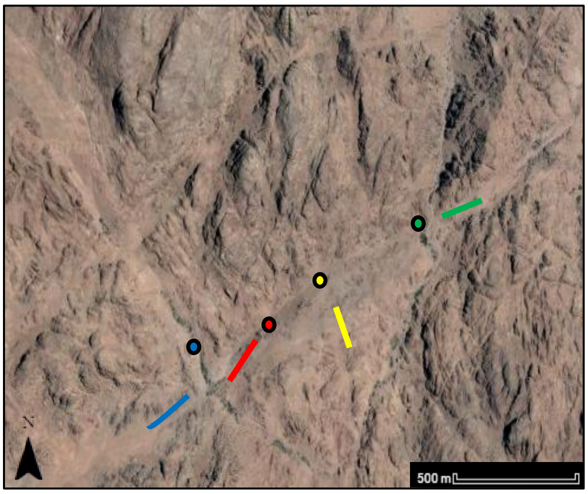


Figure 7.1. Maps of gardens and their associated transects in (a) St Katherine Town, (b) Wadi Itlah and (c) Wadi Gebel. Circles represent the gardens, lines their paired transect.

Statistical analyses

Distance 6.0 (Thomas et al. 2010) was used to estimate the densities of birds in the unmanaged habitat, with data from all five sampling rounds combined for each transect. The detection models were selected according to AIC value and were fitted with a half-normal curve. Density estimates within each gardens was the average number of birds per garden (across the five survey rounds) divided by the area of the garden. For comparison densities were all standardised to the number of individuals per 1000 m^2.

Statistical analyses were computed in R.3.0.2 (R Core Team 2013). Linear mixed-effect models were used to compare the densities in the gardens and unmanaged habitats, with average *density* as the response variable, *habitat* (gardens/unmanaged) as the fixed effect and *site* (Town/ Itlah/ Gebel) as a random effect to account for spatial variation. Model fit was based upon AIC and followed Zuur et al. (2009), with the significance of fixed effects and their interactions tested by comparing models with a likelihood ratio test (distributed as Chi-squared).

Species accumulations curves were created for gardens and unmanaged habitat in order to establish the completeness of the sample. Curves were created using the *specaccum* function in package *vegan* (Oksanen et al. 2012) and were estimated from 100 random draws from the data, sampling without replacement. Estimated species richness was calculated using Chao-bc, a bias-corrected form of Chao1. The similarity of all species found within the gardens and the unmanaged habitat was compared using the incidence-based Sørenson similarity index, calculated using SPADE with 200 iterations (Chao & Shen 2010).

A model-based approach was used to assess how community structure changed between the two habitat types. Presence/absence community data were analysed using the *manyglm* function in R package *mvabund* (Wang et al. 2012). A *manyglm* model with a binomial distribution was fitted to the data, with garden/unmanaged as the treatment effect. The significance of the treatment effect was tested using *anova.manyglm*, with the Wald test statistic and 999 resampling iterations.

7.3 Results

In total we recorded 407 bird sightings of 34 species belonging to 17 families (Table 7.1). 26 of these species were observed in the gardens and 16 in the unmanaged habitat. Estimated bird densities were significantly higher within the agricultural gardens than in the unmanaged habitat (Fig 7.1; lmer: $\chi^2 = 14.66$, df=1, $P < 0.001$), with an average density (per 1000 m^2) of 1.8 ± 0.3 within the gardens and 0.18 ± 0.04 in the unmanaged habitat.

Species accumulation rates were higher in the gardens than the unmanaged habitat, with a higher overall species richness (Fig 7.2). Estimated species richness was more than twice as high within the gardens, with Chao-bc estimates of 34 (± 16) in the gardens and 13 (± 7) in the unmanaged habitat. Within the gardens the most abundant species were Laughing Dove (14% of all garden sightings), Chiffchaff (12%), Tristram's Starling (12%), Rock Martin (11%), and White-spectacled Bulbul (9%). In the unmanaged habitat the most abundant species were Desert Lark (22%), Rock Martin (15%), Laughing Dove (10%), Sinai Rosefinch (10%) and Tristram's Starling (7%). Although three of the top five species were shared between the two habitats, the overall species similarity was 0.49 ± 0.04 (Sørenson index) and the composition of bird species differed significantly between the two habitats (manyglm: $W= 4.80$, df=22, $P=0.006$).

Comparison of the feeding guilds showed that gardens supported equal densities of granivorous species as the unmanaged habitat (Fig 7.3a; $\chi^2 = 1.58$, df=1, $P = 0.209$), but that insectivorous species occurred in significantly higher numbers within the gardens ($\chi^2 = 33.40$, df=1, $P < 0.001$). The gardens and the unmanaged habitat also supported equal densities of birds that prefer sparse, rocky, desert habitats (Fig 7.3b; $\chi^2 = 2.72$, df=1, $P = 0.099$), but birds preferring well-vegetated habitats, such as gardens, oases and woods, occurred at extremely low numbers in the unmanaged habitat and had significantly higher densities within the gardens ($\chi^2 = 7.96$, df=1, $P < 0.005$).

Resident bird species occurred at significantly higher densities within the gardens than they did in the unmanaged habitat (Fig 7.3c; $\chi^2 = 14.00$, df=1, $P < 0.001$). Gardens additionally contained high densities of migratory species that were almost entirely absent from the unmanaged habitat ($\chi^2 = 12.56$, df=1, $P < 0.001$). In total 17 migrants were observed within the gardens, five of which were over-wintering within the gardens (Table 7.1). The most common migratory species was the over-wintering Chiffchaff (*Phylloscopus collybita*), which made up 11% of all bird sightings, followed by the passage migrant Lesser Whitethroat (*Sylvia curruca*) (7% of sightings) .

Table 7.1. List of bird species and their associated functional traits. Functional trait data extracted from Svensson et al. (2010) and Hollom et al. (1988).

Family	*Species*	Common name	Migratory status	Feeding guild	Habitat preference	Present in: Gardens	Present in: Unmanaged habitat
Accipitridae	*Accipiter nisus*	Eurasian Sparrowhawk	Passage migrant	Carnivore	Mixed woods, also hunting over open ground	X	
Alaudidae	*Ammomanes deserti*	Desert Lark	Resident	Granivore	Aridground	X	X
Apodidae	*Apus apus*	Swift	Passage migrant	Insectivore	Mixed habitats		X
Columbidae	*Streptopelia decaocto*	Collared dove	Resident	Granivore	Towns and villages		X
	Columba livia (domest.)	Feral pigeon	Feral	Granivore	Towns and villages	X	X
	Streptopelia senegalensis	Laughing dove	Resident	Granivore	Gardens, oases	X	X
	Columba livia	Rock dove	Resident	Granivore	Rocky upland areas	X	X
Fringillidae	*Carpodacus synoicus*	Sinai rose finch	Resident	Granivore	Bare rocky slopes, wadis	X	X
	Bucanetes githagineus	Trumpeter finch	Resident	Granivore	Bare rocky slopes, wadis		X
Hirundinidae	*Ptyonoprogne fuligula*	Rock martin	Resident	Insectivore	Desert, dry hilly country	X	X
Motacillidae	*Motacilla alba*	White wagtail	Wintering migrant	Insectivore	Scattered vegetation, near water holes	X	
Muscicapidae	*Phoenicurus ochruros*	Black redstart	Wintering migrant	Insectivore	Mountainous regions,	X	
	Phoenicurus phoenicurus	Common Redstart	Passage migrant	Insectivore	Woodlands and parks	X	
	Erithacus rubecula	European Robin	Wintering migrant	Insectivore	Gardens and woodlands	X	
	Monticola saxatilis	Rock thrush	Passage migrant	Insectivorous	Rocky desert regions	X	
	Ficedula semitorquata	Semi-collared flycatcher	Passage migrant	Insectivorous	Orchards, gardens and woods	X	
	Saxicola torquatus	Stonechat	Migratory	Omnivore	Scrub, semi-cultivated	X	
	Oenanthe leucopyga	White-crowned wheatear	Resident	Insectivore	Rocky deserts, ravines	X	X

Nectariniidae	*Cinnyris osea*	Palestine sunbird	Resident	Nectarivore	Gardens, wadis, well-vegetated areas		X
Phasianidae	*Alectoris chukar*	Chukar partridge	Resident	Granivore	Rocky open hillsides		X
	Coturnix coturnix	Common quail	Passage migrant	Granivore	Farm crops, rough grassland	X	
	Ammoperdix heyi	Sand partridge	Resident	Granivore	Rocky, stony slopes and wadis		X
Phylloscopidae	*Phylloscopus collybita*	Chiffchaff	Wintering migrant	Insectivore	Woodland	X	
	Phylloscopus orientalis	Eastern Bonelli's warbler	Passage migrant	Insectivore	Trees and bushes	X	
Picidae	*Jynx torquilla*	Wryneck	Passage migrant	Insectivore	Gardens, orchards	X	
Pycnonotidae	*Pycnonotus xanthopygos*	White-spectacled bulbul	Resident	Insectivore + Frugivore	Gardens, orchards, date palms	X	X
Scotocercidae	*Scotocerca inquieta*	Scrub warbler	Resident	Insectivore	Deserts	X	X
Sturnidae	*Onychognathus tristamii*	Tristram's starling	Resident	Insectivore + Frugivore	Well vegetated wadis, around gardens	X	X
Sylviidae	*Sylvia atricapilla*	Eurasian blackcap	Passage migrant	Insectivore	Woodland, gardens, parks		X
	Sylvia curruca	Lesser whitethroat	Passage migrant	Insectivore	Trees and dense bushes	X	
	Sylvia rueppelli	Rüppell's warbler	Passage migrant	Insectivore	Low scrub, rocky outcrops, acacia wadis	X	
	Sylvia melanocephala	Sardinian warbler	Wintering migrant	Insectivore	Wadis, desert edges and oases	X	
Upupidae	*Upupa epops*	Hoopoe	Resident	Insectivore	Oases, orchards, palm groves	X	

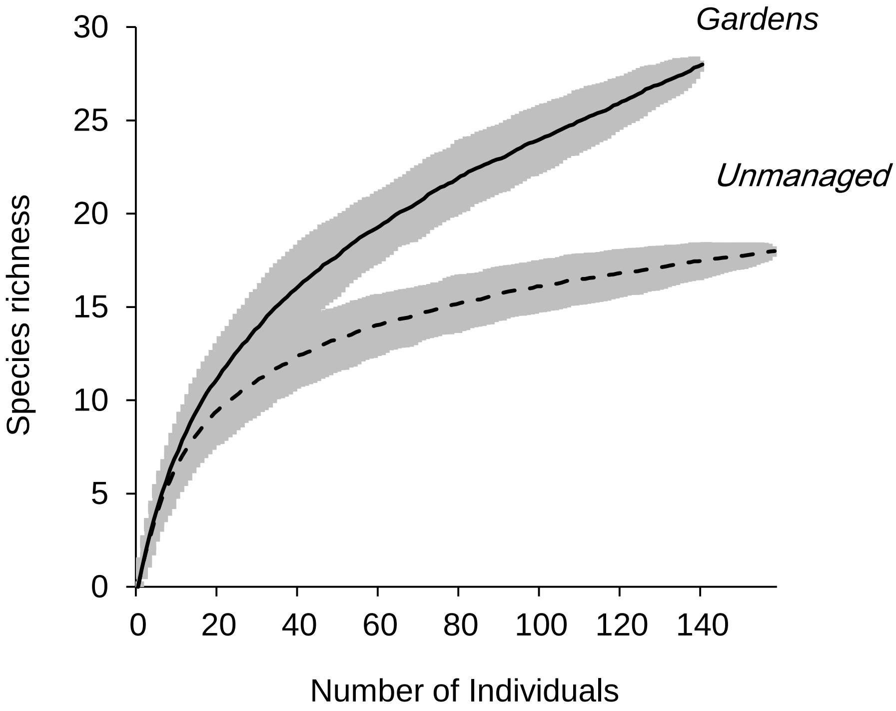


Figure 7.2. Species accumulation curves for the gardens and the unmanaged habitat. The curves represent the average of 100 random draws, sampling without replacement. The grey shaded envelopes represent the standard deviation from the random permutations of the data.

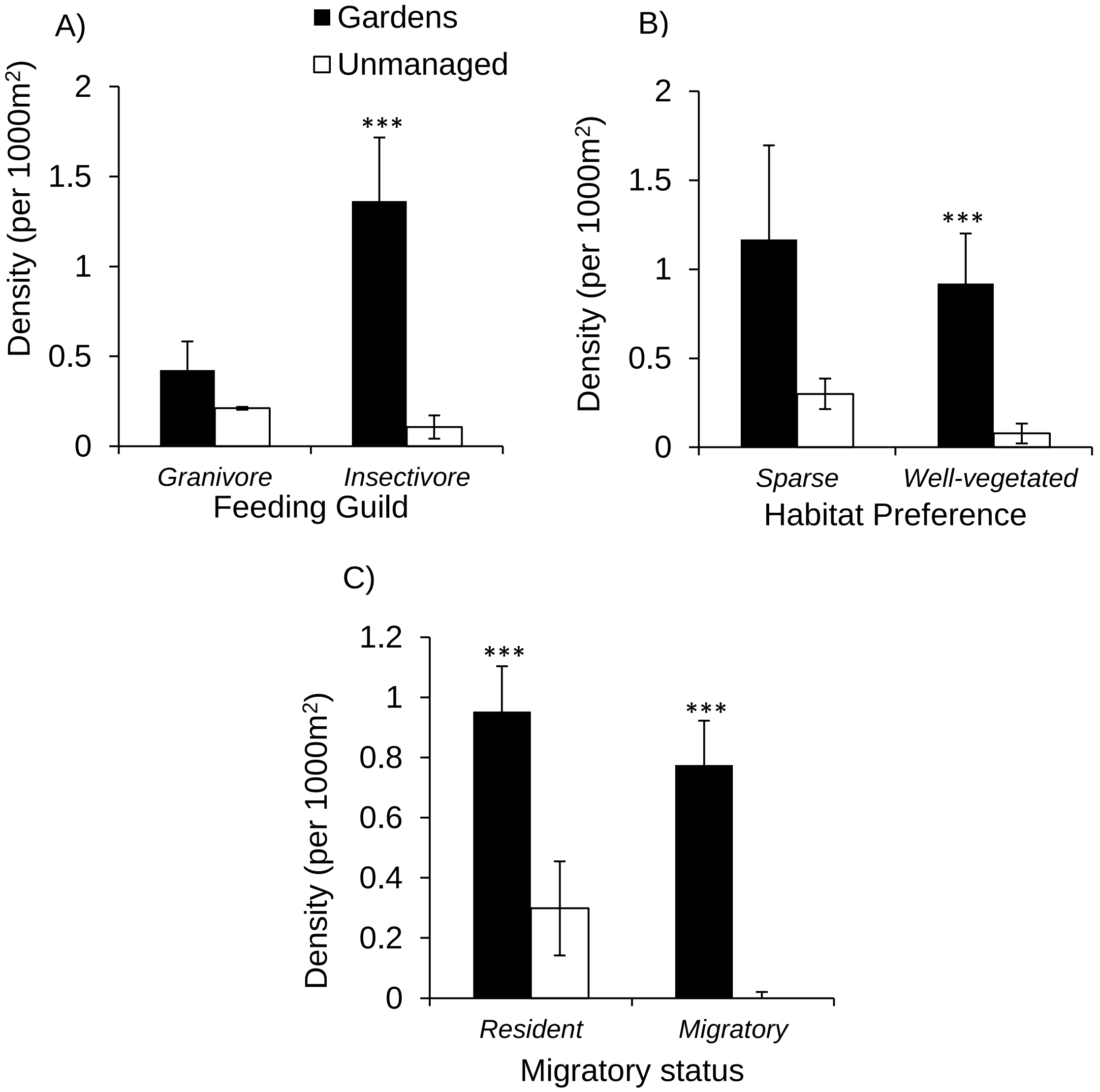


Figure 7.3. Comparison of the functional groups in gardens and unmanaged habitat, for a) feeding guilds, b) habitat preference and c) migratory status. Bars represent the average bird densities per 1000 m^2 with the standard error of the mean. Asterixes represent significant differences between the garden and unmanaged habitats.

7.4 Discussion

Estimated bird densities were higher within the gardens than the unmanaged mountain habitat. Species richness was twice as high within the gardens and the distinct community of species found within the gardens suggests that they increasing overall landscape beta diversity. There were clear differences in the functional composition of bird communities in the two habitats, with gardens supporting higher densities of migratory and insectivorous birds in addition to the desert species that were associated with the unmanaged habitat. Traditional oases in southern Tunisia have been shown to provide important habitat for breeding birds (Selmi & Boulinier 2003) and this research confirms that traditional desert agriculture can provide important habitat for resident breeding birds, whilst demonstrating its importance for migratory species.

Desert birds tend to occur at low densities due to the low availability of resources in their natural environment, with abundances positively linked to the density of vegetation (Pianka & Huey 1971; Khoury et al. 2007). In this study the actively irrigated gardens appear to elevate the availability of resources to such an extent that both resident and migratory birds are able to coexist at higher densities than those supported by the unmanaged habitat. Intensively farmed gardens in neighbouring Jordan have also been shown to boost bird numbers above those in the surrounding sand dunes (Khoury & Al-Shamlih 2006), but this was attributed to an increase in opportunistic species and not an increase in native desert species. Here the increased densities within these traditional gardens was due to the additional presence of migratory bird species and was not at the expense of desert-dwelling residents. The Bedouin gardens are low intensity and tend to contain patchy distributions of vegetation, with clusters of orchard trees interspersed with open areas of rocky soil. This heterogeneity of the garden habitat may contribute towards coexistence of birds with drastically different habitat requirements and may explain why these gardens are supporting higher densities of native desert species than the high intensity systems in Jordan.

The majority of resident species were observed in both the gardens and the unmanaged habitat, but in contrast the seventeen migratory species were found almost exclusively within the gardens (with just one sighting outside). The majority of these migrants were insectivorous passerines, with a preference for well-vegetated and wooded habitats. They were frequently observed foraging for insects in and around the flowering fruit trees and were undoubtedly benefiting from the active cultivation of the gardens. Fokidis (2011) stressed the importance of providing birds with native desert plants within ornamental gardens in the North American Sonoran desert, but in this Middle Eastern ecosystem migratory birds seem to actively benefit from the inclusion of non-native fruit trees within these agricultural gardens. The absence of migratory birds from the unmanaged desert habitat further suggests that many of these birds would not overwinter or stop off in this region were it not for the abundant resources within the cultivated gardens.

The sampling methods used within the gardens and the unmanaged habitat did differ, which means that interpretation of the density estimates must be taken with some care. However the high densities within the gardens can be directly attributed to the additional presence of insectivorous and migratory birds that were absent in the unmanaged habitat, so are unlikely to be an artefact of the differing sampling techniques. Indeed density estimates of granivorous and desert-dwelling birds were equal in the gardens and unmanaged habitat, suggesting that the two sampling techniques are achieving comparable estimates.

There were clear differences in the functional guilds within the two habitats, with gardens supporting a higher proportion of insectivores than the unmanaged habitat. As previously mentioned, many of these were migratory species, but gardens also supported several resident insectivores such as White spectacled bulbul (*P. xanthopygos*), Tristram's starling (*O. tristamii*) and Scrub warbler (*S.inquieta*), which were observed more frequently within the gardens than they were in the unmanaged habitat. In tropical habitats, the conversion of natural forest habitat into agricultural land tends to lead to decrease in insectivorous birds and a reduction in the pest control that is associated with the feeding guild (Tscharntke et al. 2008). In this arid system we find a contrasting pattern, with actively farmed agricultural gardens supporting higher densities of insectivores than the natural desert habitat. In another arid system in Mexico, more complex agricultural systems have been shown to result in an increase in bird-mediated pest control (Mellink 1991) and it is possible that the high abundance of insectivores have similar benefits in this diverse agricultural system.

The Bedouin gardens have persisted for over one thousand years (Zalat & Gilbert 2008), though recently their future has come under threat. Increasing urbanisation and a dependence on the tourism industry for income had led to many families abandoning their gardens for town life (Gilbert 2011). Current political instabilities in Egypt have devastated the local tourist industry, which has led to a resurgence of gardening with many Bedouin resuming the maintenance of gardens in the hope of supplementing dwindling incomes. Local charities have also been investing heavily in the future of the gardens by providing garden owners with the money needed for improvements of wells and garden walls. The loss of the gardens would have profound negative social and ecological implications within the region; not only are they deeply ingrained in the local culture, but we have shown here that they enhance the densities of local birds and provide important habitat for migratory birds en route to and from Europe.

Chapter 8.

General discussion:

Rainwater harvesting – an untapped solution for food security and biodiversity conservation?

Chapter 8. General discussion: *Rainwater harvesting – an untapped solution for food security and biodiversity conservation?*

8.1 Discussion

This project provides one of the first extensive assessments of how arid-land agriculture can affect patterns of biodiversity. The results show that rainwater-fed irrigation can have a dramatic impact upon the abundance and diversity of native wildlife, and that many species of plant, insect and bird actively benefit from the presence of the Bedouin agricultural gardens. Rainwater harvesting techniques help to maximise limited water resources in arid environments and have been shown to increase crop yields and enhance food security (Vohland & Barry 2009). This study extends previous work by demonstrating that these techniques can benefit native wildlife in arid regions.

Plant communities

Assessment of the plant communities in South Sinai showed that irrigated gardens support higher levels of plant diversity than the unmanaged habitat, with increased functional complexity. These results are fairly intuitive in an arid system where water is a main limiting factor, but they do contrast dramatically with patterns commonly observed in temperate and tropical agricultural systems. In tropical systems, agricultural conversion of habitat inevitably leads to a reduction in the diversity and complexity of forest ecosystems: although multi-crop agroforestry systems can reduce the impacts, agroforests still tend to represent an impoverished version of the natural forest habitat (Perfecto & Snelling 1995; Perfecto et al. 1997; Bhagwat et al. 2008). Equally, in temperate environments, agriculture tends to lead to a loss of floral diversity with negative impacts upon dependent wildlife such as pollinators (Kearns et al. 1998; Ferreira et al. 2013). In arid South Sinai, run-off agriculture and rainwater harvesting seem to provide an unusual example of an agricultural intervention that increases the complexity of plant communities.

Rainwater harvesting techniques have been shown to boost seedling recruitment and biomass accumulation in crop species (Gupta 1995; Ojasvi et al. 1999) and the results of this study show that irrigation can also benefit wild plant species. In this study system, the Bedouin gardens supported a higher diversity of native desert shrubs than the unmanaged habitat, many of which were rare or endemic species. Elsewhere in Sinai, irrigated forest plantations have been shown to increase the diversity of wild plants above those in the surrounding environment, but the majority of the extra species were agricultural weeds (Farahat & Linderholm 2012). This suggests that irrigation alone may not always benefit wild plant diversity, but in the right context, such as these low intensity Bedouin gardens, it can have a positive effect upon rare native species.

Dependent wildlife

The enhancement of the plant communities within the gardens had a direct impact upon pollinator abundance and diversity, both of which were significantly higher within the irrigated gardens than in the unmanaged habitat due to the higher availability of floral resources. As with the plants, these results strongly contrast with patterns observed in temperate and tropical environments where agricultural conversion of natural habitat typically leads to a reduction in pollinator diversity (Ricketts et al. 2008; Ferreira et al. 2013; Kennedy et al. 2013). In arid environments, rainwater harvesting techniques have the potential to enhance levels of floral resources above those found in the surrounding environment, so can have a positive impact upon dependent pollinators.

Irrigated ornamental gardens have also been shown to increase the abundance and species richness of pollinators in Israel (Gotlieb et al. 2011), although rare bee species were more likely to be found in the natural habitat than within the gardens. In this study, pollinator communities within the Bedouin gardens were similar to those in the unmanaged habitat, with the majority of species (both common and rare) occurring in both habitats. This suggests that the Bedouin gardens support high numbers of rare pollinators as well as benefiting common generalist species. Floral diversity has been linked to pollinator diversity in other agro-ecosystems (Holzschuh et al. 2007), and it seems likely that the high floral diversity of both cultivated and wild plants helps maintain the high species richness of the pollinator community within the gardens.

Birds also benefitted from the irrigated gardens and occurred at significantly higher densities within the gardens than in the unmanaged habitat. Insectivorous birds were particularly common within the gardens, presumably due to the higher availability of insects there; I have shown that flower-visiting insects are more abundant within the gardens, and previous work showed that ground arthropods are also more abundant within the gardens (Norfolk et al. 2012). Several species of migratory passerines were also observed within the gardens, but were entirely absent from the unmanaged desert habitat. Many of these migrants are woodland specialists that appear to benefit from the higher density and complexity of vegetation and trees within the gardens. It seems unlikely that such woodland migrants would stopoff in exposed desert habitat, but the presence of the rainwater-harvesting techniques facilitates the growth of dense orchard vegetation, which meets their habitat needs.

Can rainwater harvesting enhance ecosystem services?

Maintaining high levels of diversity in agricultural landscapes can have positive implications for the maintenance of ecosystem services, such as soil regulation, pollination and pest control (Tscharntke et al. 2005; Balvanera et al. 2006; Isbell et al. 2011). The results of this project show that rainwater-harvesting techniques do enhance diversity, so it seems likely that there will be implications for the quality of ecosystem services provided to crops. Previous studies have established that run-off techniques can increase soil fertility (Jia et al. 2006; Vohland & Barry 2009) and previous work in South Sinai suggests that a similar process may be occurring within the Bedouin gardens. The gardens contain a higher abundance of the ground-dwelling arthropods responsible for degradation of plant matter (Norfolk et al. 2012), as well as significantly higher concentrations of soil nitrogen, phosphorous and carbon compared to the unmanaged habitat (Norfolk 2010). In combination, the results of previous studies in Sinai and elsewhere suggest that rainwater-harvesting techniques can benefit crop growth, not only by increasing water availability, but also by increasing the quality of soil and the availability of nutrients.

The higher abundance and diversity of pollinators within the gardens also suggest that pollination services are affected by the presence of rainwater harvesting. Increased visitation by wild pollinators is typically associated with increased fruit set in crops (Garibaldi et al. 2013), and in this project we have confirmed that the visitation rate of wild pollinators enhanced fruit set within the primary crop, almond. Other studies have shown the visits of wild pollinators enhance the fruit set of several other species grown within the gardens, such as tomato (Greenleaf & Kremen 2006a) and alfalfa (Cane 2002), so it seems likely that other pollinator-dependent crops will also benefit from the enhanced pollinator community associated with the irrigated gardens.

Increased visitation rates to crops seem likely to bring agricultural benefits, but the high floral abundances found within the irrigated gardens could pose a risk to native flora if pollinators are attracted away from wild plant species. Previous research in the region has shown that the seed set of two species of native plants was not affected by the presence of the gardens, and that native plants within the gardens tended to be larger in size than those in the surrounding natural habitat (Norfolk & Gilbert 2014). This suggests that the gardens do not have a negative effect on the pollination success of wild flora, although further research to rule out dilution effects would be helpful.

8.2 General conclusions

The results of this project suggest that Bedouin gardens are having a positive impact upon wild plants, pollinators and birds within the St Katherine Protectorate. Observations from anthropologists working in the region suggest that when abandoned the gardens return to a desert like state with reduced diversity (Marx 1999). If this is the case, then the continued management of the gardens is essential for the conservation of biodiversity in this region. Gilbert (2013) discusses the lack of evidence-based management practice in Egyptian conservation, and specifically within the St Katherine Protectorate: this is a world-wide problem of both the developed and developing worlds. Furthermore, she suggests that institutionalized prejudice against Bedouin people by the Nile-Valley Egyptian culture has led to a dismissal of the value of their traditional ways of life. In contrast to these negative stereotypes, this project demonstrates that indigenous Bedouin farming practices can actively benefit biodiversity and highlights the particular importance of culturally important herbs for pollinators. Thus the message from this project is that the for maximum conservation benefit, the Nature Conservation Sector should support and encourage the continuation of traditional orchard gardens in the South Sinai region.

On a wider scale, the results of this project highlight the promising potential of rainwater harvesting in arid regions, by demonstrating that runoff water can be used to increase agricultural productivity whilst simultaneously enhancing biodiversity and ecosystem services. In addition to these ecological benefits, rainwater-harvesting techniques provide farmers with the opportunity to cultivate subsistence crops in some of the world’s harshest and driest environments. In South Sinai rainwater harvesting seems to offer a win-win solution to biodiversity conservation and food security. Although numerous studies have focussed upon the conservation potential of tropical agroforestry systems (Perfecto et al. 2005; Philpott & Armbrecht 2006; Oke & Odebiyi 2007; Bhagwat et al. 2008; Jose 2009; Jha & Vandermeer 2010; Hernandez et al. 2013), the potential of arid agro-ecosystems has received much less attention (Barrow 2014). Arid lands do not hold the same conservation significance as tropical regions, but they are home to one third of the entire human population, who are likely to suffer increasing pressures on food security in the face of predicted climate change (MEA 2005). It would be fascinating to see whether the biodiversity benefits associated with the Bedouin gardens are unique to this study system, or whether they are an inherent feature of rainwater harvesting systems worldwide. If the latter, then rainwater harvesting appears to offer an affordable method for increasing food security in arid regions, one that may simultaneously conserve biodiversity and enhance ecosystem processes.

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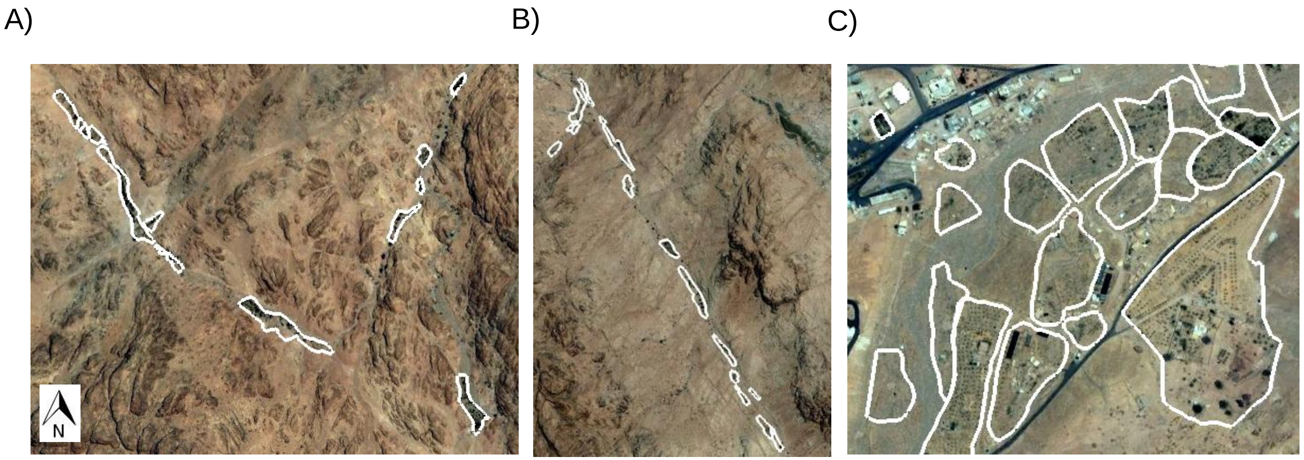
Appendices

Appendix 2.1 Co-ordinates of the gardens and control plots

		Latitude	Longitude
MOUNTAIN			
Gebel	*Gardens:*	28°32'37.12"N	33°55'59.82"E
		28°32'19.11"N	33°55'38.33"E
		28°32'33.58"N	33°55'24.12"E
	Control:	28°32'18.14"N	33°55'57.41"E
Itlah	*Gardens:*	28°34'58.40"N	33°55'9.17"E
		28°34'58.01"N	33°55'12.60"E
		28°35'12.91"N	33°55'16.39"E
		28°35'14.11"N	33°55'15.30"E
		28°35'41.03"N	33°54'60.00"
		28°35'34.93"N	33°54'57.08"E
		28°35'31.38"N	33°54'53.71"E
	Control:	28°35'33.58"N	33°55'4.06"E
TOWN			
St Katherines	*Gardens:*	28°33'28.31"N	33°56'59.38"E
		28°33'30.53"N	33°56'57.66"E
		28°33'32.80"N	33°57'1.12"E
		28°33'35.99"N	33°56'57.25"E
		28°33'31.42"N	33°56'43.21"E
		28°33'29.66"N	33°56'41.96"E
		28°33'33.15"N	33°56'54.89"E
	Control:	28°33'30.32"N	33°56'53.58"E
Rahah	*Gardens:*	28°34'26.81"N	33°56'23.11"E
		28°34'49.28"N	33°56'32.75"E
	Control:	28°34'32.28"N	33°56'39.92"E
LOW DESERT			
Sheik a wad	*Gardens:*	28°38'50.95"N	33°53'27.47"E
		28°37'53.62"N	33°52'57.62"E
	Control:	28°38'30.18"N	33°53'15.15"E
Feiran	*Gardens:*	28°41'34.99"N	33°56'32.83"E
		28°41'48.04"N	33°55'48.20"E
		28°41'45.73"N	33°55'8.61"E
	Control:	28°42'31.15"N	33°54'27.15"E
Ein Hodra	*Gardens:*	28°53'52.92"N	34°25'25.35"E
		28°53'48.32"N	34°25'22.12"E
		28°53'46.64"N	34°25'21.46"E
		28°53'43.07"N	34°25'19.36"E
		28°53'40.71"N	34°25'18.26"E
	Control:	28°54'47.45"N	34°26'3.02"E



Appendix 2.2. Satellite images demonstrating the densities of gardens within the wadis. Shown here are examples of mountain gardens in a) Wadi Gebel, b) Wadi Itlah, and c) town gardens in St Katherine, with images taken from Google Earth. Gardens are highlighted in white.

Appendix 2.3. Species list of all plants recorded in the surveys. ✔ indicates whether species was present in the gardens and/or control plots. Cultivated species only occurred within the gardens. Nomenclature follows Boulos (1999-2005).

	WILD PLANTS	Gardens	Controls		CULTIVATED SPECIES
Amaranthaceae	*Anabasis setifera* (Moq.)	✔	✔	Aizoaceae	*Mesembryanthemum* sp.
	Chenopodium murale (L.)	✔		Amarathaceae	*Beta vulgaris* (L.)
Apiaceae	*Deverra triradiata (*Hochst.)	✔		Amarylidaceae	*Allium cepa* (L.)
Asteraceae	*Achillea fragrantissima* (Forssk.)	✔	✔	Anacardiaceae	*Mangifera* sp.
	Achillea santolina (L.)	✔	✔		*Pistacia vera* (L.)
	Artemisia herba-alba (Asso.)	✔	✔	Apiaceae	*Foeniculum vulgare* (Mill.)
	Carduus getulus (Pomel.)	✔		Apocynoideae	*Nerium oleander* (L.)
	Centaurea scoparia (Sieber.)	✔	✔	Arecaceae	*Phoenix dactylifera* (L.)
	Chiliadenus montanus (Vahl.)	✔		Bambuseae	Unknown sp.
	Echinops glaberrimus (DC.)	✔		Brassicaceae	*Eruca sativa (*Mill.)
	Lactuca orientalis (Boiss.)	✔		Cactaceae	*Opuntia* sp.
	Launaea fragilis (Pau.)	✔		Cucurbitaceae	*Cucurbita pepo* (L.)
	Pulicaria undulata (Mey.)	✔		Fabaceae	*Ceratonia siliqua* (L.)
	Tanacetum sinaicum (Decne.)	✔			*Medicago sativa* (L.)
Boraginaceae	*Alkanna orientalis* (Boiss.)	✔			*Sesbania sesban (*Merr.)
	Anchusa humilis (Desf.)	✔			*Phaseolus vulgaris* (L.)
Brassicaceae	*Arabidopsis* sp.	✔			*Acacia saligna* (H.L.Wendl.)
	Diplotaxis harra (Boiss.)	✔			*Acacia nilotica* (Willd.)
	Farsetia aegyptia (Turra)	✔	✔	Lamiaceae	*Origanum syriacum* (L.).
	Matthiola arabica (Boiss.)	✔			*Rosmarinus officinalis* (L.)
	Zilla spinosa (L.)	✔	✔	Lythraceae	*Punica granatum* (L.).

Capparaceae	*Capparis spinosa* (L.)	✔	
Caryophyllaceae	*Gymnocarpos decandrus* (Forssk.)	✔	
	Paronychia sinaica (Fresen.)	✔	
	Silene schimperiana (Boiss.)	✔	
Ephedraceae	*Ephedra alata* (Decne.)	✔	
Fabaceae	*Acacia tortilis* (Hayne.)	✔	
	Astragalus sp.		✔
	Crotalaria aegyptiaca (Benth.)	✔	
	Retama raetam (Webb and Berthel)	✔	✔
Geraniaceae	*Monsonia nivea* (Decne.)	✔	
Lamiaceae	*Ballota undulata* (Fresen.)	✔	
	Phlomis aurea (Decne.)	✔	
	Stachys aegyptiaca (Pers.)	✔	✔
	Teucrium polium (L.)	✔	
	Thymus decussatus (Benth.)	✔	
Malvaceae	*Corchorus olitorius* (L.)	✔	
Moringaceae	*Moringa peregrina* (Fiori)	✔	
Myrsinaceae	*Anagallis arvensis* (L.)	✔	
Nitrariaceae	*Peganum harmala* (L.)	✔	✔
Oxalidaceae	*Oxalis corniculata* (L.)	✔	
Plantaginaceae	*Plantago sinaica* (Decne.)		✔
Poaceae	*Panicum turgidum* Forssk.	✔	
	Stipa arabica Trin. & Rupr.	✔	✔
Resedaceae	*Caylusea hexagyna* Forssk.	✔	✔
	Ochradenus baccatus Delile	✔	

Malvaceae	*Alcea striata* (DC.)
Moraceae	*Ficus carica* (L.)
	Morus alba (L.)
Myrtoideae	*Psidium* sp.
Nyctaginaceae	*Bougainvillea glabra* (Choisy.)
Oleaceae	*Fraxinus ornus* (L.).
	Olea europaea (L.)
Papaveraceae	*Glaucium corniculatum* (L.)
	Papaver sp.
Poaceae	*Avena barbata* (L.)
Rhamnaceae	*Ziziphus spina-christi (*Desf.)
Rosaceae	*Malus* sp.
	Prunus armeniaca (L.)
	Prunus dulcis (L.)
	Pyrus communis (L.)
Rutaceae	*Citrus aurantium* (L.)
	Citrus limon (Burmf.)
Solanaceae	*Nicotiana tabacum* (L.)
	Solanum lycopersicum (L.)
Vitaceae	*Vitis vinifera* (L.)

	Oligomeris linifolia Vahl	✔	
Rosaceae	*Crataegus sinaica* Boiss.	✔	
Scrophulariaceae	*Kickxia acerbiana* Taeckh and Boulos	✔	
	Scrophularia xanthoglossa Boiss.		✔
Solanaceae	*Solanum dulcamara* L.	✔	
Tamaricaceae	*Tamarix nilotica* Bunge	✔	
Zygophyllaceae	*Fagonia arabica* L.	✔	✔
	Fagonia bruguieri DC.	✔	✔
	Fagonia mollis Delile	✔	✔

Appendix 3.1. Full species list of pollinators observed in 2012 and 2013. For insects that were identified to species level, full species names are provided. Those that were separated into morphospecies are indicated as morpho sp. For several bee species I have used temporary designations (such as *Anthophora* Sinai sp1). These specimens were segregated into species by taxonomists, but the names cannot be finalised until type material has been seen.

2012	*N*	**2013**	*N*
COLEOPTERA		**COLEOPTERA**	
Acanthosomatidae		**Buprestidae**	
Acanthosomatidae morpho sp1	1	*Anthaxia scutellaris* Géné	25
Buprestidae		Buprestidae morpho sp1-2	11
Acmaeoderella sp`	1	**Coccinellidae**	
Anthaxia scutellaris Géné	7	*Coccinella septempunctata* L.	110
Chrysomelidae		**Dermestidae**	
Chrysomelidae morpho sp1 & 2	3	Dermestidae morpho sp1-3	164
Oulema sp1	51	**Pyrochroidae**	
Coccinellidae		*Pyrochroa* sp.1	106
Coccinellidae morpho sp1	23	**Scarabaeidae**	
Curculionidae		Scarabaeidae morpho sp1-2	2
Curculionidae morpho sp1-3	3	*Tropinota* sp1 & sp2	32
Dermestidae		**DIPTERA**	
Dermestidae morpho sp1-6	74	**Bombyliidae**	
Pyrochroidae		Bombyliidae morph sp1-2	29
Pyrochroa sp1	17	**Caliphoridae**	
Scarabaeidae		Caliphoridae moropho sp1-3	8
Scarabaeidae morpho sp1	3	**Conopidae**	
Tropinota sp1	2	Conopinae sp1-2	6
DIPTERA		**Sarcophagidae**	
Acroceridae		Sarcophagidae morpho sp1-9	40
Acroceridae morpho sp1	1	**Syrphidae**	
Anthomyiidae		*Eristalinus aeneus* (Scopoli)	14
Anthomyiidae morpho sp1-4	26	*Eristalis tenax* (Linnaeus)	9
Bombyliidae		*Eristalinus taeniops* (Wiedemann)	6
Bombyliidae morpho sp1-2	6	*Eumerus vestitus* Bezzi	2
Calliphoridae		*Eupeodes corollae* (Fabricius)	83
Calliphoridae morpho sp1-5	57	*Ischiodon aegyptius* (Wiedemann)	18
Chloropidae		*Scaeva albomaculata* (Macquart)	8
Chloropidae morpho sp1	1	*Sphaerophoria rueppellii* Wiedemann	13
Drosophilidae		*Sphaerophoria scripta* (Linnaeus)	38
Drosophilidae morpho sp1	21	*Syritta fasciata* (Wiedemann)	428
Muscidae		**Tephritidae**	
Muscidae morpho sp1-4	2	*Capitites augur* (Frauenfeld)	21
Platypezidae		*Euarestella iphionae* (Efflatoun)	8
Platypezidae morpho sp1	4	*Goniurellia spinifera* Freidberg	1
Sarcophagidae		*Katonaia aida* Hering	1
Sarcophagidae morpho sp1-9	37	*Trupanea amoena* (Frauenfeld)	3

Scathophagidae	
Scathophagidae morpho sp1-4	7
Sepsidae	
Sepsidae morpho sp1	33
Syrphidae	309
Eristalinus aeneus (Scopoli,)	8
Eristalis arbustorum (Linnaeus)	1
Eristalis tenax (Linnaeus)	5
Eristalinus taeniops (Wiedemann)	2
Eumerus vestitus Bezzi	1
Eupeodes corollae (Fabricius)	92
Ischiodon aegyptius (Wiedemann)	27
Melanostoma scalare (Fabricius)	2
Paragus tibialis (Fallén)	2
Scaeva albomaculata (Macquart)	2
Sphaerophoria rueppellii Wiedemann	7
Sphaerophoria scripta (L.)	7
Syritta fasciata (Wiedemann)	95
Tephritidae	
Acanthiophilus helianthi (Rossi)	1
Capitites augur (Frauenfeld)	2
Carpomya incompleta (Becker)	3
Dacus ciliatus (Loew)	1
Euarestella iphionae (Efflatoun)	1
Oxyaciura tibialis (R.D.)	1
HEMIPTERA	
Acanthosomatidae	
Acanthosomatidae morpho sp1-2	25
Cicadellidae	
Cicadellidae morpho sp1	1
Lygaediae	
Lygaeus saxatilis (Scopoli)	1
HYMENOPTERA	
Andrenidae	
Andrena sp1	9
Andrena sp2	2
Apidae	
Amegilla mucorea Klug	26
Anthophora (Heliophila) sp 1	3
Anthophora (Heliophila) sp 2	7
Anthophora (Heliophila) sp 3	2
Anthophora (Heliophila) sp 4	2
Anthophora caelebs Gribodo	3
Anthophora pauperata Walker	21
Apis mellifera L.	419
Xylocopa (Notoxylocopa) sp	1

Trupanea pulcherrima (Efflatoun)	3
HEMIPTERA	
Acanthosomatidae	
Acanthosomatidae morpho sp1-3	20
Lygaeidae	
Lygaeus saxatilis (Scopoli)	1
HYMENOPTERA	
Andrenidae	
Andrena sp1	14
Panurgus sp1	25
Apidae	
Amegilla mucorea (Klug)	11
Amegilla savignyi (Lepeletier)	6
Amegilla Sinai sp1	1
Anthophora (Heliophila) concinna (Klug)	7
Anthophora (Heliophila) Sinai sp1	3
Anthophora crassipes Lepeletier	19
Anthophora hermanni Schwarz & Gusenleitner	4
Anthophora pauperata Walker	4
Anthophora senescens Lepeletier	1
Anthophora Sinai sp1	103
Anthophora Sinai sp2	7
Apis cerana Fabricius	2
Apis mellifera Linnaeus	267
Tetraloniella sp1	3
Xylocopa sulcatipes Maa	28
Braconidae	
Braconidae morpho sp1	3
Chalcididae	
Chalcididae morpho sp1	5
Chrysididae	
Chrysididae morpho sp1-3	9
Colletidae	
Colletes nanus Friese	5
Colletes perezi Morice	21
Colletes pumilus Morice	1
Colletes tuberculatus Morawitz	3
Hylaeus (Dentigera) sinaiticus (Alfken)	155
Hylaeus (Hylaeus) Sinai sp1	10
Hylaeus (Paraprosopis) xanthopoda (Vachal)	4
Hylaeus (Prosopis) albonotatus (Walker)	12
Crabonidae	
Ammatomus sp1	2
Astata sp1	1
Bembecinus hebraeus de Beaumont	2
Bembix arenaria Handlirsch	1

Chrysididae	
Chrysis sp1 and sp2	4
Colletidae	
Colletes sp1	7
Hylaeus sinaiticus (Alfken)	90
Hylaeus xanthopoda (Vachal)	4
Crabronidae	
Ammatomus sp1	1
Bembecinus bytinskii de Beaumont	2
Bembix capensis Lepeletier	1
Bembix oculata Panzer	6
Cerceris sablosa (Panzer)	5
Cerceris tricolorata Spinola	7
Oxybelus sp1	7
Philanthus coarctatus Spinola	4
Tachysphex sp1	1
Evaniidae	
Evaniidae morpho sp1-3	3
Halictidae	
Halictus smaragdulus Bachal	47
Halictus sp1	15
Halictus sp2	6
Halictus sp3	4
Lasioglossum sp1	1
Lasioglossum sp2	1
Lasioglossum sp3	1
Lasioglossum sp4	1
Lasioglossum sp5	1
Lasioglossum sp6	1
Nomioides turanicus Morawtiz	5
Nomioides squamiger Saunders	2
Nomioides rotundiceps Handlirsch	11
Pseudapis sp1	1
Pseudapis sp2	1
Ichneumonidae	
Ichneumonidae morpho sp1-2	3
Megachilidae	
Anthidium amabile Alfken	1
Anthidium bischoffi Mavromoustakis	3
Chalicodoma montenegrense Dours	2
Hoplitis hofferi Tkalců	9
Icteranthidium ferrugineum Fabricius	4
Megachile walkeri Dalla Torre	79
Megachile flabellipes Pérez	1
Megachile montenegrensis Dours	13
Megachile sp1	2

Bembix oculata Panzer	2
Bembix sp1	1
Cerceris alboatra Walker	6
Cerceris sabulosa (Panzer)	33
Cerceris tricolorata Spinola	9
Diodontus sp1	1
Gastrosericus sp1	3
Oxybelus sp1	74
Palarus histrio Spinola	1
Philanthus coarctatus Spinola	25
Philanthus triangulum (Fabricius)	10
Prosopigastra fumipennis Gussakovskij	1
Tachysphex sp1	4
Halictidae	
Ceylalictus variegatus (Olivier)	5
Halictus tibalis Walker	17
Halictus (Seladonia) smaragdulus Vachal	115
Halictus falx Ebmer	9
Halictus pici Perez	12
Lasioglossum (Evylaeus) erraticum (Blüthgen)	2
Lasioglossum kowitense (Cockerell)	1
Lasioglossum (Evylaeus) subaenescens asiaticum (Dalla Torre)	3
Lassioglossum (Dialictus) collopiense (Perez)	14
Nomioides rotundiceps Handlirsch	59
Nomioides squamiger Saunders	6
Nomioides turanicus Morawtiz	65
Pseudapis nilotica (Smith)	5
Megachilidae	
Anthidium sp1	1
Hoplitis (Alcidamea) sp1	1
Hoplitis (Anthocopa) sp1	4
Hoplitis (Hoplitis) sp1	1
Hoplitis (Platosmia) africana (Warncke)	8
Hoplitis (Platosmia) gerofita (Warncke)	2
Hoplitis (Alcidamea) epeoliformis (Ducke)	2
Hoplitis (Alcidamea) hofferi Tkalců	43
Megachile concinna Smith	3
Megachile inexspectata Rebmann	1
Megachile doriae Magretti	1
Megachile insignis van der Zanden	13
Megachile minutissima Radoszkowski	1
Megachile Sinai sp1	7
Megachile Sinai sp2	4
Megachile tenuistriga Alfken	1
Megachile (Eutricharaea) walkeri Dalla Torre	97
Osmia (Helicosmia) alfkenii Ducke	1

Megachile sp2	15
Megachile sp3	1
Osmia sp1	8
Vespidae	
Odynerus sp1	5
Vespidae morpho sp1	3
LEPIDOPTERA	
Hesperiidae	
Spialia doris (Walker)	1
Lycaenidae	
Deudorix livia (Klug)	1
Agrodiaetus loewii Zeller	2
Lampides boeticus (L.)	171
Leptotes pirithous (L.)	39
Tarucus rosacea (Austaut)	74
Polyommatus icarus (Rottemburg)	1
Nymphalidae	
Danaus chrysippus (L.)	2
Vanessa cardui (L.)	3
Pieridae	
Colias croceus (Geoffroy)	1
Colotis fausta (Olivier)	1
Pieris rapae (L.)	1
Pontia daplidice (L.)	1
Sphingidae	
Macroglossum stellatarum (L.)	1

Osmia (Helicosmia) laticella van der Zanden	6
Stelis sp1	1
Scoliidae	
Scolia carbonaria (L.)	38
Sphecidae	
Chalybion flebile (Lepeletier)	1
Podalonia tydei (Le Guillou)	1
Scotia sp1	2
Tiphiidae	
Meria sp1	3
Vespidae	
Celonites fischeri Spinola	1
Eumeninae morpho sp1-5	17
Vespa orientalis Linnaeus	8
Jugurtia sp1	3
Quartinia sp1	16
Quartinia sp2	9
LEPIDOPTERA	
Hesperiidae	
Spialia doris (Walker)	4
Lycaenidae	
Agrodiaetus loewii Zeller	3
Iolana alfierii Wiltshire	10
Lampides boeticus (L.)	175
Tarucus rosacea (Austaut)	11
Nymphalidae	
Danaus chrysippus (L.)	3
Vanessa cardui (L.)	1
Pieridae	
Belenois aurota (Fabricius)	24
Colias croceus (Geoffroy)	4
Colotis fausta (Olivier)	2
Pieris rapae (L.)	2
Pontia daplidice (L.)	14
Pontia glauconome Klug	1
Sphingidae	
Macroglossum stellatarum (L.)	6

Appendix 4.1. List of plant species within the visitation networks.

	Plant species
A	*Astralagus* sp.
AC	*Allium cepa*
AK	*Arabidopsis kneuckeri*
AM	*Anchusa milleri*
AO	*Alkanna orientalis*
AP	*Anarrhinum pubescens*
AP1	*Asperugo procubens*
AS	*Achillea santolina*
AS1	*Alcea striata*
AT	*Astragalus tribuloides*
BB	*Bituminaria bituminosa*
BO	*Borago officinalis*
BU	*Ballota undulata*
BV	*Beta vulgaris*
CA	*Convolvulus arvensis*
CG	*Carduus getulus*
CH	*Caylusea hexagyna*
CI	*Colutea istria*
CP	*Capparis spinosa*
CP	*Cucurbita pepo*
CS	*Centaurea scoparia*
CS1	*Crataegua sinaica*
CS2	*Crepis sancta*
DA	*Diplotaxis acris*
DH	*Diplotaxis harra*
EG	*Echinops glaberrimus*
EG1	*Erodium glaucophyllum*
ES	*Eruca sativa*
FM	*Fagonia mollis*
FS	*Ferula sinaica*
FV	*Foeniculum vulgare*
GC	*Gypsophila capillaris*
GS	*Gomphocarpus sinaicus*
HA	*Helianthus annuus*
HB	*Hyoscyamus boveanus*
HP	*Hyoscyamus pusillus*
IC	*Ipomea cairica*
IL	*Isatis lusitanica*
L	*Lamiacae* unknown sp.
LC	*Lantana camara*
LN	*Launaea nudicaulis*
LP	*Lavandula pubescens*
LS	*Launaea spinosa*

M	*Mesembryanthemum* sp.
MA	*Matthiola arabica*
ML	*Matthiola longipetala (livida)*
ML1	*Mentha longifolia*
MLS	*Mentha longifolia schimperi*
MN	*Monsonia nivea*
MS	*Medicago sativa*
NR	*Nicotiana rustica*
OB	*Ochradenus baccatus*
OL	*Oligomeris linifolia*
OS	*Origanum syriacum*
P	*Papaver somniferum*
P1	*Papaver* sp.
PA	*Phlomis aurea*
PC	*Petroselinum crispum*
PD	*Prunus dulcis*
PG	*Punica granatum*
PH	*Peganum harmala*
PO	*Portulaca oleracea*
PR	*Paracaryum rugulosum*
PV	*Phaseolus vulgaris*
R	*Rosa* sp.
RC	*Rosa canina*
RO	*Rosmarinus officinalis*
SA	*Stachys aegyptiaca*
SM	*Salvia multicaulis*
SN	*Solanum nigrum*
SX	*Scrophularia xanthoglossa*
TS	*Tanacetum sinaicum*
UK1	*Asteraceae* sp.1
UK2	*Asteraceae* sp.2
VS	*Verbascum sinaiticum*
VV	*Vitis vinifera*
ZS	*Zilla spinosa*

Appendix 4.2. The most common flower visitors observed in gardens and control plots.

High Mountain (>1800m)						Low Mountain (1500m)					
Garden	N	(% visits)	Unmanaged	N	(% visits)	Garden	N	(% visits)	Unmanaged	N	(% visits)
Megachile (Eutricharaea) walkeri Dalla Torre	84	(17 %)	*Megachile walkeri*	60	(15 %)	*Syritta fasciata*	281	(20 %)	*Syritta fasciata*	24	(13 %)
Hylaeus (Dentigera) sinaiticus (Alfken)	59	(12 %)	*Lassioglossum (Dialictus) collopiense* (Perez)	56	(14%)	*Apis mellifera* L.	155	(11 %)	*Apis mellifera*	15	(8%)
Omophlus sp.	33	(6%)	*Anthophora pauperata* Walker	16	(4 %)	*Lampides boeticus*	101	(7 %)	*Halictus smaragdula*	11	(6 %)
Halictus (Seladonia) smaragdula (Vachal)	25	(5 %)	*Capitites augur* (Frauenfeld)	16	(4 %)	*Coccinella septempunctata*	78	(5 %)	*Halictus tibialis*	11	(6 %)
Lampides boeticus L.	23	(5%)	*Hylaeus sinaiticus*	16	(4%)	*Hylaeus sinaiticus*	72	(5 %)	*Quartinia* sp.	9	(5%)
Eupeodes corollae (Fabricius)	17	(5 %)	*Halictus smaragdula*	15	(4 %)	*Attagenus* sp.	70	(5 %)	*Coccinella septempunctata*	8	(4 %)
Syritta fasciata Wiedemann	15	(3 %)	*Halictus tibialis*	13	(3 %)	*Hylaeus* sp.	48	(3 %)	*Lampides boeticus*	6	(3 %)
Coccinella septempunctata L.	13	(3 %)	*Omophlus* sp.	12	(3 %)	*Megachile walkeri*	47	(3 %)	*Hoplitis (Anthocopa)* sp.	6	(3 %)
Capitites augur (Frauenfeld)	13	(3 %)	*Eupeodes corollae*	11	(3 %)	*Anthophora pauperata*	36	(2 %)	*Anthophora pauperata*	5	(3 %)
Halictus tibialis Walker	13	(3 %)	*Quartinia* sp.	10	(3 %)	*Halictus smaragdula*	27	(2 %)	*Pontia daplidice* L.	5	(3 %)

Appendix 5.1 Species list of the cultivated and wild flora that received insect visits.

		Visited in		Number of flowers (2013)			
Species	Family	2012	2013	Mean[a]	±	S.E.	Cumulative[b]
Cultivated							
Beta vulgaris L.	Amaranthaceae	X	X	1046.88	±	608.09	50250
Foeniculum vulgare Mill.	Apiaceae	X	X	740.83	±	218.23	35560
Origanum syriacum L.	Lamiaceae	X	X	325.00	±	173.40	15600
Olea europaea L.	Oleaceae	X	X	312.50	±	342.33	15000
Petroselinum crispum (Mill.) Fuss	Apiaceae	X	X	208.33	±	228.22	10000
Mentha longifolia schimperi (Briq.) Briq.	Lamiaceae		X	188.75	±	131.94	9060
Allium cepa L.	Amaranthaceae	X	X	156.04	±	102.99	7490
Rosmarinus officinalis L.	Lamiaceae	X	X	109.19	±	63.94	5241
Salvia officinalis L.	Lamiaceae		X	100.00	±	109.54	4800
Eruca sativa Mill.	Brassicaceae	X	X	83.58	±	33.81	4012
Limonium sp.	Plumbaginaceae	X	X	44.79	±	24.58	2150
Salvia multicaulis Vahl	Lamiaceae		X	32.25	±	15.03	1548
Medicago sativa L.	Fabaceae	X	X	28.88	±	10.88	1386
Mentha longifolia L.	Lamiaceae	X	X	25.00	±	27.39	1200
Phaseolus vulgaris L.	Fabaceae		X	24.38	±	11.07	1170
Punica granatum L.	Lythraceae	X	X	23.06	±	13.14	1107
Borago officinalis L.	Boraginaceae		X	19.58	±	18.33	940
Mesembryanthemum sp.	Aizoaceae	X	X	17.08	±	8.33	820
Portulaca oleracea L.	Portulaceae	X	X	12.17	±	5.15	584
Alcea rosea L.	Malvaceae	X	X	8.44	±	3.76	405
Colutea istria Mill.	Fabaceae	X	X	6.10	±	3.40	293
Rosa sp.	Rosaceae		X	4.17	±	3.19	200
Solanum lycopersicum L.	Solanaceae		X	2.75	±	2.23	132
Cucurbita pepo L.	Cucurbitaceae	X	X	1.98	±	1.27	95
Nicotiana rustica L.	Solanaceae		X	1.06	±	0.88	51
Helianthus annuus L.	Asteraceae		X	0.21	±	0.23	10
Wild							
Achillea santolina L.	Asteraceae		X	1256.25	±	577.66	60300
Chenopodium album L.	Amaranthaceae		X	1200.00	±	929.45	57600
Caylusea hexagyna (Forssk.) M.L.Green	Resedeaceae	X	X	478.75	±	257.01	22980
Alkanna orientalis (L.) Boiss	Boraginaceae	X	X	96.94	±	41.65	4653
Ochradenus baccatus Delile	Resedeaceae	X	X	93.75	±	61.90	4500
Fagonia mollis Delile	Zygophyllaceae	X	X	67.04	±	31.84	3218
Salvia sp.	Lamiaceae	X		66.67	±	51.09	3200
Artemisia judaica L.	Asteraceae	X		66.17	±	24.68	3176
Zilla spinosa (L.) Prantl	Brassicaceae	X	X	58.77	±	33.03	2821
Peganum harmala L.	Nitrariaceae	X	X	53.73	±	35.23	2579
Echinops glaberrimus DC.	Asteraceae	X	X	53.33	±	40.54	2560
Diplotaxis harra (Forssk.) Boiss.	Brassicaceae	X	X	48.60	±	23.14	2333
Fagonia arabica L.	Zygophyllaceae	X		43.06	±	20.67	2067
Matthiola arabica Boiss.	Brassicaceae	X	X	36.98	±	19.06	1775

Stachys aegyptiaca Pers.	Lamiaceae	X	X	23.42	±	8.06	1124
Monsonia nivea (Decne.) Decne. ex Webb	Geraniaceae	X	X	19.88	±	6.47	954
Tanacetum sinaicum (Fresen.) Decne. ex K. Bremer and C.J.Humphries	Asteraceae	X		10.71	±	6.36	514
Centaurea scoparia Sieber ex Spreng.	Asteraceae	X		8.19	±	5.83	393
Anchusa milleri Spreng.	Boraginaceae	X		6.46	±	2.95	310
Launaea nudicaulis (L.) Hook.f.	Asteraceae	X		6.25	±	3.83	300
Hyoscyamus boveanus (Dunal) Asch. & Schweinf.	Solanaceae	X	X	4.27	±	4.68	205
Matthiola longipetala (Vent.) DC.	Brassicaceae	X		4.13	±	2.51	198
Cleome arabica L.	Cleomaceae	X		2.29	±	1.48	110
Carduus getulus Pomel	Asteraceae	X	X	1.33	±	0.74	64
Gomphocarpus sinaicus Boiss.	Apocynaceae	X	X	1.04	±	1.14	50
Pulicaria incisa (Lam.) DC.	Asteraceae	X		0.42	±	0.46	20
Launaea fragilis (Asso) Pau	Asteraceae	X		0.04	±	0.05	2
Glaucium corniculatum (L.) J.H.Rudolph	Papaveraceae	X		0.02	±	0.02	1
Achillea fragrantissima (Forssk.) Sch.Bip.	Asteraceae	X		0.02	±	0.02	1
Arabidopsis kneuckeri (Bornm.) Schulz	Brassicaceae	X		0.02	±	0.02	1
Ephedra alata Decne.	Ephredraceae	X		0.02	±	0.02	1
Pulicaria undulata (Forssk.) C.A.Mey.	Asteraceae	X		0.02	±	0.02	1

Appendix 5.2 Top ten most abundant pollinator species visiting cultivated and wild flowers.

2012						2013					
Cultivated	*N*	%	Wild	*N*	%	Cultivated	*N*	%	Wild	*N*	%
Apis mellifera L.	404	27	*E. corollae*	57	19	*S. fasciata*	268	28	*S. fasciata*	51	16
Lampides boeticus L.	164	11	*A. mellifera*	16	5	*L. boeticus*	92	9	*A. mellifera*	39	12
Syritta fasciata Wiedemann	83	6	*T. rosaceus*	14	5	*A. mellifera*	87	9	*Anthophora pauperata* Walker	39	12
Megachile (Eutricharaea) walkeri Dalla Torre	75	5	*S. fasciata*	12	4	*Coccinella septempunctata* L.	69	7	*Osmia laticella* van der Zanden	20	6
Hylaeus (Dentigera) sinaiticus (Alfken)	61	4	*Attagenus* sp A	11	4	*Attagenus* sp. A	67	7	*Tropinota* sp.1	18	5
Tarucus rosaceus (Austaut)	60	4	Calliphoridae sp. E	11	4	*Seladonia smaragdula* (Vachal)	50	5	*H. sinaiticus*	15	5
F.Chrysomelidae *Oulema sp. A*	51	3	*Ischiodon aegyptius* (Wiedemann)	9	3	*Oxybelus* sp. A	43	4	*Coccinella septempunctata L.*	12	4
Eupeodes corollae (Fabricius)	39	3	*L. boeticus*	6	2	*Scolia carbonaria* L.	27	3	*E. corollae*	10	3
Leptotes pirithous (L.)	38	3	*H. sinaiticus*	6	2	*Halictus tibialis* Walker	23	2	*H.tibialis*	10	3
F.Dermestidae *Attagenus* sp. A	32	2	*Sphaerophoria rueppellii* Weidemann	6	2	*Attagenus sp. B*	16	2	*Amegilla mucorea* Klug	9	3

Appendix 6.1. Species list of the flowering ground vegetation recorded within the orchards.

	% total floral abundance
Cultivated	**60**
Rosmarinus officinalis L.	44
Eruca sativa Mill.	15
Mesembryanthemum.sp	0.5
Fragaria vesca L.	0.5
Wild	**40**
Diplotaxis harra (Forssk.) Boiss.	20
Alkanna orientalis (L.) Boiss.	8
Arabidopsis kneuckeri (Bornm.) Schulz	5
Monsonia nivea Decne. ex Webb	2
Stachys aegyptiaca Pers.	2
Anchusa milleri Lam. ex Spreng.	1.5
Ochradenus baccatus Delile	0.8
Fagonia mollis Delile	0.4
Zilla spinosa (L.) Prantl	0.3
Launaea sp	0.01