ASSESSING THE DRIVERS OF POLLINATOR AND NATURAL ENEMY COMMUNITIES IN PIGEONPEA AND FIELD BEAN CROPS

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Thesis submitted for the degree of Doctor of Philosophy

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September 2010

Declaration

I confirm that this is my own work and the use of all material from other sources has been properly and fully acknowledged.

Mark Otieno

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Dedication

To my mum Margaret, wife Miriam and son Austin

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Acknowledgements

First, I would like to thank the Felix Trust through the University of Reading for their full sponsorship of my studies and for exceptionally providing funds to cover an extra year of my course. I am really grateful. My heartfelt gratitude also goes to my key supervisor, Dr. Simon G. Potts who offered a constant source of inspiration and great mentorship throughout my studies. My sincere thanks also go to Dr. Alice Mauchline for her tremendous help during the preparations to begin my PhD and for helping with application for funding for this study. Thank you very much for your advice and guidance. I am greatly indebted to Dr. Andy wilby, Dr. Ben Woodcock and Dr. Ioannis Vogiatzakis for their tremendous supervisory advice. I learnt so much from your great skills and experience. Thank you.

I would also like to thank Mr. Harrison Kioko and Mr. Kyalo Wambua for their invaluable help during fieldwork in Kenya. Without them, it would have been very difficult to achieve all that was achieved during fieldwork. Many thanks also to Dr. Connal Eardley, Prof. Laurence Packer and Dr. Mary Gikungu for sharing their great knowledge on pollinators and for helping with identification of bees. I thank Mr. Mbaluka Kimeu for his help with plant identification. Many appreciations to the farmers of Kibwezi and Mr. John Miller for allowing constant access and use of their farms to run my experiments and for sharing a great wealth of local knowledge. You made it happen. Many thanks to Mr. Stuart Roberts for his training on how to sample bumblebees and Ms. Steph Harris for her incredible support during fieldwork in Grazeley, Berkshire. Thank you all!

Thesis abstract

1. Arthropods are key components in agro-ecosystems providing critical ecosystem services to crops, such as pollination and natural pest control. The structure and functioning of these arthropod communities can be influenced by farm management practices and the context of the farm within the wider landscape, yet this is poorly understood.
2. The impact of both local management and landscape context on insect pollinators and natural enemy communities was investigated in pigeonpea fields across a gradient of landscape contexts in lower Eastern Kenya.
3. In terms of local management, pesticide application was the main negative predictor of both pollinator and predator abundance within pigeon pea crops. Landscape structure at 1 km spatial scale had a major impact on the abundance of pollinators. Habitat complexity and configuration had positive impacts on bee abundance and species richness which was also linked to an increase in pigeonpea fruit set.
4. Pigeonpea relies on insect pollinators for its production. Therefore continued pesticide application poses a risk to sustainable yield production. Conservation strategies that target a reduction in pesticide inputs and maintain complex landscapes can support the presence of pollinators and predators and thereby enhancing crop production.
5. The effect of distance from field boundary on spillover of arthropod-mediated ecosystem services was assessed on field bean crop in England. Distance from the boundary of field bean crops had significant negative effects on generalist species of both pollinator and natural enemy but not specialists.

6. Field bean specialised bee species and aphid specialised ladybird beetles abundance was determined by the availability of specific resources regardless of landscape composition. Consideration of the role of these species in field bean production is more critical rather than the overall diversity of floral visitors and natural enemy communities.

Chapter 1 – Introduction and literature review

1.1 Ecosystem services and crop production

Ecosystem services are critical elements of ecological functioning that are important to humans, for example by promoting crop production and therefore food supply security (Kremen et al. 2002). These services can be direct and simple (e.g. crop pollination and biological control), or indirect and complex (e.g. soil fertility, carbon sequestration, detoxification of wastes, climate regulation, water purification – (Daily, 1997), and have been defined as "the conditions and processes through which natural ecosystems and the species that make them up, sustain and fulfil human life" (Daily, 1997).

Ecosystems, and their related functions, have historically undergone human mediated modifications. However, the scale of effects of these changes only became apparent after the Millennium Ecosystem Assessment (2005) established the scientific basis for actions needed to conserve and sustain them. The assessment identified major threats to ecosystem services that arise from human activity (mainly through land use change), particularly those that have changed profoundly in the last 50 years. In particular, changes in land use have resulted in simplification of ecosystems, habitat fragmentation, population isolation, and a reduction in the level of provision of some ecosystem services (Millennium Ecosystem Assessment, 2005).

The continued provision of many ecosystem services is linked to effective conservation and management of biological diversity, and is thus an important component of sustainability, particularly in human-dominated landscapes (Kremen, 2004). Pollination and natural control of pest species are among the most important ecosystem services, underpinning crop production and wild plant reproduction. These two services deliver significant and measurable benefits for human survival that warrants a powerful ecological and economic basis for conservation of ecosystems and biodiversity (Kremen et al. 2004). The main service providing organisms for pollination and bio-control are insects, and many species have been seriously affected by land use change and habitat fragmentation (Kearns et al. 1998; Landis et al. 2000; Steffan-Dewenter et al. 2005; Bale et al. 2008).

1.2 Pollinator communities and pollination services

Across the world, pollinator communities comprise a diverse assemblage of insects (e.g. bees, butterflies, moths and flies), birds (e.g. sunbirds and humming birds) and even bats, all of which can play a significant role in plant reproduction. The yield of insect pollinated crops has been shown to positively correlate with increasing diversity and abundance of pollinator species and functional groups (Hoehn, 2008).

In a review Klein et al. (2007) found 87 of the leading global food crops to be entirely, or partly, dependent on animal pollination. These crops make up 35% of global food production and as such a reduction in their productivity could have major economic consequences and impact on food security (Aizen et al. 2009). Approximately 75% of all global crop species are reliant upon insect pollination, mostly by bees (Klein et al. 2007). The annual global value of animal-mediated pollination services is estimated to be €153 billion, representing about 10% of the total economic value of global agriculture (Gallai et al. 2009). However, recent studies have documented insect pollinator declines (Beismeijer et al. 2006; Potts et al. 2010a). The main causes of these declines have been attributed to agricultural intensification, fragmentation of semi-natural habitats; agro-chemicals, climate change, parasites and disease infection (Potts et al. 2010).

Increasing agricultural intensification has resulted in the loss of forage resources (i.e. wild flowers) and nesting resources commonly available in non-cropped areas e.g. in post war Britain (Robinson and Sutherland, 2002). Perhaps the most affected pollinators are wild species (i.e. not the domesticated honey bee (*Apis mellifera* L., Hymenoptera, Apidae)) which tend to be more dependent on resources in semi-natural and natural areas. Research has demonstrated that the availability and distance to semi-natural and natural areas are important in determining the number of species and abundance of pollinators on agricultural fields (Ricketts et al. 2008). Increasing agricultural intensification reduces the interspersion of non-cropped areas that support wild ranging bees (mainly solitary bees), which are important to crop pollinators (Winfree et al. 2009).

Agro-chemicals include a variety of chemicals used in agriculture such as fertilizers, pesticides (including insecticides, fungicides and herbicides) (Kevan, 1999; Thompson, 2001). Pesticides have a specific impact on insect pollinators mainly by poisoning resulting in significant population losses (El Hassani et al. 2005; Brittain et al. 2010). The impact of pesticides on wild ranging bees is still poorly understood, yet these bees are crucial for reproduction in some crops (Kearns et al. 1998).

Climate is a major driver that determines the distribution and diversity of organisms across the world (Currie, 2004). Insects are among those organisms that hibernate or aestivate in response to unsuitable seasons (e.g. cold winter temperate or tropical droughts) for the active parts of their lifecycle and emerge when conditions are suitable and food resources are abundant (Bale and Hayward, 2010). A changing climate can cause a mismatch between floral blooming and pollinators' emergence, because of alteration of environmental cues (Memmott et al. 2007), thus affecting pollinators especially those specialised on specific plant species for their survival. Work by Inouye and Barr (2006) based on a three decade data set has revealed disruptions in pollination service provision in response to timing of flowering season and change in floral abundance. This increases the risk of disruption of the coevolved synchrony between flowers and their pollinators, which may, by implication, lead to extinctions of both the plants and insects that are obligate to each other (Schweiger et al. 2010).

The problems of parasites and diseases on pollinators are of global concern. Parasites such as V*arroa sp.*, tracheal mites, African hive beetles, and diseases such as Nosema are known to affect native pollinators, and are mainly introduced by invasive species in most of Europe and North America (Thomson, 2004). Plant invasive species may alter community composition of natural habitats, reducing or increasing food and nesting resources available to pollinators (Stokes et al. 2006). Due to co-evolution, native pollinators usually have their physiology closely matched to exploit the floral resources (mostly nectar and pollen) presented to them by the plants they visit (Stokes et al. 2006).

1.3 Natural enemies and natural pest control

Natural pest regulation is an ecosystem service that involves natural consumption of pest organisms (prey) by other organisms (predators), resulting in suppression of the population of the pest species (Landis et al. 2000). This phenomenon has been utilized to promote food production by using parasites, predators and pathogens to regulate the populations of pests, parasites and diseases in agro-ecosystems, a practice referred to as conservation biological control (DeBach and Rosen, 1991). Globally, an estimated 95% of the approximately 100,000 species of potential arthropod pests in agricultural fields and forests are regulated to some extent by natural enemies (DeBach and Rosen, 1991). The value of biological control to global crop production is estimated at US$ 400 billion per year (Costanza et al. 1997), a significant contribution towards the production of food and other goods.

Arthropods are a key group of natural pest enemies that contribute significantly to pest regulation by attacking crop pests and reducing yield losses (Landis et al. 2000). Although it has been established that most polyphagous parasitoids and predators of pests are effective regulators of pest populations in many occasions (Henneman and Memmott, 2001), they can have devastating impacts on native communities (Howarth, 1991). One example of natural enemy that escaped their intended target pest in Europe and North America is *Harmonia axyridis* (Coleoptera: Coccinellidae) (Michaud, 2002), which has become invasive and has massive impacts on native natural enemies (Koch et al. 2006). There is a huge potential for crops to benefit from biological control, however, these benefits are sometimes overlooked when a pest has been successfully controlled as a result of the release of an efficient natural enemy (van Lenteren and Bueno, 2003).

Perhaps the greatest threat to natural enemies, and their potential to benefit crop production, is the indiscriminate use of pesticide agro-chemicals (Landis et al. 2000) and landscape scale effects of habitat fragmentation (Tscharntke et al. 2007). Most crops are entirely dependent on pesticides to maintain production (van Lenteren and Bueno, 2003). As such the scale of the impact of pesticides on non-target organisms is likely to be great, although the extent of this impact remains to be established in many regions of the world (Landis et al. 2000). Habitat fragmentation is linked to low species diversity resulting from reduced abundance of individuals (Tscharntke et al. 2007). This follows a reduction in suitable habitat areas for particular species and spatial separation between them leading to isolation that limits movement between populations. As a result, the population declines due to a reduction in the chances for reproduction, higher probability of extinction risk and an increased probability of inbreeding depression (Steffan-

Dewenter and Tscharntke 2001) and scarcity of food items within the fragmented habitats (Elzinga et al. 2007).

Agricultural land use (local drivers) combined with the context of a field within the wider landscape (landscape drivers) can thus have profound impacts on both pollinator and natural enemy communities. These two sets drivers are discussed in details here.

1.4 Local drivers: field boundary features, farm management and impact on biodiversity

Agricultural fields are commonly characterized by field boundary habitats demarcating crop boundaries (Marshall and Moonen, 2002). The common types of these habitats are field margins (grass or wild flower strips), hedges, fences, uncropped strips, ditches and conservation headlands, among others (Marshall et al. 2006)**.** Modern farming requirements combined with increasing capacity of agricultural machinery has led to a reduction in the field area:boundary ratio as crop fields continue to expand. As a result, organisms residing and utilising these habitats are faced with increasing pressure. Although some farmers perceive field boundarys as sources of diseases, insect pests, and weeds, field boundarys have considerable conservation potential for both beneficial organisms (i.e. species providing ecosystem services) and other native species (den Belder et al. 2002; Holland et al. 2004).

Invertebrates are numerically one of the most dominant forms of native wildlife utilising field boundarys. They are important to crop production, either as beneficial insects (pollinators and natural enemies) or as pests reducing yields (den Belder et al. 2002). Whereas field boundarys mainly provide refugia for insects, crop fields may provide them with additional resources such as food (especially for pests). Some predators may track pest populations into the field, but most studies suggest that these foraging distances into crops are somewhat limited to a few tens of metres (see Holland et al. 2004). Enhanced diversity at field boundary may result into enhanced interactions and potentially stronger spillovers of organisms and their related ecosystem services extending deeper into crop fields (see Rand et al. 2006; Rand and Louda, 2006). The focus of spillover studies has largely been on patterns of change in diversity and abundance of organisms as a result of fluctuations in environmental conditions (Rand et al. 2006). Insects may either move across the edge in order to acquire resources or disperse in either direction responding to factors such as competition or other changes in ambient conditions, such as food resource availability (Fagan et al. 1999). This phenomenon has been determined for many guilds of vertebrate natural enemies, but few studies have investigated insect natural enemies (but see Rand and Louda 2006; Eilers and Klein, 2009). Furthermore spillover dynamics of pollinators is still poorly understood (Steffan-Dewenter and Westphal 2008).

Of the few studies looking at insect natural enemies, results show that high densities occur at the edge of crop fields, resulting in insects spilling over into adjacent systems (see Rand and Louda, 2006). Based on predictions by Oksanen (1990), it can only be expected that the direction of spillover effects on beneficial insects and pests will be from high to low quality semi-natural field boundary habitats into the crops habitats.

Reconciling sustainable food production and conservation remains a major challenge to farmers, governments and other stakeholders (see Gliessman, 2007). Agricultural activities such as tillage, grazing and extensive usage of pesticides and fertilizers have been shown to negatively impact on the population and abundance of wild species of animals (McLaughlin and Mineau, 1995).

The impacts of agro-chemicals and other farm management practices have been investigated in key indicator species in cultivated landscapes (Robinson and Sutherland, 2002). For example, research conducted on spider communities under varying management regimes has established a higher abundance and occasionally species diversity correlating with low land use intensity (Schweiger, 2005). Similarly, studies on carabid beetles have established similar results, with evidence of higher abundance, and species richness on less intensively managed fields (Pfiffner and Luka, 2003).

For pollinators, an increase in agricultural intensification is perhaps one of the most important drivers for simplification to their community structure. Agricultural intensification can lead to the destruction of pollinator nesting sites (especially for ground nesting bee species; Winfree et al. 2009), and cause changes in the foraging landscape by interrupting life cycles (e.g. killing insects in their early development stages; Weibull and Östman, 2003). The overall outcome of these practices could be reduced diversity and abundance of pollinators and a concurrent decline in pollination service delivery. The use of agro-chemicals such as pesticides is necessitated by an increasing demand for food production. Whereas pesticides can save a valuable amount of crops from pest damage, mounting evidence suggest they have negative impacts on pollinators (Brittain et al. 2010). Although experiments have widely demonstrated significant reduction in the number of some key pollinators, such as butterflies and bees, by pesticide usage (Weibull and Östman, 2003; Brittain et al. 2010), little is still known about the impacts of pesticides on wild pollinator communities (but see Brittain et al. 2010). It is important to establish the impacts of pesticides on wild pollinator communities and other beneficial insects such as natural enemies and in small holder agricultural systems where knowledge of wider pesticide impacts is often limited (Ngowi et al. 2007).

1.5 Landscape drivers: land-cover transformation and impact on biodiversity

Over recent decades, landscapes have been transformed tremendously by changes in land use and modification of land-cover largely due to human activities such as deforestation, agricultural expansion and urbanization (Lambin, 1997). Land-cover transformation is mainly associated with both fragmentation of habitats and reduction in habitat areas which can have major impacts on insect dispersal, especially for those species with greater ranges to move between these isolated habitats in search of food resources or for reproduction purposes (Fahrig, 2003). Due to such movements, community structure and diversity of organisms are influenced by spatial scales larger than the local habitat within which they can be found at any one instant, necessitating the adoption of landscape level approaches to understand interactions and changes in community patterns. For pollinators and natural enemies, increased level of complexity of the surrounding landscape may play a positive role in increasing their populations and species diversity (Thies and Tscharntke, 1999; Weibull et al. 2000; Steffan-Dewenter et al. 2002). Complex habitats present a greater number and variety of niches for functionally diverse communities of pollinators and natural enemies (Vickery et al. 2002; Kleijn et al. 2006).

Since landscape context has been shown to impact on species diversity, several studies have taken an applied approach and investigated how this affects beneficial insects. Many of these studies have found close positive correlations between landscape context and species richness (Banaszak 1992; Krauss et al. 2004), while others have established the effects of landscape context on the delivery of ecosystem services. For example, Bianchi et al (2006) found positive effects of landscape factors on the control of *Mamestra brassicae* L. (Lepidoptera: Noctuidae) by natural enemies. Carré et al. (2009) demonstrated the negative effects of loss of semi-natural habitats on changes in bee community composition. The existing knowledge on the effects of landscape context on organisms is largely drawn from studies that focused on individual species rather than community level dynamics (e.g. Collinge et al. 2003). In order to understand the scale of impacts of landscape context on organisms, especially insects, it is crucial to consider aspects of local management in tandem with landscape context.

The relative complexity (usually related to high heterogeneity of habitats within an area) or simplicity (associated with high homogeneity of an area), coupled with variations in farm management practices are likely to have direct impacts on the way insects interact with their habitat and as such deliver ecosystem services (Kleijn and Verbeek, 2000). Structurally complex landscapes could potentially enhance local diversity of beneficial insect communities (especially pollinators and natural pest enemies) in cultivated landscapes. Therefore, for sustainability of delivery of ecosystem services, it is important to understand the impact of landscape context on pollinators and natural enemies in cultivated landscapes (Steffan- Dewenter et al. 2002; Carré et al. 2009).

1.6 Statement of research problem and justification

Understanding ecosystem processes, and the scales at which these processes drive local communities of species, is critical for biodiversity conservation in agricultural landscapes. Drivers of change within communities, such as landscape context and farm management have been widely investigated on pollinator and natural enemy communities, particularly in developed systems in Europe and North America (Steffan-Dewenter et al. 2002; Dauber et al. 2005; Bianchi et al. 2006; Tscharntke et al. 2007; Eilers and Klein, 2009). However in developing countries, where great diversity and densities of these invertebrates are likely to contribute towards a high proportion of crop production (Ricketts et al. 2008), the status of insect pollinators and natural enemies within different landscape contexts are largely unknown. It is crucial to investigate the dynamics of the communities of these insects at multiple spatial scales under contrasting management regimes in order to promote optimal provision of ecosystem services.

The dynamics of insect communities and their linked ecosystem services are often poorly understood. Understanding these relationships between land cover, land management and biodiversity would yield important insights towards conservation measures that would profit both pollinators and natural enemies. This is particularly so in a non-European / North American context, where comparatively little research has been undertaken.

Pollination and natural pest control are among the most commonly studied ecosystem services delivered to crops by insects, however, their dynamics have rarely been examined in tandem at both landscape and local scales. Moreover, the interactions between landscape context and local farm management factors on local insect diversity and their capacity to deliver these ecosystem services is particularly poorly understood in an Afro-tropical context.

Establishing the extent to which suppression of pest population occurs into the field from the edge could be an important step in providing insight into conservation of field boundary habitats to enhance the productivity of crops. Furthermore, the current knowledge on this subject is drawn from a handful of studies that placed more focus on abundance of arthropod natural enemies (e.g. French et al. 2001; Baldissera et al. 2004; Rand and Louda 2006), widely ignoring functional implications of such patterns on ecosystem service delivery.

Knowledge on the dynamics of insect-mediated pollination services with increasing distance from field boundarys is also scarce. It is widely believed that increasing distance from the field boundary leads to a general decrease in pollination (see Steffan-Dewenter and Tscharntke, 1999 and Chacoff and Aizen, 2006). This knowledge is based on a handful of crops and wild plants.

There is therefore need to extend this knowledge to a wider variety of economically important crops in order to establish whether they show similar patterns in the decline of these ecosystem services with distance from the field boundary. It is not clear whether there are tradeoffs in delivering ecosystem services and interactions between different guilds of beneficial insects. Furthermore, the extent and efficiency of pollination with increasing distance from field boundary still remains to be investigated.

1.7 Specific aims

My thesis investigates the impacts of landscape structure, field boundary features and local farm management on pollinator and natural enemy community diversity. The focus is on how these factors affect the delivery of key ecosystem services to pigeonpea and field bean crops.

The specific objectives were to determine:

1. The effect of local farm management and landscape drivers of pollination and biological control services in a pigeonpea agro-ecosystem in lower Eastern Kenya, Africa.
2. The impact of landscape complexity, configuration and pesticide application on functional responses of bee communities in pigeonpea cropping systems in lower Eastern Kenya.
3. The effects of distance from field boundary on pollinator and natural enemy diversity and delivery of related services to field beans *(Vicia faba* L.: Leguminosae) in the South East England, UK.

Chapter 2 – General methods

2.1 Description of study areas

The field studies were located in Kenya and England. The first (2007) and third (2009) years of fieldwork were conducted in Kibwezi District, lower Eastern Kenya (2°15'S and 37°45'E – Fig. 1).

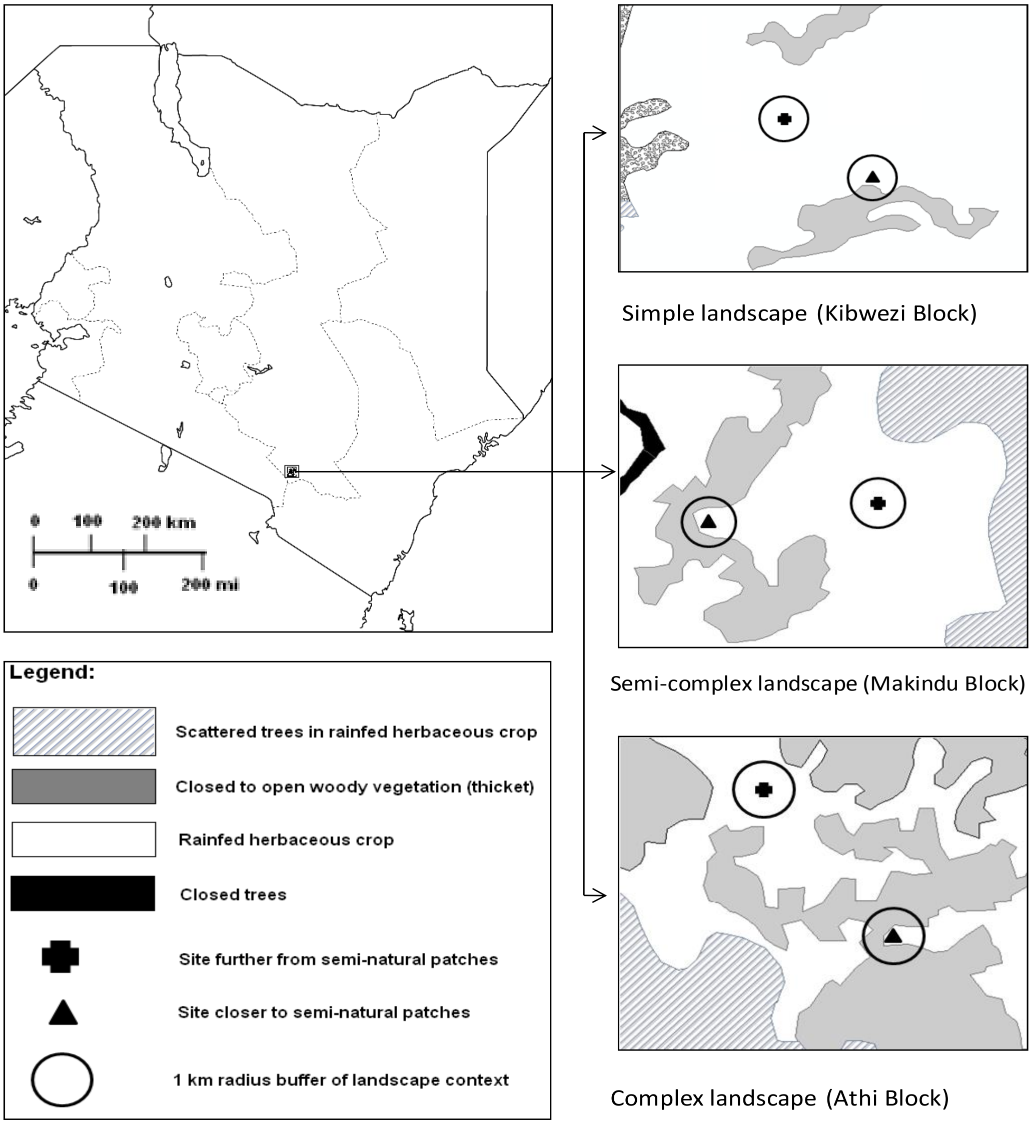

Figure 1: A map of Kenya example gradient of landscape structural complexity. Study site is at the centre of the 1 km sampling radius shown by the circles.

The Kenyan study area was located 150 km South East of Nairobi, generally characterized as undulating (723-1015 m above sea level) and the soils are mainly regosols and ferrosols. The climate is warm and semi-arid, with mean annual temperatures of 30°C and mean annual rainfall of less than 500mm (Mbuvi, 2009). There are two rainy seasons (long rains between March-May and short rains between October and December) separated by long spells of dry weather. Habitat types in the area can be broadly classified as riparian forest, swampy palm and sedge, *Acacia* woodlands, wooded grasslands, and open grasslands. Vegetation comprises a complex of plant communities of trees (mainly *Acacia-Commiphora* vegetation), and a number of shrubs and herbs (Bogdan, 1958). Maize (*Zea mays L.* Poaceae) is the most widely grown crop in the area both on large (> 10 ha) and small scale farm holdings (< 1 ha). Other crops grown include pigeonpeas (*Cajanus cajan* (L.) Millsp: *Leguminosae*), beans (*Phaseolus vulgaris* L.: *Leguminosae*.), cowpea (*Vigna unguiculata* L.: *Leguminosae*), green grams (*Vigna radiata* L.: *Leguminosae*), okra (*Abelmoschus esculentus* Moench.: Malvaceae) and a number of fruit trees such as mangoes (*Mangifera indica L.*: Anacardiaceae) and papaya (*Carica papaya* L. Caricaceae).

Plate 1: Pigeonpea crop

The English study ((2008) was based in Grazeley, Reading (Berkshire County) in South East England at 51º 23'N, and 0 º 59'W (Fig 2). The study area is characterized by intensively managed fields interspersed by wild flower field margins and hedgerows. This area falls within the UK lowlands (42 m above sea level), with an annual rainfall less than 650 mm and mean annual temperature between 9 -11 ºC (The Met Office – 2010 http://www.metoffice.gov.uk/climate/uk/so/print.html). Winter wheat (*Triticum aestivum* L.: Poaceae), oilseed rape (*Brassica napus* L.: Brassicaceae) and field beans (*Vicia faba* L.: *Leguminosae*) were observed to be the most common crops cultivated within the study area on large scale (>20 ha).

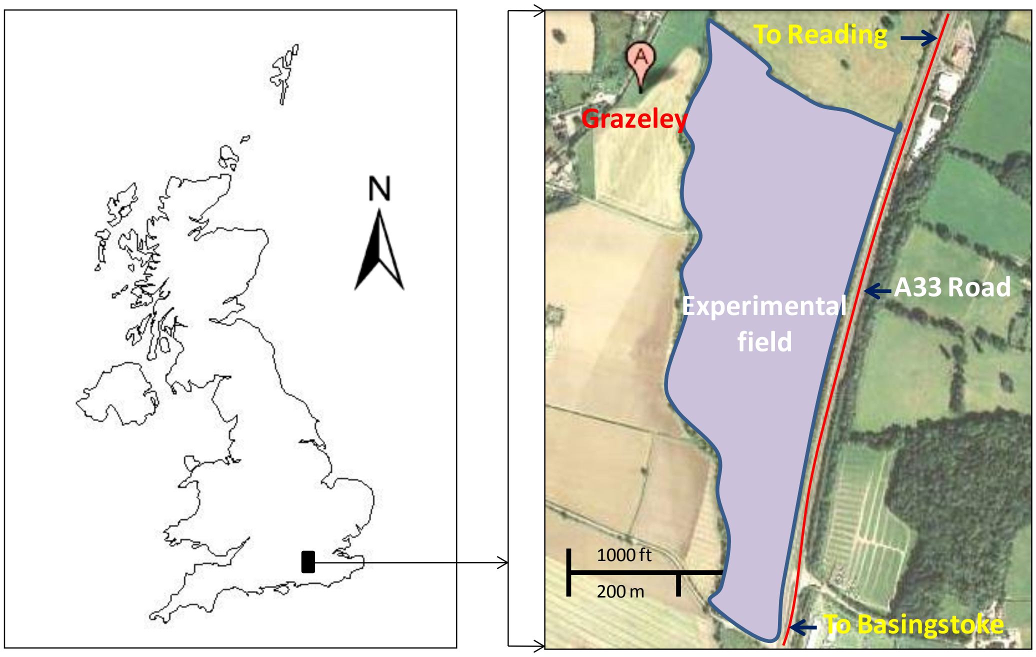

Figure 2: Map of UK showing study area and field site.

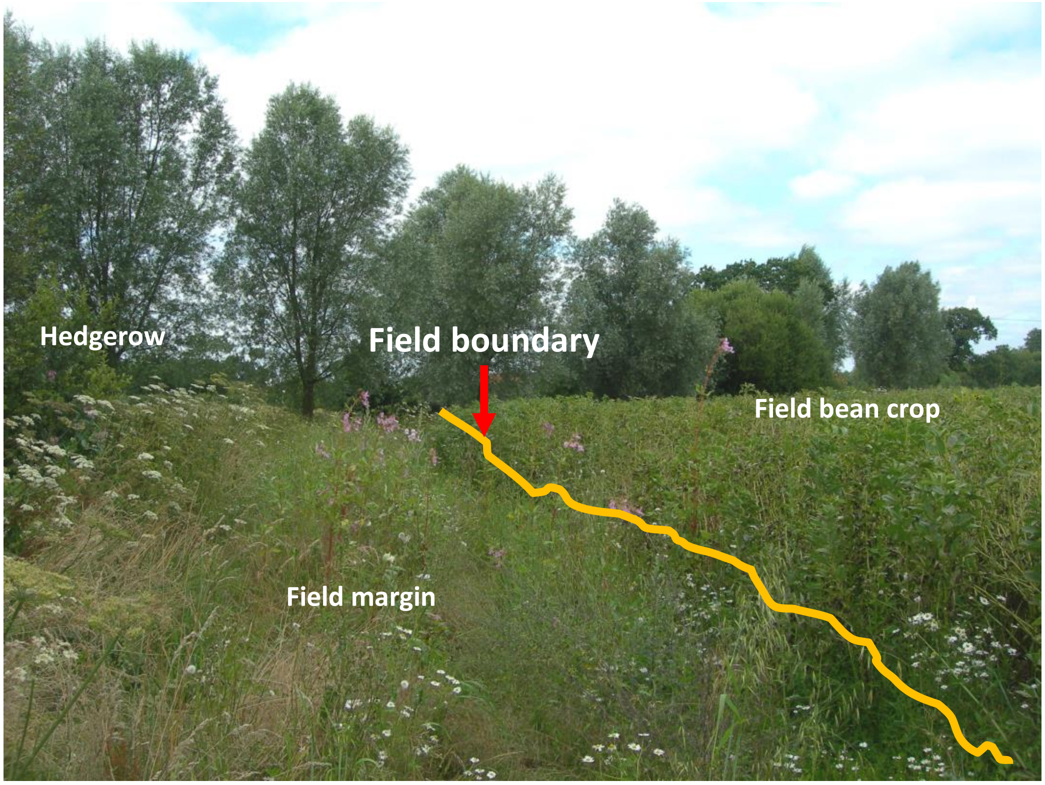

Plate 2: Typical field bean crop.

2.2 Study plants and associated insects

2.2.1 Pigeonpea

Pigeonpea is an erect annual or short-lived perennial plant reaching a height of 1-3 m (Price, 1998). Globally, it is among the most important pulse crops, accounting for about 5% of legume production (Hillocks et al. 2000). It is dominant and widely grown in the dry lower Eastern regions of Kenya, and is cultivated in more than 150,000 ha in total (Johansen et al. 1993). It is mainly grown for human dietary protein provision and fodder for animals (Price, 1998). Pigeonpea is commonly intercropped with cereals such as maize (*Zea mays* L.), sorghum (*Sorghum bicolor* L. Moench), millet (*Panicum miliaceum* L.), as well as legumes, such as beans (*Phaseolus vulgaris* L.) and cowpea (*Vigna unguiculata* L. (Walp.)) (Omanga et al. 1996). A large number of insect pest species attack pigeonpea, key among them, *Helicoverpa armigera* (Hübner) (Noctuidae: Lepidoptera) and *Maruca vitrata* (Fabricius) (Pyralidae: Lepidoptera) (Shanower and Romeis, 1999). It is estimated that pests cause an annual yield loss of about 17% - 27% (Silim-Nahdy et al. 1999). Although its flowers are mainly self-pollinating, out-crossing does occur (Shiying et al. 2002). Insects, especially bees, have been found to contribute between 5 and 40% of the total cross pollination (McGregor, 2002).

2.2.2 Field beans

Field bean (*Vicia faba*) is a common pulse crop grown in most parts of the world, with its origin possibly in central Asia, Mediterranean or South America (Cubero, 1974; Hebblethwaite, 1983; Bond et al. 1985). It is used as either food in developing countries or animal feed in developed countries (Muehlbauer and Tullu, 1997). This crop is considered superior to other pulse crops as it has a high nutritional value. It is also used as a source of green manure, vegetable and skimmed milk substitute (Muehlbauer and Tullu, 1997).

Field bean is an annual herb that grows up to 2 m in height (Bond et al. 1985). It comprises of two sub species: *paucijuga* and *eu-faba*. The latter comprises of three varieties based on seed sizes; *major* or large seeded, *equina* or intermediate and *minor* or small seeded variety (Hebblethwaite, 1983). In English its names are given according to variety type. For example, *minor* and *equina* are commonly referred to as "field beans" where as large *major* is known as "broad beans" (Bond et al. 1985; Polhill and van der Maesen, 1985). The crop generally requires a cool season for best development. Two growing seasons are recognized in the UK, i.e. winter beans sown in autumn and spring beans sown in early spring (Lockhart and Wiseman, 1988). It requires evenly distributed rainfall with an annual range of between 650 and 1000 mm (Kay, 1979) and takes between 90 to 220 days to mature (Bond et al. 1985).

The flower structure is adapted to insect pollination. About 30 - 35% of out crossing is aided by insects (especially bumblebees), but published results range between 4 and 84% (Hebblethwaite, 1983; Bond et al. 1985). More recent studies show that the incidence of insect pollination rarely goes below 50% and averages at 80% (see Stoddard, 1991). Hand pollination experiments have established some level of protogyny (a characteristic of some plants where female flower parts (pistil) mature before the male parts (stamen) are ready so as to avoid self fertilization) (Hebblethwaite, 1983). Common flower visitors include bumblebees (*Bombus spp*.: Apidae), honey bees (*Apis spp*.: Apidae), solitary bees (*Anthophora spp*.: Apidae, *Osmia spp*.; Megachilidae, *Andrena spp*.: Andrenidae, *Eucera spp*. Apidae and *Tetralonia spp.* Apidae). Long tongue bumblebees have been found to be the most efficient pollinators (Kendall and Smith, 1976). Nectar robbing, a process by which a floral visitor takes nectar from a flower without touching its reproductive part and therefore does not benefit the plant's pollination requirement (Inouye, 1980), has been reported on field beans (Bond, 1985). This behaviour is especially common in short tongue bees (Robert, *pers comm.*). The flowers normally open in the afternoon (Hebblethwaite, 1983), at which time, insect visitation rates are generally highest.

Bardner (1983) and Cammell and Way (1983) reviewed known insect pests of field beans. Some of the economically important pests that inflict direct damage to this crop include aphids (*Aphis fabae* (Scopoli), *Aphis craccivora* (Koch), *Acyrthosiphon pisum* (Harris), *Myzus persicae* (Sulzer) and *Meguora viciae* (Buckton) (Hemiptera: Aphididae), Thysanoptera (Thripidae), various species of Lepidoptera larvae, Coleoptera (Bruchidae and Curculionidae) among other pests. It is estimated that over 50 species of pests attack field beans (Hebblethwaite, 1983). Some of these pest species cause serious economic damage, others cause only minor damage. Studies aimed at establishing the insect associations in field beans have categorized them as herbivores, predacious or parasitic (Nuessly et al. 2004). It is the predacious and parasitic ones that are beneficial to farmers because they can naturally suppress insect plant herbivore populations, thereby reducing the cost of production. Such insect taxa are normally referred to as natural pest control agents. Predacious insects tend to be more generalist, while parasitic ones are quite restricted in their host ranges (Snyder, 2009). Since aphids are the most serious pests of most pulse crops including field beans, literature on their bio-control is huge (see Pickett and Bugg, 1998; Noma et al. 2008; Snyder, 2009).

Chapter 3 - Local management and landscape drivers of pollination and biological control services in a Kenyan agro-ecosystem

Abstract

Arthropods that have a direct impact on crop production (i.e. pests, natural enemies and pollinators) can be influenced by both local farm management and the context within which these fields occur in the wider landscape. However, the contributions and spatial scales at which these drivers operate and interact are not fully understood, particularly in Africa. The impact of both local management and landscape context on insect pollinators and natural enemy communities and on their capacity to deliver related ecosystem services to an economically important tropical crop, pigeonpea was investigated. The study was conducted in nine paired farms across a gradient of landscape contexts in Kibwezi, Kenya. Results show that proximity of fields to semi-natural habitats negatively affected pollinator and chewing insect abundance. Within fields, pesticide use was a key negative predictor of pollinator, pest and foliar active predator abundance. On the contrary, fertilizer application significantly enhanced pollinator and both chewing and sucking insect pest abundance. At 1 km spatial scale of fields, significant negative effects of number of semi-natural habitat patches within fields dominated by mass flowering pigeonpea on pollinator abundance were found. For service provision, a significant decline in fruit set when insects were excluded from flowers was recorded. This study reveals the interconnections of pollinators and predators with pigeonpea crop. For sustainable yields and to conserve high densities of both pollinators and predators of pests within pigeonpea landscapes, it is crucial to target the adoption of less disruptive farm management practices such as reducing pesticide and fertilizer inputs.

3.1 Introduction

Historically, there has been a juxtaposition between achieving sustainability in food production and conserving biological diversity, a problem that is particularly pertinent to Africa where growing populations place ever greater pressure on finite biological resources (Abalu and Hassan, 1998; Steffan-Dewenter et al. 2005). In the context of developing countries, the importance of the relationship between agricultural biodiversity, local management and landscape context is poorly studied, while potentially being of great importance for human livelihoods (Abalu and Hassan, 1998). For many crops, agricultural production benefits from both pollination and natural pest control services that are provided by invertebrates (Landis et al. 2000; Ricketts et al. 2008). Insect pollinators and arthropod natural enemies are major groups of invertebrates that significantly contribute to the crop production by delivering pollination and natural pest regulation services.

Insect communities associated with agriculture, whether pollinators, natural enemies or pest species, have been shown to be affected by both local management practices and the context of the farm in the wider landscape (Bianchi et al. 2006; Ricketts et al. 2008). Here, the term ‘landscape context’ is used to refer to the land cover and land use surrounding a site (Bianchi et al. 2006). The impact of local management practices, such as pesticide application and cropping regimes, on invertebrates has been well documented for many agricultural systems. However, the landscape context within which these management practices operate is likely to be of fundamental importance in the maintenance of both pollination and bio-control ecosystem services (Bianchi et al. 2006; Ricketts et al. 2008). To date, attempts to determine the relative effects of landscape context and local management have largely focused on individual species of crops such as Cacao (Cassano et al. 2009) and coffee (Klein et al. 2003); insects such as bees (Carre et al. 2009; Feon et al. 2010), beetles (de la Peña et al. 2003; Purtauf et al. 2005;

Woodcock et al. 2010), spiders (Öberg, 2007; Pluess et al. 2010), and single ecosystem services such as pollination (Kleijn et al. 2006; Brittain et al. 2010; Hagen and Kraemer, 2010) and bio-control (Bianchi et al. 2006; Gardiner et al. 2010; Eilers and Klein, 2010). The interactions between landscape context and farm management for insect communities is also typically well documented only in developed countries, particularly in Europe and North America. There is therefore a pressing need to elucidate the relationship between landscape context, local farm management practices, and the components of insect biodiversity that underpin the provision of ecosystem services, such as pollination and bio-control in developing nations, such as those within Africa.

In this study, the focus was on the impact of local management and landscape context on Kenyan arthropod natural enemies and pollinators associated with pigeonpea, one of the most economically important crops in Kenya (Kimani et al. 1994). The Hypotheses were that: (1) landscape factors are more important drivers of pollinator and natural enemy abundance and species richness and delivery of their related services than local factors (i.e. management), reflecting the large foraging distances that many insects operate over e.g. ballooning spiders (Thomas et al. 2003b), pollinators (Greenleaf et al. 2007) and Carpenter bees (Pasquet et al. 2008); (2) the abundances of arthropod natural enemies and pollinators will correlate positively with the observed provision of their respective ecosystem services.

3.2 Methods

3.2.1 Study design

The study was conducted in Kibwezi District of Kenya between February and June 2007. Three broad regional blocks of agricultural intensification were identified (Fig. 1). These blocks were: (1) Kibwezi Block (2°31'S, 38°01'E), the region characterized by a high proportion of arable land (fields with more than 50% of arable land), and low proportion of semi-natural habitat patches (fields with less than 50% of semi-natural habitats); (2) Makindu Block (2°18'S, 37°57'E), the region characterized by almost equal proportions of arable land and semi-natural habitats; and (3) Athi Block (2°13'S, 38°03'E), the region characterized by a low proportion of arable land and a high proportion of semi-natural habitat patches. In all cases semi-natural habitats were considered to be patches of vegetation that although altered by land management were composed predominantly of native plants and animals.

3.2.2 Landscape context

Within each of the above three regional blocks, three pairs of sites were selected to reflect local landscape conditions necessary to fulfil the pairing of a simple and complex landscape, each pair of farms was separated by at least 2 km and were of approximately the same elevation. Within each pair, the site situated in a simple landscape context was characterised as being far away from semi-natural habitats and surrounded within a 1 km radius by mainly arable land combined with a low proportion of non-crop habitats. The other site of the pair was located within a complex landscape context, which was characterised as close to semi-natural habitats and surrounded by a heterogeneous mixture of crop and non-crop areas within a 1 km radius. This radius was chosen to reflect the typical flight and foraging distances of many insects including pollinators (Greenleaf et al. 2007). In Kibwezi block, the average distance to semi-natural vegetation patches was much further (Mean = 956 m ± 465 m SE) than that in Athi block (Mean = 175 m ± 81 m SE). Likewise, the proportion of semi-natural habitat patches within 1 km radii in the complex landscape farm pairs was much higher in Athi (Mean = 0.42 ± 0.05 SE) than Kibwezi (Mean = 0.15 ± 0.06 SE). Farms where pigeonpeas were intercropped with maize were selected to typify the most common cropping practice of the area. Site details are given in Appendix 1.

In all cases, site selection was based on elevation and land use maps overlaid within ArcGIS 9.2. For elevation the SRTM data (Farr et al. 2007) was used, while land use maps were produced from Landsat 7 Enhanced Thematic Mapper image (2003) combined with additional ground-truthing of site conditions in January 2007. Landscape context and structure were evaluated on rasterized land cover maps (1:500,000). Landscape metrics were then calculated using the Patch Analyst tools in ArcGIS 9.2 (Elkie et al. 1999). The landscape parameters chosen were assessed by generating a correlation matrix, excluding collinear metrics and selecting major factors that have been previously shown to be important in structuring insect communities (Table 1) (Barbaro et al. 2005; Steffan-Dewenter et al. 2005). The landscape metrics retained for further analysis were: the number of semi-natural habitat patches (the measure of spatial heterogeneity within a landscape) of semi-natural vegetation and the proportion of the landscape which was arable land (see Elkie et al. 1999 for full descriptions of these metrics).

Table 1: Correlation matrix of the landscape metrics abbreviated as follows: *No.patch* is the number of semi-natural habitat patches; *MPS* is mean patch size; *MPI* is mean proximity index; *ED* is edge density; *Prop.arable* is proportion of arable land; *IJI* is interspersion and Juxtaposition Index and *Prop.SN* is proportion of semi-natural patches.

	No.patch	MPS	MPI	ED	Prop.arable	IJI	Prop.SN
No.patch	1						
MPS	-0.86	1					
MPI	0.16	-0.5	1				
ED	-0.28	-0.07	0.59	1			
Prop.arable	-0.72	0.76	-0.43	0.02	1		
IJI	0.71	-0.37	-0.16	-0.39	-0.45	1	
Prop.SN	0.72	-0.76	0.43	-0.02	-1	0.45	1

3.2.3 Local management

Structured questionnaire surveys (Appendix 2) were conducted on each farm to assess local management practices. These provided presence/absence information on whether these farms used pesticides and fertilizers. Seven fields were found applying pesticides on pigeonpea, but the remaining 11 did not. Farmers were found to use pesticides preventively regardless of whether or not pest populations are high. Fertilizer application was scored as: (1) low to medium input characterized by the application of organic based manures e.g. green, compost and farmyard and (2) high input characterized by application of large amounts of inorganic fertilizers. Thirteen fields were scored as low to medium input and 5 fields were scored as high input. Both pesticide and fertilizer application were main farm management practices across the study area. For those farmers who did not use these agro-chemicals on their fields, financial restriction was usually the main reason.

Pesticide and fertilizer data were used as explanatory variables representing local farm management practices. These two variables were selected based on the knowledge that they have profound impacts on biodiversity and ecosystem services (see Kevan, 1999; Mozumder and Berrens, 2007).

3.2.4 Measurement of pollinator and natural enemy diversity

At each site, invertebrate sampling and service provision measurements were made along three parallel, 50 m line transects. Each of these transects was located at least 5 m from the edge of the experimental field to avoid edge effects (Laslett, 1982), while individual transects were spaced 10 m apart. All transects taken in the fields had the same orientation i.e. North to South. Sampling was done from February to June 2007, covering pre-flowering, peak flowering and post-flowering stages of pigeonpea phenology.

Pantraps were used to assess insect pollinator (especially bee) diversity and abundance. A cluster of three pantraps (UV bright yellow, blue and white – Westphal et al. 2008), were placed at both ends of each 50-m transects and half filled with water and 1 drop of soap to break surface tension of water. These pans were left for 24 hours before insects caught were collected and preserved in 70% ethanol before identification.

For natural enemies, two main groups were identified i.e. (i) ground (surface) active predators, and (ii) foliar active predators. Pitfall traps were used to mainly sample surface active predators (e.g. predatory beetles, spiders, scorpions). One pitfall trap was sunk at both ends of each transect. The pitfalls were filled with water only, thus avoiding the use of any chemical preservative that could have attracted invertebrates associated with such products (Southwood and Henderson, 2000). These pitfalls were left open for three days before specimens were collected. For foliar active natural enemies (e.g. ladybird beetles and leaf spiders) were collected using beat trays. On each transect, five pigeonpea plants were chosen at random and on each plant, a 50 cm long branch was selected. The branch was vigorously shaken to release any insects to a beat tray held beneath (Southwood and Henderson, 2000). Invertebrates were immediately transferred to insect killing jars impregnated with ethyl acetate and later to storage vials containing 70% ethanol. Both pitfall traps and beat trays also captured pest species that were treated and preserved as described for each method above. These pests were categorized as either chewers or suckers based on their feeding specialisation. For the pollinators, natural enemies and pests, identification to recognizable taxonomic unit (RTU - Oliver and Beattie 1993; Bolger et al. 2000) was undertaken with specialist help from experts at the National Museums of Kenya. A reference collection of the materials is deposited at the National Museums of Kenya and also at CAER laboratory, at the University of Reading.

3.2.5 Measurement of pollination services

Insect pollinator exclusion experiments were undertaken based on protocols described by Ricketts (2008). These were conducted to quantify the contribution of insects to the pollination of pigeonpea for each of the three transects at 5 m, 25 m and 45m points. At each of these points, one plant of the same approximate height and size (~1.5 m tall) was selected. On each of these plants, one branch with fresh flower buds was selected and covered with a Tulle (fine cloth netting) bag to keep out insect pollen vectors, while allowing self and wind pollination. A second branch was covered with an osmolux bag to keep out both wind and insect pollen vectors. A third branch was not covered and thus was left open to both wind and insect pollination. The bags were left on these plants until fruiting when the number of pods formed was quantified. Pollination due to insects, as measured by fruit set, was calculated following the formula described by Ricketts et al. (2008):

Insect Pollination = Open pollination [control] – (Self pollination + Wind pollination [Tulle bags]).

In addition, insect pollinator visitation rates to flowers were quantified by observing a cluster of 10 freshly open flowers on three randomly chosen plants per transect. The flowers were observed continuously for 10 minutes of every hour from 08h30 to 16h30 on a weekly basis during peak flowering season between April and June 2007.

3.2.6 Measurement of pest control service

The impact of natural enemies on foliar insect pests was assessed by recording the extent of leaf damage on pigeonpea plants. It was not possible to control for variation in herbivore pressure which was assumed to be uniform across all sites. A total of five plants were chosen per transect and for each of these the total number of leaves were counted on one randomly selected branch of 50 cm length. Out of the total number of leaves on each of these branches, the number of leaves damaged as a result of chewing insects was recorded, and from this the percentage of leaves damaged per branch was calculated.

3.2.7 Data analysis

Data were analysed in R version 2.10.1 (R Project, 2010). Linear mixed effects models (lmer, lme4 package) were run for the following response variables: pollinator (1) abundance and (2) species richness; floral visitor (3) abundance and (4) species richness; surface active predator (5) abundance and (6) species richness; foliar active predators (7) abundance and (8) species richness; chewing insect pest (9) abundance and (10) species richness; Sucking insect pest (11) abundance and (12) species richness; (13) fruit set and (14) leaf damage.

All response variables were continuous and each was tested using the same mixed effects model structure comprising of both fixed and random explanatory variables. The optimal model structure was specified using random intercept and slope models with different combinations of random effects (phenology, region, area and site) and comparing the Akaike Information Criterion (AIC) numbers of these models (Bates, 2005; Crawley, 2007). The best fit model was one with the lowest AIC number with random variables comprising of field site nested within region and nested within crop phenology (i.e. pre-flowering, peak flowering and post flowering seasons). Each model was fitted with Poisson errors (Bates, 2005; Crawley, 2007). There were 5 fixed effect explanatory factors included in each model. These were: (1) the presence or absence of pesticide usage; and (2) the score of fertilizer application; (3) local proximity to non-crop habitat (considered as a local site effect); (4) the number of semi-natural habitat patches within 1 km radii; (5) the proportion of arable land within 1 km radii; where explanatory variables (1) and (2) were local management effects. (3) was a categorical variable classifying sites as being either near or far from non-cropped habitat and was considered as a local site effect describing sites as either high quality (closer to non-cropped habitat patches) or low quality (further away from non-cropped habitat patches); (4) – (6) described landscape structure and considered as landscape scale effects.

To determine the delivery of each tested ecosystem service (fruit set and leaf damage), pollinator and natural enemy abundance and species richness were included as fixed terms in addition to the above explanatory and categorical variables in the model. For all models, stepwise deletion of least significant effects within the mixed models were used to achieve minimum adequate models for each of the above response variables based on the assessment of their explanatory power. Only those factors whose deletion caused a significant reduction in the explained deviance of the model were retained in the minimum adequate model (Crawley, 2007). Interactions between each landscape effect term with the local and management factor and between the local site effect and each of the management effects were tested. Paired sample t-tests were used to assess the difference between fruit set with and without pollinators in the system.

3.3 Results

3.3.1 Pollinator communities and pollination service

3.3.1.1 Pollinator abundance and species richness

A total of 477 insect pollinators representing 98 species were recorded. The most dominant were honey bees (Apidae: Hymenoptera) which constituted 29.56%, carpenter bees (Apidae: Hymenoptera) (27.46%), sweat bees (Halictidae: Megachilidae) (16.56%), blow flies (Calliphoridae) (12.79%), leaf cutter bees (Megachilidae: Hymenoptera) (9.44%) and bee flies (Bombyliidae) (3.35%).

At local site level, proximity of fields to semi-natural patches had a significant impact on pollinator abundance. Significantly more pollinators were found on sites further from semi-natural patches than those in close proximity to these patches ($P<0.001$ – Fig. 3). Among the fields in the study area that used pesticides, there was strong negative effects of pesticides use on pollinator abundance ($P<0.001$ – Table 2). On the contrary fertilizer application had significant positive impact on the abundance of pollinators. Significant negative interactions were found between the proximity of fields to semi-natural habitat patches and (1) pesticide use ($P<0.001$) and (2) fertilizer application ($P=0.005$). Species richness was not correlated with any of the fixed landscape factors, local site effects or farm management practices. For measures of landscape structure at 1 km spatial scale, pollinator abundance was negatively correlated with the number of semi-natural habitat patches ($P<0.001$ - Table 2). In addition, the number of semi-natural habitat patches was found to negatively interact with (1) pesticide use (P<0.001) and (2) fertilizer application (P<0.05).

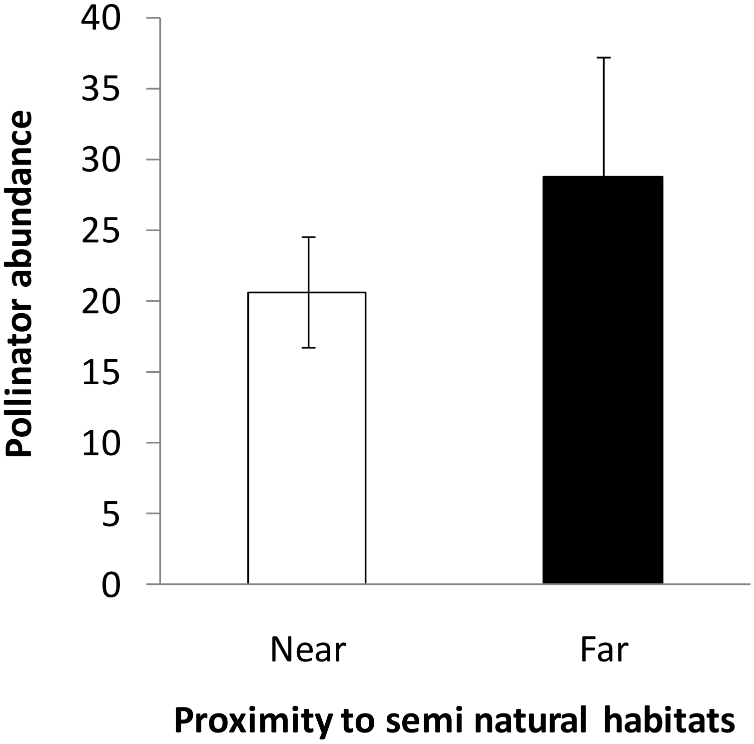

Figure 3: Abundance of pollinators on pigeonpea fields closer to or further from semi-natural habitat patches.

3.3.1.2 Insect visitation rates and fruit set

A total of 393 visits were recorded across sites (Mean per branch 1.90± 0.10SE). Insect floral visitation did not significantly respond to any of the fixed terms, and there were no significant interactions. However, insect visitation rates were positively correlated with fruit set (P<005). Fruit set per branch across fields declined significantly (t = 2.10, P<0.001), when pollinators were excluded from the system (Mean fruit set in fields with pollinators = 88.67±4.81 SE and without pollinations = 22.72±3.23 SE).

3.3.2 Natural enemy abundance, species richness and bio-control service

A total of 1279 arthropods from 149 species recognized as predators of pests based on published information and expert opinion were recorded. Of these, 511 were foliar active predators comprising mainly of sphecid wasps (Sphecidae: Hymenoptera) (59.49%), flies (Diptera) (13.11%), paper wasps (Vespidae: Hymenoptera) (10.37%), ladybird beetles (Coccinellidae: Coleoptera) (7.63%), lace wing larvae (Neuroptera) (4.11%) and Assasin bugs (Reduviidae) (2.15%). Surface active predators were 747 individuals comprising 63.99% spiders, 28.51% ground beetles and 4.82% camel spiders. The others (antlion, rove beetles and springtails) constituted less than 1%.

At local site level, pesticide use had a significant negative effect on foliar-active predator abundance ($P<0.05$ – Table 2). Proximity of fields to semi-natural patches and fertilizer application had no impact on predator abundance. At landscape level, foliar-active predators did not respond to any factor. However, their abundance was negatively affected by an interaction between the proportion of arable fields and pesticide use ($P<0.05$). Species richness was not affected by any of the tested fixed factors. For surface active predators, there were no changes in their abundance and species richness at both landscape and local scales. No significant interactions between fixed terms ($P>0.05$) were detected.

3.3.3 Pest abundance and species richness

A total of 2754 arthropods from 323 species recognized as pests of pigeonpea were recorded. Of this number, 65.1% were found to be chewers and 24.7% to be suckers. The main chewers were;

bean weevils (Chrysomelidae: Coleoptera) (n=851), black crickets (Gryllidae: Orthoptera) (n=142), blister beetles (Meloidae: Coleoptera) (n=188), bush crickets (Tettigoniidae: Orthoptera) (n=111), grasshoppers (Acrididae: Orthoptera) (101), leaf beetles (n=174), leaf miners (Agromyzidae: Diptera) (n=121), stemborers (Pyralidae) (n=200). Sucking insects consisted of; aphids (Aphididae: Hemiptera) (n=49), fruitflies (Drosophilidae: Diptera) (n=263), leaf hoppers (Cicadellidae: Homoptera) (n=171), plant bugs (Miridae: Hemiptera) (n=44), stink bugs (Pentatomidae: Hemiptera) (n=51) and treehoppers (Membracidae: Hemiptera) (n=44).

Chewers abundance and species richness were negatively affected by proximity of fields to semi natural patches ($P<0.001$ and $P<0.05$ respectively). Pesticide application and fertilizer both had significant negative effects on chewers abundance ($P<0.001$ and $P<0.05$ respectively) and species richness ($P<0.001$ for both factors). Proximity to semi-natural habitat patches interacted negatively with pesticide application to affect both chewer abundance and species richness ($P<0.001$ in both cases). Furthermore, proximity of fields to semi-natural patches negatively interacted with fertilizer application to effect chewers abundance ($P<0.01$ – Table 2). Landscape factors did not have significant effects on chewer abundance or species richness.

For suckers, their abundance was only affected by the proportion of arable fields ($P<0.001$), while species richness positively responded to fertilizer application ($P<0.001$ – Table 2). No other factor or interactions significantly impacted on chewers. Furthermore, neither chewers nor suckers abundance nor species richness correlated with leaf damage or fruit set.

Table 2: Z-values and levels of significance from linear mixed effects models relating to responses of pollinator, natural enemy and pest abundance and species richness and insect floral visitation rates, fruit set and leaf damage in pigeonpea fields to five predictive factors: (i) the number of semi-natural patches (*No.patch*); (ii) the proportion of arable land (*Prop.Arable*); (iii) the local proximity to non-crop habitats (*Prox.toSN*); (iv) the presence or absence of pesticide usage (*Pestic.*); and (v) fertilizer application (*Fert*).

(a) Pollinator abundance				
Response	**Mixed effects: minimum adequate model factors**	*Est.*	*SE*	*Z*
Pollinator abundance	No.patch	0.35	0.13	2.69**
	Prox.to SN	1.53	0.23	6.72***
	Pestic.	2.71	0.87	3.12**
	Fert.	1.70	0.47	3.66***
	No.patch:Pestic.	-1.35	0.41	-3.31***
	Prox.to SN:Pestic.	-2.20	0.37	-5.91***
	No.patch:Fert.	-0.34	0.17	.-2.05*
	Prox.to SN:Fert.	-1.82	0.37	-4.93***
Visitation rates	None	•	•	•
Pollinator spp. Richness	None	•	•	•
(b) Natural enemy abundance and species richness				
Surface active abundance	None	•	•	•
Surface active species richness	None	•	•	•
Foliar active abundance	Prop.arable	0.50	0.30	1.67
	Pestic.	-1.24	0.56	-2.22*
	Prop.arable:Pestic.	1.60	0.69	2.33*
Foliar active species richness	None	•	•	•
(c) Pest abundance and species richness				
Chewers abundance	Prox.to SN	0.56	0.09	6.14***
	Pestic.	0.69	0.11	6.57***
	Fert.	0.67	0.10	6.43***
	Prox.to SN:Pestic.	-1.25	0.12	.-10.07***
	Prox.to SN:Fert.	-0.59	0.18	-3.22**
Chewers species richness	Prox.to SN	0.24	0.11	2.26*
	Pestic.	0.26	0.11	2.39*
	Fert.	0.36	0.11	3.38***
	Prox.to SN:Pestic.	-0.66	0.15	-4.43***
	Prox.to SN:Fert.	-0.12	0.16	-0.77
Suckers abundance	Prop.arable	1.11	0.24	4.63***
	Pestic.	-0.48	0.53	-0.90
	Prop.arable:Pestic.	-0.43	0.62	-0.69
Suckers species richness	Fert.	0.48	0.13	3.78***
(d) Delivery of services				
Fruit set	Floral visits	1.00	8.00	9.47*

3.4 Discussion

3.4.1 Pollinator communities and pollination services

Contrary to hypothesis 1, both landscape structure and local farm management were important drivers of pollinator abundance. It is widely reported that an increase in the proportion and number of semi-natural habitat patches enhances the diversity and abundance of pollinators (Ricketts et al. 2008; Landis et al. 2005; Bianchi et al. 2006). However, the findings of this study are not consistent with this pattern, as there was a negative effect of the number of semi-natural habitat patches on pollinator abundance at the landscape scale. This is likely to be due to the fact that the study area falls within the semi-arid landscapes of Kenya, where semi-natural vegetation is characterized by drought hardened plants which are of comparatively low value to pollinators (Bogdan, 1958). Areas of low semi-natural habitat patches are characterised by arable fields dominated by crops. This could contribute to pollinators switching to high rewarding crops, like pigeonpea when they are in flowering phase (Shanower and Romeis, 1999). Furthermore, pigeonpea mass flowers and so may attract large numbers of nectar and pollen feeding invertebrates from the surrounding semi-natural habitats into crop fields. Mass flowering has been demonstrated to be an effective determinant of high densities of pollinators, especially bees, regardless of landscape context (Westphal et al. 2003). This fact is further supported by the finding that fields that were closer to semi-natural habitat patches at the local scale had significantly fewer pollinators than those further away (largely dominated by pigeonpea crop). Pesticide usage was a negative predictor of pollinator abundance and is likely that pollinating insect populations are being severely negatively impacted as a consequence.

The enhancement of pollinators by fertilizer application is possibly due to an increase in floral attractiveness with greater crop growth responding to additional nutrient availability. However, this also caused a dis-service because it enhanced the proliferation of chewing pest populations. Although fertiliser use may result into massive increase in yield of the crop, the tradeoff is challenging and could be a matter of probability to the farmers. An increase in pest population triggers farmers to increase the amount of pesticide inputs, most of which will also result in high levels of mortality for non-target arthropods such as pollinators (see Müller et al. 2005). This is where farmer education and increased awareness could be targeted as an intervention to conserve pollinator species and other beneficial arthropods.

The negative interactions between the number of semi-natural habitat patches at landscape scale and also proximity of fields to semi-natural habitat patches at local scale with the number of pesticide applications confirm the significant impacts of pesticide application on pollinators. Fields that were dominated by the mass flowering pigeonpeas further from semi-natural habitat patches and did not apply pesticides had significantly more pollinators than fields in similar contexts that applied these chemicals. A similar effect was found in fields that applied fertilizers. For pigeonpea fields, it may be important to reduce pesticide and minimize fertilizer inputs in order to conserve pollinator populations, given the significant contribution of pollinating insects to fruit set. However, it is worth noting that in the context of developing countries, particularly in Africa, attaining sustainable pesticide use is extremely difficult given the large inter-annual variability in pest damage and high financial risks associated with non-use of pesticides (Ngowi et al. 2007). In addition, poverty, disease, high levels of illiteracy and an ever increasing population leads to little attention being paid to pesticide regulation. As a result, the future of pollinator communities and consequently their functioning in the ecosystem is likely to be under serious threat from uncontrolled pesticide use (Kevan, 1999).

3.4.2 Natural enemies, pests and pest control service

Farm management played a key role in foliar-active predator abundance and both the abundance and species richness of chewing and sucking pests. Pesticide application had a strong effect in suppressing pest population. However, there were no complementary effects of pesticides and natural enemy abundance or species richness in pest suppression. Instead, an antagonistic effect of pesticide use on natural enemies, particularly on foliar active predators was found. This is not surprising because they have direct contact with leaves and may be more exposed to direct application of pesticides or ingest the compounds via prey items (e.g. predatory beetles - Theiling and Croft, 1988). In the context of these findings, this effect is likely to have been responsible for lack of a correlation between natural enemies and leaf damage. Furthermore, lack of correlation between natural enemy abundance or species richness and leaf damage could mean that there was generally a low proportion of chewing insects following pesticide application at the time this study was conducted.

The interaction between the proportion of arable fields and pesticide application provide evidence that both landscape and farm management factors affected foliar active predator abundance. Fields with a high proportion of arable fields and did not use pesticides had significantly more foliar active predators than those in a similar context but used pesticides.

The presence of a high concentration of pigeonpea in fields further away from semi-natural habitat patches could have resulted in a range of pest communities being attracted. These fields also had high abundance of pollinators and foliar active predators. To enhance the populations of beneficial insects while eradicating pests can be fragile and needs a lot of knowledge and understanding (Kearns et al. 1998). To maintain a high density of both pollinators and foliar pest predators in pigeonpea fields, it is relevant to target a reduction of both pesticides and fertilizer inputs.

Proximity of pigeonpea fields to semi-natural habitat patches and pesticide usage appear to be important local drivers of pollinator, foliar active predator and pest abundance. Although pesticides save a significant amount of crops and therefore economic returns, their cost on biodiversity, especially beneficial insects, cannot be ignored. It is important to apply a careful approach to pesticide usage within set guidelines and regulations and, most importantly, adopt integrated pest management programs to limit the costs to biodiversity and the environment while increasing the benefits of pesticides in crops. This would enhance sustainable pigeonpea yields and conserve high communities of both pollinators and predators of pests within pigeonpea landscapes. In addition local farmer education on the benefits of insect mediated ecosystem services to crop production would greatly boost the conservation of these beneficial insects.

Chapter 4 - Impacts of landscape structure and local management on bee diversity, functional traits and visitation to pigeonpea crops in Eastern Kenya

Abstract

Bees in agricultural fields are critical to crop pollination. The landscape structure surrounding these fields combined with farm management practices can have profound impacts on bee communities. However the extent to which these two sets of drivers impact on bee communities and their ability to deliver pollination services is not fully understood, particularly in African agro-ecosystems. This study investigated the functional responses of bee communities to these drivers in a series of six paired farms along a gradient of landscape structure in Kibwezi, Kenya between April and July 2009. Results show that total bee abundance and species richness are positively affected by habitat complexity. Similarly, the abundance of social, polylectic and nest renting bees was positively impacted by habitat complexity. In addition, the abundance of polylectic bees was further affected by habitat configuration. A negative response of total bee abundance to the number of insecticide application was recorded. A similar response was also seen in the abundance of polylectic and carpenter bees. An increase in both total bee abundance resulted in a corresponding increase in fruit set. Similarly, the abundance of social and carpenter bees correlated with fruit set. From these findings, conservation management targeting maintenance of high landscape complexity within pigeonpea crop fields, and a reduction in insecticide inputs in terms of number of application would enhance bee abundance, species richness and the abundance of social, polylectic, carpenter and nest renting bees.

4.1 Introduction

Pollination is a crucial ecosystem service delivered to crops by bees and other insects and is critical to human well being (Zhang et al. 2007). Colonies of managed bees have become common place in many crop fields in the U.S.A and Europe as farmers use them to supplement native bees and so increase agricultural output (Potts et al. 2010a). However, in the past five decades, there have been severe declines in the numbers of managed bee colonies, including honey bees (*Apis mellifera*) and unmanaged bees (Biesmeijer et al. 2006; Potts et al. 2010b). This threatens the stability of biotic interactions which these insects maintain (Loreau et al. 2001). The significance of these declines is particularly difficult to quantify, especially for wild bees because the biology of most of these bees are poorly understood, although some contribute significantly to the pollination of crops (Klein et al. 2007).

It is widely accepted that agricultural intensification, resulting in habitat loss and fragmentation, causes alterations in the spatial-temporal distribution of resources for insect pollinators, and is one of the main causes of their decline (Ricketts et al. 2008; Winfree et al. 2008; Potts et al. 2010b; Bommarco et al. 2010). The landscapes resulting from fragmentation are commonly characterized by mosaics of semi-natural habitat patches interspersed with agricultural fields. These semi-natural habitats are often of different configurations (i.e. different sizes of land cover types and remnants of natural vegetation – Concepción et al. 2007). The configuration of semi-natural patches is of critical importance in shaping community patterns and interactions among organisms because it determines niche diversity and community structure resulting from fragmentation (Díaz et al. 1998). In addition, landscape effects on local community patterns can be influenced by the complexity of habitats (Tscharntke et al. 2005) which can range from simple habitats (characterized by expanse agricultural fields, monocultures and low proportion of semi-natural patches) to more complex habitats (high proportion of uncultivated habitats and remnants of natural vegetation) (Concepción et al. 2007). Diversity is expected to increase with an increase in landscape complexity due to recolonisation of uncultivated habitats surrounding fields (Tscharntke et al. 2005; Concepción et al. 2007). Due to complex interrelationships among organisms, habitat patches within the landscape (whether simple or complex) need to be connected for species movement between them foraging for resources (With and Crist, 1995; Haddad, 1999; Diekötter et al. 2008). Landscape structure is therefore significant in understanding diversity patterns and species dispersal ability because different organisms require different resources in a variety of habitats within the landscape. With increasing agricultural intensification, habitat transformation and fragmentation, it is even more relevant to understand the long-term impacts of these factors on species diversity, functional responses and ecosystem functioning. Several studies have established close correlations between increasing agricultural intensification and declining abundance and diversity of insect pollinator species (e.g. Kremen et al. 2002; Steffan-Dewenter, 2003; Neumann and Carreck, 2010; vanEngelsdorp et al. 2010).

There is increasing need to monitor the responses of insect populations and community diversity to environmental changes in order to prescribe appropriate measures for their conservation. Monitoring the abundance and species richness responses has been traditionally used in assessing the changes in species composition within insect communities (Moretti et al. 2009; Williams et al. 2010). However, trophic interactions involve multiple species and hence it is appropriate to monitor functional responses of insect communities to environmental changes (Vandewalle et al. 2010). Insects exhibit various life history traits depending on how they adapt to habitat conditions and can thus be grouped into functional traits (de Bello et al. 2010). These are core characteristics that organisms exhibit in response to the environment and cross taxomomic boundaries and have direct effects on ecosystem functioning. Trait diversity, commonly comprising of the range and abundance of functional traits of organisms, can affect ecosystem services and human livelihoods

(Kremen et al. 2005). Functional trait diversity is widely used in monitoring plant responses to environmental changes, but rarely on animal biodiversity monitoring programs (Vandewalle et al. 2010). Traits are useful because common patterns in their response to environmental pressures can be identified within one habitat. These patterns could be applicable to other habitats with completely different species that share similar distributions of traits within their populations and therefore, increasing the generality of findings beyond the specific habitat within which they were undertaken (Williams et al. 2010).

As bees are known to forage across wider landscapes up to several hundred metres (Greenleaf et al. 2007; Pasquet et al. 2008), their taxonomic and functional diversity are likely to be influenced by: (1) landscape context of crop fields; (2) habitat quality (both within the fields and the adjacent natural/semi-natural habitats) as a local site effect; and (3) management practices, such as insecticide application used on individual fields. All three of these factors may influence the diversity of bee communities in agricultural landscapes and thus their capacity to pollinate crops. However, in a tropical setting, the impacts of these factors have rarely been considered (but see Klein 2002, 2003ab; Ricketts et al. 2004; Hoehn et al. 2008; Carvalheiro et al. 2010; Hagen and Kraemer, 2010). This study investigated the effects of these three factors in driving both taxonomic and functional trait diversity of bees and pollination in pigeonpea (*Cajanus cajan*. (L.) Millsp.: Leguminosae), an economically important crop in Kenya. The predictions were: (1) an increase in landscape complexity and configuration increases bee species and functional trait diversity; (2) habitat quality at local site has a positive influence on both bee species and functional trait diversity; (3) insecticide application has negative impacts on bee species and functional trait diversity and (4) an increase in bee species and functional trait diversity results in increased pigeonpea fruit set.

4.2 Materials and methods

4.2.1 Landscape scale factors

The study was undertaken in Kibwezi District, Kenya from April to June 2009. Six site pairs were selected and placed 1 km radius buffers around each site. Site selection was conducted as described in Chapter 3, Section 3.2.2. Non-collinear landscape metrics were selected from a correlation matrix (Table 3) for the 6 site pairs. These were (1) Edge density of semi-natural patches, which is an edge metric defined by the amount of habitat patch edge within a landscape area (i.e. 1 km radius in this case). Edge density measures landscape configuration, and is important in making comparisons between landscapes of variable complexities and sizes. This metric is useful for pollinators because it provides an indication of the diversity and abundance that can be supported by a particular patch type of natural and semi-natural habitats (Chacoff and Aizen, 2006); (2) Mean Shape Index, which is a measure of patch complexity taking into account the perimeter and area of each patch type within the landscape (Elkie et al. 1999; McGarigal and Marks, 1995), and is a useful measure of landscape structure on pollinators (Coulson et al. 2005).

Table 3: Correlation matrix of landscape metrics generated by Patch Analyst within ArcGIS 9.2 at 1 km spatial scale. *MPI* refers to mean proximity index, *MNN* indicates mean neighbourhood distance between habitat patches, IJI refers to interspersion and juxtaposition index, *NP* refers to the number of semi-natural habitat patches, *MPS* refers to mean number of patches, *ED* refers to edge density and *MSI* indicates mean shape index of each habitat patch.

	MPI	*MNN*	*IJI*	*NP*	*MPS*	*ED*	*MSI*
Mean proximity index	1.00						
Mean nearest neighbour distance	0.66	1.00					
Interspersion Juxtaposition Index	-0.11	0.57	1.00				
number patches	0.64	0.99	0.62	1.00			
mean patch size	-0.53	-0.66	-0.41	-0.65	1.00		
Edge Density	0.57	0.32	-0.38	0.30	0.02	1.00	
Mean shape index	0.35	0.43	0.17	0.42	-0.77	0.37	1.00

4.2.2 Local habitat quality and farm management factors

Local habitat quality was also measured at each field site as a local effect that could act as sources of pollinator communities on pigeonpea fields. This was measured in terms of plant percentage cover and species richness within the adjacent semi-natural habitats along 5 transects separated by 10 m and each measuring 100 m in length. This data was later used as fixed effects in the analysis.

To assess the local management used on each site, face to face interviews with farmers were conducted. Variations in levels of fertilizer application and pesticide usage were found to be the main management practices used in the study area. Key among these practices was insecticide usage, which emerged as the most consistent practice either used or not used by farmers. Insecticide application data was therefore used as an indication of the intensity at which the local landscape surrounding each study plot was managed by recording the number of applications and volume of each application (standardized into litres/hectare).

4.2.3 Bee abundance, diversity and pollination services

To measure the abundance of bees visiting flowers, and to assess the amount of pollination service they delivered to pigeonpeas, 100 m long transects were laid in a North to South orientation, each separated by a minimum of 10 m from each other at each site. Five of these transects were within the crop field, five in the semi-natural patches immediately next to the crop and one transect at the interface between the crop field and the semi-natural habitat measuring about 2 m wide. This habitat was consistent in all our study sites and was either a planted hedge or fence with wild plants to mark the boundary of crop fields. Each transect was walked for 10 minutes, twice a day (between 09h00 and 16h00) recording insect flower visitors, 2 m either side.

Pollination service delivery was quantified using exclusion experiments to prevent bees and other insects from visiting flowers to determine the proportion of this service that is attributable to bees (Ricketts et al. 2008). Three plants were selected in each transect within the crop at 5 m, 50 m and 95m totalling to 180 plants (3 plants per transects x 5 transects x 12 sites = 180). Each selected plant would have at least two branches (50 cm long each) with unopened flower buds. One of these branches was covered with a fine cloth netting (Tulle bag) to stop insect pollen vectors. The other branch was left open as a control (open pollination). No other floral visitors were observed apart from insects. The number of pods set on both the experimental and control plants were counted at the end of the experiment and the amount of pollination due to insects quantified following the formula from Ricketts et al. (2008):

Insect Pollination = Open pollination [control] – Self pollination [Tulle bags].

4.2.4 Traits

Data from the above transects were used for trait analysis. The composition of functional traits was measured following published methods used in functional diversity studies (e.g. Michener, 2000; Mason et al. 2007; Moretti et al. 2009; Williams et al. 2010). Ecological and life history traits information was compiled for all species from existing literature, and where gaps existed, from expert opinion. Four of the most ecologically relevant traits were identified and the data was pooled to these trait guilds for further analysis. The traits considered were social specialty, feeding specialisation, nesting specialisation and body size (Table 4). For sociality, three groups were identified as social bees (e.g. *Apis mellifera* and stingless bees), solitary bees (e.g.

Megachilidae and *Ceratina sp.*) and bees with variable sociality, showing primitively social behaviour (e.g. *Halictus spp.*, see Yanega, 1990). Polylectic bees (i.e. generalist bees utilizing a range of pollen and nectar sources e.g. honey bees – Morreti et al. 2009) and oligolectic bees (i.e. specialist bees on pollen and nectar sources - Morreti et al. 2009) were identified to assess feeding specialisation. For nest specialisation, five groups were identified as carpenter bees (e.g. *Xylocopa* spp.), mining bees (e.g. *Amegilla spp*., *Halictus spp*.), mason bees (e.g. *Megachile spp*.), nest renting bee (e.g. *Ceratina spp.* and *Apis mellifera* L.) and bees with no nests (largely comprising of cleptoparasites e.g. *Thyreus spp.*) (see Table 4 for definitions). We did not consider the species whose trait was difficult to determine and consensus on expert opinion was not reached. Intertegular distance (ITD) was measured as the proxy for body size by measuring the distance (in mm) between the sockets of articulation of the wings to the thorax on all specimens using a compound microscope at x60 magnification.

Table 4: Bee functional trait description and subsets under each trait used for analysis. Trait groups were determined based on published literature and through consensus of researcher expert opinion (Moretti et al. 2009). Each trait category was calculated from pooled bee abundance per site. Different subsets of traits per trait group were analysed to determine the response of each to landscape structure and local site conditions/ management.

Trait groups	Categories	Definition
Social status	Solitary	Does not live in colonies
	Social	Colonial life form
	Variable	Shows primitive social life history
Feeding specialization	Oligolectic	Forages on limited resources and requires specific components of the habitat.
	Polylectic	General forager utilizing a broad range of resources.
Nest specialization	Carpenter	Excavates (drills nests in wood).
	Miners	Excavate nests in the ground.
	Mason	Mason, builds nests with mud.
	Renters	Nest in existing tunnels and cavities.
	No nest	Cleptoparasites or parasitic bees.
Bosy size	ITD	Body size as a measure of dispersion. ITD (µm) is the distance between the two wing insertion sockets in the thorax of an insect that can be transformed into a function of dispersal ability (Cane, 1987).

4.2.5 Data analysis

Data were summed for each field from the entire sampling period and analysed using linear mixed effects models in R for Windows version 2.10.1 (R Project, 2010). The data were first tested for normality using Shapiro-Wilk test procedures and where non-normality was detected; they were $\log_{10}$ (n+1) transformed to achieve normality. A total of twelve response variables were tested. These were: (1) the total bee abundance; the species richness of bees (2) solitary bees, (3) social bees, (4) bees with variable sociality, (5) oligolectic bees, (6) polylectic bees, (7) carpenter bees (8), miners, (9) masons, (10) renters, (11) bees with no nests, and (12) body size (measured by intertegular distances – Cane, 1987), (13) fruit set. Each model comprised of both fixed and random landscape, local and management factors. The fixed landscape factors were: (1)

Edge Density and (2) Mean Shape Index. Both these were continuous variables. Local habitat quality factors were (3) plant percentage cover and (4) plant species richness both as continuous variables. The management factors were (5) number of insecticide applications, and (6) insecticide input (litres) per hectare. The site location was used as a random factor to control for spatial variance in all the models run using maximum likelihood to increase parameter estimates (Crawley, 2007). Stepwise deletion tests were used, leaving the least significant fixed terms in each subsequent model until minimum adequate model was achieved with terms whose deletion caused significant changes in the deviance. The difference in fruit set with pollinators included and excluded from flowers was determined using paired sample t-test. This test was used because the paired comparisons came from the same data points. All means are given with ± SE.

4.3 Results

4.3.1 Bee abundance, diversity and pollination services

A total of 1,008 bee visitors from 31 genera were recorded visiting flowers. *Megachile spp.* (Megachilidae: Hymenoptera), *Apis mellifera* (Apidae: Hymenoptera) *Xylocopa spp.* and *Ceratina spp.* accounted for the majority of visits (71%).

There was no evidence that plant cover and species richness adjacent to experimental fields had an impact on overall bee abundance and species richness. The number of pesticide applications had a significant negative impact only on total bee abundance (P=0.007 – Fig. 4a) but not species richness. At landscape scale habitat complexity had a significant positive effect on both total bee abundance and species richness (P=0.002 – Fig. 4b and P=0.027 respectively – Table 5). No other landscape factor had significant effects on these two response variables.

A significant decline in fruit set was detected when pollinators were excluded from the system (t = -7.88, p <0.001 (Mean fruit set with pollinators = 42.08±3.76 and without = 24.58± 2.86)). In addition, total bee abundance significantly correlated with fruit set (P=0.022).

4.3.2 Bee traits

The mean abundance of solitary bees was 25.2±4.9 while that of social bees was 30.0±7.9 and bees with variable sociality were 11.5±2.5. For feeding specialisation, a mean abundance of 75.5±14.9 polylectic bees and 1.20±0.5 of oligolectic bees was recorded. Nest renting bees were the most abundant group of all the five nesting trait guilds selected. The abundance of these bees was 21.3±5.1 while that of mining bees was 15.4±3.0. Carpenter bee abundance was 20.1±3.9 and mason bees 19.9±4.1.

At local scale, plant cover and vegetation composition had no detectable effects on any trait categories. However, farm management involving the number of insecticide application on pigeonpea crop had significant negative effects on the abundance of bees in two sub-trait categories: polylectic bees (P=0.005 – Table 5) and carpenter bees (P=0.028 –Fig. 4c).

Habitat complexity at landscape scale had significant positive effects on social bees (P=0.009), polylectic bees (P=0.002) and nest renting bees (P=0.007). In addition, polylectic bees were significantly affected by habitat configuration (P=044 – Fig. 4d). No other trait category responded to any landscape variables.

A significant positive correlation was detected between social bees and fruit set (P=0.048). Similarly, carpenter bees correlated positively with fruit set (P= 0.006). None of the other traits measured correlated with fruit set.

Table 5: Outputs of linear mixed effects models showing results of the impact of landscape complexity (Mean Shape Index) and configuration (Edge Density); local habitat quality (percentage plant cover) and management (number of insecticide application (number of sprays) and volume of application (input per litre) on the abundance of bees and functional traits. *Est.* refers to the parameter estimates for each main fixed effects. *SE* is the standard error, *T* is the test statistic and *P* is the probability of each main effect in the model.

Response	**Mixed effects: minimum adequate model factors**	*Est.*	*SE*	*T*	*P*
(a) Bee abundance and species richness					
Total bee abundance	Edge Density	-0.02	0.01	-2.45	0.058
	Mean Shape Index	0.70	0.12	5.71	0.002
	No. Pesticide sprays	-0.10	0.02	-4.42	0.007
Bee species richness	Edge Density	-0.01	0.01	-2.07	0.093
	Mean Shape Index	0.26	0.09	3.09	0.027
	No. Pesticide sprays	-0.04	0.01	-2.51	0.054
(b) Functional traits					
(i) Sociality					
Social	Mean Shape Index	0.58	0.15	3.80	0.009
	No. Pesticide sprays	-0.07	0.03	-2.17	0.073
Solitary	None				
Variable	None				
(ii) Lecty					
Polylectic	Edge Density	-0.02	0.01	-2.67	0.044
	Mean Shape Index	0.74	0.12	6.05	0.002
	No. Pesticide sprays	-0.10	0.02	-4.74	0.005
Oligolectic	None				
(iii) Nesting					
Carpenter	Mean Shape Index	0.43	0.18	2.37	0.056
	No. Pesticide sprays	-0.10	0.04	-2.88	0.028
Mason	None				
Miner					
Renter	Mean Shape Index	0.71	0.19	3.81	0.007
(iii) Body size					
Body size	None				
(iv) Pollination service delivery					
Fruit set	Total bee abundance	0.42	0.14	2.94	0.022
	Social bees	0.35	0.15	2.39	0.048
	Carpenter bees	0.43	0.11	3.89	0.006

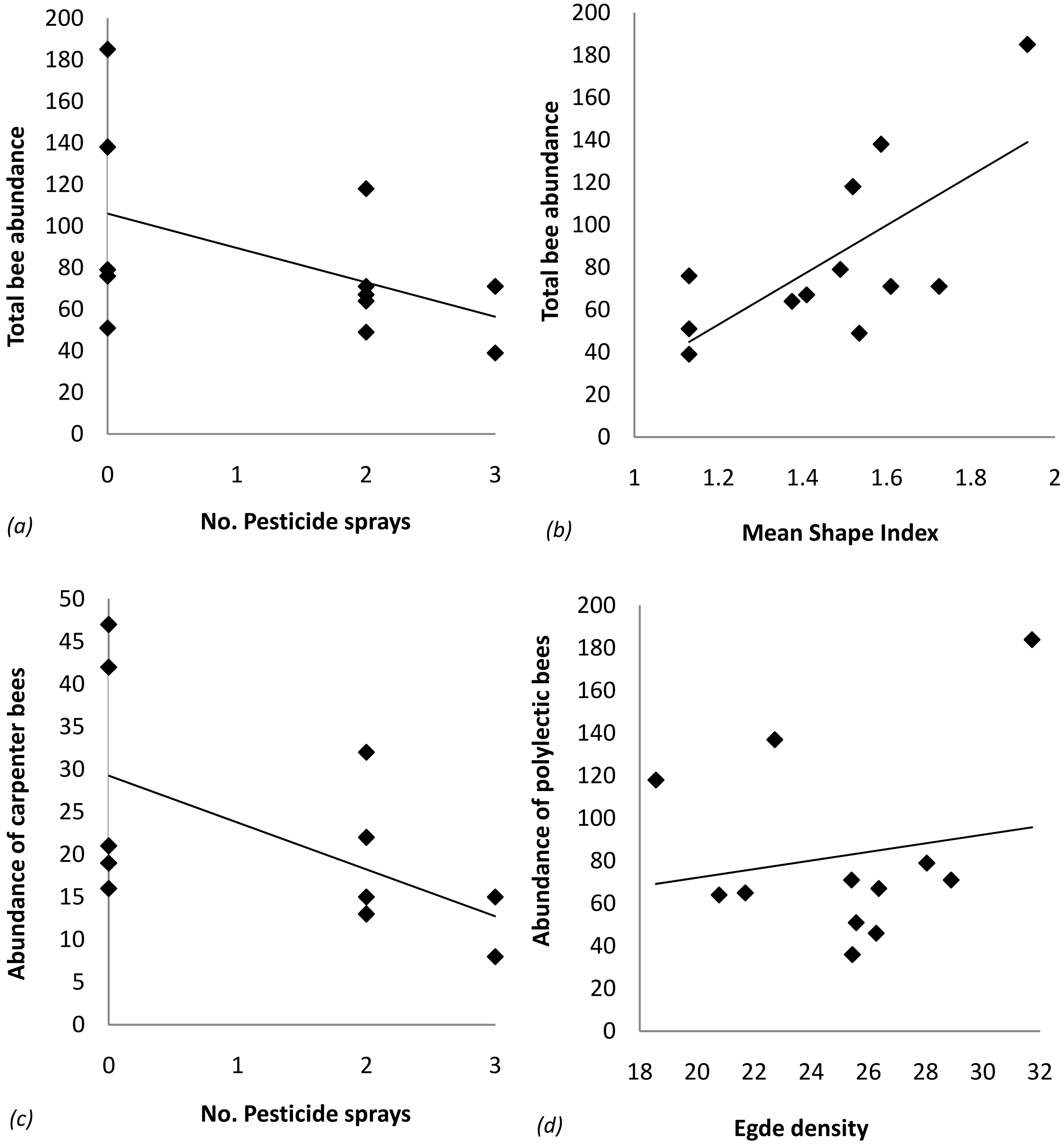

Figure 4: Relationship between main effects of landscape structure and local management on total bee abundance (a-b) and the abundance of carpenter bees (c) and polylectic bees (d). Values at "0" on the x-axis (e.g. 2a) indicate fields with no pesticide application.

4.4 Discussion

4.4.1 Bee abundance, diversity and pollination services

At landscape scale, habitat complexity played a key role in driving bee abundance and species richness within pigeonpea fields. These results can be explained by diversity of resources (e.g. nectar, pollen and nesting) normally associated with complex landscapes (Concepción et al. 2007). As a result, pigeonpea growing in more complex landscapes benefitted from an increased number and species of bees. Relating the results from fruit set data to abundance, it is clear that pigeonpea requires bees to increase its yields. These results are similar to those of Klein (2002) drawn from a tropical context which found increased fruit set in highland coffee as a result of increased diversity of pollinating bees.

Bee species diversity and interactions with their habitats are known to be driven by a combination of factors both in their immediate local surroundings (e.g. forage and nesting availability), those at landscape level (e.g. when they need additional resources that are not available within their immediate habitats - Steffan-Dewenter, 2002; Potts et al. 2005; Williams and Kremen, 2007), all within their flight ranges (Pasquet et al. 2008).

The number of pesticide applications negatively impacted on bee abundance. This finding emphasizes knowledge that pesticide usage leads to the loss of bees through disruptions of trophic interactions and persistence of toxicity (McLaughlin and Mineau, 1995; Steffan-Dewenter et al., 2005; Brittain, 2010).

Within the study area, pigeonpea is a critical crop due to its ability to withstand long droughts and grow on nutrient poor soils. However, the contribution of bees to the production of this crop is widely unappreciated. And until now, the effects of pesticide application on bees in these landscapes were not known. Significantly fewer bees recorded on less complex landscapes and on fields that apply more pesticides justify the need to target maintaining high quality fields combined with reduced number of pesticide application to promote the conservation of bee species diversity. As revealed in this study, an increase in bee abundance and species richness results in increased pigeonpea fruit set. Therefore farmers within these Kenyan semi-arid landscapes categorised as areas of low agricultural potential (Waiganjo and Ngugi, 2001) would benefit from improved yields.

4.4.2 Bee traits

The use of functional traits in understanding ecosystem functioning has recently gained interest from ecologists because it is recognised as a critical component of biodiversity (e.g. Lavorel and Garnier, 2002). In this study, habitat complexity played an important role in shaping the community of social bees, polylectic and nest renting bees. In addition, the abundance of polylectic bees was further driven by habitat configuration. These patterns can be attributed to an increase in diverse resources within the bees foraging landscapes and availability of a variety of substrates to rent their nests (Cane, 2001; Williams et al. 2010). The abundance of social bees was driven by habitat complexity. This subset of bee sociality trait is the most diverse in tropical regions (Roubik, 1992; Michener, 2000) and contributes to the efficient pollination of many plants including key crops such as coffee (Klein et al. 2003a; Ricketts, 2004) and fruits and vegetables. The persistence of social bees in this study area is critical for crop production because apart from paying substantial visits to pigeon pea flowers and being responsible for increased fruit set, they also have a potential to benefit other crops such as tomatoes, carrots, squashes, okra and onions that are commonly grown as intercrops (personal field observation). The effects of landscape complexity were also consistent for the abundance of polylectic bees which responded positively to this driver. This subset of feeding specialisation trait was dominant in all study sites. Their general foraging behaviour underpins a key strategy for survival and adaptability to new forage sources especially in a changing environment and fragmentation that might introduce new plants and floral resources (Cane, 2001) and hence more stable than oligolectic counterparts that are more resource restricted in diet breadth (Williams et al. 2010). The positive effects of increasing habitat complexity on nest renting bees demonstrate the important role of this driver in structuring the communities of renters (e.g. honey bees) within pigeonpea landscapes that play a substantial role in the pollination of many crops (Cane, 1991; Petanidou and Ellis, 1996; Potts et al. 2003). It is absolutely critical for these habitats to be configured in a way that bees can easily access resources (Tscharntke et al. 2002; Steffan-Dewenter, 2002).

The number of insecticide application had a greater impact on the abundance of polylectic and carpenter bees. It is likely that forage substrate (nectar and pollen) is poisoned by continued insecticide application. Similar findings were reported by Johansen et al. (1983) on leafcutter bees and attributed the rapid decline in these bees abundance on pesticide poisoning. Carpenter bees play a critical role in pigeonpea pollination as their visitation rates positively correlated with fruit set. A decline in their population will severely impact on yields.

This study has revealed that structurally complex landscapes (with high patch complexity and configuration) dominated by pigeonpea crop combined with insecticide application have a direct effect on bee communities and their ability to deliver pollination service to this crop. Based on these findings, conservation management targeted at maintaining a high complexity of habitats within pigeonpea landscapes would enhance the abundance and species richness of bees and increase key trait guilds such as social and polylectic bee abundance as well as carpenter and nest renting guilds that are key visitors to pigeonpea flowers. Reducing the number of insecticide application will particularly enhance the abundance of polylectic bees. Similarly, such a reduction would enhance the abundance of carpenter bees. The negative impacts of insecticide application on bee functional traits poses a particular threat to the existence of bee communities within these small scale farming systems and the contribution of these bees to the production of pigeonpea. So far, this is the first report of such an effect in a tropical context and on this crop. Considering the practicalities of adopting these result findings in enhancing pigeonpea production in Kenya, a multi-stakeholder intervention is required. This should combine small scale individual farmers to cooperative, NGOs and Ministry of Agriculture extension officers operating within these regions to facilitate the adoption of landscape level management of bees that deliver pollination services.

Chapter 5 - Spillover of ecosystem services from field boundaries: pollination and pest regulation in a field bean crop.

Abstract

Edge habitats surrounding intensively managed agricultural fields are of considerable conservation value for insect communities, and can provide key ecosystem services such as pollination and natural pest control to crops. In this study I investigate the effect of plant position within a bean field on the amount of pollination and natural pest control it receives. Sampling was done at increasing distances from the field boundary (wild flower rich field margin and hedgerow) along transects in a typical field bean crop in South East England. Results indicate significant declines in generalist pollinator and predator abundance and species richness with increasing distance from field boundary. However the abundance of *Bombus hortorum* (specialist visitor to field bean flowers) and *Coccinella septempunctata* and *Adalia bipunctata* (Aphid specialists) did not significantly decline with increasing distance from field edge. Parasitoid abundance increased significantly with increased isolation from field boundary and positively correlated with aphid abundance. Fruit set significantly dropped when pollinators were excluded from the system but this was not affected by distance from field edge. These results show that the abundance of specialist species is determined by the availability of specific resources regardless field boundary features. It is therefore important to consider the role of specialist pollinators and natural enemies in field beans production systems rather than the overall diversity of floral visitors and natural enemy communities while investigating spillovers of related ecosystem services. Furthermore it is critical to take into account the specific characters and habitat requirement of these specialists in order to understand their individual responses to landscape elements and how these affect their capacity to deliver ecosystem services.

5.1 Introduction

Increasing agricultural intensification can be linked to severe declines in biodiversity which in turn negatively impacts on ecosystem service provision (Steffan-Dewenter et al. 2007). Insects are a key group of arthropods that have an essential role in ecosystem functioning, yet they are among the most severely affected by agricultural intensification (Wilby and Thomas, 2002; Hendrickx et al. 2007; Geiger et al. 2010; Féon et al. 2010). In Europe Agri-environment schemes provide financial incentives to farmers (European Commission, 2005) to promote conservation of wildlife and habitats and thus amerolating the effects of biodiversity loss (Ryszkowski et al. 2009), disruption of trophic interactions (Clough et al. 2010) and habitat loss. In many farmlands, crop boundaries are often characterized by semi-natural habitats, such as field margins (grass or wild flower strips), hedges, fences, uncropped strips, and conservation headlands demarcating the edge of the field (Marshall, 2006).

Field boundary habitat can have a major influence on many species, especially invertebrates, which are the most numerically dominant forms of native wildlife utilizing these habitats (Kleijn et al. 2006; Thomas et al. 2008). Arthropods are important subset of invertebrates that are critical to crop production, either as beneficial insects (pollinating crops or regulating pest populations) or as pests reducing crop yields (den Belder et al. 2002; Mandelik and Roll, 2009; Fernandes et al. 2010; Holzschuh et al. 2010; Sigsgaard, 2010). Field boundaries provide key habitats for many insects (Haenke et al. 2009; Delattre et al. 2010) and cross edge dispersal between these habitats and adjacent crop fields occur, a phenomenon referred to as spillovers. The distance to which these spillovers extend from field boundary habitats into the crop depends to a large extent on an arthropod foraging range (Rand et al. 2006; Rand and Louda, 2006). Many studies investigating arthropod spillovers have largely focused on patterns of change in species diversity as a result of fluctuations in environmental conditions (Rand et al. 2006). Arthropods (e.g.

natural enemies of pests or pollinators) may disperse from their habitat in order to acquire additional resources (e.g. alternative hosts, pollen or nectar) or as a response to factors such as competition or other changes in ambient conditions, such as food availability (Fagan et al. 1999; Kleijn et al. 2006).

Increased natural enemy density at field boundaries can be an important factor in reducing prey populations within adjacent habitats (both crop and natural/semi natural) (Dennis and Fry, 1992; Sunderland and Samu, 2000; Fernandes, et al. 2010). Profiling the extent of suppression of natural prey populations into the field from the boundary is an important step in understanding the effect of distance from field boundary on this ecosystem service. Current knowledge on this subject are mainly drawn from a handful of studies focusing on the abundance of arthropod natural enemies (e.g. French et al. 2001; Baldssera et al. 2004; Rand and Louda, 2006), as opposed to functional implications in terms of the degree of pest control they deliver. Also such studies have largely been confined to boundary fauna, rather than investigating the shifts in their related services with increasing distance from field boundary (but see Bowie et al. 1999). On the other hand, the decline in insect mediated pollination services (as measured by fruit set and seed set) with increasing distance from field boundary has been drawn from only a few crops and wild plants (e.g. Dramstad and Fry, 1995; Steffan-Dewenter and Tscharnke, 1999; Bäckman and Tiainen, 2002; Morandin and Winston, 2006). It is therefore warranted to extend these studies to investigate the patterns of insect pollinators and natural enemy communities from the field boundary towards crop interior and to quantify how much ecosystem services (such as pollination and natural pest control) are affected by this same factor. Furthermore, it is important to conduct such investigations on commercially important crops, especially in conventional fields because they are more intensively managed and therefore pose a more serious threat to wildlife than organic fields (Kremen et al. 2002).

This study focused on field beans crop in South East England and examined how the position of a bean plant in a field affected how much pollination and natural pest control it received. The prediction was that (1) the abundance and species richness of floral visitors will decline with increasing distance from field boundary. The reason for this is that field margins planted with wild flowers would support more floral visitors than crop because they provide a diversity of resources that is needed by most pollinators (Carvell et al. 2007); (2) natural enemy and field bean pest abundance and species richness will increase with increasing distance from field boundary because some species of natural enemies are known to track their prey (Zang et al. 2009); (3) a decline in floral visitor abundance and species richness will result in reduced fruit set, and (4) an increase in natural enemy abundance and species richness will result in increased pest suppression.

5.2 Materials and methods

5.2.1 Study design.

The study was undertaken at Grazeley Farm, Berkshire, UK at 51^{0}23’N and 0^{0} 59’W (Fig. 2). In 2008, five 150 m × 70 m replicate plots were identified each separated by 150 m between them (Fig. 5). In each plot, 4 transects, each measuring 32 m in length were laid from the edge towards the centre of the field. On each transect, 5 sampling points were located at 2, 4, 8, 16 and 32 m. Farm boundary type was kept roughly constant for all the replicates (as a grass/flower margin, hedgerow and fence). Sampling was conducted for eight weekly rounds from May to August 2008 to coincide with the flowering and fruiting periods of field beans.

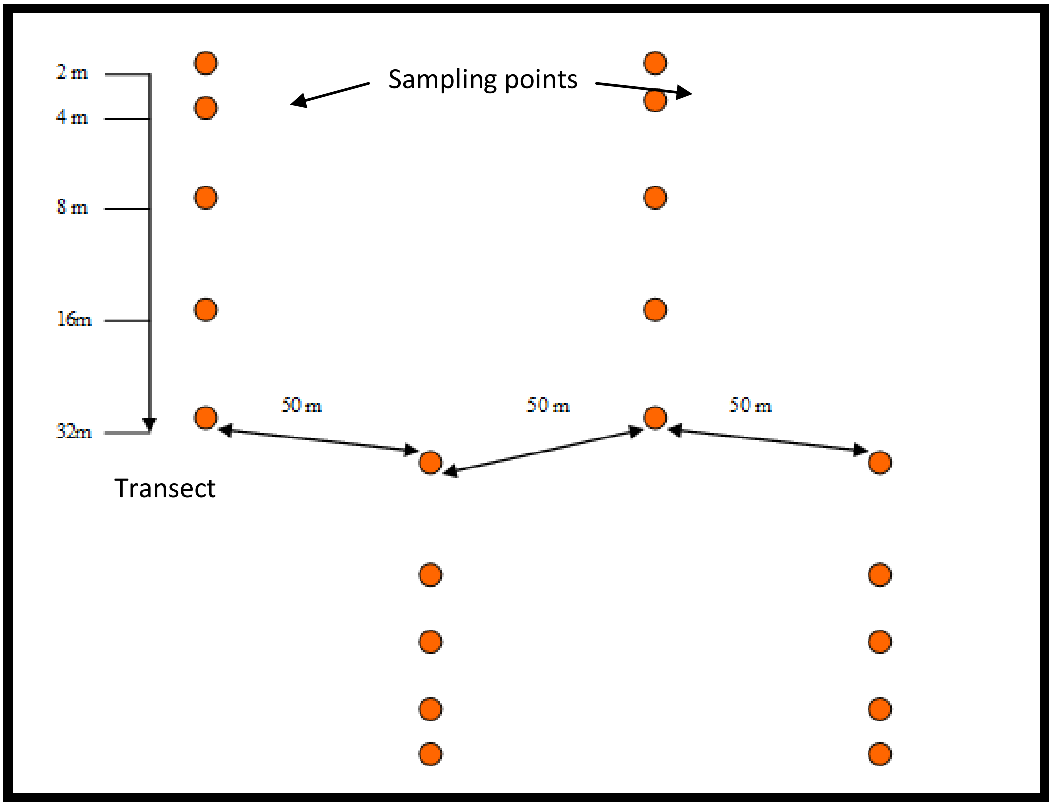

Figure 5: Outlay of transects per plot.

To estimate the abundance and species richness of insects visiting flowers, two plants were selected at random at each of the distances. One of the plants was continuously observed for 10 minutes each round (two rounds per day) between 0900h to 1700h recording all flower visitors. At the beginning of each observation, environmental variables (temperature, humidity and cloud cover) were recorded using Kestrel® 3500 Environmental Meter at the top of the vegetation. Two distinct floral visiting behaviours were observed among visitors (predominantly by bumblebees); some bees visited the flowers from the open end of the corolla and so made direct contact with the floral reproductive surfaces (i.e. legitimate visitation), while others bit off the base of the corolla tube from the outside and sucked nectar (i.e. nectar raiding). These visitation behaviours were recorded separately for analysis purposes. Individual visitors were identified to species and morphospecies level for some predators and parasitoids.

The abundance and species richness of natural enemies of pests were also determined on this plant by hand searching technique. Here natural enemies refer to invertebrates in the families of Coccinellidae (Coleoptera), Syrphidae (Diptera), spiders (Araneae), and parasitoids of which the former two are typically specialist predators of Aphids, while the spiders are more generalist invertebrate predators. The plant was searched for 6 minutes at a time and the species identity and abundance of natural enemy invertebrates found on the leaves, stems and seed pods recorded. Similarly, the number of insects known to be pests of field beans (Aphididae: Hemiptera, Curculionidae: Coleoptera and Bruchidae: Coleoptera) on each plant was recorded. The rate of population growth of aphids per week was calculated from the first date of sampling using the formula: Average Growth Rate (Agr) $=ln(Nt /N0)$ = rt: where ln is the natural log, Nt aphid population in the successive week, N_0 is the population in the previous week and rt is the rate of growth (Odum and Barrett, (2005). It was difficult to quantify the number of parasitoids present on leaves and stems, therefore, their abundance and species richness were assessed from parasitized aphids outlined below.

5.2.2 Pollination and pest control services

In order to quantify pollination service delivery, visitation rate data were used alongside seedset data derived from bagging experiments on the second plant. This plant was bagged using a fine mesh cloth at the onset of flowering in May 2008 when the crop was about 10% into flowering (Stuart *pers. Comm.*). The bags were regularly checked and moved upwards as fruits set and the plant continued to grow taller. The bags were left throughout flowering and fruiting periods until a week before harvesting. The first plant was open to all insect visitors. At the end of the experiment, the pods formed on both the bagged and open plants were harvested for further analysis. Pods set by open plants were counted and averaged to give an approximate fruit set per plant. Fruit set due to insects (*IP*) was determined by subtracting the number of pods set by the bagged plant (*SP*) from the average number of pods set by the open plants (*OP*). i.e. $IP = OP\text{-}SP$ (following Ricketts et al. 2008). The harvested pods were brought back to the lab where their wet weight was determined, and then dried in the oven at 105^0C for 24 hours before the dry weight was determined. The total number of seeds per pod was counted and dry weight calculated per seed.

Pest control service was quantified by visually counting aphid mummies on each experimental plant. From these, a sub sample of 200 mummies were taken for incubation until parasitoids emerged to quantify emergence rates and to detect any shifts in their abundance and species richness with increasing distance from the edge. The mummies were incubated in the oven at 25^0C for 14 days. The emerged parasitoids were preserved in 70% ethanol for later counting and identification to species. As a further proxy for pest control, the abundance of pests on the experimental plants was quantified using the same protocol for predators and analysed to detect any significant relationships.

5.2.3 Data analysis

Data collected over the 8 week period were summed up for each parameter at each sampling point (i.e. 5 data points per transect x 4 transects per replicate x 5 replicates). All these data were analyzed in R for Windows version 2.10.1 (R Project, 2010). Linear mixed effects models (lmer, lme4 package) were used, an approach suited to spatial nesting that retains the variance structure of the raw data (Crawley, 2007). Model specification was undertaken to obtain the optimal structure for data analysis using random intercept and slope models with different combinations of random effects (replicate, transect and the position within transect) and comparing the Akaike Information Criterion (AIC) numbers of these models (Bates, 2005; Crawley, 2007). The best fit model was one with the lowest AIC number which had distance from field boundary as a fixed continuous variable and transect nested within replicate as random effects fitted with poisson errors. Markov-chain Monte Carlo (MCMC) simulations were used with a chain length fixed at 5000 based on highest-posterior-density (HPD) and confidence interval fixed at 95% to assess the significance of model fixed effects (Plummer et al. 2006). For each model, declines were tested with one response variable at a time. A total of six response factors: (1) floral visitor abundance; (2) pest predator abundance; (3) parasitoid abundance; (4) pests abundance; (5) pollination service and (6) pest control were analysed for the variables shown in Table 6. Simple regression models were used to test for linear associations and relationships between predator and parasitoids abundance and species richness with that of prey and floral visitors' abundance and species richness with yields parameters. The predator to prey and aphid mummy and aphid ratios (Cohen, 1977) were also calculated as indicators of the strength of pest regulation. These ratios were analysed with lmer linear mixed effects models using binomial error distribution with logit link because they have strict upper and lower bounds (Schwartzberg et al. 2008). Student t-test was used to determine the difference in means between fruit set on open and bagged plants.

5.3 Results

5.3.1 Abundance and species richness of floral visitors, natural enemies and pests

5.3.1.1 Floral visitors

A total of 382 insect visits to the flowering field bean plants were recorded from 8 different species, comprising bumblebees (*Bombus spp.*, Apidae: Hymenoptera), honey bees (*Apis mellifera*, Apidae: Hymenoptera and solitary bees (*Andrena spp.* Andrenidae: Hymenoptera). *Bombus hortorum* L. (Apidae) was the main floral visitor accounting for 88% of all the 202 legitimate visitations recorded. Although the long tongue bees (e.g. *B. hortorum*) are the known legitimate visitors to field beans flowers (Free and Williams, 1976), occasional legitimate visitation by some short tongue bee species such as *B. terrestris* L. (Apidae) and *B. Lucorum* L. (Apidae) were observed. Other short tongue bees (*Apis mellifera* L., *Andrena wilkella* K., *B. pratorum* L. (Apidae)) raided nectar from flowers. Both total insect flower visitation and legitimate visits significantly declined with increasing distance from field boundary (t = -12.82, P=0.05 and t = -8.80, P=0.05 – Table 6). Although *B. hortorum* visitation followed this trend, their decline from the field boundary was not significant (P>0.05 – Fig. 6).

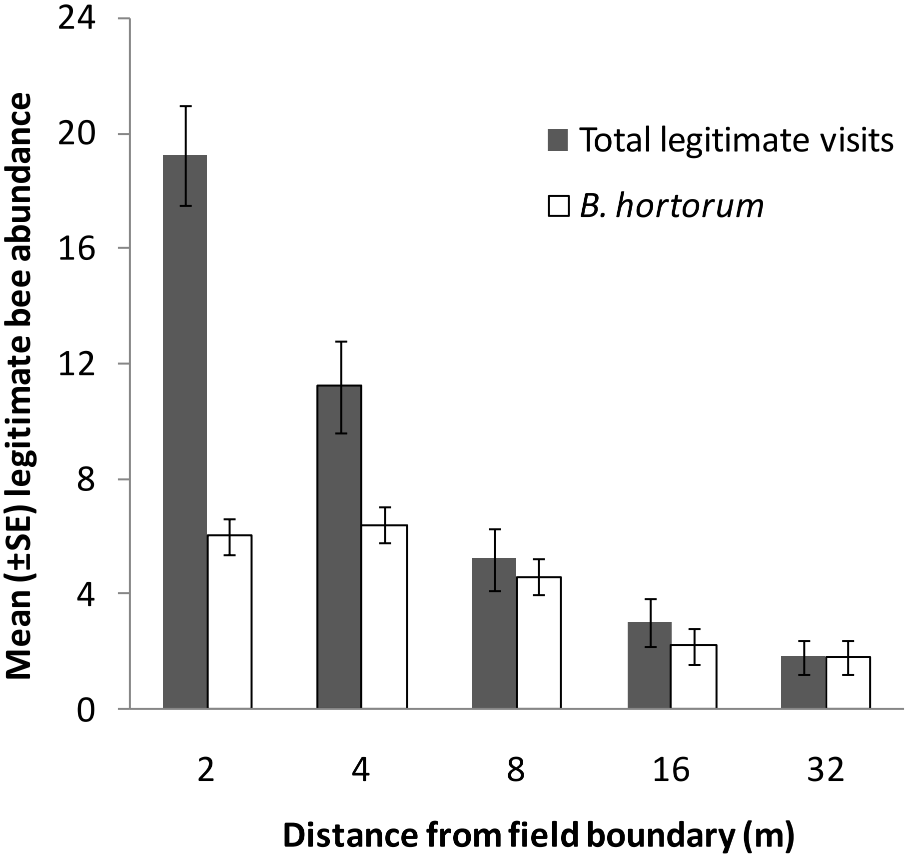

Figure 6: Mean abundance (±SE) of legitimate flower visitation by all insects and *B. hortorum* with increasing distance from field boundary.

Flower visitor species richness (combining both legitimate and nectar raiders) was significantly higher at the field boundary (t = -6.80, P=0.05 – Fig. 7) and systematically reduced towards field interior. Incidents of nectar raiding, recognized from holes punched by bee mandibles on the bases of floral petals, were observed along the whole transects, but this was not significantly affected by distance.

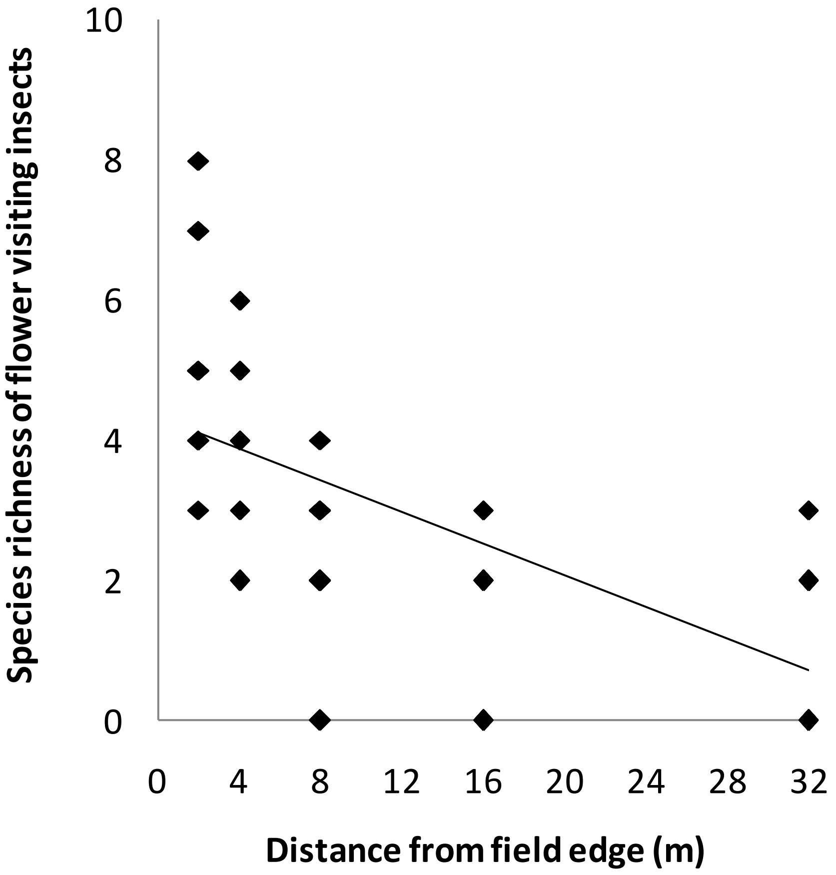

Figure 7: Relationship between species richness of flower visiting insects and plant distance.

5.3.1.2 Pests and natural enemies

A total of 24,622 individual pests were recorded. The abundance of these pests was mainly influenced by that of the black bean aphid (*Aphis fabae* Scopoli (Aphididae: Hemiptera). Other common species encountered were *Myzus persicae* Sulzer (Aphididae: Hemiptera), mainly found on leaves and stems and *Bruchus pisorium* L. (Bruchidae: Coleoptera) and *Sitonia lineatus* L. (Curculionidae: Coleoptera) found on pods. In contrast to predator distribution into the crop interior, pest abundance increased with increasing distance from the boundary (t = 4.04, P=0.05 – Fig. 8). This was largely as a result of higher abundance of *A. fabae*, which systematically increased with increasing distance from field boundary (t = 4.87, P=0.05 – Table 6). *Bruchus pisorium* and *S. lineatus* were restricted to the edge of the field, although their distribution into the crop did not significantly decline. Similarly, pest species richness did not significantly decline (P>0.05) with increasing distance from the edge.

A total of 205 individual pest predators (4 species) mainly Coccinellidae (Coleoptera) (74%) and Syrphidae (Diptera) (26%) were observed. Three adult species of Coccinelids (*Adalia bipunctata* L., *Coccinella septempunctata* L. and *Harmonia axyridis* Pallas (all larvae and adults)) and one species of hover fly larvae (*Syrphus ribesii* L. Syrphidae: Diptera) were the most abundant of all the known natural enemies of field bean pests observed. The overall abundance of pest predators significantly declined with increasing distance from field boundary (t = - 2.74, P<0.05 – Fig. 8). The abundance of *S. ribesii* larvae increased systematically with increasing distance from the field boundary, although this increase was not significant (P>0.05). However, there were no significant declines found in the abundance of *C. septempunctata* and *A. bipunctata* with increasing distance from field boundary. Pest predator species richness significantly declined from the field boundary towards the crop interior (t = -7.86, P=0.05).

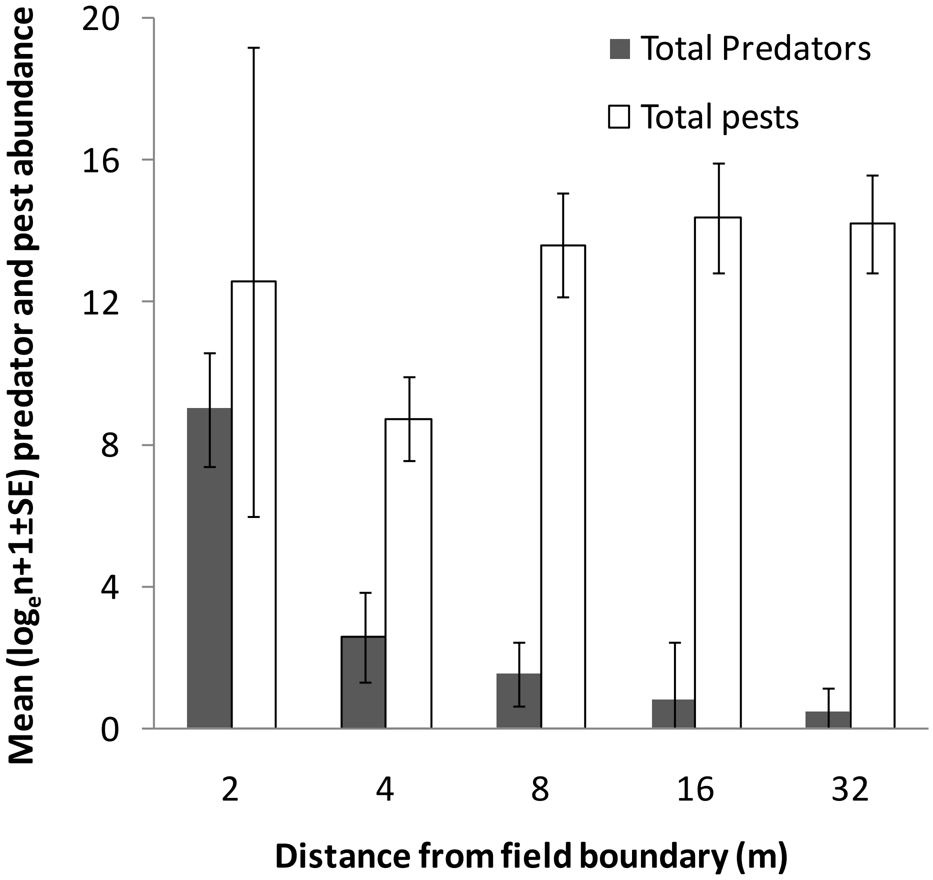

Figure 8: Mean abundance (±SE) of total pests and total predators with increasing distance from field boundary.

A total of 125 individuals of parasitoid wasps from 4 species (*Aphidius spp.* Hymenoptera, *Diaeratiella spp*. Braconidae: Hymenoptera, *Ephedrus spp.* Aphidiidae: Hymenoptera: and *Praon spp.* Braconidae: Hymenoptera) and 1unidentified species were recovered from 200 Aphid mummies (mainly from parasitized *M. persicae*) representing a 62.5% emergence rate. *Diaeratiella spp*. and *Aphidius spp.* were the most abundant species representing 40% and 30% of all the recovered parasitoids. Parasitoid abundance recovered from parasitized mummies systematically increased with increasing distance from the field boundary (t = 6.24, P=0.05 – Fig. 9), mainly due to *Diaeratiella spp*. (t = 3.63, P=0.05) and *Aphidius spp.*

(t = 4.87, P=0.05). However, species richness of these parasitoids did not respond to increasing distance from the field boundary (P>0.05).

Table 6: Results of linear mixed effects models relating to effects of distance from field boundary and insect pollinators, predators, and pests of field beans and related yield parameters.

Category	**Response factor**	**t-value**	**β**
Floral visitors	Total legitimate visits	-8.80*	-0.98
	Total overall visits	-12.82*	-1.20
	B. hortorum	•	•
	B. pascuorum	•	•
	B. terrestris/lucorum	•	•
	Floral visitor species richness	-6.80*	-1.45
Pest predators	Total predators	-2.74*	-0.76
	Syrphus spp.	•	•
	Coccinella septempunctata	•	•
	Adalia bipunctata	•	•
	Predators species richness	•	•
Parasitoids	Total parasitoids	6.24*	0.84
	Aphidius spp.	4.87*	1.00
	Diaeratiella spp.	3.63*	0.33
	Ephedrus spp.	•	-
	Praon spp.	•	-
	Parasitoid species richness	•	-
Pests	Total pests	4.04*	0.79
	Aphis fabae	4.87*	1.23
	Myzus persicae	•	•
	Bruchus pisorum	•	•
	Sitonia linaetus	•	•
	Pest species richness	•	•
Pollination service parameters	Number of pods/plant	•	•
	Seeds per pod	•	•
	Dryweight per seed	•	•
Pest control service parameters	*Myzus persicae* mummies	9.06*	1.13
	Predator:prey ratio	z = -3.56***	-0.07
	Aphid mummy:aphid ratio	•	•

*Note: *=P<0.05; ***=P<0.001;* ß=Slope, •=non significant

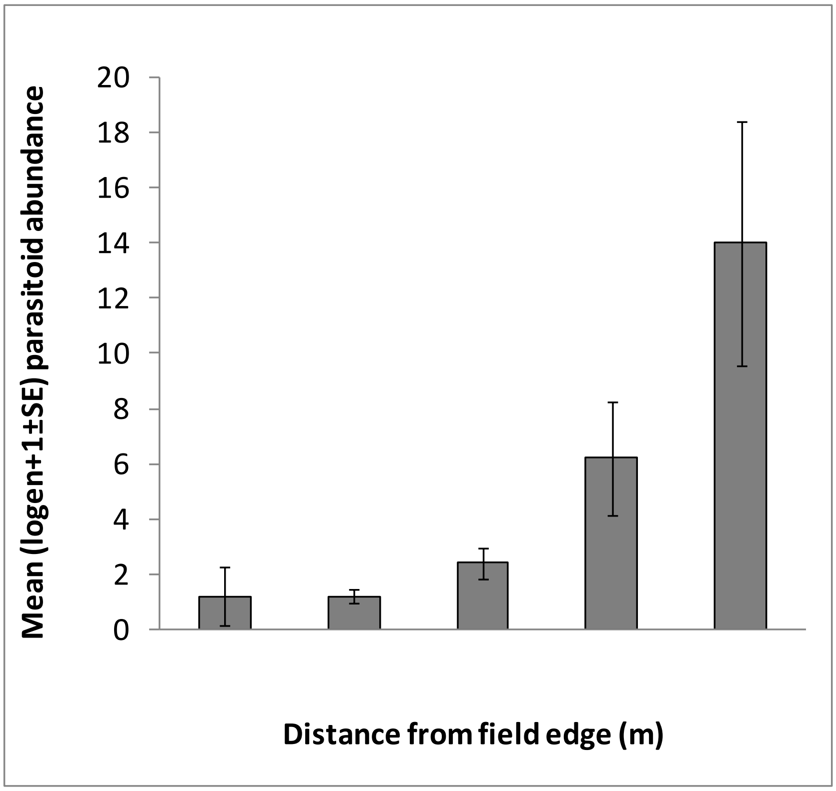

Figure 9: Mean parasitoid abundance (±SE) with increasing distance from field boundary

5.3.1.3 Pollination and pest control services

The crop yield parameters tested (number of pods per plant, number of seeds per pod and dry weight of seeds) remained unaffected with increasing distance from field boundary. However, bagged plants produced significantly less pods than open plants (t = -11.59, P<0.001- Table 7) across all replicates. There was no evidence that flower visitor abundance and species richness were directly affecting these yield parameters (P>0.05).

Table 7: Mean pod set per plant with increasing distance from field boundary.

	Mean of fruit set per plant (Mean±SE)	
Distance from field boundary (m)	**Bagged plant**	**Open plant**
2	10±1.46	24.2±2.46
4	13.2±2.68	19.0±1.78
8	10.4±1.07	22.8±1.08
16	10.4±1.23	18.8±2.08
32	10.0±1.00	17.4±1.40

Although the population growth rate did not significantly change with distance from field boundary, that of *M. persicae* was far lower and grew slower than that of *A. fabae*. However, parasitized mummies of *M. persicae* were the only evidence of biological control, although dense aggregations of *A. fabae* made it difficult to find mummified individuals. The abundance of *M. persicae* mummies was significantly enhanced with an increase in distance from crop edge ($t = 9.06$, $P=0.05$). With reference to predator to prey ratio; this was found to significantly decline with increasing distance from field boundary ($z = -4.23$, $P=0.001$; $\beta=-0.10$ – Fig. 10). Aphid mummy to aphid did not show any significant changes with increasing distance from field boundary.

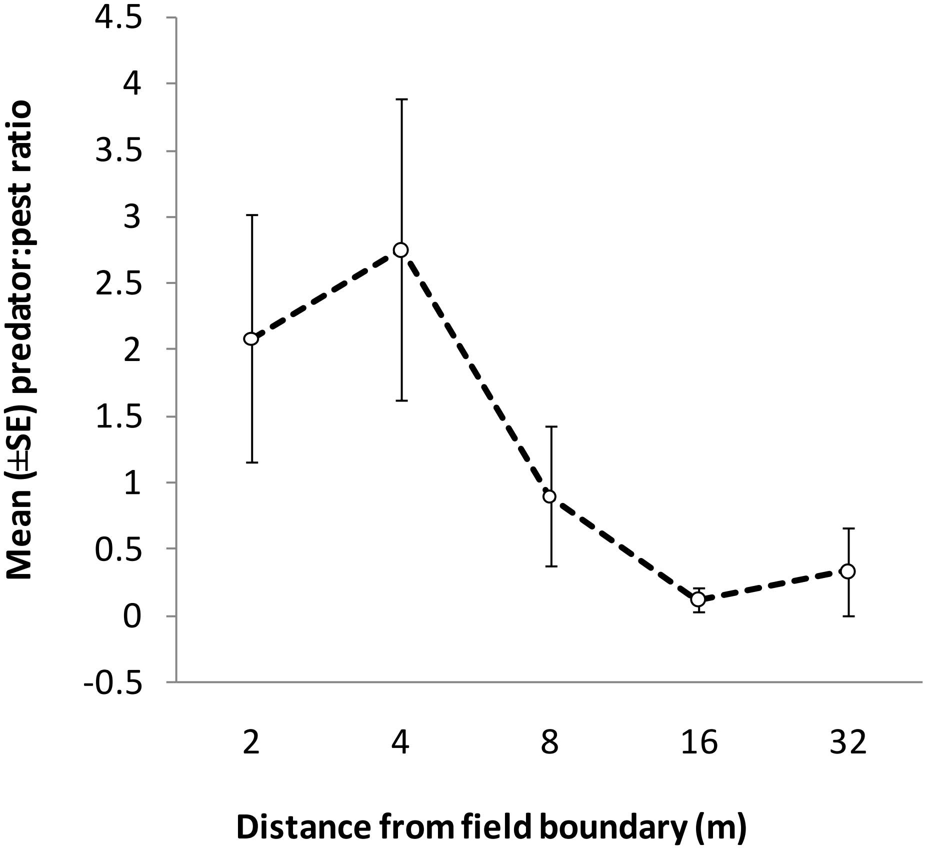

Figure 10: Decline in predator:prey ratio with increasing distance from field boundary.

5.4 Discussion

The sharp decline in the abundance of floral visitors observed towards the field interior strongly suggest that these generalist species are utilizing resources outside the crop field where there was a mixture of flowering plants at the field margin. However, there was no effect of distance on the *B. hortorum* (the main legitimate visitor). This result might be due to the specialization of this species in utilizing field bean flowers (Free and Williams, 1976), although 32 m represents a very small flight distance for bumblebees which can fly several hundreds of metres (Greenleaf et al. 2007). This finding indicates that the specialists are not driven by the presence of diverse resources (e.g. presence of pollen and nectar sources), but rather by the availability of the specific requirements within their narrow breadth and would disperse across the landscape in search of these resources (Jauker et al. 2009; Bommarco et al. 2010). Whereas the short tongue bees largely visited flowers of the plants in the field margin, the long tongue *B. hortorum* concentrated on visiting field bean flowers deeper into crop. It is likely that this was why there was no pollination deficit anywhere in the test area of the crop despite a negative gradient of overall legitimate visitation from the field boundary towards field interior. This is confirmed by the lack of a significant effect of distance from field boundary on *B. hortorum* visitation.

For predators of pests, although overall abundance declined significantly with increasing distance from field boundary, both the aphid specialist predators *A. bipunctata* and *C. Semptempunctata* remained unaffected. These two species are effective predators that can significantly regulate aphid populations (e.g. *A. fabae* and *M. persicae*) (Lommen et al. 2008). This finding is interesting because it also indicates the fact that specialist species are not solely driven by the diversity of resources, but by their specific requirement within the habitat (Fahrig, 2003). Aphid abundance showed a significant surge in population with increasing distance from field boundary which was opposite to the distribution of total predators at same distance intervals. This is likely to have been caused by increased predation closer to the field boundary perhaps due to the complementary effects of generalist predators present in those habitats in addition to predation by *A. bipunctata* and *C. Semptempunctata*. This is further supported by the significant decline in the predator to prey ratio. However, as distance from field boundary increases, aphids appear to be relieved from heavy predation but the abundance of the two specialist predator species remains significantly unaffected. This is likely to be due to uniform foraging by the specialist predators across the crop. In addition, it is possible that the population of specialist predators was far lower than detectable threshold required to exert substantial suppression of aphid population (Collins et al. 2002).

Parasitoids are known to be closely linked with their hosts both spatially and temporally (Vollhardt et al. 2008; Zang et al. 2009). This explains the increase in their abundance from the field boundary towards the crop interior similar to the trend recorded for aphids. Unlike most generalist natural enemies, most parasitoids are co-evolved with their host species and may not require alternative hosts (Vollhardt et al. 2008). Furthermore the absence of a high abundance of parasitoids closer to the field boundary implies that they are not dependent on alternative vegetation to overwinter such as that at the field boundary but possibly on their hosts instead (Stary´, 1970. This ensures a steady lifecycle almost entirely dependent on hosts rather than on the vegetation composition or other habitat elements at the field boundary or in the surrounding landscapes (Vollhardt et al. 2008). Moreover these patterns could be explained by the fact that parasitoids are specialised in utilising honeydew from aphid hosts rather than rely on nectar provided by the high diversity of flowers on the field boundary (Wa¨ckers et al., 2008).

These results suggest that distance from field boundary towards crop interior is only important in determining the abundance and species richness of generalist floral visitors and predators. Specialist species appear to be driven by the availability of specific resources within the habitat regardless for the composition of the landscape and hence their spill over is not driven by field boundary features. In order to understand the effects of landscape context and field boundary features on the functioning of pollinators and natural pest enemies underpinning crop production, it is important to take into account the taxon specific characters for each relevant species within these landscapes.

Chapter 6 - General Discussion

6.1 Summary of findings

6.1.1 Local site conditions and farm management

This study has generated important insights into how both pollinator and natural enemy communities in both pigeonpea and field beans are affected by landscape context and farm management in two different systems; Kenya and the United Kingdom. In the Kenyan system, pesticide application was the main important driver of pollinator abundance in pigeonpea crop. Particular effects were on bee communities where an increase in the number of insecticide application, corresponded with significant declines in the abundance of polylectic and carpenter bees. Proximity of fields to semi-natural habitats had a negative impact on pollinator abundance. However, fertilizer application significantly increased the abundance of pollinators. For natural enemies, foliar active predator abundance responded negatively to pesticide application. Similar effects were recorded for both chewing and sucking pests.

6.1.2 Landscape structure

At 1 km spatial scale, landscape structure had major impacts on the abundance of both pollinators and foliar active pest predators. For pollinators, the number of semi-natural habitat patches within fields interspersed within mass flowering pigeonpea fields had negative effects on their abundance. Habitat complexity positively correlated with total bee abundance and species richness. Furthermore, the complexity of semi-natural habitat patches positively enhanced the abundance of social bee, polylectic bees and nest renting bees.

Foliar active predator abundance was negatively impacted by an interaction between the proportion of arable fields and pesticide application. A similar response was also detected in the abundance of both chewing and sucking insects.

Significant decline in fruit set was recorded when pollinators were excluded from this system. These results indicate that pigeonpea requires insects for its production. Therefore continued pesticide application poses a risk to sustainable yield production. Conservation strategies that target a reduction in pesticide inputs and maintaining landscape complexity may enhance the existence of pollinators and predators and ensure stable crop production.

6.1.3 Spillover of ecosystem services from boundaries

In the UK study, the distance from field boundary had significant effects on both generalist pollinator and predator abundance and species richness in field bean crop. However, the abundance of the long tongue *B. hortorum* specialised on visiting field beans flowers was not affected by distance from field boundary. Similarly, aphid specialist *A. bipunctata* and *C. Septempunctata* abundance was not affected by distance from field edge. On the other hand, parasitoid abundance increased significantly with increasing distance from field boundary. This was largely due to the significant increase in the abundance of *Aphidius spp.* and *Diaeratiella spp.* A similar response was detected for pest abundance largely due to the significant increase in *Aphis fabae* abundance.

When pollinators were excluded from the system, the number of pods set significantly reduced. On the other hand, the abundance of *Myzus persicae* increased significantly with an increase in distance from the field boundary. However, predator to prey ratio declined significantly with distance.

6.2 Implications of findings

6.2.1 Impacts of local site conditions and management on pollination and pest control

Of all the measured local site factors, insecticide use emerged as a major factor linked to reduced abundance of pollinating insects and natural enemies. The compounds in these chemicals are widely used in controlling pest species attacking crops, but have also been demonstrated to have severe negative impacts on non target species such as pollinators and natural enemies (Theiling and Croft, 1988; Brittain et al. 2010). In an African context, the problem of insecticide use is of particular concern since their application depends on many variables such as household income, level of literacy and availability (Mbuvi, 2009). Added to this, there is very little understanding that the presence of beneficial insects among the pool of insects regarded as pests, could provide financial benefits for the growers. Furthermore, the contribution of pollinating insects and natural enemies to crop production is widely unappreciated and hence there is little effort to conserve them. However, this thesis reports direct benefits of pollinating insects to pigeonpea yields and the consequences of insecticides to their survival. A reduction in the pesticide is necessary to the sustainability of pigeonpea production.

It should be considered that while a reduction in pesticides may increase the population sizes of beneficial insects, which would potentially offset the decrease in yield associated with an increase in pest numbers. Most pigeonpea farmers operate at a subsistence level and it might be particularly challenging to sell them the idea of reduced pesticide application because they may not want to take the risk for what may be tentative benefits. One possibility of implementing this would be through farmer education and possibly explore links with organic production sales to the west, where people are effectively paying for the increased risk linked with the absence of pesticide use.

6.2.2 The effects of landscape structure on pollination and biological pest control

The results from pigeonpea studies showed strong effects of landscape structure on both pollinators and natural enemies of pests. In previous studies it has been reported that both pollinator abundance and species richness respond positively to increasing complexity in landscape structure, in terms of the proportion of semi-natural patches, within a given spatial scale (Landis et al. 2005; Bianchi et al. 2006; Ricketts et al. 2008). Contradictory results were found for insect pollinators in the presented 2007 study that showed a negative response of their abundance to both the number of semi-natural patches at 1 km spatial scale. These results may have been due to pollinators switching to the high rewarding mass flowering pigeonpea fields from the surrounding low resourced semi-arid vegetation habitats. Therefore, appropriate management of these pigeonpea fields i.e. by reducing pesticide application to a minimum would profit pollinator species.

Based on the 2009 study on bee communities in the same region, landscape structure was critical in determining both abundance and species richness of bees within pigeonpea fields. Habitat complexity was of particular significance in affecting bee communities. Similar results were recorded for bee functional traits where strong positive impacts of increasing complexity on social, polylectic and nest renting bees were found. These bees capacity to effect pigeonpea pollination would rapidly be compromised as a result of direct losses in their populations as a consequence of habitat homogenisation and loss. It has been shown that habitat homogenisation results into disruption of species interactions and food webs leading to unstable trophic levels (Laliberté and Tylianakis, 2010). As a precautionary measure, it is critical to conserve landscape complexity in order to maintain the diversity within these habitats to ensure the preservation of bee communities. Given that bees were found to be key to increased pigeonpea yields, adopting landscape level conservation may help ensure the provision of pollination services.

Natural enemy abundance responded negatively to the interaction between the proportion of arable land and pesticide use. Significantly more foliar active predators found within fields dominated by a high proportion of arable fields without pesticides in the system show the benefits and conservation potential for these natural enemies relived from pesticide pressure.

6.2.3 Spillover of ecosystem services from boundary to field bean crop.

The results from the field beans study demonstrate the importance of understanding how generalist and specialist species are driven by different components of crop and surrounding habitat features. Generalist species form the majority of diversity of insects within field bean crop fields. However, their abundance is more concentrated on the field boundary and therefore of little direct benefit to field bean production. Specialist species are on the contrary unlimited by the composition of field boundary features and forage deep within crop. It is important to examine the specific habitat requirements and diet breadth of these species in order to fully understand their functioning and how dynamics in their population impacts on ecosystem service delivery.

6.2.4 Insect conservation and ecosystem services in agriculture

6.2.4.1 Importance of scale

From the pigeonpea study, both landscape context and local farm management contributed to driving pollinator and natural enemy communities. For the conservation of these insect communities, it is important to understand how the dynamics of habitat change both at the local scale and landscape level affects their survival.

At a local scale, factors such as habitat quality (both measured in terms of plant abundance and species richness) and farm management have profound effects on both insect abundance and species richness (Steffan-Dewenter and Tscharntke, 2001). Good quality habitats can support a large diversity of insects because they present more niches and trophic interactions (Moilanen and Hanski, 1998; Thomas et al. 2001), and as a result the magnitude of ecosystem services is increased in these habitats. The 2009 study examining the effect of habitat quality on bee communities and pigeonpea pollination did not find any significant effects as predicted. This result can be attributed to the local bio-physical conditions of each study site. First, the study area falls within a semi-arid zone with poor soils and scarce precipitation (e.g. rainfall) impeding plant growth (Bogdan, 1958). As a result only drought resistant species with few resources (floral and foliage) grow. In these conditions, insects may easily switch to agricultural fields or may forage across wide geographical areas to acquire these resources, therefore having a short resident time in these poor habitats. It should be noted that these conditions can drastically change during short spurts of rainfall when the vegetation surrounding these crop fields, especially annual plants recover very quickly and insect abundance increases (Mr. Wambua, *pers. Comm.*).

Farm management used on crop fields is an important local factor that determines the insect community on these fields. In both 2007 and 2009, the effect of pesticides was an important factor negatively affecting both the pollinators and foliar active pest predators on fields that applied them. Conserving beneficial insects and eradicating pests at the same time requires an understanding of local site conditions that shape the community structure and interactions at larger spatial scales (Kearns et al. 1998). Farmer education is necessary to increasing the knowledge base of the potential impacts of losing agricultural biodiversity.

Some insects have large foraging ranges that extend beyond the single habitat level (Steffan-Dewenter et al. 2002). An understanding of how landscape structure in terms of complexity and configuration affect patterns of change within insect communities is crucial to underpinning the ecosystem functioning of these insects (Steffan-Dewenter and Tscharntke, 1999; Krauss et al. 2003). The 2007 and 2009 studies both showed significant effects of landscape structure to pollinators and natural enemies. Landscape structure largely depends on the habitat and crop types in the region. In addition, environmental factors, local site conditions (e.g. habitat quality) and management are major contributors to landscape elements (Wassen et al. 2003). These factors should be taken into account when planning conservation measures. In order to fully understand insect diversity at landscape scale, it is critical that all stakeholders be involved with land use both in cropped areas and non cropped areas (e.g. farmers, policy makers and land managers; Jackson et al. 2007). Engaging all these stakeholders ensures the conservation of species in the wider landscape (Bawa et al. 2004) and is relevant in Kenya where most agricultural fields are still traditionally managed.

6.2.4.2 Managing habitats for ecosystem services

At the global scale, biodiversity is under threat from many forces such as increasing demand for natural resources to meet the needs of an ever increasing human population, poverty and climate change (Shah, 2010). Increasing species losses consequently leads to a reduction in the amount of ecosystem services (Wilkie et al. 2006). Management of habitats by addressing the forces driving species losses is critical for the sustainability of ecosystem service delivery. In the present world it is difficult to attain a win-win scenario which simultaneously tackles the problem of species loss while at the same time promotes sustainable utilization of natural resources (Jackson et al. 2007). In some parts of the world, this has been tried with success (e.g. the agri-environment schemes in Europe – European Commission, 2005 and countryside stewardship – Countryside commission, 2005). There is evidence that these management prescriptions are benefiting biodiversity more than fields that don't apply them (Knop et al. 2006; Carvell et al. 2007; Wilson et al. 2007).

To obtain a win-win scenario, tradeoffs between conserving biodiversity and ecosystem services and sustaining human livelihoods must be made and requires cooperation and understanding about the rationale behind such interventions (Adams et al. 2004; Sandker et al. 2009). Achieving positive success with these types of interventions is particularly challenging in Africa and other developing nations, which still struggle with poverty leading to high extraction rates of natural resources (Collier, 2007). In addition existing policies (e.g. the Agricultural policy in Kenya) fail short of addressing key issues of conserving biodiversity within agricultural landscapes. Lack of clear guidelines on resource utilization poses a threat to biodiversity and ecosystem services. One solution to this would be that all stakeholders agree on strategies that will promote a win-win biodiversity conservation and agricultural production, perhaps even explore links with the west to ensure stability in ecosystem service provision.

In view of my study findings, promoting both pollination and natural pest control in pigeonpea, requires the adoption of conservation level management practices that serve both conservation and livelihood interests. A better understanding of patterns and flows of ecosystem services within these landscapes is needed to target specific conservation efforts to a particular service (Ricketts, 2004; Kremen, 2005; Zang et al. 2007). In addition, managing these ecosystem services involve decisions on what tradeoffs to make when planning land use (Zang et al. 2007).

For example a decision on utilizing land solely for crop production will involve compromising biodiversity conservation and potentially a reduction in the associated ecosystem services while the opposite effect would result from conserving habitats for species protection. The optimal balance would be adopting both land use types through strategies such as agroforestry (Ashley et al. 2006) and land sparing (Perfecto and Vandermeer, 2010).

It should be noted that implementing these strategies to manage ecosystem services can be challenging particularly in the African context faced with insecure land tenure systems (Koohafkan, 2002). This is further complicated by increasing land subdivisions into even smaller farm holdings within a region. From a policy point of view, it would be possible to prescribe management practices that come with incentives such as tax credits (e.g. agri-environment schemes model; European Commission, 2005) to encourage farmers to adopt these management strategies.

6.2.4.3 The importance of experimental rather than correlative approaches in field studies.

The approaches used to collect data in this thesis are more correlative than experimental. The main advantage of correlative approaches is that it presents an opportunity to examine how a target organism responds to various variables in its native range (Morin et al. 2009). This may be useful for closer examination of the dynamics of the target species and identify ecological factors limiting its population (e.g. Blossey, 1995; Sheppard, 2006). A drawback of using this approach is that it is difficult to control for confounding variables such as changes in insect community structure, variability in land use, micro climate, soils and aspect or even between year variations, which can have profound impacts on the data (e.g. Swirepik and Smyth, 2003; Denoth and Myers, 2005; Morin et al. 2009). On the other hand, experimental approaches allows for a more controlled environment within which to conduct investigations because the conditions are less prone to fluctuations. Experimental studies can also allow for prior designing and presents an advantage of deciding which conditions each candidate should be exposed to (e.g. den Boer-Hazewinkel, 1987). Good science combining both approaches, e.g. measuring the effectiveness of predators and parasitoids by manipulating prey items in controlled and field conditions or efficiency of a specific pollinator species on pigeonpea or field beans fruit set. This is useful in establishing the effectiveness of insect communities and more understanding the patterns of changes within local and landscape scales.

Chapter 7 - General conclusions and recommendations

7.1 Conclusions

7.1.1 Sustainable pigeonpea production

The great diversity and densities of insects known to be effective pollinators is likely to account for large proportions of crop production within an African context (Ricketts et al. 2008). Based on the results from the pigeonpea study, it is clear that the survival of pollinators and predators of pests are at risk from increasing insecticide application. In contrast to many developed countries, where pollinators are often actively conserved (e.g. flower rich field margins as part of agri-environment schemes) and there are clear policy frameworks that support these efforts, there is little attempt to manage or protect them in most developing countries.

For bee communities and sustainable delivery of pollination services, conservation management targeted at maintaining a high landscape complexity within pigeonpea landscapes, combined with a reduction in insecticide inputs, could enhance both taxonomic and functional diversity and potentially increase pigeonpea yields. The negative impacts of the number of insecticide application on social, polylectic and nest renting bees and pose particular threats to the existence of these functional groups within the small scale Kenyan farming systems and the contribution of these bees to the production of this crop. This is the first report of such an effect on pigeonpea crop.

Similarly, there is a huge potential for farmers to benefit from bio-control in Afrotropical agricultural systems. Based on my findings, foliar active predators could best be conserved by targeting a reduction of pesticide inputs in order to enhance natural pest control not only for pigeonpeas, but also for many crops commonly grown as intercrops or in succession after the main crop has been harvested, a widely practiced activity in small scale Kenyan farms.

7.1.2 Enhancing field bean yields

Based on the effects of distance from the field boundary on insect abundance and species composition in field bean agricultural systems, it is highly recommended that habitat requirement of specialist pollinators and natural enemies be put into consideration in order to enhance production. Perhaps less habitat disruption (e.g. tillage and pesticide use (Kromp, 1999)) may potentially benefit conservation of these pollinators and natural enemies. These measures may potentially enhance yields due reduced pest pressure and increased pollination effected by a high abundance and diversity of pollinating insects and natural enemies of crop pests.

7.2 Recommendations and future direction

1. In order to fully benefit from the potential of insect-mediated pollination, an assessment of requirements of the most economically important crops grown in Africa is essential. It is critical to establish the status of pollinators in terms of their habitat requirements and population dynamics, diversity within different landscape contexts, and agricultural zones within Kenya to identify priority areas to focus conservation attention.
2. There is need to investigate pollination efficiencies of different floral visiting species and how each of these species respond to landscape structure and local farm management practices in pigeonpea landscapes in order to target specific conservation measures to particular species. Many bee species, most of which are new to Science (See Appendix 3 and 4), were recorded but it is not clear how these species respond to changes in their environment. It is also not known whether extinctions are occurring as a result of poor land management practices, and whether the most efficient pollinators are the ones being lost.

3. It is essential to monitor the use and impacts of pesticides on pollinator and natural enemy community dynamics in order to develop pesticide application protocols and regimes that have minimum disruption of pollination and pest control service provision.

4. There is need for wider public understanding and awareness programs to educate local farmers and stakeholders on the importance of pollinators and natural enemies and the need to protect them.

5. For the field beans study, the influence of landscape context on the local spillover of ecosystem services to this crop should be investigated on contrasting farm management systems and longer distances from semi-natural/natural habitats than the one used in this study to establish the optimal combinations of distance and farm management practices that result in maximum yields.

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Appendices

Appendix 1: Site Description

Site	**General description**	**Semi-natural vegetation composition (within 100m radius)**	**Cropping system**	**Major crops (within 100m**
Athi Bridge 1A	site surrounded by patches of natural vegetation	none-farm is more than 50 hactares), but Acacia-commiphora vegetation formation	Monocropping	Pigeon pea
Athi Bridge 1B	site surrounded by mostly other farms, generally dorminated by arable weeds	Acacia, Balanites, arable weeds and herbs, shrubs and thickets	Monocropping	Pigeon pea
Athi Bridge 2A	Patches of thicket vegetation	Acacia, thickets, convolulacea, Ipomoeaceae, arable weeds,	Intercropping	Pigeon pea, green grams, sorghum
Athi Bridge 2B	Dense thickets at farm site dorminated by arable weeds	Acacia, arable weeds, grasses, hedgerow	Mixed cropping	Pigeon pea, maize, cowpeas
Athi Bridge 3A	Patches of dense thickets, pockets of open grassland	Dense shruby thickets, grassland	Mixed cropping	Pigeon pea, maize
Athi Bridge 3B	Open to closed bushland	Scrub vegetation, closed Acacia bushland, herbacious layer, arable weeds	Mixed cropping	Pigeon pea, maize, green grams,
Kibwezi 1A	Site in proximity to other farms, but surrounded by scattered patches of shruby	Scattered shrubs	Intercropping	Pigeon pea, maize, green grams,
Kibwezi 1B	Site is within Acacia-Commiphora vegetation matrix, borded by a river,	Acacia-Commiphora vegetation, arable weeds	Intercropping	Pigeon pea, maize, green grams, cotton
Kibwezi 2A	Predominantly surrounded by Acacia-commiphora thickets	Acacia-Commiphora vegetation, hedgerow,	Mixed cropping	Pigeon pea, cow peas, maize, beans
Kibwezi 2B	Site surrounded by Acacia thickets	Acacia thickets, open grassland overgrown by wild flowering plants, herbaceous	Mixed cropping	Pigeon pea, cow peas, chick pea,
Kibwezi 3A	Open vegetation, with Acacia bushes and scattered Baobab trees, grassland	Acacia thickets, open grassland, shrubs, Baobab trees, herbaceous layer	Intercropping	Pigeon pea, pumpkin, cow peas,
Kibwezi 3B	Farm surounded by alternating vegetation (scattered trees, grassland, shrub thickets)	Open grassland, baobab, scattered shrub thickets	Intercropping	Pigeon pea, pumpkin, cow peas,
Makindu 1A	Generally cropped area, dotted with Acacia woodland	Acacia woodland, open shrubs, lantana thickets, hedgerow	Intercropping	Pigeon peas, maize, cow peas,green
Makindu 1B	Site in proximity to other farms, but surrounded by scattered patches of shruby	Shrub thickets	Mixed cropping	Piegeon peas, maize, beans, cow
Makindu 2A	Dense thickets at farm site dorminated by arable weeds	Dense bushland, hedgerow,	Mixed cropping	Pigeon peas, maize, sweet potatoes
Makindu 2B	Farm generally surrounded by other farms	Scattered Acacia bush	Mixed cropping	Piegon peas, maize, green grams
Makindu 3A	Site primarily located among other farms and build area	Arable weeds, fence	Mixed cropping	Pigeon pea, maize, millet, cow peas
Makindu 3B	Site wihin urban environment	Open lawns/grasses, short herbs	Intercropping	Pigeon pea, maize

Appendix 2: Questionnaires.

PLEASE, DO NOT WRITE YOUR NAME IN THIS FORM.

SECTION A: BACKGROUND

1) How old are you?...years.
2) Gender:- Male [ ] Female [ ]
3) What is your District of birth?..
4) Village of residence now?..
5) Marital status? Single [ ] Married [ ] Divorced [ ] Widowed [ ]
6) How many people stay in your household?...

SECTION B: FARM MANAGEMENT

1) How big is your farm? *(please state in acres)*………………………………………..…………………………………………
2) What main crop do you grow most of the year?...
3) How important is this crop to you?

i. ……………………………………………………………………………………
ii. ……………………………………………………………………………………
iii. ……………………………………………………………………………………

4) Do you apply fertilizer or manure on your farm? *(Please tick the ones that apply)*

(i) Fertilizer only [ ] (ii) Manure only [ ] (iii) Both fertilizer and manure [ ]

5) a) (i) What type of fertilizers do you apply? *(Please list the brand names).*

……
…………………………………………………………………………………………..

(ii) How many times do you apply these fertilizers?

…………………………………………………………………………………………………..

b) If you use manure, what type do you use? *(Please tick the ones that apply)*

(i) Farmyard manure [ ] (ii) Compost manure [ ] (iii) Green manure [ ]

6) How many times do you apply these manures from planting to harvesting?

…………………………………………………………………………………………………

7) Do you grow pigeonpeas on your farm Yes [ ] No [ ]

a) (i) If yes, on what area of the farm? *(Please state in acres)*
…………………………………………………………………………………………………

(ii) How do you benefit from growing pigeonpeas?

……………………………........................……………………………………………………
……………………………………………...

b) If no, why don't you grow pigeonpeas?

....……...................

…………………………………………………………………………………………………

8) What other crops do you grow apart from pigeonpeas? *(Please list)*
…………………………………..…………………………………………………………………
…………………………………………………………………………………………….

9) a) Do you experience any pest problems on pigeonpeas? Yes [ ] No [ ]

b) If yes, at what growth stage are the pests most serious? *(please tick one)*

(i) pre-flowering [ ] (ii) budding [ ] (iii) flowering [ ] (iv) fruiting [ ]

c) Could you please list some of the common pests that you encounter on pigeonpeas?
………………………………………………………………………………………………………
……………………………………………………………………………………………

d) How do you control/manage these pests?

i. ………………………………………………………………………………………...
ii. ……………………………………………………………………………….……..
iii. …………………………….……………………………………………….……….
iv. ………………………………………………………………………………..…...

10) a) Do you use any chemical sprays to control pigeonpea pests? Yes [ ] No [ ]

b) If yes, how many times in its growth period do you spray your pigeonpeas with chemicals?

……………………… and at growth stages………………………………………………..

c) What are these chemical sprays? *(Please tick the ones that apply)*

(i) Herbicides [ ] (ii) Insecticides [ ] (iii) Fungicides [ ] (iv) Other *(please specify)*

…………………………………………………………………………………………

d) Please list the common brands (names) of pesticides that you use on pigeonpeas.

…...……………………………………………………………………………………..
....

11) a) Do you apply any other treatment on the pigeonpeas? Yes [ ] No [ ]

b) If yes, what kind of treatment?

…………………………………………………………………………….……………..

…………………………………………………………………………………………

12) What do you use in preparing your farm for pigeonpea planting? *(Please tick the ones that apply)*

(i) Tractors [ ] (ii) Ox-plough [ ] (iii) Hand hoe [ ] (iv)Other *(please specify)*……………………………….....................………...

13) How many operations do you conduct on your pigeonpea crop from planting to harvesting?... and at what stages
....……………………………………………………………………
…………………………………………………………...

14) How much yield do you harvest from your pigeonpeas? *(please state in kilograms)*
………………………………………………………………………………………………..

………………………………………………………………………………………….

Appendix 3: New bee Tribe, Genus and species from 2007 pigeonpea survey

Schwarzia emmae Eardley (Biastini: Halictidae)

Appendix 4: Bee species found within pigeonpea fields in 2009

Pigeonpea and wild flower visitors in Kibwezi, Kenya
April – June 2009

Megachilidae
1: *Pachyanthidium cordatum* (Smith)

Apidae
2: *Xylocopa imitator* (Smith)

Apidae
3: *Tetraloniella sp.*

Apidae
4: *Pachymelus conspicuus* (Smith)

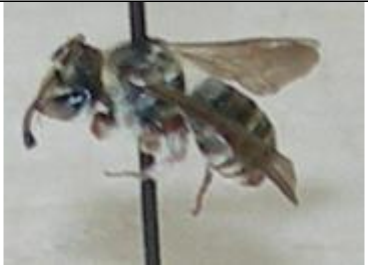
Megachilidae
5: *Megachile sp.*

Megachilidae
6: *Megachile sp.*

Megachilidae
7: *Megachile sp.*

Apidae
8: *Xylocopa senior* (Vachal)

Megachilidae
9: *Megachile sp.*

 **Apidae** **10: *Macrogalea candida* (Smith)**	 **Megachilidae** **11: *Euaspis abdominalis* (Fabricius)**	 **Apidae** **12: *Hypotrigona sp.***
 **Apidae** **13: *Hypotrigona sp.***	 **Apidae** **14: *Pachymelus conspicuus* (Smith)**	 **Halictidae** **15: *Lipotrichea sp.***
 **Apidae** **16: *Tetraloniella sp.***	 **Halictidae** **17: *Pseudapis sp.***	 **Megachilidae** **18 : *Megachile discolor* (Smith)**
 **Apidae** **19: *Thyreus sp.***	 **Apidae** **20: *Braunsapis sp.***	 **Apidae** **21: *Thyreus pictus* (Smith)**

 **Megachilidae** **22: *Megachile sp.***	 **Megachilidae** **23: *Megachile sp.***	 **Halictidae** **24: *Lasioglossum sp.***
 **Apidae** **25: *Amegilla caelestina* (Cockerell)**	 **Halictidae** **26: *Systropha sp.***	 **Megachilidae** **27: *Megachile sp.***
 **Megachilidae** **28: *Megachile sp.***	 **Apidae** **29: *Ceratina sp.***	 **Halictidae** **30: *Pseudapis sp.***
 **Apidae** **31: *Macrogalea candida* (Smith)**	 **Apidae** **32: *Ceratina sp.***	 **Apidae** **33: *Ceratina sp.***

 **Megachilidae** **34:** ***Megachile sp.***	 **Megachilidae** **35:** ***Megachile sp.***	 **Apidae** **36:** ***Xylocopa erythrina*** **(Gribodo)**
 **Megachilidae** **37:** ***Megachile sp.***	 **Megachilidae** **38*:*** ***Coelioxys sp.***	 **Apidae** **39:** ***Amegilla cymatilis*** **(Eardley)**
 **Apidae** **40:** ***Amegilla sp.*** **(Dours)**	 **41: Unknown**	 **Megachilidae** **42:** ***Megachile flavipennis*** **(Smith)**
 **Apidae** **43:** ***Ceratina sp.***	 **Megachilidae** **44:** ***Megachile sp.***	 **45: Unknown**

 **Megachilidae** **46:** ***Megachile sp.***	 **Megachilidae** **47:** ***Megachile sp.***	 **Megachilidae** **48:** ***Megachile sp.***
 **Apidae** **49:** ***Anthophora?***	 **Megachilidae** **50:** ***Megachile sp.***	 **Apidae** **51:** ***Apis mellifera*** **(Linnaeus)**
 **Apidae** **52:** ***Tetralonia sp?***	 **Apidae** **53:** ***Xylocopa sp.***	 **Apidae** **54:** ***Xylocopa inconstans*** **(Smith)**
 **Apidae** **55:** ***Xylocopa somalica*** **(Magretti)**	 **Apidae** **56:** ***Xylocopa flavicollis*** **(DeGeer)**	 **Apidae** **57:** ***Amegilla acaensis*** **(Fabricius)**

 **Apidae** **58:** ***Xylocopa sp.***	 **Apidae** **59:** ***Amegilla sp.***	 **Apidae** **60:** ***Amegilla sp.***
 **Apidae** **61:** ***Apis mellifera***	 **Apidae** **62:** ***Amegilla sp***	 **Apidae** **63:** ***Amegilla sp***
 **Megachilidae** **64:** ***Megachile sp.***	 **Megachilidae** **65:** ***Pachyanthidium sp***	 **Megachilidae** ***66: Pachyanthidium***
 **Apidae** **67:** ***Apis mellifera***	 **Megachilidae** **68:** ***Megachile sp.***	 **Apidae** **69:** ***Amegilla sp.***