The effect of urbanization and agriculture on predacious arthropod diversity in the Highveld grasslands

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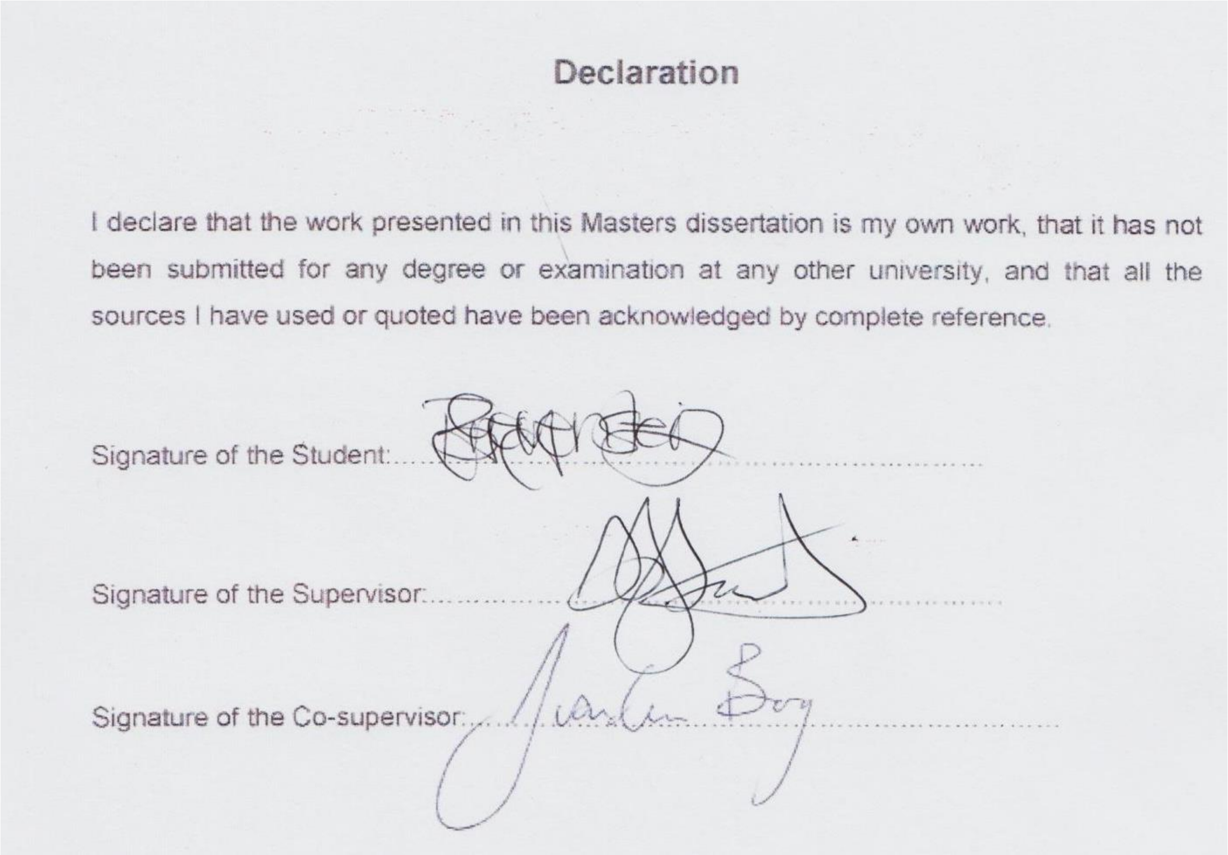

Declaration

I declare that the work presented in this Masters dissertation is my own work, that it has not been submitted for any degree or examination at any other university, and that all the sources I have used or quoted have been acknowledged by complete reference.

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Abstract

Biodiversity provides vital ecosystem services and more diverse ecosystems are known to be more stable and resilient in the face of disturbance. Predacious arthropods provide a valuable ecosystem service to control pest numbers. We hypothesised that anthropological activities, which result in land-use change and habitat fragmentation, deplete this species pool. However, current knowledge is lacking regarding the diversity and abundance of arthropod predators in crop fields within the agricultural and urban environments of South Africa. We studied the diversity, abundance and species assemblages of predaceous arthropods of the Chrysopidae, Mantodea, Araneae and Coccinellidiae at different intensity levels of disturbances within urban and agricultural sites in the grasslands of South Africa. Study sites included agricultural (maize field-field margin-untransformed grassland) and urban (ruderal-fragmented-untransformed grassland) gradients. The optimal sweep net sampling time was also determined within these agricultural ecosystems. Sweep nets were used to sample fields and field margins during the following daily times (07:00, 12:00, 15:00 and 17:00). Sampling was replicated four times. Shannon diversity index values did not differ between the different sweep net sampling times. Predacious arthropods were however more abundant within maize fields in the mornings (07:00) and adjacent fields around midday (12:00). The results indicated there were no statistical differences despite the higher abundance of predators at the previously mentioned times. The study done on the agro-ecosystem zones did however indicate that the field margins were the most diverse and abundant in terms of predacious arthropod diversity along a maize field-field margin-untransformed grassland gradient. Despite this our results indicated that urban areas had a similar species richness of arthropods as agro-ecosystems, but predator species were more abundant in urban areas. With an increase in distance away from disturbance, in both the agricultural and urban environments, an increase was noticed in diversity and abundance. The maize fields were the least diverse and abundant for these predacious arthropods. Surrounding agricultural landscapes could therefore play an important role in the maintenance of predacious arthropod diversity as well as future integrated pest management strategies. This study generated baseline data to monitor the effects of anthropological activities on predator diversity and abundance in urban- and agro-ecosystems.

Keywords: Biodiversity; predacious arthropods; diversity; agriculture; urbanization; ruderal, fragmented grasslands; maize; field margin; untransformed grasslands.

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Chapter 1: Introduction

1.1 Importance of biodiversity

Life exists as a result of the delicate balance of many systems that sustain life on earth. Ecosystems can be as big and complex as the biosphere but it can also exist at a smaller scale such as an agricultural field (Van As *et al.*, 2012). A key factor in all ecosystems is biodiversity, which is defined as the diversity of biological resources (Collins and Qualset, 1999).

Biodiversity is seen as a resource that has important ecological and evolutionary potential that is required for many vital ecosystem services which allow earth's delicate balance to remain in a state of equilibrium (Begon *et al.*, 2006). Biodiversity is also an essential part of the earth's natural capital that sustains life. High biodiversity is maintained by allowing species to inhabit different niches to avoid competition, which in turn decreases the potential threats of disturbances and invasions, and promotes sustainability (Neher and Barbercheck, 1999).

The different components of biodiversity are species, genetic, ecosystem and functional diversity (Miller and Spoolman, 2012). Scale is an important aspect of biodiversity. It is therefore important to define what is included. For example, the genetic diversity or community types can differ at different scales within an ecosystem and therefore biodiversity can function at large or small scales (Begon *et al.*, 2006). Biodiversity is critical not only for ecosystems services and maintaining the balance of life, but it is also important for its economic potential such as the ability to generate energy and food resources (Miller and Spoolman, 2012).

Examples of the benefits derived from increased biodiversity include increased productivity of grassland communities (Begon *et al.*, 2006), decomposition of organic matter by arthropods (Jonsson and Malmqvist, 2000) and reducing the loss of nutrients in wetland communities (Holmes *et al.*, 2003).

Despite the obvious benefits of biodiversity, economic perspectives has an influence on the conservation of biodiversity as this is where funds for conservation actions are derived from (Begon *et al.*, 2006). The effectiveness of conservation could be increased when cost-benefit analysis of biodiversity can be assigned. It is however, difficult to assign economic values to biodiversity and ecosystem services since these systems are very complex (Begon *et al.*, 2006).

1.1.2 Predators within the food web

Ecosystems are not discrete (Van As *et al*., 2012). They are often closely related to other ecosystems but, some aquatic and terrestrial ecosystems have more defined boundaries. Despite this, all ecosystems include individuals at different levels, each performing a particular task, which not only creates a food web but also establishes the delicate balance of life that is required. Different functional groups, i.e. primary producers, consumers and decomposers, perform these different tasks (Van As *et al.*, 2012).

Primary producers are the basis of the ecosystem and include all vegetation types. An area’s vegetation is determined by factors such as soil nutrients (which are partly determined by decomposers) and climate. Ultimately, this creates different biomes such as forests or grasslands (Miller and Spoolman, 2012). Consumers collectively include not only herbivores that feed on plants but also carnivores that feed on herbivores. All consumers therefore invoke a form of predation (Dugatkin, 2009). This is the situation for all of the taxonomic kingdoms of living organisms. Consumers are usually categorized as generalists or specialists, which refers to how specific they are with regards to their food resource. Some species have evolved in such a way that they only consume one or two other species, while generalists tend to consume a variety of resources, depending more on the abundance of the resource than the type of species itself (Begon *et al.*, 2006). Predation has an influence on basically every aspect of a community’s foraging ability, making it a very important part of a functioning ecosystem (Dugatkin, 2009). Decomposers are the last link in a simplified food web. They feed on dead biotic matter which includes plants and animals and recycles the nutrients back into the soil for primary producers to grow (Van As *et al.*, 2012).

Biodiversity within agro-ecosystems refers not only to heterogeneous plant communities, but also to insect species assemblages (Alteri, 1999). In an agro-ecosystem, most arthropods are consumers. Arthropods fulfil the roles of herbivores (termed as pests in the agricultural environment), but are also present as predators (e.g. spiders, lacewings, coccinellids and praying mantids) in all ecosystems. It has been shown that a decrease in biodiversity of agro-ecosystems, especially with the implementation of monocultures, increases the effects of pest species (Alteri, 1999). Stability in the insect communities of agro-ecosystems is depended not only on diverse plant communities, but also on the diversity of insect trophic structures (Alteri, 1999). This diversity is increased when surrounding natural vegetation is retained (Alteri, 1999; Deuli and Obrist, 2003). For example, Deuli and Obrist (2003) reported that 63% of all animal species depend on semi-natural habitats that surround agricultural activities in Switzerland's Limpach Valley.

Many different species of predacious arthropods exist and even though they seem to be vastly abundant, they each have a different role to play which contributes not only to the health of an ecosystem but also to its resilience (Calcagno *et al*., 2011). Natural enemies in agro-ecosystems seem to be affected by plant diversity as it affects their fitness in terms of climate and the variety of prey available (Lundgren *et al.*, 2009). Diverse plant communities create habitats for a greater variety of prey species which is beneficial to natural enemies, especially when pest abundance in crop fields is low (Lundren *et al.,* 2009). The variety in vegetation seems to also allow natural enemies the opportunity to select a niche of optimal suitability which, includes plant species with better associated nutrients, especially during oviposition. The selection of nutritious plant species at this time is said to decrease the likelihood of mortality of predatory larvae (Lundren *et al.,* 2009). This indicates the importance of habitat variability within monoculture crops systems.

A study of arthropod diversity in coffee agro-ecosystems suggests that although monoculture agro-ecosystems have a negative effect on arthropods, these systems are more diverse than expected and should be considered in conservation efforts (Perfecto *et al.*, 1997). Botha *et al*. (2015) also reported that maize and surrounding zones are rich in diversity with up to 117 arthropod families recorded. A total of 576 morpho-species were recorded in maize fields alone while, the surrounding landscape had 2054 morpho-species of arthropods. Truter *et al*. (2014) also reported high species richness (six species per 20 plants or 288 morpho-species) inside mono-cropped maize fields. It was also reported that beetle, ant and non-formicid Hymenoptera species richness on coffee bushes where almost the same as that of trees in a tropical forest (Perfecto *et al.*, 1997). Arthropods being both abundant and diverse in coffee agro-ecosystems provide an example of the resilience of species in intensely disturbed habitats.

1.1.3 Resilience

Resilience is described as the capacity of a community to recover and return to its original state after a displacement or disturbance has taken place but also to withstand a current disruptive pressure (Begon *et al.*, 2006). The more biologically diverse an ecosystem is the more stable and resilient it will be in the face of disturbance (Begon *et al*., 2006). Even though species may seem functionally redundant under a certain set of biophysical conditions, more species are required to sustain various functions at multiple times and places, especially in an ever changing environment (Calcagno *et al*., 2011). Gause's principle of competitive exclusion states that two species that have the same niche requirements cannot exist in the same ecosystem and confirms the need for multiple niches to be present in an ecosystem to avoid one species from dominating (Kormondy, 1976).

The way in which the world changes is mostly due to anthropological activities that cause disturbances which in turn result in declining biodiversity (Miller and Spoolman, 2012). These activities change the natural state of the environment in such a way that it influences ecosystems and their functions. An example of anthropological activities that influence the resilience of species is logging. Some arthropod species are more resilient to this kind of disturbance than others. Galling insects (Hemiptera and Diptera) have been found to have a high resilience to logging when pioneer plant species are preserved, which indicates the importance of conserving one species to ensure a future for another species (Malinga *et al.*, 2014). However, these activities can be so intense that the natural resilience of an ecosystem is pushed to a limit where the need for conservation arises. The strive towards sustainability of the natural environment necessitates an increase in resilience, which is achieved through the conservation of biodiversity (Deutsch *et al*., 2005).

1.2 Threats to biodiversity

Activities that threaten biodiversity include mining, habitat fragmentation, urbanization, agriculture, pollution, poaching, logging, overgrazing and climate change (Miller and Spoolman, 2012). It is suggested that future climate change will lead to biodiversity loss through the effects of increased temperature on species distribution and survival, leading to species extinctions and changes in distribution (Hellman *et al.*, 2013).

The implications of such changes will especially impact on arthropods that are comparatively more susceptible to changes in temperature than other organisms (Chidawanyika *et al.*, 2012; Wilson and Maclean, 2011). A recent study indicated that climate change is most likely to pose a greater threat to lepidopteran species than other arthropods and due to the lack of knowledge concerning the conservation status of insects, climate change poses an ever greater threat to insects (Wilson and Maclean, 2011). Agricultural pests could also be influenced by climate change resulting in species moving to more suitable environments which could lead to an increase in the numbers of pest species as well as changes in status of current pest species (Begon *et al.*, 2006).

Although it is thought that economic strive and the environment are two conflicting forces it is clear that a sustainable future can only be achieved if these two forces work together especially, with the impeding threat of climate change.

1.3 Agriculture as a disturbance

Agricultural land occupies approximately 75% of the land surface on earth and it therefore competes with the natural habitat of biodiversity (Young, 1999). An example of the effect of agriculture on arthropod communities was indicated by a study done on ant abundance. It indicated a decline in abundance as the intensification and management regime increased in coffee agro-ecosystems (Philpott *et al.*, 2006). This is not an uncommon phenomenon since a recent study also reported that biodiversity has been reduced significantly at global scale by land-use changes such as agriculture (Katayama *et al.*, 2015).

However, agriculture is important to humans for food, feed, fibre and fuel resources as an essential part of mankind's survival and economic strive (Connor *et al.*, 2011). It is known that biodiversity within agricultural ecosystems are important as it influences soil processes, local hydrological processes, recycling of nutrients, detoxification of noxious chemicals, pollination of crops and other vegetation, control of agricultural pests and dispersal of seeds (Altieri and Nicholls, 1999). This aids in the effectiveness of agricultural systems.

Although some studies have shown negative effects associated with agriculture, numerous other studies have illustrated that the effects of agricultural land-use on species richness/diversity can also be positive. According to studies done by Burel *et al*. (1998), the intensification of agriculture does not always lead to a decrease in species richness. The latter study indicated that species richness and diversity of carabid beetles and herbaceous plants did not change significantly along an agricultural intensification gradient while a decrease in species richness was only noted in two Diptera families (Chironomidae and Empididae). In contrast to this, another study has found that there is a significant decrease in arthropod diversity and abundance between the maize field and surrounding semi-natural habitats and that those arthropods are influenced by the diversity of plant species (Botha *et al.*, 2015). Furthermore, this study indicated that the Grasslands biome had higher arthropod species richness with regards the surrounding semi-natural vegetation of agro-ecosystems than that of the Savanna biome in South Africa (Botha *et al.*, 2015). This indicates the need for more studies to be done in agro-ecosystems especially in different plant biomes.

Beyond land use intensification, agriculture poses other potential threats to biodiversity. The most important of these are the use of pesticides and genetically modified crops (Connor *et al.*, 2011).

1.3.1 Pesticides

Since pest damage to crops reduce crop yield, pesticides need to be applied to suppress pest numbers (Collins and Qualset, 1999). There are concerns regarding the impact of pesticides on the environment and human health (Veres *et al.*, 2013). Beyond chemical compounds that are released into the environment, pesticides also kill beneficial arthropods which in some cases can lead to a rapid increase in pest species abundance, known as pest resurgence (Begon *et al.*, 2006).

New strategies of managing pest populations in the agricultural sector are required. The search for alternative pest management strategies is driven by the increasing resistance that pests express to chemical insecticides, the possible long term effects of genetically modified Bt crops as well as the growing demand for organically produced food (Chidawanyika *et al*., 2012). The use of natural enemies of pests is regarded as a sustainable solution since it promotes sustainability (Collins and Qualset, 1999).

The use of natural enemies as biological control agents can be seen as conservation biological control and is beneficial to both farmers and biodiversity. There has been great success in implementing conservation biological control especially in *Brassica* vegetables in China (Liu *et al.*, 2014). Aphid numbers were successfully suppressed by coccinellids (specifically *Harmonia axyridis*, which is an invasive species in South Africa) and stone flies, but spiders are still considered as important natural enemies of *Brassica* crops despite their inability to suppress aphids (Liu *et al.*, 2014). Recent studies indicated that the landscape complexity of the agricultural ecosystem is the determining factor of the success of conservation biological control (Jonsson *et al.*, 2015).

1.3.2 Genetically modified crops

South Africa is the 9th largest producer of genetically modified crops in the world with nearly 2.14 million hectares of genetically modified maize planted annually (James, 2014). These maize hybrids express delta-endotoxins of the entomopathogenic bacterium, *Bacillus thuringiensis* (Bt), throughout the maize plant (Pilcher *et al*., 1997; Lundren and Wiedenmann, 2002; Lundren and Wiedenmann, 2004). There is a concern over the unknown adverse effect of genetically modified crops on natural enemies and food webs in agricultural fields (Romeis and Meissle, 2011; Pilcher *et al.*, 1997; Lundren and Wiedenmann, 2002; Lundren and Wiedenmann, 2004).

However, Cry1Ab-expressing Bt maize shows no significant threats to Coccinellidae feeding on aphids that are known to contain traces of these insecticidal Cry proteins (Romeis *et al*., 2012). This conclusion is supported by field studies conducted which, showed that Cry1Ab-expressing Bt maize does not cause harm to Coccinellidae species under field conditions (Obrist *et al.*, 2006). A recent study did find that Bt toxins can be transferred to predators such as *Chrysoperla* spp. (Neuroptera: Chrysopidae) in laboratory studies, yet field surveys showed little to no toxin levels in species such as *Hippodamia* sp. (Coleoptera: Coccinellidae) and *Nabis* sp. (Hemiptera: Nabidae) (Obrist *et al.*, 2006).

1.3.3 Biological control

Biological control implies the use of beneficial species (natural enemies) at different trophic levels to suppress pest species in agricultural fields (Begon *et al.*, 2006). Classical biological control implies importation of natural enemies from different geographic regions to be used beyond their normal dispersal zones, thus making them introduced species. Conservation bio-control, contrasts this classical biological control by manipulating natural enemies to increase their presence and also to sustain a population of native generalist enemies thus avoiding introduction of possibly invasive species into a new geographic region (Begon *et al.*, 2006).

However, the rise in resistance to pesticides and GM crops increases the importance of natural enemies as biological control agents, especially as the mechanisms of resistance are not yet understood (Heckel, 2012). In the method where classic biological control programmes are implemented, predators can suppress pest populations enough to reduce economic damage. This requires better understanding of the dynamics of predators and their prey to develop effective pest management strategies (Jacometti *et al*., 2010). In Australia, lacewings are used in such biological control systems since they are generalists and thus contribute to the control of pest populations (New, 2002). It is however, cautioned that one predator species on its own is not sufficient to suppress pests below economically important infestation levels (New, 2002). A variety of predators, especially native species, have a greater chance to reduce pest populations as they all have different physiological temperature and moisture requirements (Chidawanyika *et al*., 2012).

1.4 Urbanization as a disturbance

It is estimated that half of the world's population live in urban areas (Miller and Spoolman, 2012). Urban areas expand due to two factors, i.e. immigration and natural increase which implies a higher birth rate than mortality rate. The immigration towards cities can either be ascribed to factors areas such as jobs, food, housing and education that attract people but, people can also be pushed to move to urban areas by circumstances such as poverty, war, and decline in agricultural activities (Miller & Spoolman, 2012).

Urbanization can be considered a threat to biodiversity. Urbanization is a process that includes large scale disturbance that destroys natural habitat by replacing these habitats with large buildings, paved roads and many other impermeable surfaces (Gardiner *et al.*, 2013). Urbanization is usually accompanied by other effects on the natural environment which is beyond the physical manifestations of buildings and roads. These effects can be changes in for example, remnants of native plant communities which are turned into gardens that consist of non-native species that results from preference instead of necessity (Gardiner *et al.*, 2013), a rise in pollution levels from motor vehicles, industries and sewerage. Another effect is known as the "urban heat island" which is caused by urban areas consisting of many surface types that retain heat and so increase the ambient temperature within cities (Baker *et al.*, 2002; Miller and Spoolman, 2012).

Urban sprawl, which is the expansion of cities or urban areas at the edges through low-density developments, is the cause of decreasing rural or semi-natural landscapes (Miller and Spoolman, 2012). Environmental impacts that are associated with urban sprawl include loss of agricultural lands, fragmentation of natural areas, increased pressure on resources such as water and infrastructure, and higher levels of pollution of both water and air (Sarzynski, 2012).

In some cases urbanization does include fragments and isolated remnants of what were once presumably pristine natural habitats (Gardiner *et al.*, 2013; Cilliers *et al.*, 1999). These areas are vital for conservation and the sustainability of biodiversity throughout the ever expanding urban matrix (Gardiner *et al.*, 2013; Miller and Spoolman, 2012; Blair and Launer, 1997; Bolger *et al.*, 2000; Burkman and Gardiner, 2014; McKinney, 2002).

1.4.1 Island biogeography theory

The island biogeography theory states that species diversity of isolated areas such as islands is affected by the size of the island, locality, as well as evolutionary changes in the long-term that could occur (Miller & Spoolman, 2012). The biodiversity of islands, according to this theory, is depended on two factors. These include the rate at which new species immigrate to the island which is influenced by its locality, and the mortality rate of species on the island. The latter is influenced by the biophysical aspects of the island as well as its size (Begon *et al.*, 2006; Miller & Spoolman, 2012).

Urban green spaces can be seen as islands in an ocean of anthropogenic structures. A study of tenebrionid beetles in Italy indicated that the species richness or diversity of these beetles can be explained by the biogeography island theory which indicates that the island size, or in this case patch size and isolation invokes a higher mortality than immigration rate (Fattorini, 2014). The surrounding forest area did not supply the urban environment with sufficient immigrants, perhaps due to the patch isolation and limited connectivity of these patches to the forests. The authors also suggest that the behaviour of these beetles could have also influenced their results as these beetles consist of urban avoiders and adapters. (Fattorini, 2014). A recent study found that even small habitats (despite their limited habitat availability) such as street medians have a diverse arthropod community, which mostly consists of ants (Youngsteadt *et al.*, 2015). These ants are able to sustain ecosystem functions such as refuse consumption in urban patches and it was estimated that these arthropods remove 4 - 6.5 kg of refuse per side walk per annum.

Philpott *et al.* (2014) stated that predatory arthropods were more abundant in vacant lots than in forest samples whilst, the species richness of spiders was higher than any other predacious arthropod in the urban areas. However, a strong correlation was found between species richness of spiders and the habitat type of vacant lots (Philpott *et al.*, 2014). This indicates that there were more factors influencing arthropods than could be explained by the island biogeography theory and that different arthropods react differently to disturbed environments (Gardiner *et al.*, 2013; Philpott *et al.*, 2014).

1.5 Conservation

Conservation is defined as the act of preserving the natural environment or keeping it from harm, decay or loss for the future (Hawkins, 1983). However in reality it is stated that, "conservation as a whole is losing the war not because of any lack of commitment or focus but because of the sheer scale, growth, and complexity of the problems it faces" (Balmford and Cowling, 2005). One of the biggest problems regarding conservation is that only the interest exhibited by the biological sciences is not enough, conservation includes multiple disciplines and is coherently about people and their perspectives as well as choices (Balmford and Cowling, 2005; Poiani *et al.*, 2000). Despite the problems facing conservation, a need for more knowledge with regards to ecosystem function (Balmford and Cowling, 2005) and original methods of incorporating conservation into everyday situations are required. This will ultimately aid in shifting the world we know into a more sustainable direction (Miller and Spoolman, 2012; Balmford and Cowling, 2005).

1.5.1 Conservation in agriculture

The presence of predatory arthropods in agricultural landscapes increases the effectiveness of biological control of pest populations and should therefore be the focus of conservation strategies (Henn *et al.*, 2009).

Field margins in Europe have been noted to increase the biodiversity in agro-ecosystems as well as to act as refuge for beneficial arthropods such as natural enemies of pests (Marshall, 2002). Beyond the field margin, the habitat surrounding maize fields serve as reservoirs for biodiversity (Kanya *et al.,* 2004). Thus, non-farmed areas in the surrounding crop landscape plays a vital role as habitats for various arthropods due to the agricultural ecosystems being composed of intensely cultivated areas together with patches of natural or semi-natural areas that host both beneficial and pest insects (Blackshaw and Vernon, 2006).

The percentage of land that is temporarily covered by perennial grass has been found to have a significant effect on the species composition of butterflies while, the composition of different plant species is the most influential factor for carabid species (Weibull and Ostman, 2003). This shows that different species are influenced by different factors as well as different habitat management techniques (Landis *et al.*, 2000). To ensure effective conservation actions these factors and management practices should be implemented after further investigation.

Agricultural land comprises of a large portion of South Africa's landscape in comparison to its conservation areas. Non-cultivated land and conservations areas should be actively preserved for the benefits of natural enemies and the abundance of its unknown biodiversity (Wessels *et al*., 2003).

1.5.2 Conservation in urban areas

Ecosystems within urban areas have the capacity to be resilient enough to permit species to occupy and survive in urban areas (Gardiner *et al* 2013). This capacity can to a certain extent lead to the restoration of some of these disrupted ecosystems within urban areas. Urban green spaces can aid in urban ecosystems regaining functions as well as services which could ultimately contribute to the increase of species diversity by acting as a refuge in a disturbed environment (Gardiner *et al.*, 2013).

Green spaces, also known as vacant lands, are able to support diverse and rare species of arthropods, making these green spaces an important aspect of conservation, especially in the case of arthropod preservation (Gradiner *et al.*, 2013; Bolger *et al.*, 2000; Gibbs. and Hochuli, 2002; Blair and Launer, 1997). Arthropods are good indicators of how biodiversity is affected by different types of land use changes, as they have quick reproductive cycles, are present in almost all biomes on earth and are thus worthy of conservation (Chidawanyika *et al*., 2012). Due to the biodiversity present in urban green spaces, these areas should be conserved and incorporated into future conservation planning (Gardiner *et al.*, 2013). Urban biodiversity is not readily incorporated into conservation planning as the fragmentation and in some cases isolated existence, of these green spaces is not sustainable enough for conservation to take place (Miller and Spoolman, 2012). However, it was found that urban green spaces that formed part of a more diverse system of vacant patches had an increased diversity of arthropods such as pollinators (Sattler *et al.*, 2011). This higher diversity is ascribed to the abilities of arthropods to re-colonize an area that is within close proximity to a source population. Urban green areas are suspected to act as sink populations with the surrounding agricultural fields as the source for these species (Sattler *et al.*, 2011).

It is therefore suggested that using urban green spaces as part of conservation efforts could effectively aid in the quality of the ecosystems found in urban vacant lands through the preservation of biodiversity (Burkman and Gardiner, 2014). Conservation efforts of this nature must however be vigilant as these urban green spaces act as islands which have as stated by the island theory, certain prerequisites for individuals to continue surviving in this environment.

1.5.2.1 Corridors and linkages

Urban green belts and corridors can be seen as a passage way for animals to pass through urban areas "untouched" by the anthropological disturbances of cities (Begon *et al.*, 2006). This promotes migration and decreases the probability of extinction by isolation of small populations.

Urban planning can include the already existing urban landscape belts such as rivers to promote corridors (Xu *et al.*, 2015; Austin, 2012). Fragmented urban green spaces, when linked to each other, can create a pathway for animals. These pathways are most effective when they are composed of remnants of original ecological environments and rehabilitated urban fragments. When this is done these pathways are less likely to compromise the original landscape composition and have a higher likelihood of being ecologically sustainable and a viable method for animals to manoeuvre through urban areas (Xu *et al.*, 2015). A study on garden shrews showed that ecological corridors in urban areas are effective for species with low dispersal abilities and should be included in future urban planning (Vergnes *et al.*, 2013). In accordance, urban green spaces might aid in maintaining urban corridors which might increase the connectivity of city centre's (a possible pool population) to the surrounding natural or agricultural areas which effectively might be the source population for these arthropods (Gardiner *et al.*, 2013).

A unique approach to creating urban corridors stems from the idea of using green roofs (Millet, 2004). A study indicated that green roofs support generalist arthropod species however research is needed to evaluate the biodiversity associated with green roofs to improve these potential habitats to include rare species as well to increase their conservation status (Williams *et al.*, 2014).

1.5.2.2 Sink and source theory

The sink-source theory stipulates that animals in a habitat of good quality and resource abundance have a higher reproductive rate than those within areas of less preferable habitat (Begon *et al*., 2006). However, due to the size of the habitat or overpopulation, some individuals might explore new habitats which do not have similar abundances of resources and habitats types. Therefore, the first habitat can become the source (individual migratory source) for the second habitat in which the reproductive rate of the animals do not exceed their mortality rate (Begon *et al*., 2006; Hansen, 2011; Loehle, 2012; Rosenheim, 2001). However, it is noteworthy that other factors beyond habitat quality influences the movement of animals between sink and source habitats (Rosenheim, 2001). Lacewings, for example, are depended on large numbers of aphids as food, while they are themselves also subject to predation in their larval form. It was noted that lacewing adult immigration sustained smaller populations in cotton fields and that predators of lacewing larvae influenced the position where adult lacewings lay their eggs (Rosenheim, 2001).

Nevertheless, when comparing the diversity of arthropods in natural, urban and agricultural areas, it was found that community composition was quite similar (Sattler *et al*., 2011). The latter study suggested that agricultural sites might act as source populations for urban arthropod communities (Sattler *et al*., 2011). Agriculture is therefore not only important for food production but also for sustaining the biodiversity of arthropods. Sattler *et al*. (2011) also suggested that some species might be vanishing from intensified agricultural areas and that their new survival strategy could be to find new habitats in urban areas.

The aim of this study was to determine the effect of different disturbance intensities, sampling times within different environments on the diversity and abundance of predacious arthropod. The specific objectives and hypotheses will be discussed in each of the chapters that follow.

1.7 Selecting representative arthropod groups

Predators were selected for this study since these species proved representative of the diversity that exists in the predatory trophic level of agro-ecosystems (Veres *et al*., 2013). Four groups were selected, namely spiders, coccinellids, lacewings and praying mantids. The selected predators for this study included the surface dwelling and fast moving spiders, the sit-and-wait for prey praying mantids and the fly-and-seek coccinellids and lacewings (Holwell *et al*., 2007; Dippenaar-Schoeman and Van den Berg, 2010). These chosen groups provide a well-rounded picture of the predator functional group diversity within both the agro- and urban ecosystems (Kremen *et al*., 1993).

Predator groups have an important ecological “status”. Spiders are good bio-indicators of ecological disturbances (Haddad *et al*., 2013), while lacewings are described as good bio-indicators of ecosystem health. The coccinellids in agro-ecosystems play an important role as resource biota that contributes to productivity through pollination, decomposition and biological control (Altieri, 1999). Although praying mantids are less abundant, including them in a diversity study provides a more rounded and broader overview which includes a variety of other ecological and behavioural factors which could have previously been overlooked. A scarcer species might fill niche gaps which abundant species may not be able to fill and utilise.

1.7.1 Lacewings (Insecta: Neuroptera: Chrysopidae)

Lacewings (especially the green lacewings, *Chrysoperla* spp.) are regularly used as bio-indicators of ecosystem health, making them a good “control” group to include into any biodiversity study (Deutsch *et al*., 2005). Lacewings usually prey on arthropods such as aphids, coccids, caterpillars and mites amongst others (Thierry *et al*., 2008). Larvae of the common green lacewing is especially considered to be a generalist predator and considered as an important component in agro-ecosystems (Kazemi and Mehrnejad, 2011).

Different species of lacewings have different overwintering strategies. Some spend winter as motionless instars in a cocoon while others spend the winter as inactive adults (Thierry *et al*., 2008). The overwintering strategy is important since it determines the abundance of lacewings in spring and thus the abundance of predators that will consume aphid colonies that occur on young plant growth early in the growing season. This difference could influence not only the species richness and abundance of lacewings that are captured during surveys but, also their role in biological control systems (Thierry *et al*., 2008). Some chrysopid species, especially of the genus *Chrysoperla*, have been successfully mass-reared and used in biological control of agricultural pests (Kazemi and Mehrnejad, 2011).

1.7.2 Praying mantids (Insecta: Mantodea)

There are about 1 500 species of Mantodea that have been described worldwide (Perez, 2005). Praying mantids are generalist predators and include some species that are cannibalistic. Their ecological functions in ecosystems are thus complicated by the fact that some species can alternate between these different food resource utilisation strategies or use all of the resources at once. Their eggs hatch in late spring and nymphs largely occupy the lower grassy vegetation layer (Holwell *et al*., 2007). It has been found that Mantids affect aphid abundance but their net effect on aphid biomass was reported to be larger (up to 45% reduction) (Holwell *et al*., 2007). There is still little known about Mantid abundance in agro-ecosystems as well as urban environments.

The height of vegetation has an influence on different species of mantids as some prefer taller vegetation where they sit and wait for prey, while others prefer lower vegetation (Ilyse *et al*., 1983). This behaviour and preference for certain vegetation types could therefore influence the abundance of mantids in an agro-ecosystem. Praying mantids can therefore have a positive influence on pest supression in agricultural ecosystems but this is dependent on the level of interspecific competition within the ecosystem itself (Hurd, 2009). Mantid abundance is influenced not only by the abundance of food resources but also by rainfall and temperature (Hurd, 2009). Their diversity and abundance in urban environments could therefore be markedly influenced by the higher temperatures that are usually observed in urban environments. The dispersal habits of mantids indicated that they are patch orientated instead of evenly dispersed (Holwell *et al*., 2007).

1.7.3 Spiders (Arachnida: Araneae)

The spider guilds can be divided into web dwellers and wanders (Dippenaar-Schoeman, 2014). The majority of spiders live on the ground, however, some species live in burrows, under stones and others can "comfortably live underwater with a bubble of air surrounding their bodies" (Cloudsley-Thompson, 1958). Web builders, of which there are only 28 families in South Africa, construct a number of different types of webs such as: funnel, mesh, gumfoot, orb, retreat, sheet and space webs (Dippenaar-Schoeman, 2014). Some spider species are active during the day and dependent on their sight while, others are more sensitive to touch and therefore hunt at night (Cloudsley-Thompson, 1958). This difference within taxa is a strategy that reduces inter-specific competition (Van As *et al*., 2012).

Spiders are valuable in nature because they are able to devour a large number of prey in a relatively short amount of time. This large consumption ability is due to their distensible abdomens, the fact that they can slow down their metabolic rates and "overcome" starvation in times when prey is not abundant and since they can feed on adults, eggs, nymphs and larvae of their prey (Dippenaar-Schoeman and Van den Berg, 2010).

Some spider species are specialists and only feed on ants or termites but most are generalists. Spiders fulfil a critical role in ecosystems as pest control agents (Haddad *et al*., 2013) and are considered to be good bio-indicators for measuring ecological disturbance and pollution effects (Haddad *et al*., 2013).

1.7.4 Ladybirds (Insecta: Coleoptera: Coccinellidae)

Coccinellids are ecologically and morphologically diverse and consist of over 6000 species that differ in size from 0.8 mm to 18 mm (Seago *et al*., 2011). Coccinellids play an important role as biocontrol agents in urban gardens and agricultural fields (Seago *et al*., 2011). When coccinellids are threatened by other natural enemies, they exude noxious alkaloid-based compounds (Seago *et al*., 2011).

Due to the predatory behaviour of coccinellids towards mites and hemipterous insects, 90% of coccinellid species are beneficial (Iperti, 1999). Only 10% are phytophagous on crops or exhibit fungivorous behaviour (Iperti, 1999). The use of coccinellids as biological control agents is essential to control aphid infestations on crops (Grez *et al*., 2010). Coccinellids provide key ecosystem services in all agricultural landscapes by suppressing pest numbers (Isaacs *et al*., 2009). It is complex to measure the value of these services but it has been noted that coccinellids have a substantial financial value for crop pollination and pest suppression (Losey and Vauhan, 2006). It is therefore of great importance to maintain Coccinellidae communities for the ecosystem services they provide such as biological control (Landis *et al*., 2012).

1.8 Reference

ALTIERI, M.A. & NICHOLLS, C.I. 1999. Biodiversity, ecosystem function and insect pest management in agricultural systems. (In Collins, W.W. & Qualset, C.O., eds. *Biodiversity in agroecosystems.* Boca Raton: CRC Press. p. 69-82).

ALTIERI, M.A. 1999. The ecological role of biodiversity in agroecosystems. *Agriculture, Ecosystems and Environment*, 74:19-31.

AUSTIN, G.D. 2012. Multi-functional ecological corridors in urban development. *Spaces and flows: An international conference on urban and extra urban studies*, 2(3):211-230.

BAKER, L.A., BRAZEL, A.J., SELOVER, N., MARTIN, C., MCINTYRE, N., STEINER, F.R., NELSON, A. & MUSACCHIO, L. 2002. Urbanization and warming of Phoenix (Arizona, USA): Impacts, feedbacks and mitigation. *Urban Ecosystems*, 6(3):183-203.

BALMFORD, A. & COWLING, R.M. 2005. Fusion or failure? The future of conservation biology. *Conservation Biology*, 20(3):692-695.

BEGON, M., TOWNSEND, C.R. & HARPER, J.L. 2006. Ecology: From individuals to ecosystems. Oxford: Blackwell Publishing Ltd.

BLACKSHAW, R.P. & VERNON, R.S. 2006. Spatiotemporal stability of two beetle populations in non-farmed habitats in an agricultural landscape. *Journal of Applied Ecology*, 43:680-689.

BLAIR, R.B. & LAUNER, A.E. 1997. Butterfly diversity and human land use: Species assemblages along an urban gradient. *Biological Conservation*, 80:113-125.

BOLGER, D.T., SUAREZ, A.V., CROOKS, K.R., MORRISON, S.A. & CASE, T.J. 2000. Arthropods in urban habitat fragments in southern California: Area, age and edge effect. *Ecological Applications*, 10(4):1230-1248.

BOTHA, M., SIEBERT, S.J., VAN DEN BERG, J., MALIBA, B.G. & ELLIS, S.M. 2015. Plant and arthropod diversity patterns of maize agro-ecosystems in two grassy biomes of South Africa. *Biodiversity and Conservation.* 24:1791-1824

BUREL, F., BAUDRY, J., BUTET, A., CLERGEAU, P., DELLETTRE, Y., LE COEUER, D., DUBS, F., MORVAN, N., PAILLAT, G., PETIT, S., TENIAL, C., BRUNEL, E. & LEFEUVRE, J. 1998. Comparative biodiversity along a gradient of agricultural landscapes. *Acta Oecologica*, 19:47-60.

BURKMAN, C.E. & GARDINER, M.M. 2014. Urban greenspace composition and landscape context influence natural enemy community composition and function. *Biological Control*, 75:58-67.

CALCAGNO, V., HECTOR, A., CONNOLLY, J., HARPOLE, S.W., REICH, P.B., SCHERER-LORENZEN, M., LOREAU, M., ZAVALETA, E.S., WILSEY, B.J., WEIGELT, A., VAN RUIJVEN, J., TILMAN, D., SCHMID, B. & ISBELL, F. 2011. High plant diversity is needed to maintain ecosystem services. *Nature*, 477:199-204.

CHIDAWANYIKA, F., MUDAVANHU, P. & NYAMUKONDIWA, C. 2012. Biologically based methods for pest management in agriculture under changing climates: challenges and future directions. *Insects*, 3:1171-1189.

CILLIERS, S.S., VAN WYK, E. & BREDENKAMP, G.J. 1999. Urban nature conservation: vegetation of natural areas in the Potchefstroom municipal area, North West Province, South Africa. *Koedoe*, 42(1):1-30.

CLOUDSLEY-THOMPSON, J.L. 1958. Spiders, scorpions, centipedes and mites: the ecology and natural history of woodlice (*Myriapods* and *Arachnids*). London: Pergamon Press.

COLLINS, W.W. & QUALSET, C.O. 1999. Biodiversity in agroecosystems. London: CRC Press LLC.

CONNOR, D.J., LOOMIS, R.S & CASSAMAN, K.G. 2011. Crop Ecology: Productivity and management in agricultural systems. Cambridge: Cambridge University Press.

DEULI, P. & OBRIST, M.K. 2003. Regional biodiversity in an agricultural landscape:the contribution of seminatural habitat islands. *Basic Applied Ecology*, 4:12-138.

DEUTSCH, B., PAULIAN, M., THIERRY, D. & CANARD, M. 2005. Quantifying biodiversity in ecosystems with green lacewing assemblages. *Agro Sustainable Development.*, 25:337-343.

DIPPENAAR-SCHOEMAN, A. 2014. Field guide to the spiders of South Africa. Pretoria: LAPA Publishers.

DIPPENAAR-SCHOEMAN, A. & VAN DEN BERG, A. 2010. Spiders of the Kalahari. Pretoria: Agricultural Research Council.

DUGATKIN, L.A. 2009. Principles of animal behaviour. New York: W.W Norton & Company.

FATTORINI, S. 2014. Island biogeography of urban insects: Tenebrionid beetles from Rome tell a different story. *Jornal of Insect Conservation*, 18:729-735.

GARDINER, M.M., BURKMAN, C.E. & PRAJZNER, S.P. 2013. The value of urban vacant land to support arthropod biodiversity and ecosystem services. *Environmental Entomology*, 42(6):1123-1136.

GIBBS., H. & HOCHULI, D.F. 2002. Habitat fragmentation in an urban environment: large and small fragments support different arthropod assemblages. *Biological Conservation*, 106:91-100.

GREZ, A.A., TORRES, C., ZAVIEZO, T., LAVANDERO, B. & RAMÍREZ, M. 2010. Migration of coccinellids to alfalfa fields with varying adjacent vegetation in central Chile. *Ciencia e Investigación Agraria*, 37(2):111-121.

HADDAD, C.R., DIPPENAAR-SCHOEMAN, A.S., FOORD, S.H., LOTZ, L.N. & LYLE, R. 2013. The faunistic diversity of spiders (Arachnida:Araneae) of the South African grassland Biome. *Transactions of the Royal Society of South Africa*, 68(2):97-122.

HANSEN, A.J. 2011. Contribution of source-sink theory to protected area science. Sources, sinks, and sustainability across landscapes. Cambridge: Cambridge University Press.

HAWKINS, J.M. 1983. The Oxford paperback dictionary. Oxford: Oxford University Press.

HECKEL, D.G. 2012. Learning the ABCs of Bt: ABC transporters and insect resistance to *Bacillus thuringiensis* provide clues to a crucial step in toxin mode of action. *Pesticide Biochemistry and Physiology*, 104:103-110.

HELLMAN, J.J., CORNEJO, C.R., SHUFELDT, G. & JAVELINE, D. 2013. Expert opinion on ciimate ciiange and threats to biodiversity. *BioScience*, 63(8):665-674.

HENN, T., WEINZIERL, R. & KOEHLER, P.G. 2009. Beneficial insects and mites. Cooperative Extension Service Document ENY-276, Institute of Food and Agricultural Sciences, University. of Florida, Gainesville.

HOLMES, D.R.Z, WHITE, W.E., PEACOCK, A.D. & TILMAN, D. 2003. Plant diversity, soil microbial communities and ecosystem function: are there any links? *Ecology*, 84:2042-2050.

HOLWELL, G.I., BARRY, K.L. & HERBERSTEIN, M.E. 2007. Mate location, antennal morphology, and ecology in two praying mantids (Insecta:Mantodea). *Biological Journal of the Linnean Society*, 91:307-313.

HURD, L.E. 2009. Encyclopedia of Insects Second edition. London: Elsevier.

INTERGOVERMENTAL PANEL ON CLIMATE CHANGE *(*IPCC*)*. 2007. *Synthesis Report.* p1-52

ILYSE, H., RATHET, L. & HURD, L.E. 1983. Ecological Relationships of Three Co-occurring Mantids, *Tenodera sinensis* (Saussure), *T.angustipennis* (Saussure), and *Mantis religiosa* (Linnaeus). *American Midland Naturalist*, 110(2):240-248.

IPERTI, G. 1999. Biodiversity of predaceous coccinellidae in relation to bioindication and economic importance. *Agriculture, Ecosystems and Environment*, 74(2):323-342.

ISAACS, R., TEULL, J., FIEDLER, A., GARDINER, M.M. & LANDIS, D. 2009. Maximizing arthropod-mediated ecosystem services in agricultural landscapes: The role of native plants. *Frontiers in Ecology and the Environment*, 7:196-203.

JACOMETTI, M., JORGENSEN, N. & WRATTEN, S. 2010. Enhancing biological control by an omnivorous lacewing: Floral resources reduce aphid numbers at low aphid densities. *Biological Control*, 55:159-165.

JAMES, C. 2014. *Global status of commercialized biotech/GM crops: 2014. ISAAA Brief No. 49.* Ithaca: ISAAA.

JONSSON, M., STRAUB, C.S., DIDHAM, R.K., BUCKLEY, H.L., CASE, B.S., HALE, R.J., GRATTON, C. & WRATTEN, S.D. 2015. Experimental evidence that the effectiveness of conservation biological control depends on landscape complexity. *Journal of Applied Ecology*, 52:1274-1282.

JONSSON, M. & MALMQVIST, B. 2000. Ecosystem process rate increases with animal species richness: evidence from leaf-eating, aquatic insects. *Oikos*, 89:519-523.

KANYA, J.I., NGI-SONG, A.J., SÉTAMOU, M.F., OVERHOLT, W. & OCHORA, J. 2004. Diversity of alternative hosts of maize stemborers in Trans-Nzoia district of Kenya. *Environmental Biosafety Research*, 3:159-168.

KATAYAMA, N., OSAWA, T., AMANO, T. & KUSUMOTO, Y. 2015. Are both agricultural intensification and farmland abandonment threats to biodiversity? A test with bird communities in paddy-dominated landscapes. *Agriculture, Ecosystems and Environment*, 214:21-30.

KAZEMI, F. & MEHRNEJAD, M.R. 2011. Seasonal occurrence and biological parameters of the common green lacewing predators of the common *pistachio psylla, Agono scenapistaciae* (Hemiptera:Psylloidea). *European Journal of Entomology,* 108:63-70.

KORMONDY, E.J. 1976. Concepts of ecology. New Jersey: Prentice-Hall.

KREMEN, C., COLWELL, R.K., ERWIN, T.L., MURPHY, D.D., NOSS, R.F. & SANJAYAN, M.A. 1993. Terrestrial arthropod assemblages: their use in conservation planning. *Conservation Biology*, 7(4):796-806.

LANDIS, D.A., GRADINER, M.M. & TOMPKIND, J. 2012. Using native plant species to diversify agriculture. UK: Wiley-Blackwell.

LANDIS, D.A., WRATTEN, S.D. & GURR, G.M. 2000. Habitat management to conserve natural enemies of arthropod pests in agriculture. *Annual Review of Entomology*, 45:175-201.

LOEHLE, C. 2012. A conditional choice model of habitat selection explains the source-sink paradox. *Ecological Modelling*, 235-236:59-66.

LOSEY, J.E. & VAUGHAN, M. 2006. The economic value of ecological services provided by insects. *BioScience*, 56:311-323.

LIU, Y., SHI, Z., ZALUCKI, M.P. & LIU, S. 2014. Conservation biological control and IPM practices in *Brassica* vegetable crops in China. *Biological Control*, 68:37-46.

LUNDGREN, J.G. & WIEDENMANN, R.N. 2002. Coleoptera-specific Cry3Bb toxin from transgenetic corn pollen does not affect the fitness of a non-target species, *Coleomegilla maculate* (Coleoptera: Coccinellidae). *Environmental Entomology*, 31:1213-1218.

LUNDGREN, J.G. & WIEDENMANN, R.N. 2004. Nutritional suitability of corn pollen for the predator *Coleomegilla maculat* (Coleoptera: Coccinellidae). *Journal of Insect Physiology*, 50:567-575.

LUNDGREN, J.G., WYCKHUYS, K.A.G. & DESNEUX, N. 2009. Population responses by *Orius insidiosus* to vegetational diversity. *BioControl*, 54:135-142.

MALINGA, G.M., VALTONEN, A., NYEKO, P. & ROININEN, H. 2014. High resilience of galling insect communities to selective and clear-cut logging in a tropical rainforest. *International Journal of Tropical Insect Science*, 34(4):277-286.

MARSHALL, E.J.P. 2002. Introducing field margin ecology in Europe. *Agriculture, Ecosystems and Environment*, 89:1-4.

MCKINNEY, M.L. 2002. Urbanization, biodiversity, and conservation. *BioScience*, 52(10):883-890.

MILLER, T.G. & SPOOLMAN, S.E. 2012. Living in the environment. Toronto: Brooks/Cole Cengage learning.

MILLET, K. 2004. *Birds on a cool green roof.* Chicago Wilderness Magazine.

NEHER, D.A. & BARBERCHECK, M.E. 1999. Biodiversity in agro-ecosystems. New York: CRC Press.

NEW, T.R. 2002. Prospects for extending the use of the Australian lacewings in biological control. *Acta Zoologica Academiae Scientiarum Hungaricae*, 48(2):209-216.

OBRIST, L.B., DUTTON, A., ALBAJES, R. & BIGLER, F. 2006. Exposure of arthropod predators to Cry1Ab toxin in Bt maize fields. *Ecological Entomology*, 31:143-154.

PEREZ, B. 2005. Calling behaviour in the female praying mantis, *Hierodula patellifera*. *Physiological Entomology*, 30:42-47.

PERFECTO, I., VAN DER MEER, J., HANSON, P. & CARTIN, V. 1997. Arthropod biodiversity loss and the transformation of a tropical agro-ecosystem. *Biodiversity and Conservation*, 6:935-945.

PHILPOTT, S.M., COTTON, J., BICHIER, P., FRIEDRICH, R.L., MOORHEAD, L.C., UNO, S. & VALDEZ, M. 2014. Local and landscape drivers of arthropod abundance, richness, and trophic composition in urban habitats. *Urban Ecosystems*, 17:513-532.

PHILPOTT, S.M., PERFECTO, I. & VANDERMEER, J. 2006. Effects of management intensity and season on arboreal ant diversity and abundance in coffee agroecosystems. *Biodiversity and Conservation*, 15:139-155.

PILCHER, C.D., OBRYCKI, J.J., RICE, M.E. & LEWIS, L.C. 1997. Preimaginal development, survival, and field abundance of insect predators on transgenic *Bacillus thuringiensis* corn. *Environmental Entomology*, 26:446-454.

POIANI, K.A., RICHTER, B.D., ANDERSON, M.G. & RICHTER, H.E. 2000. Biodiversity conservation at multiple scales: functional sites, landscapes and networks. *BioScience*, 50(2):133-147.

ROMEIS, J., ALVAREZ-ALFAGEME, F. & BILGER, F. 2012. Putative effects of Cry1Ab to larvae *Adalia bipunctata*. *Environmental Sciences Europe*, 24(18):18-25.

ROMEIS, J. & MEISSLE, M. 2011. Non-target risk assessment of Bt crops: Cry protein uptake by aphids. *Journal of Applied Entomology*, 135:1-6.

ROSENHEIM, J.A. 2001. Source–sink dynamics for a generalist insect predator in habitats with strong higher-order predation. *Ecological Monographs*, 71(1):93-116.

SEAGO, A.E., GIORGI, J.A., LI, J. & SLIPINSKI, A. 2011. Phylogeny, classification and evolution of ladybird beetles (Coleoptera: Coccinellidae) based on simultaneous analysis of molecular and morphological data. *Molecular Phylogenetics and Evolution*, 60(1):137-151.

SARZYNSKI, A. 2012. Bigger is not always better: A comparative analysis of cities and their air pollution impact. *Urban Studies*, 49:3121-3138.

SATTLER, T., OBRIST, M.K., DEULLI, P. & MORETTI, M. 2011. Urban arthropod communities: Added value or just a blend of surrounding biodiversity? *Landscape and Urban Planning*, 103:347-361.

THIERRY, D., PAULIAN, M. & CANARD, M. 2008. Comparison between green lacewing assemblages (Neuroptera:Chrysopidae) in the lower valley of the Danube (Romania) and Loire (France). *Journal of Entomological Research Society*, 10(2):43-53.

TRUTER, J., VAN HAMBURG, H. & VAN DEN BERG, J. 2014. Comparative diversity of arthropods on Bt maize and non Bt maize in two different cropping systems in South Africa. *Environmental Entomology*, 43(1):197-208.

VAN AS, J., DU PREEZ, J., BROWN, L. & SMIT, N. 2012. The Story of Life and the Environment: an African Perspective. Cape Town: Random House Struik.

VERES, A., PETIT, S., CONORD, C. & LAVIGNE, C. 2013. Does landscape composition affect pest abundance and their control by natural enemies? A review. *Agriculture, Ecosystems and Environment*, 166:110-117.

VERGNES, A., KERBIRIOU, C. & CLERGEAU, P. 2013. Ecological corridors also operate in an urban matrix: A test case with garden shrews. *Urban Ecosystems*, 16:511-525.

WEIBULL, A. & OSTMAN, O. 2003. Species composition in agroecosystems: The effect of landscape, habitat, and farm management. *Basic and Applied Ecology*, 4:349-361.

WESSELS, K.J., REYERS, B., VAN JAARSVELD, A.S. & RUTHERFORD, M.C. 2003. Identification of potential conflict areas between land transformation and biodiversity conservation in north-eastern South Africa. *Agriculture, Ecosystems and Environment*, 95:157-178.

WILLIAMS, N.S.G., LUNDHOLM, J. & MACLVOR, J.S. 2014. Do green roofs help urban biodiversity conservation? *Journal of Applied Ecology*, 51:1643-1649.

WILSON, R.J. & MACLEAN, I.M.D. 2011. Recent evidence for the climate change threat to Lepidoptera and other insects. *Journal of Insect Conservation*, 15:259-268.

XU, X., NI, Y. & HE, J. 2015. Urban green corridor planning. *Applied Mechanics and Materials*, 744:2067-2070.

YOUNG, A. 1999. Is there really spare land? A critique of estimates of available cultivated land in developing countries. *Environment, Development and Sustainability*, 1:3-18.

YOUNGSTEADT, E., HENDERSON, R., SAVAGE, A., ERNST, A.F., DUNN, R.R. & FRANK, S.D. 2015. Habitat and species identity, not diversity, predict the extent of refuse consumption by urban arthropods. *Global Change Biology*, 21:1103-1115.

Chapter 2: Optimal sweep net sampling time

2.1 Abstract

Climatic factors fluctuate substantially during a single day and affects efficacy of arthropod sampling. Arthropods are especially influenced by temperature. It is essential to know the optimal sampling times during which sampling of certain species should be done to allow for accurate arthropod biodiversity studies in agro-ecosystems. This study considered the diversity of predacious arthropods of the Chrysopidae, Mantodea, Araneae and Coccinellidiae in and around agricultural fields in the extensively cultivated Highveld grasslands of South Africa. Sweep nets were used to sample fields and field margins daily during the following times (07:00, 12:00, 15:00 and 17:00). Sampling was repeated three times during two years which consisted of three replicates at each of three zones (maize field, field margin and untransformed grassland) at three sites. Shannon diversity index values did not differ significantly between the different sweep net sampling times. Predacious arthropods were, although not significantly so, more abundant within maize fields in the mornings (07:00) and adjacent fields around the afternoon (15:00). This study revealed no particular time to be better suited for sampling the diversity and assemblages of the four groups of predacious arthropods.

2.2 Introduction

Climate is depended on season, longitude, latitude, air pressure and altitude. These factors also affect climatic events during a single day. Insects are very susceptible to changes in temperature as they have small bodies and their body temperatures are usually in balance with the ambient temperature of the environment in which they occur (Chidawanyika *et al*., 2012). These arthropods compensate for changes in environmental temperatures by means of physiological or behavioural changes. Environmental factors such as temperature directly affect the survival, development, reproduction and dispersal of pest insects (Chidawanyika *et al*., 2012). It is important to take into account the possible effect of temperature on trap catches of both epigeal and other arthropods (Saska *et al.,* 2013). Ground dwelling arthropods are especially influenced by temperature, which in turn has been shown to affect the densities at which they are collected by means of different trapping methods (Saksa *et al.*, 2013). Although it was indicated by Saska *et al.* (2013) that temperature has the largest influence on the arthropod catches compared to the weather variables, other fundamental factors such as flexibility in diel rhythms, thermoregulation and body size could also influence the daily activity patterns of arthropods (Baars, 1979). External factors such as vegetation and ground cover essentially may also influence the activity patterns of arthropods (Hatten *et al.,* 2007*).* It was suggested that the response of the total catch to temperature variations can differ from the responses found at species level (Saksa *et al.*, 2013). For example, Saksa *et al.* (2013) reported that carabid catches doubled where an increase of 8°C was noted with regards to the maximum temperature during the day, while an increase of 14°C to the minimum temperatures yielded the same results.

Studies on the effect of temperature on behaviour and daily activity patterns of *Harmonia axyridis (*Coleoptera: Coccinellidae*)* (Pallas, 1773) showed that it was more active during summer than in spring but, ultimately that it was more active during night time (Flowers *et al.*, 2007). This indicates the importance of temperature and plasticity in the activity patterns of specific arthropod species which, could influence the outcome of studies conducted during times when arthropods may be less active or inactive (Flowers *et al.*, 2007). A recent study indicated that circadian rhythm influenced the locomotive and striking behaviour in the praying mantid species *Hierodula patellifera* (Serville, 1839) (Mantodea: Mantidae) (Schirmer *et al.*, 2014). The study indicated that the highest locomotive activity was during crepuscular (Schirmer *et al.*, 2014). Clear activity patterns have also been reported for spiders. Three spiders species of the genus *Cupiennius* (Araneae: Ctenidae) were found to become active approximately 20 min after the onset of darkness (Schmitt *et al.*, 1990). This indicated nocturnal behaviour but also highlighted that sexes behave differently as males of the three spider species were more active than females, allegedly due to their need to find mating opportunities (Schmitt *et al.*, 1990).

Other studies suggest that circadian rhythms have different effects on arthropods such as a decrease in visual sensitivity experienced by horseshoe crabs at dawn. This suggests that not only does temperature have an influence on the activity of arthropods but also daily ambient illumination (Battelle, 2013). According to Saunders (2009), arthropod evolution is based on temperature and light. Insects from higher latitudinal altitudes, where longer nights and cooler temperatures prevail, have evolved to be increasingly sensitive to longer periods of darkness and cooler temperatures, which leads to their adaptive mechanism of diapause cycles (Saunders, 2009). The diel pattern of activity of organisms has also been reported to influence trap catches and outcomes of biodiversity surveys (Saunders, 2009). Araneae and some coccinellid species were found to be more abundant in sweep net samples collected at night compared to samples collected during the day in cereal fields, indicating that nocturnal behaviour of species could influence sample collection (Vickerman and Sunderland, 1975).

It is stated that arthropods have periods of activity which is genetic (Saunders, 1976). An example of this periodicity is the pupal eclosion cycles in species such as that found in *Drosophila melanogaster* (Diptera: Dropsophillidae) (Saunders, 1976). Arthropod behaviour is influenced by their internal chemical balances. For example this can lead to a male cricket not responding to cercal stimulation due to the phase which it is in with regards to its reproduction cycle (Matthews and Matthews, 1978). This indicates the importance of different factors that could influence the activity of arthropods in agro-ecosystems. The movement of arthropods could have influenced the results found in arthropod studies. The dispersal of arthropods can be influenced by environmental factors such as temperature, wind and gradients of the earth but also factors such as food resources, mate location and ovipositioning (Matthews and Matthews, 1978). Despite this some species have activity patterns that contrast this for example, lacewings are primarily nocturnal and thus their behaviour and activity patterns could possibly be influenced by inter-specific competition (Dugatkin, 2009), instead of temperature.

Limited information exists regarding optimal timing of sweep net sampling with the purpose of surveying general predator abundance. Sampling methods are not only influenced by the density of arthropods but also by their daily activity patterns (Saksa *et al.*, 2013). For example, an increase in flight activity was noted for predatory lacewings between 16:00 and 22:00, while hardly any captures were made during daylight hours (Keulder and Van den Berg, 2013). The beating sheet method was described as most effective for predatory insects and spiders in cotton but that the seasonal timing as well as time of day influenced the number of arthropods captured (Wade *et al.*, 2006). For example, the relative abundance of beetle *Rhyzobius lophanthae* (Stephens, 1832) (Coleoptera: Coccinellidae) and leafhopper *Oliarus lubra (*Kirkaldy, 1907) (Hemiptera: Cixiidae) species differed significantly between the sample collected during the day for the beat sheet method (Wade *et al.*, 2006). However, overall the arthropods did not significantly differ between the sampling periods found by Wade *et al*. (2006).

Sweep net sampling was reported to be more accurate than pitfall traps for capturing foliage dwelling arthropods (Cooper *et al.*, 2012). According to Xue *et al.* (2012), sweep net sampling is a useful method of collecting arthropods as it is cost effective and not intrusive but it is laborious and is depended on the skills of the person sweeping. Sweep net sampling tends to be more effective in open habitats such as grasslands and savannas instead of forest or areas that contain vegetation with tall plant structures. A comparitive study of D-vac and sweep net sampling indicated that the sampling methods yielded different results with regards to species richness and the types of species collected (Doxon *et al*., 2011). D-vac samples were dominated by the orders Hemiptera, Diptera and Hymenoptera and the only collections of Dermaptera (earwigs) and Opiliones (harvestmen) (Doxon *et al*., 2011). Sweep net samples were dominated by Hemiptera, Orthoptera and Araneae and included the only specimens of Phasmatodea (stick insects) and Heteronemiidae (brown lacewings) (Doxon *et al*., 2011).

As the target group of predacious arthropods of this larger study were leaf-dwelling organisms in the Araneae, Coleoptera, Mantodea and Neuroptera, sweep net sampling was considered an appropriate sampling technique. However, it is known that temperature and light influence the activity and movement of arthropods. Uninformed choices of sampling times might therefore influence the representativeness of the sample. The aim of this study was to determine the optimal time of day to collect the highest species richness, abundance and diversity of predacious arthropods by means of sweep net sampling.

2.2.1 Aim, objective and hypothesis

The aim of this chapter was to determine optimal sweep net sampling time to capture the highest diversity of the selected predacious arthropod groups.

The specific objective was to determine if either of the following times (i.e. 07:00, 12:00, 15:00 and 17:00) yielded a statistically significant higher abundance or diversity of the selected predacious arthropods.

Hypothesis: If arthropods are susceptible to small changes in temperature, a greater abundance or diversity of predator arthropods could be collected at a certain time of day as the temperature changes at different times during the day, thereby possibly influencing the abundance and diversity of predacious arthropods.

2.3 Materials and Methods

2.3.1 Study sites

The study was conducted in the agricultural area of the Ventersdorp district, which is situated in the Dry Highveld Grassland Bioregion of the Grassland Biome (Mucina and Rutherford, 2006). Three maize fields bordered by untransformed grasslands were selected (Table 2.1). Sampling was done at each of these three fields (Figure 2.1) and involved sampling in three different zones of the maize agro-ecosystem.

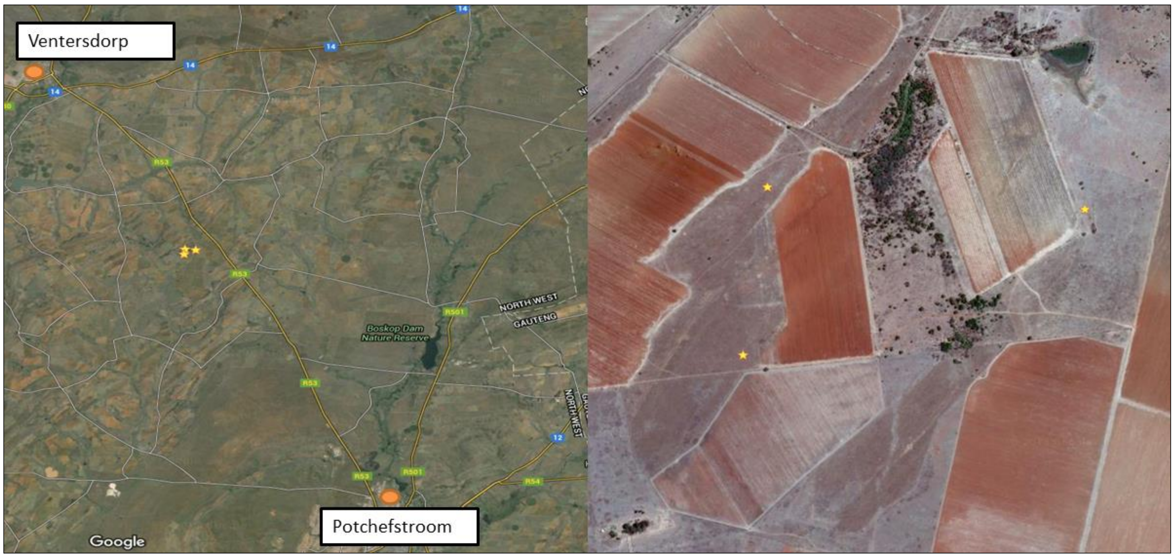

Figure 2.1: Localities (indicated by yellow stars) of the three maize fields and surrounding grasslands used for arthropod sampling.

Each of the sites consisted of a maize field, field margin and surrounding untransformed grasslands (Figure 2.2). Before samples were taken, the temperature was measured by means of a digital thermostat (Exo Terra Digital Thermometer). Sweep net sampling was done at the following times: 07:00, 12:00, 15:00 and 17:00. This process was replicated three times over two years (2014-2015).

The same sites (fields) were used for all three replicates. A fourth site was transformed from a maize (2014) to a sunflower field (2015) as part of a crop rotation strategy and was discarded from further analysis.

Table 2.1: Localities of maize fields in the Venterdorp district and the specific time and temperature at sampling.

Localities:	**Year**	**Time of day**			
		07:00	**12:00**	**15:00**	**17:00**
A **26°28’00.4” S;** **26°56’14.3” E**	**2014**	16°C 3 March	18°C 18 March	31°C 24 March	18°C 26 March
	2015	18°C 3 March	33°C 5 March	28°C 5 March	25°C 5 March
		19°C 10 March	30°C 13 March	30°C 10 March	28°C 12 March
B **26°27'58.4”S;** **26°55'45.1” E**	**2014**	17°C 26 March	21°C 3 March	32°C 18 March	22°C 24 March
	2015	19°C 5 March	28°C 3 March	28°C 6 March	31°C 10 March
		22°C 16 March	31°C 10 March	31°C 16 March	29°C 13 March
C **26°28'13.6" S;** **26°55'42.9" E**	**2014**	16°C 24 March	24°C 26 March	27°C 19 March	24°C 18 March
	2015	22°C 6 March	36°C 3 March	27°C 6 March	27°C 3rd March
		19°C 13 March	27°C 12 March	33°C 13 March	29°C 10 March

To minimize the possible influence of soil and climate, the same study sites were used for the three years of sampling. Sampling was conducted during the soft-dough stage of maize development, approximatley two to four weeks after flowering. Plant-dwelling predacious arthropods of maize during the reproductive stage of plant growth were therefore the target organisms (Truter *et al.*, 2014). Arthropod biodiversity on maize is high after anthesis in comparison to the rest of the growing period (Dively, 2005). Other studies have indicated that arthropod diversity during the plant growth stage captured different trophic levels and could thus be a good method to sample an extensive arthropod community (Eckert *et al.*, 2006).

Figure 2.2: Sampling site indicating the zones in the maize agro-ecosystem.

The mean temperatures for the various sampling times 07:00, 12:00, 15:00 and 17:00 were 18.6°C, 27.5°C, 29.6°C and 25.8°C respectively. The mean daily temperature during the sampling period was 25.4°C (Table 2.2). The highest recorded temperature was 36°C and the lowest was 16°C (Table 2.1).

Table 2.2: Mean temperatures recorded during the three sampling periods.

Sampling Year	Time of day			
	07:00	12:00	15:00	17:00
2014	16.3 °C	21°C	30°C	21.3°C
2015 (1st collection)	19.2°C	30.8°C	29.5°C	28.2°C
2015 (2nd collection)	20°C	29.3°C	31.3°C	28.7°C

Morning temperatures were lower during the 2014 sampling period. In 2014, sampling was delayed for approximately three weeks due to continuous rain. The mean annual rainfall for the Ventersdorp area is approximately 500mm, with January to March being the wettest. The mean monthly rainfall for the three months in 2014 and 2015 was 88mm and 100mm respectively (Table 2.3).

Table 2.3: Monthly rainfall and mean monthly precipitation for January to March at sampling sites during each of the two years of arthropod sampling.

	Year of sampling	
	2014	2015
January	117 mm	142 mm
February	81 mm	72 mm
March	66 mm	88 mm
Mean monthly	88 mm	100 mm

2.3.2 Sweep net sampling

Arthropod samples were collected by means of sweep nets (Appendix 1). Each sample plot was 100 m^2. Three plots were sampled in each of the agro-ecosystem zones, i.e. maize field, field margin and untransformed grassland. Nine samples were therefore taken per locality at the specified times. Sample plots were 50 m apart in each zone per study site (Figure 2.3). A total of 81 samples were taken per time of day (3 repeats x 3 sites x 3 zones x 3 replicates), equating to 324 plot samples in total.

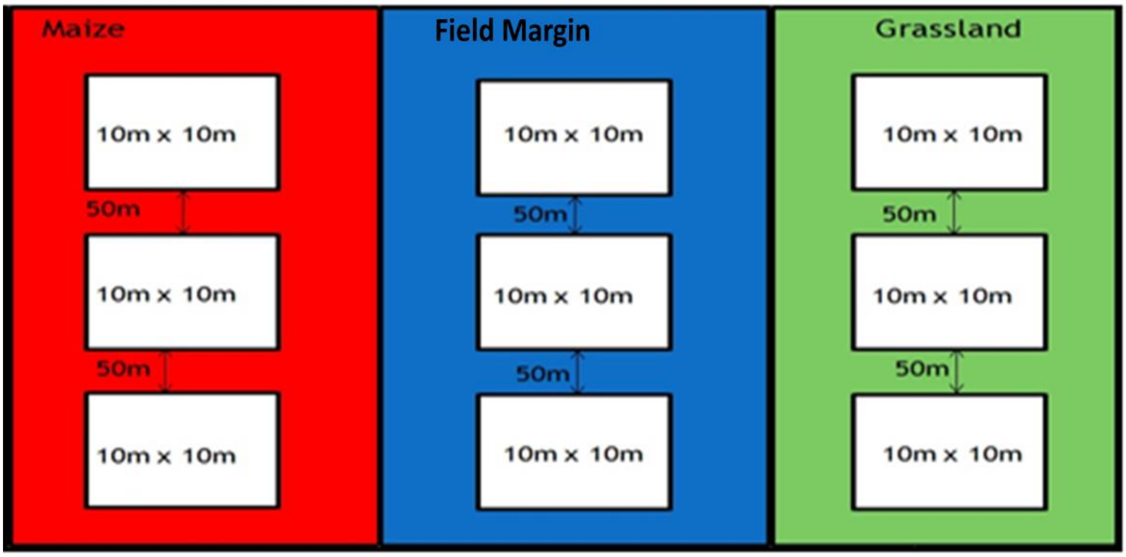

Figure 2.3: Layout of sample plots per study site. Arthropods were collected in each zone of the agricultural landscape at specific time intervals.

Each plot was demarcated as a square by means of a pre-measured rope. Each plot in the grassland area was sampled systematically by sweeping from side to side. In the maize fields the sweeping was conducted with an up and down motion. Each sample consisted of about 30 sweeps, regardless of motion. The samples were then placed into marked bags and refrigerated before the collected arthropods were identified to morpho-species level (for more detail see Appendix 1).

For identification purposes the bag from each plot was emptied into a bowl with 70 % ethanol and from here all lacewings, praying mantids, coccinellids and spiders were transferred into separate containers with 70% ethanol. Collection bottles were marked with different coloured adhesives to distinguish between the four predator groups (see Appendix 1). All specimens were identified to species level with the assistance of the Biosystematics Division at the Agricultural Research Council (ARC) in Pretoria, Gauteng. Praying mantids were identified by means of comparison with the Ditsong museum collection in Pretoria as there was no expert availiable for their identification.

2.3.3 Data analyses

The data did not exhibit normality/normal distribution due to variation in and size of the predator populations in the different agro-ecosystem zones, and did not display constant variance. Shapiro-Wilk and Levene’s tests, as well as Q-Q and Box-plots confirmed that data did not satisfy the assumptions required to analyse variance.

Linear mixed models (LMM) are employed when data is not distributed normally, including data generated by counts which allows for a variety of correlation patterns to be modelled. Population numbers of one zone were related to the population numbers of other zones of the agro-ecosystem, as individuals may switch between zones during the day due to hunting or feeding strategies. Data was also collected from fixed points over time. This created a dependence of data (covariance) which, is corrected by LMM. Multiple measurements per plot per zone were performed every three hours of the day over seasons, i.e. sampling was replicated over time and across populations and therefore correlated data arose in the statistical analyses.

Significant differences between population means were acquired by applying pairwise comparisons on Estimated Marginal Means (EMM). EMM represent the transformed means of populations/samples of unequal size, enabling the comparison of sample means across different treatments, revealing significant differences between means. EMM represent a statistical prediction of population means and are values derived from linear mixed models applied to data with a non-normal distribution. By employing Sidak adjustments, multiple population and predator group means could be accurately compared and differences over time (07:00, 12:00, 15:00, 17:00) and habitats (maize field, field margin, untransformed grassland) determined.

IBM SPSS version 21 was used for all statistical analyses

Abundance, species richness and the Shannon-Wiener Diversity Index was determined for each of the predator groups using Primer software (Hammer *et al.,* 2001). Diversity measures were subjected to Mixed Model Analysis to determine whether significant variation existed between the three diversity measures across the different sampling times, predator groups and zones of the agro-ecosystem. The Mixed Model Analysis also included factorial analysis to determine if there was any interaction between the time of day and the abundance and diversity of the predator groups. Mixed Model Analysis (statistical modelling that using Bayesian inference) was performed (Ellis and Steyn, 2003). Mean abundance graphs were compiled using Statistica software (Statsoft Inc. 2013).

2.4 Results

2.4.1 Time of day and predator groups

Predator abundance did not differ significantly for time of day but, lacewing abundance was noted to increase from 0.6 individuals at 07:00 to two individuals per sample collected at 17:00 This increase was however not significant (Figure 2.4). The number of species present at the different sampling times did not differ significantly. The presence of certain species were noted to vary at the different sampling times. A list of the spiders, lacewings, coccinellids and praying mantid species collected at the different sampling times during this study is provided in Appendix 2. Overall the species richness was 0.5 predator species per sample collected at any given time during the day.

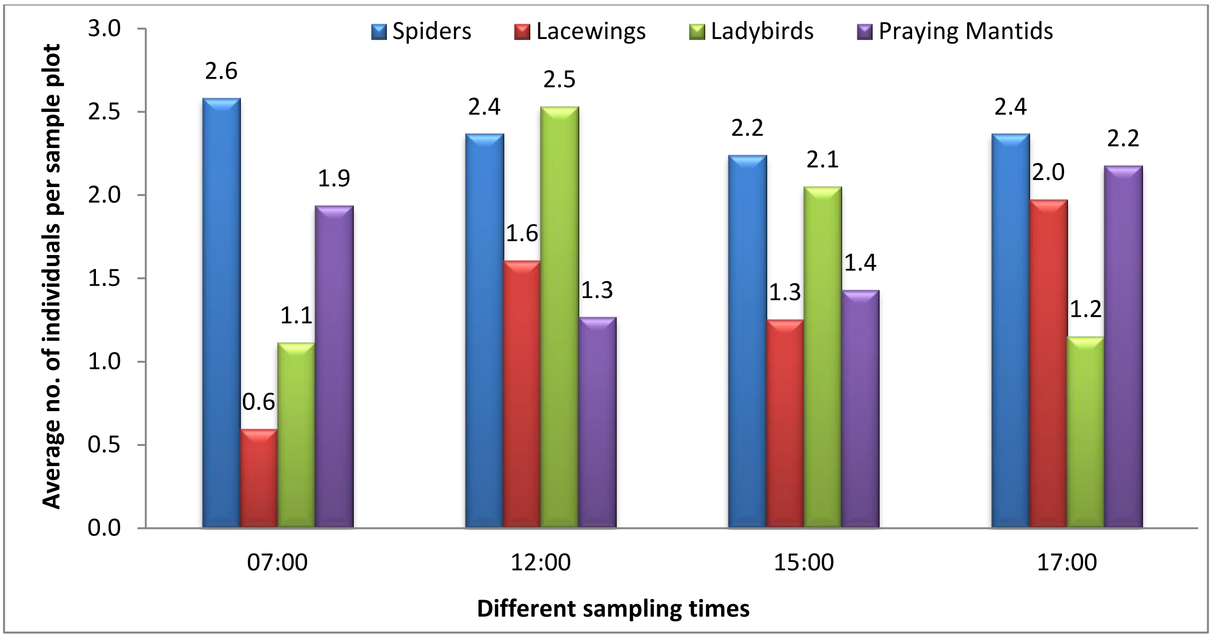

Figure 2.4: Overall mean abundance of arthropod predator individuals recorded at specific sampling times within the agro-ecosytem.

The Mixed Model Analysis which was done to investigate possible interactions between the time of day and predatory group, indicated that time of day at which sampling was done did not have a significant effect on the abundance, species richness or Shannon diversity index of predacious arthropods (Table 2.4). Statistically, significant differences were observed for the abundances, species richness and the Shannon diversity index values between the four different arthropod groups (Table 2.4). The model also showed no significant interactions. Therefore, no specific time of day could be identified for optimal collection of individuals of a specific group. A significant difference was detected between predatory groups such as the diversity of spiders and lacewings and lacewings and coccinellids respectively. This indicated that as one predator species increases in diversity (spiders) the other species (lacewings and coccinellids) either declined or increased.

Table 2.4: Mixed Model Analysis of the interactions between time of day and predatory group for different indices (significant differences indicated by asterisk when $p<0.05$).

	Index					
	Abundance		**Species richness**		**Shannon-Wiener Diversity Index**	
Factor	F-value	P-value	F-value	P-value	F-value	P-value
Time	0.549	0.619	0.953	0.414	0.943	0.419

Group	5.781	0.001*	104.437	0.000*	74.058	0.000*
Group x Time	1.361	0.201	0.430	0.920	0.458	0.903

Even though no significant differences were observed between the overall abundance, richness or diversity of the arthropods at the different sampling times (Figure 2.5), the data indicated that the dominant spider and lacewing species during all sampling times were *Misumenops rubrodecoratus* and *Chrysoperla congrua* respectively (see Appendix 2). Numbers of the dominant coccinellid species differed between the morning (07:00) and afternoons (12:00-17:00) with *Hippodamia variegata* dominating in the morning period and *Cheilomenes lunata* in the afternoon (12:00-17:00). A similar trend was observed for the praying mantis species with *Episiscopus chalybeus* being most abundant in the morning (07:00) and late afternoon (17:00) while *Galepsus* sp. dominated during the mid-afternoon (12:00-15:00).

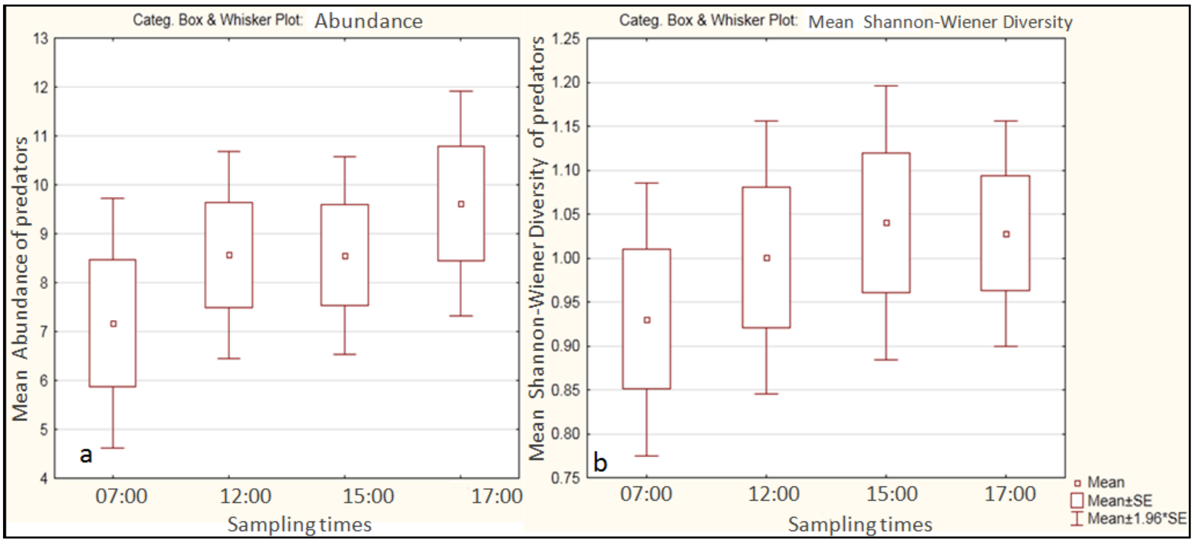

Figure 2.5: Mean abundance (a) and Shannon-Wiener Diversity Index (b) values for the predacious arthropods in the agro-ecosystem at different sampling times

2.4.1.1 Lacewings

Lacewings were analysed using Mixed Model analysis however, only the abundance was analysed since the lacewings collected only consisted of two different species, which is not definitive for diversity criterions. No statistical differences (P-value) were observed. Lacewings were more abundant in agro-ecosystems in the late afternoon (17:00) with an average of two per sample plot collected while the early mornings (07:00) lacewings were the least abundant (Figure 2.6).

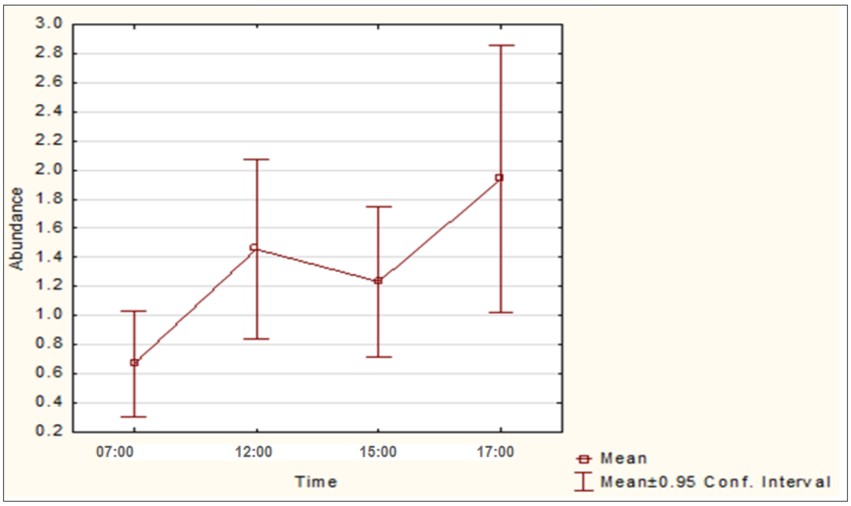

Figure 2.6: Mean abundance of lacewing in the agro-ecosystems at different sampling times.

2.4.1.2 Spiders

The overall spider abundance and richness did not differ significantly (P-value) between different sampling times within the agro-ecosystem. The mean Shannon-Wiener Diversity index did however, indicate a slight, but non-significant increase in spider diversity during the afternoon (15:00) (Figure 2.7).

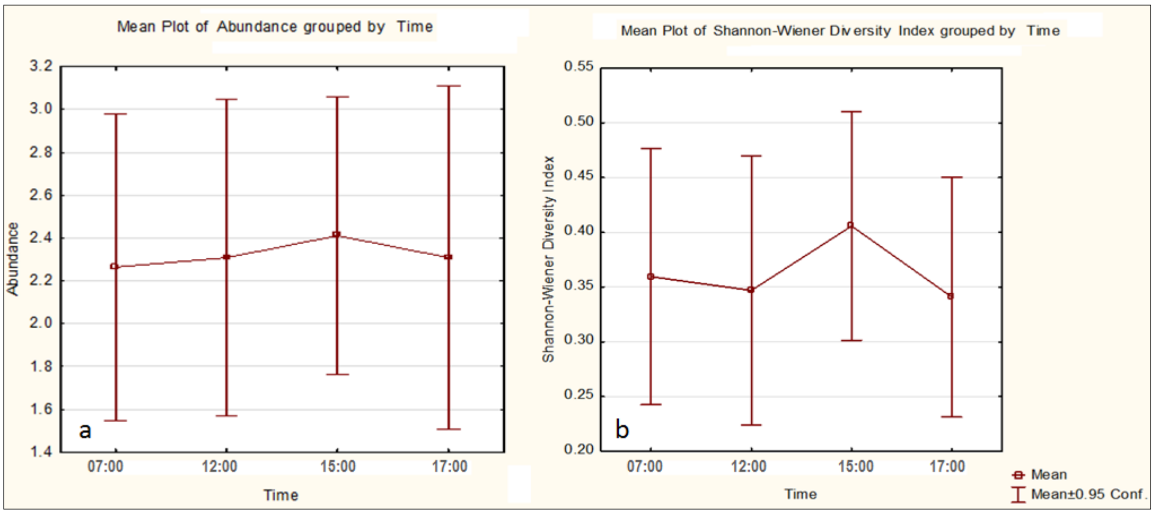

Figure 2.7: Mean abundance (a) and Shannon-Wiener Diversity Index (b) values for the spiders in the agro-ecosystem at different sampling times.

On average, the spider diversity was higher around 15:00, even though it was not statistically significant.

2.4.1.3 Coccinellids

No significant differences were observed at the different sampling times with regards to the diversity or abundance of coccinellids in the maize agro-ecosystem (Figure 2.8).

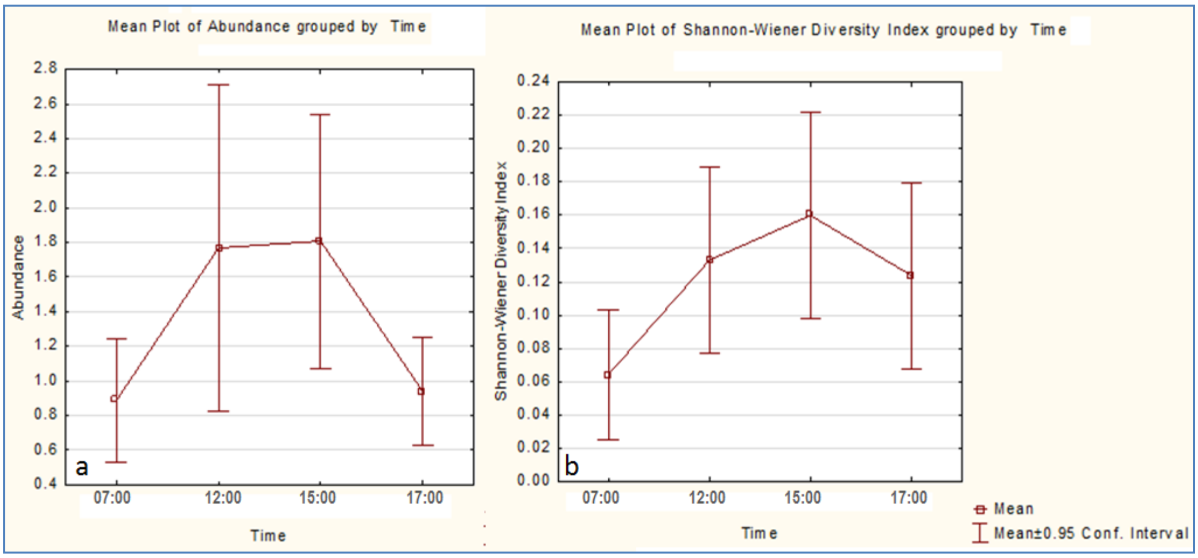

Figure 2.8: Mean abundance (a) and Shannon- Wiener Diversity Index (b) values for the coccinellids in the agro-ecosystem at different sampling times.

Although the abundance of the coccinellids was highest at midday (12:00), their diversity was highest in the afternoon (15:00) (Figure 2.8). Early morning sampling (07:00) produced the lowest abundance and diversity in samples.

2.4.1.4 Praying Mantids

As with the other predators, there were no statistical differences for the praying mantids in terms of their diversity at the different sampling times. However, mean abundance per sample plot was the highest during late afternoon (17:00) while the most diverse samples were collected in the afternoon (15:00) (Figure 2.9), however these differences were not significant.

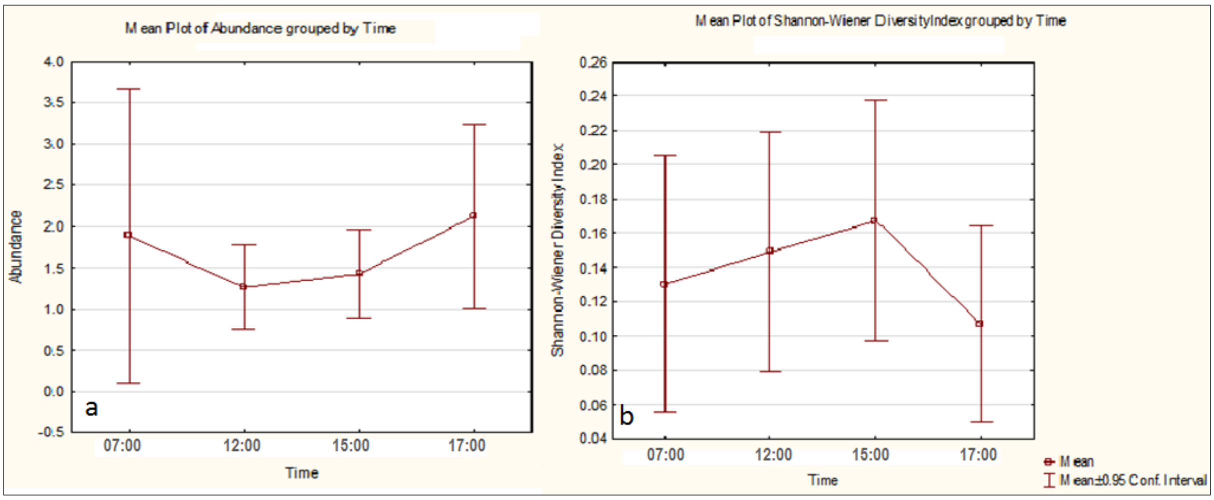

Figure 2.9: Mean abundance (a) and Shannon- Wiener Diversity Index (b) values for the praying mantids in the agro-ecosystem at different sampling times.

2.4.6 Time of day with regards to zones

2.4.6.1 Maize fields

No significant differences were observed between sampling times within the maize fields for diversity or abundance for the predatory arthropods (Table 2.5; Figure 2.10). Shannon- Wiener diversity index indicated the highest difference even though it was not statistically significant.

Table 2.5: Mixed Model analysis for differences in arthropod diversity values between different sampling periods (07:00, 12:00, 15:00 and 17:00).

	Index					
	Abundance		**Species Richness**		**Shannon-Wiener Diversity Index**	
Factor	F-value	P-value	F-value	P-value	F-value	P-value
Time	0.399	0.754	0.252	0.860	0.852	0.471

Significant p values at $p<0.05$

The highest mean abundance, although not statistically significant, was at 07:00 in the morning with an average of 2.5 predators collected per sample plot, while midday (12:00) yielded the least number of predators with an average of 1.75 individuals collected (Figure 2.10 (a)). The Shannon-Wiener diversity index of arthropods collected in the morning (07:00) had an average of 0.25, while the Shannon-Wiener diversity index had an average of 0.45 collected in the late afternoon at 17:00 (Figure 2.10 (b)).

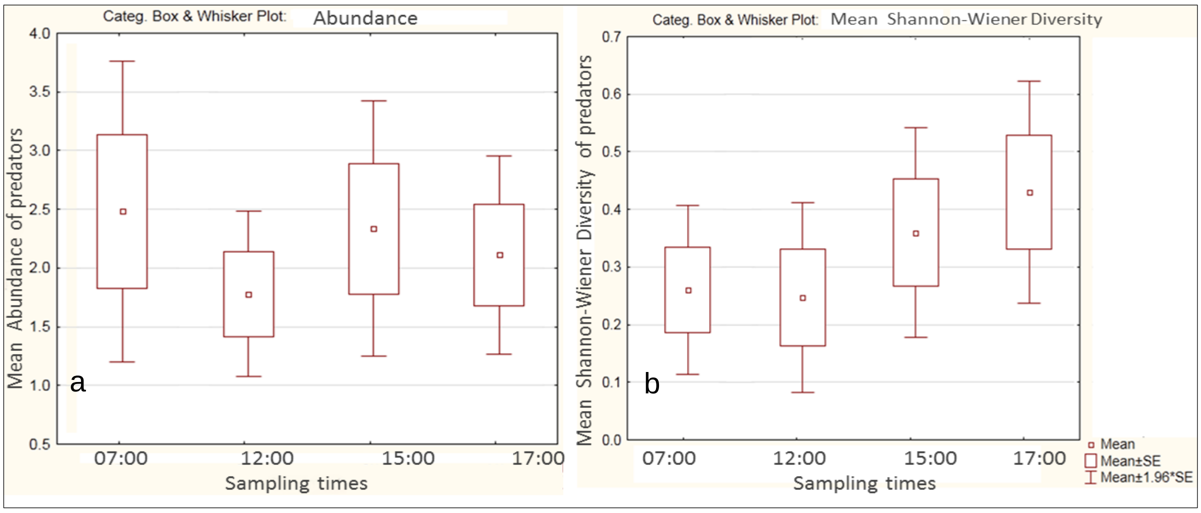

Figure 2.10: Mean abundance (a) and Shannon- Wiener Diversity Index (b) values for predatory arthropods within the maize field at different sampling times.

Coccinellidae was the most abundant predator group within the maize field with the highest average abundance of 1.5 individuals collected per sample plot at 07:00, while praying mantids were the least abundant with an average of 0.1 individuals also collected at 07:00 (Figure 2.11). Lacewings reached peak abundance in maize fields at 07:00 while spiders peaked at 17:00.

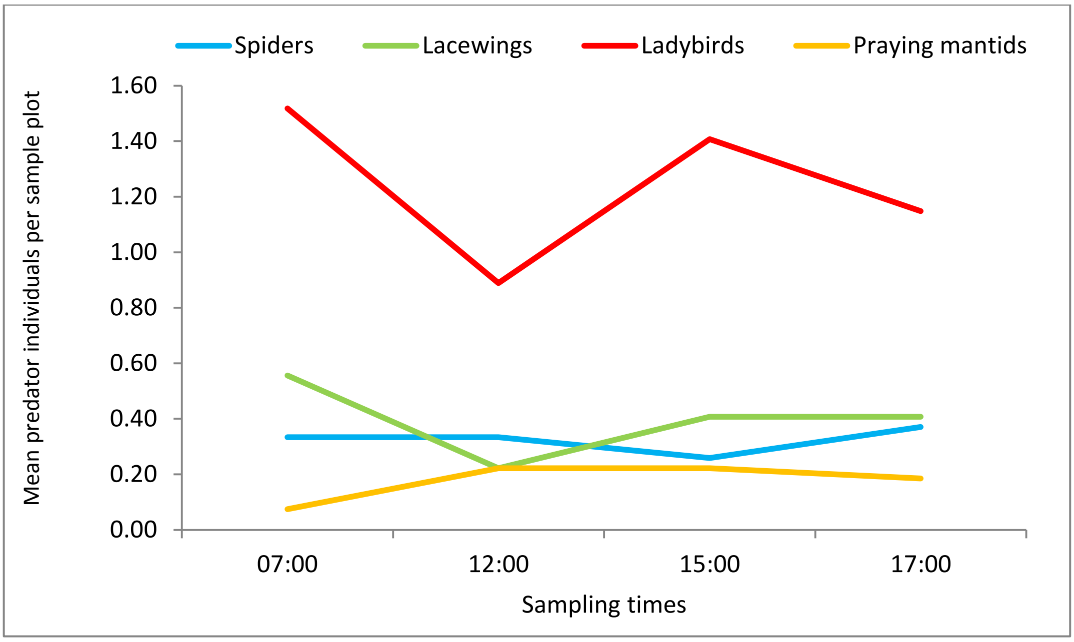

Figure 2.11: Mean number of individuals collected within the maize fields at the different sampling times for each predator group.

2.4.6.1.1 Spiders

Analysis indicated that there were no statistical differences for the spiders within the maize field with regards to the different sampling periods (Table 2.6).

Table 2.6: Mixed Model analysis for differences in spider species richness and abundnace values between different sampling periods (07:00, 12:00, 15:00 and 17:00).

	Index					
	Abundance		**Species Richness**		**Shannon-Wiener Diversity Index**	
Factor	F-value	P-value	F-value	P-value	F-value	P-value
Time	0.114	0.952	0.207	0.891	0.841	0.479

Significant p values at $p<0.05$

The mean abundance of spiders in the maize fields were highest at 17:00 with 0.35 individuals per sample plot while 15:00 was the least abundant time for spiders with an average of 0.25 individuals (Figure 2.12 (a)). The diversity of spiders was also higher at 17:00 with an average of 0.07 while at midday their diversity was zero (Figure 2.12 (b)).

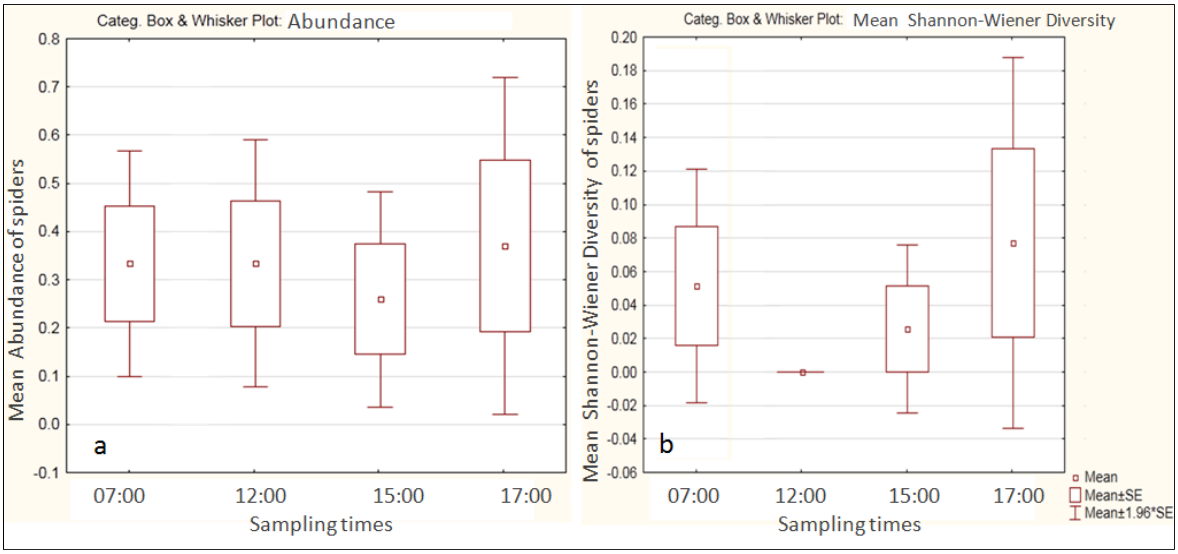

Figure 2.12: Mean abundance (a) and Shannon-Wiener Diversity Index (b) values for spiders within the maize field at different sampling times.

2.4.6.2 Lacewings

No significant differences were found between the different periods for the abundance of lacewings in the maize fields (Table 2.7).

Table 2.7: Mixed Model analysis for differences in lacewing abundance values between different sampling periods (07:00, 12:00, 15:00 and 17:00).

	Index	
	Abundance	
Factor	F-value	P-value
Time	0.579	0.657

Significant p values at $p<0.05$

Lacewings were most abundant in early morning samplings with an average of 0.55 individuals per sample plot while the midday sampling time only yielded an average of 0.25 individuals per sample plot (Figure 2.13).

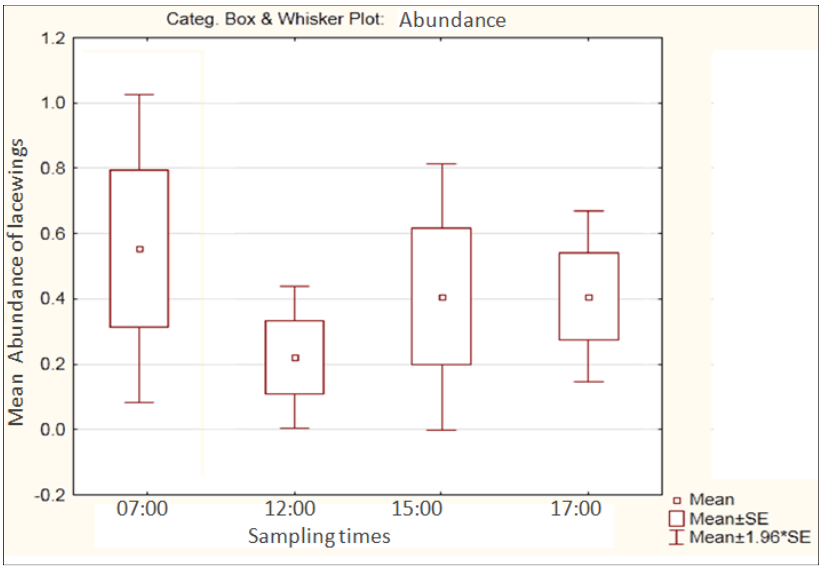

Figure 2.13: Mean abundance of lacewing individuals in maize field at different sampling times.

2.4.6.1.3 Coccinellids

No significant differences were found between the sampling times with regards to the coccinellids within the maize fields (Table 2.8).

Table 2.8: Mixed Model analysis for differences in coccinellids species richness and diversity index values between different sampling periods (07:00, 12:00, 15:00 and 17:00).

	Index					
	Abundance		**Species Richness**		**Shannon-Wiener Diversity Index**	
Factor	F-value	P-value	F-value	P-value	F-value	P-value
Time	0.469	0.705	0.137	0.935	0.747	0.547

Significant p values at p<0.05

Coccinellids were slightly more abundant at 07:00 with an average of 1.5 individuals per sample collected while the average individuals collected at midday was one individual per sample (Figure 2.14 (a)). The midday sampling might have yielded a lower abundance of coccinellids in the maize field but this time of day did have the higher average Shannon-Wiener diversity index (0.155) while the early morning sampling had the lowest mean Shannon-Wiener diversity index value (0.0525) (Figure 2.14 (b)).

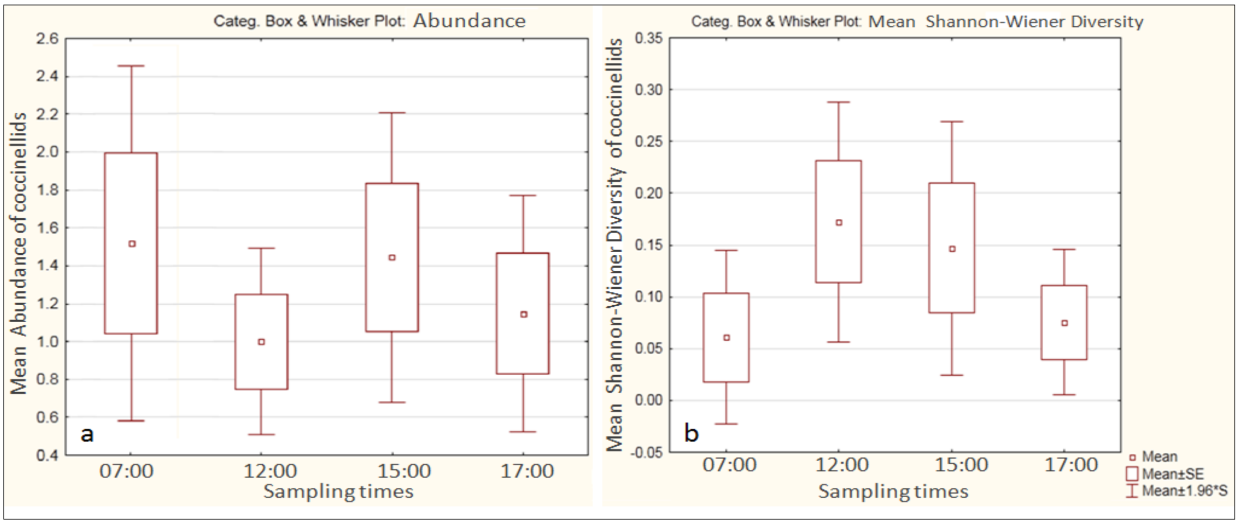

Figure 2.14: Mean abundance (a) and Shannon-Wiener Diversity Index (b) values for coccinellids within the maize field at different sampling times.

2.4.6.1.4 Praying mantids

The abundance and species richness of praying mantids did not significantly differ with regards to diversity or abundance between the different sampling times within the maize fields (Table 2.9).

Table 2.9: Mixed Model analysis for differences in praying mantids species richness and diversity index values between different sampling periods (07:00, 12:00, 15:00 and 17:00).

	Index					
	Abundance		**Species Richness**		**Shannon-Wiener Diversity Index**	
Factor	F-value	P-value	F-value	P-value	F-value	P-value
Time	0.324	0.808	0.376	0.772	0.994	0.406

Significant p values at $p<0.05$

A slightly higher abundance of mantids were observed between 12:00-15:00 while the early morning collections had the lowest abundance of mantids in the maize fields (Figure 2.15). The diversity of mantids collected never exceeded more than two species and therefore the diversity for these predators in the maize fields were not graphically depicted. Both the species richness as well as the Shannon Wiener Diversity index requires a larger number of species to be represented by box and wisker plots as this graph indicates the variance within the data. When this data does not exceed two different species, the variance and mean values within the data are low which is also depicted on these graphs. Therefore there was simply not enough variation in species of praying mantids within maize fields at the different sampling times to be graphically depicted.

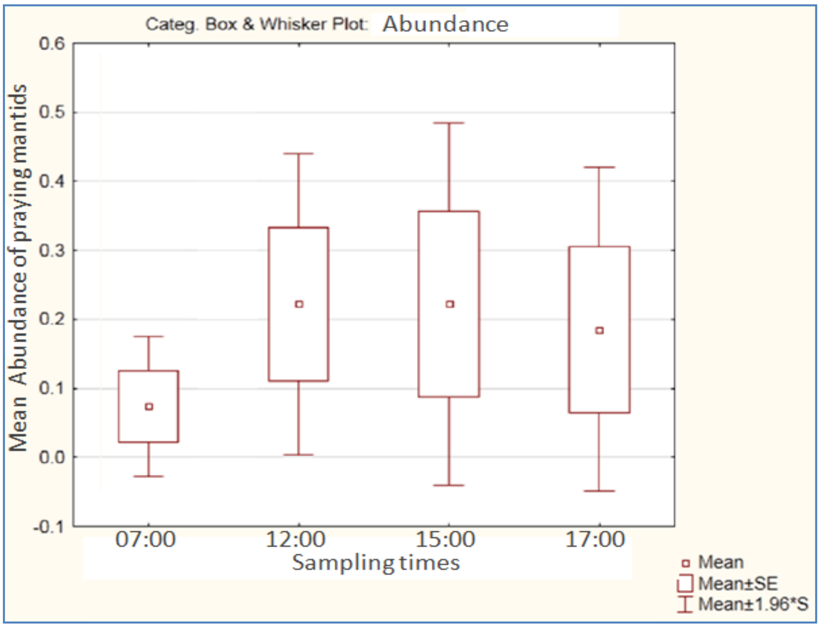

Figure 2.15: Mean abundance value for praying mantids within the maize field at different sampling times.

2.4.6.2 Field Margin

No significant differences were observed between sampling times within the field margin for diversity or abundance of predatory arthropods (Table 2.10; Figure 2.16). Abundance indicated the most variation even though it was not statistically significant.

Table 2.10: Mixed Model analysis for differences in arthropod richness and diversity index values between different sampling periods (07:00, 12:00, 15:00 and 17:00) within the field margin.

	Index					
	Abundance		**Species Richness**		**Shannon-Wiener Diversity Index**	
Factor	F-value	P-value	F-value	P-value	F-value	P-value
Time	0.328	0.805	0.980	0.405	0.581	0.656

Significant p values at $p<0.05$

The highest average abundance, although not statistically significant, was at 17:00 with an average of almost 12 predators collected per sample plot, while early mornings (07:00) yielded the least number of predators with an average of 9 individuals collected (Figure 2.16 (a)). The diversity of arthropods was lowest in the late afternoon (17:00) while the most diverse time of day in the field margin was at midday (12:00) (Figure 2.16 (b)).

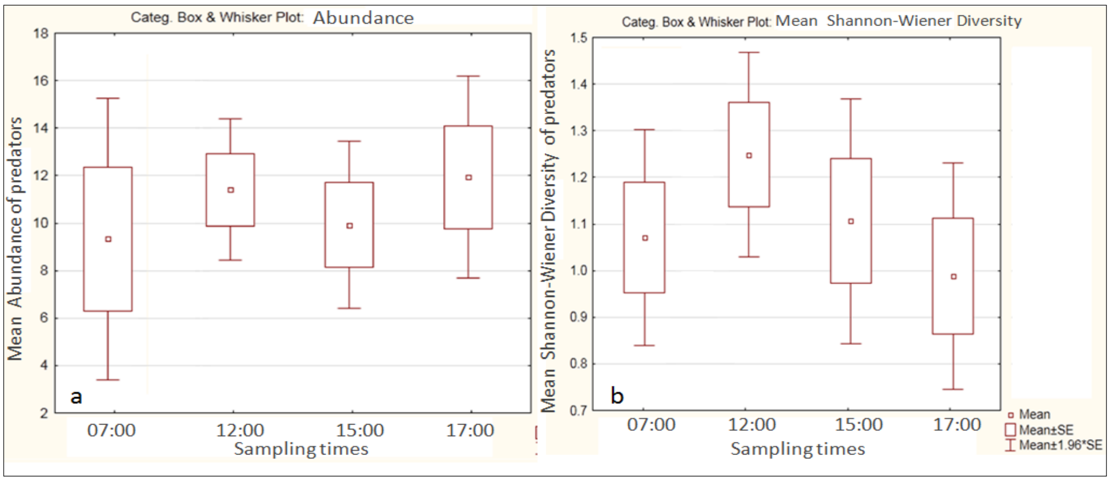

Figure 2.16: Mean abundance (a) and Shannon- Wiener Diversity Index (b) values for predatory arthropods within the field margin at different sampling times.

Spiders were the most abundant predator group within the field margin with the highest mean abundance of 3.8 individuals collected per sample plot at 17:00. Praying mantids were most abundant with a mean of four individuals collected at 07:00 (Figure 2.17). Lacewings were most abundant in field margin at 12:00 with an average of three individuals per sample, while coccinellids were least abundant predator in the field margin, especially at 07:00 with an average of one individual per sample collected.

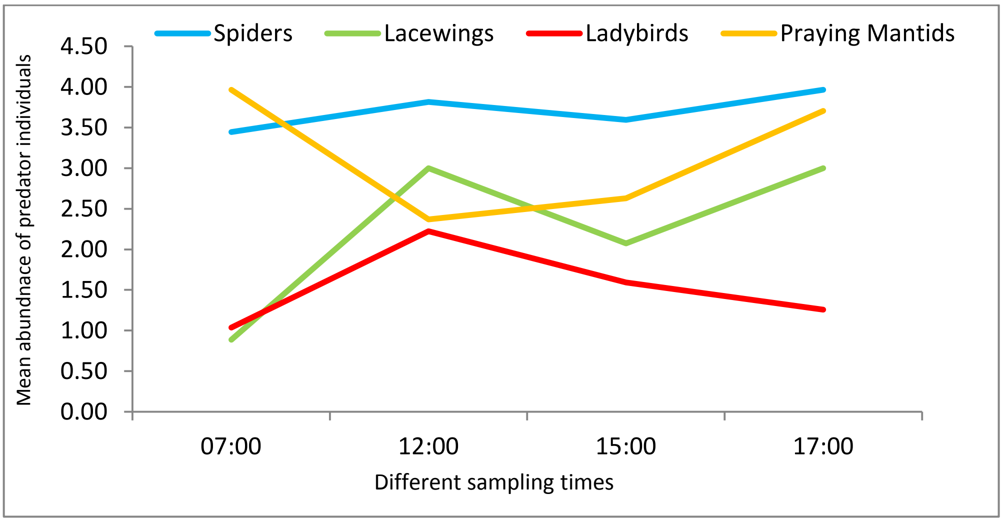

Figure 2.17: Mean number of individuals collected within the field margin at the different sampling times for each predator group.

2.4.6.2.1 Spider

Analysis indicated that there were no significant differences for the spiders within the field margin with regards to the different sampling periods (Table 2.11).

Table 2.11: Mixed Model analysis for differences in spider species richness and diversity index values between different sampling periods (07:00, 12:00, 15:00 and 17:00).

	Index					
	Abundance		**Species Richness**		**Shannon-Wiener Diversity Index**	
Factor	F-value	P-value	F-value	P-value	F-value	P-value
Time	0.085	0.968	0.566	0.647	0.330	0.803

Significant p values at $p<0.05$

The average abundance of spiders in the field margin was the highest at 17:00 with 3.9 individuals per sample plot while 07:00 was the least abundant time for spiders with an average of 3.4 individuals (Figure 2.18 (a)). The Shannon-Wiener diversity index value of spiders was higher at 12:00 with an average of 0.58 while the late afternoon the Shannon-Wiener diversity index decreased to 0.46 (Figure 2.18 (b)).

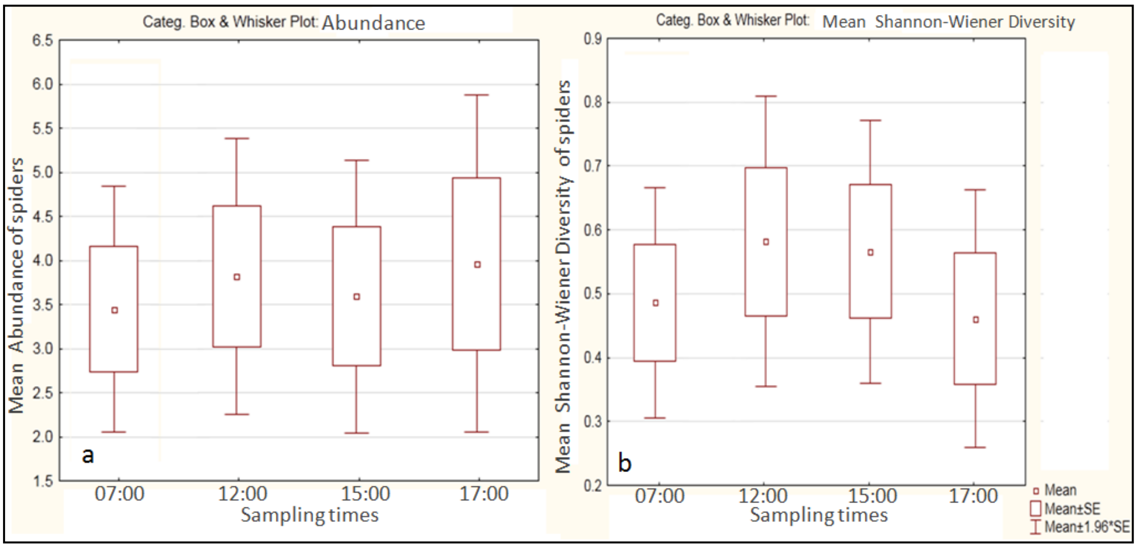

Figure 2.18: Mean abundance (a) and Shannon- Wiener Diversity Index (b) values for spiders within the field margin at different sampling times.

2.4.6.2.2 Lacewings

No statistical differences were found between the different periods for the abundance of lacewings in the field margin (Table 2.12).

Table 2.12: Mixed Model analysis for differences in lacewing abundance values between different sampling periods (07:00, 12:00, 15:00 and 17:00) within the field margin.

	Index	
	Abundance	
Factor	F-value	P-value
Time	1.441	0.276

Significant p values at $p<0.05$

Lacewings were most abundant late in the afternoon with an average of three individuals per sample plot while the early morning sampling time only yielded an average of one individual per sample plot (Figure 2.19).

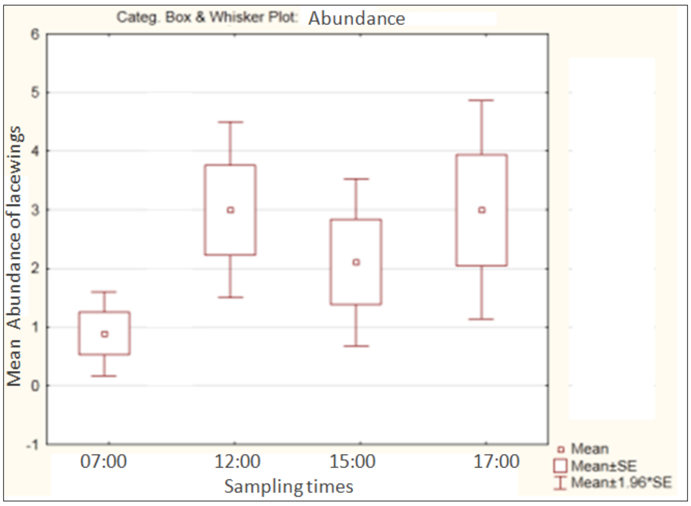

Figure 2.19: Mean abundance of lacewing individuals in field margin at different sampling times.

2.4.6.2.3 Coccinellids

No statistical differences were found between the sampling times with regards to the coccinellids within the field margin (Table 2.13).

Table 2.13: Mixed Model analysis for differences in coccinellids species richness and diversity index values between different sampling periods (07:00, 12:00, 15:00 and 17:00).

	Index					
	Abundance		**Species Richness**		**Shannon-Wiener Diversity Index**	
Factor	F-value	P-value	F-value	P-value	F-value	P-value
Time	1.120	0.390	0.621	0.621	0.501	0.691

Significant p values at $p<0.05$

Coccinellids were more abundant at 12:00 with an average of 2.5 individuals per sample collected while the average individuals collected at 07:00 was one individual per sample (Figure 2.20 (a)). The early morning sampling yielded the lower abundance and mean Shannon-Wiener Diversity Index value (0.155) of coccinellids in the field margin, while the midday sampling had the highest mean Shannon-Wiener diversity index value (0.225) (Figure 2.20 (b)).

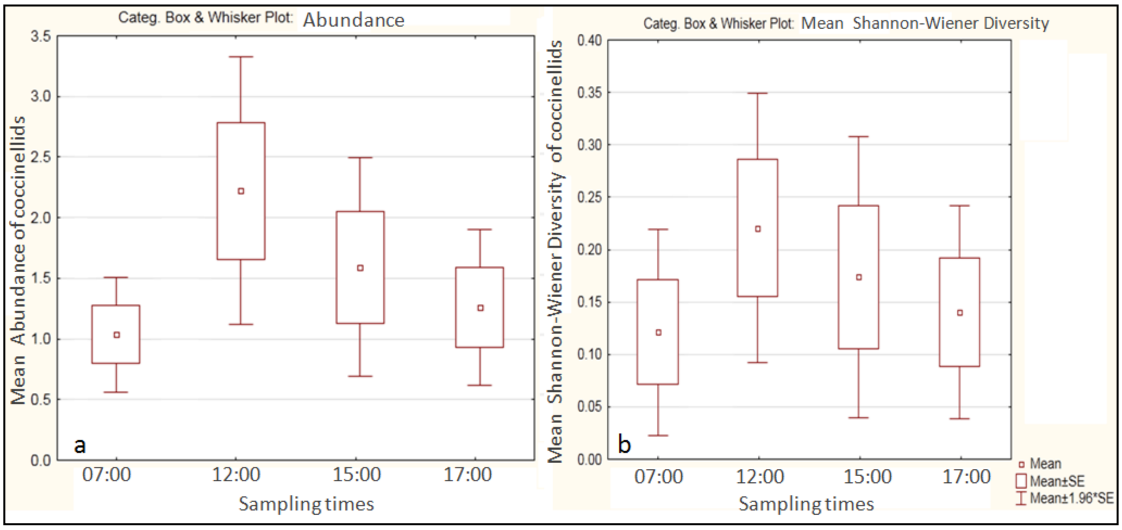

Figure 2.20: Mean abundance (a) and Shannon- Wiener Diversity Index (b) values for coccinellids within the field margin at different sampling times.

2.4.6.2.4 Praying mantids

The abundance and species richness of praying mantids did not differ significantly with regards to diversity or abundance between the different sampling times within the field margins (Table 2.14).

Table 2.14: Mixed Model analysis for differences in praying mantids species richness and diversity index values between different sampling periods (07:00, 12:00, 15:00 and 17:00).

	Index					
	Abundance		**Species Richness**		**Shannon-Wiener Diversity Index**	
Factor	F-value	P-value	F-value	P-value	F-value	P-value
Time	0.332	0.803	0.026	0.994	0.383	0.766

Significant p values at p<0.05

A higher abundance of mantids were observed at 07:00 (four individuals per sample plot) while the midday collections had the lowest abundance of mantids (2 individuals per sample plot) in the field margin (Figure 2.21). The mean diversity per sample plot for the mantids was highest at 15:00 while the lowest abundance was at 17:00.

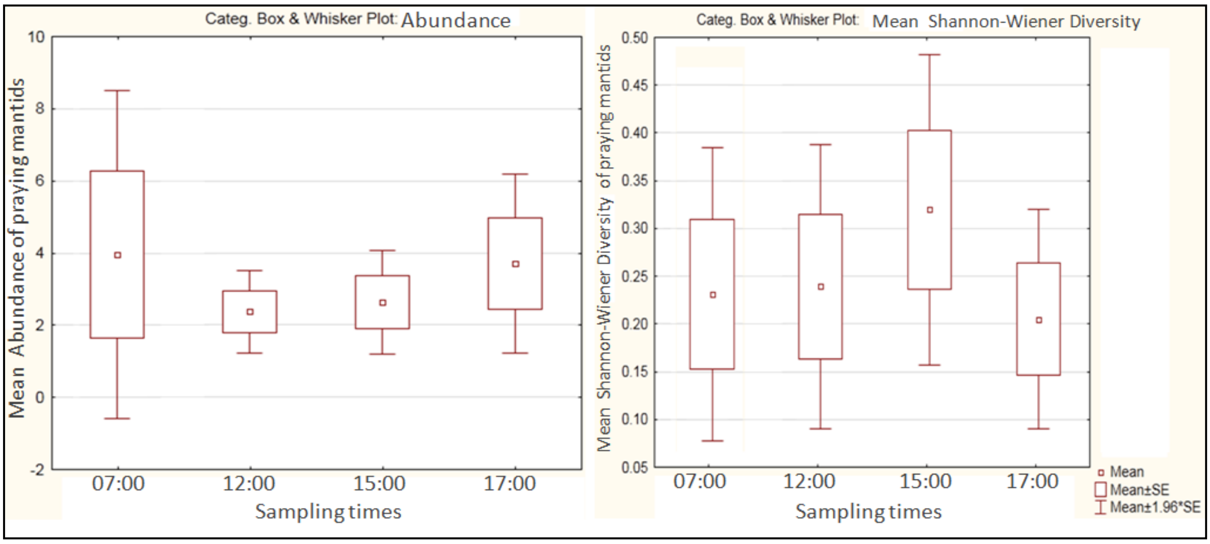

Figure 2.21: Mean abundance value for praying mantids within the field margin at different sampling times.

2.4.6.3 Untransformed grassland

No significant differences were observed between sampling times within the untransformed grasslands for diversity or abundance for the predatory arthropods (Table 2.15; Figure 2.22).

Table 2.15: Mixed model analysis for differences in arthropod richness and diversity index values between different sampling periods (07:00, 12:00, 15:00 and 17:00) within the untransformed grasslands.

	Index					
	Abundance		**Species Richness**		**Shannon-Wiener Diversity Index**	
Factor	F-value	P-value	F-value	P-value	F-value	P-value
Time	0.316	0.814	0.131	0.937	0.334	0.802

Significant p values at $p<0.05$

The highest mean abundance, although not statistically significant, was at 12:00 with an average of almost 9.5 predators collected per sample plot, while early mornings (07:00) yielded the least number of predators with an average of seven individuals collected (Figure 2.22 (a)). The diversity of arthropods was lowest in the late afternoon (17:00) while the field margin had the highest diversity at midday (12:00) (Figure 2.22 (b)).

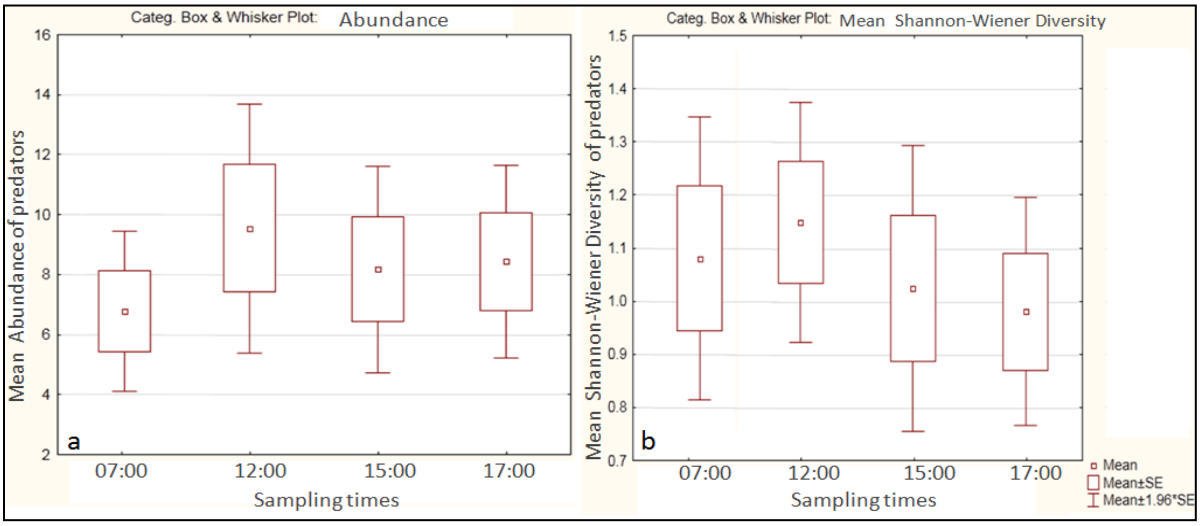

Figure 2.22: Mean abundance (a) and Shannon- Wiener Diversity Index (b) values for predatory arthropods within the untransformed grasslands at different sampling times.

On average spiders were the most abundant predator group within the untransformed grasslands with the highest average abundance of 3.7 individuals collected per sample plot at 07:00, whilst the coccinellids were the most abundant predator group at midday with an average of 4.25 individuals per sample plot (Figure 2.23). Lacewings and praying mantids were most abundant in untransformed grasslands at 17:00 with an average of 2.4 and 2.5 individuals per sample collected respectively. The praying mantids were the least abundant predator group in the untransformed grasslands, especially at 07:00 with an average of less than one individual per sample collected.

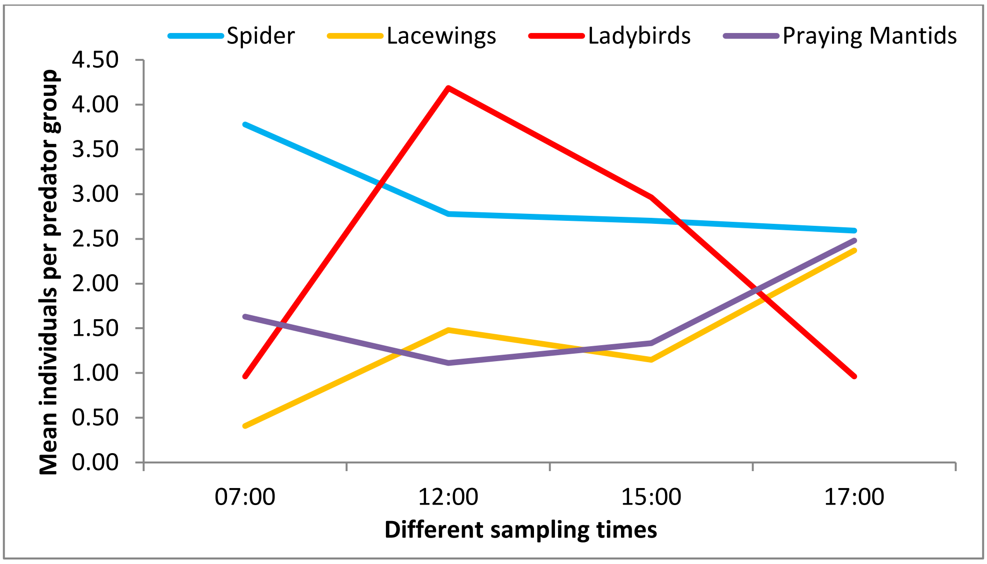

Figure 2.23: Mean number of predator individuals collected within the untransformed grasslands at the different sampling times for each predator group.

2.4.6.3.1 Spider

There were no significant differences in spider diversity and abundance within the untransformed grasslands during the different sampling times (Table 2.16, Figure 2.24).

Table 2.16: Mixed Model analysis for differences in arthropod richness and diversity index values between different sampling periods (07:00, 12:00, 15:00 and 17:00) within the untransformed grasslands.

	Index					
	Abundance		**Species Richness**		**Shannon-Wiener Diversity Index**	
Factor	F-value	P-value	F-value	P-value	F-value	P-value
Time	1.057	0.371	1.883	0.137	1.360	0.259

Significant p values at $p<0.05$

The highest diversity and abundance values for the spiders were recorded during the early morning sampling time with an average of 3.75 spider individuals per sample while the 17:00 sampling time yielded an average of 2.51 spider individuals (Figure 2.24).

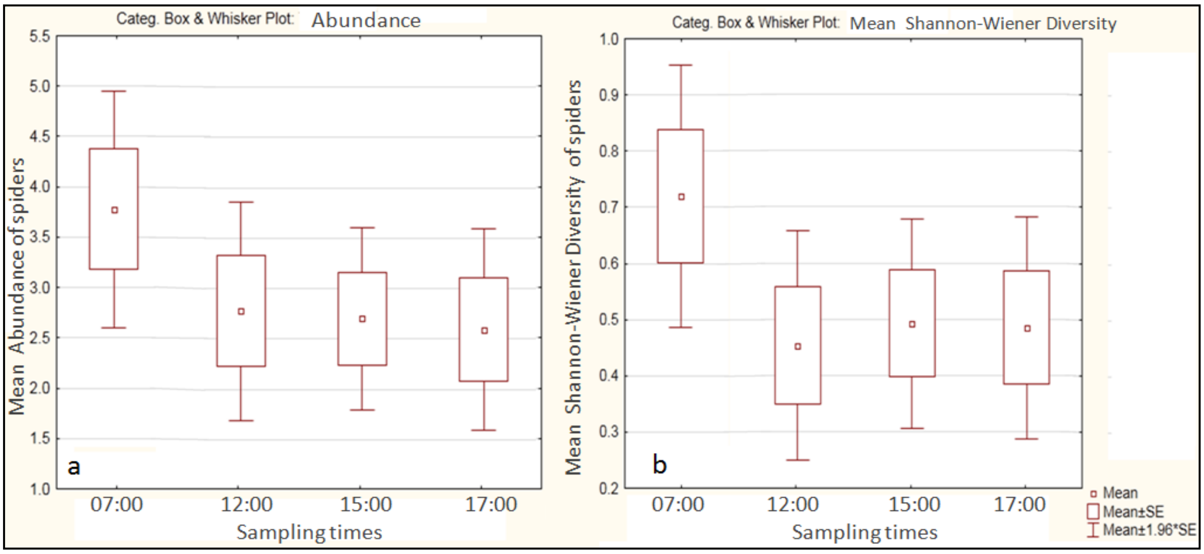

Figure 2.24: Mean abundance (a) and Shannon- Wiener Diversity Index (b) values for spiders within the untransformed grasslands at different sampling times.

2.4.6.3.2 Lacewings

There were no significant differences in lacewing diversity and abundance within the untransformed grasslands during the different sampling times (Table 2.17, Figure 2.25).

Table 2.17: Mixed Model analysis for differences in lacewing abundance between different sampling periods (07:00, 12:00, 15:00 and 17:00) within the untransformed grasslands.

	Index	
	Abundance	
Factor	F-value	P-value
Time	1.435	0.336

Significant p values at p<0.05

The highest abundance values for the lacewings were recorded during the late afternoon sampling time with an average of 2.4 individuals per sample while least abundant sampling time was 07:00 which yielded an average of 0.4 individuals (Figure 2.25).

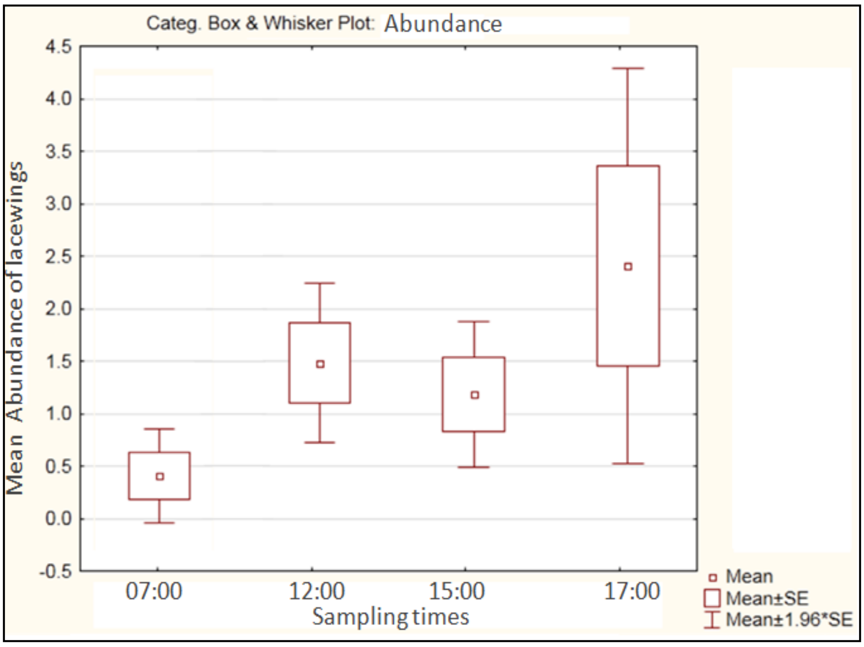

Figure 2.25: Mean abundance values for lacewings within the untransformed grasslands at different sampling times.

2.4.6.3.3 Coccinellids

There were no significant differences in coccinellid diversity and abundance within the untransformed grasslands during the different sampling times (Table 2.18, Figure 2.26).

Table 2.18: Mixed Model analysis for differences in coccinellid richness and diversity index values between different sampling periods (07:00, 12:00, 15:00 and 17:00) within the untransformed grasslands.

	Index					
	Abundance		**Species Richness**		**Shannon-Wiener Diversity Index**	
Factor	F-value	P-value	F-value	P-value	F-value	P-value
Time	0.975	0.457	0.981	0.448	0.649	0.610

Significant p values at $p<0.05$

The highest diversity and abundance values for the coccinellids were recorded during the afternoon sampling time (15:00) with an average of 4.2 individuals per sample while the 07:00 and 12:00 sampling time yielded the least number of coccinellids with an average of one individual per sample plot (Figure 2.26). The lowest Shannon-Wiener diversity index value was during the 12:00 sampling time.

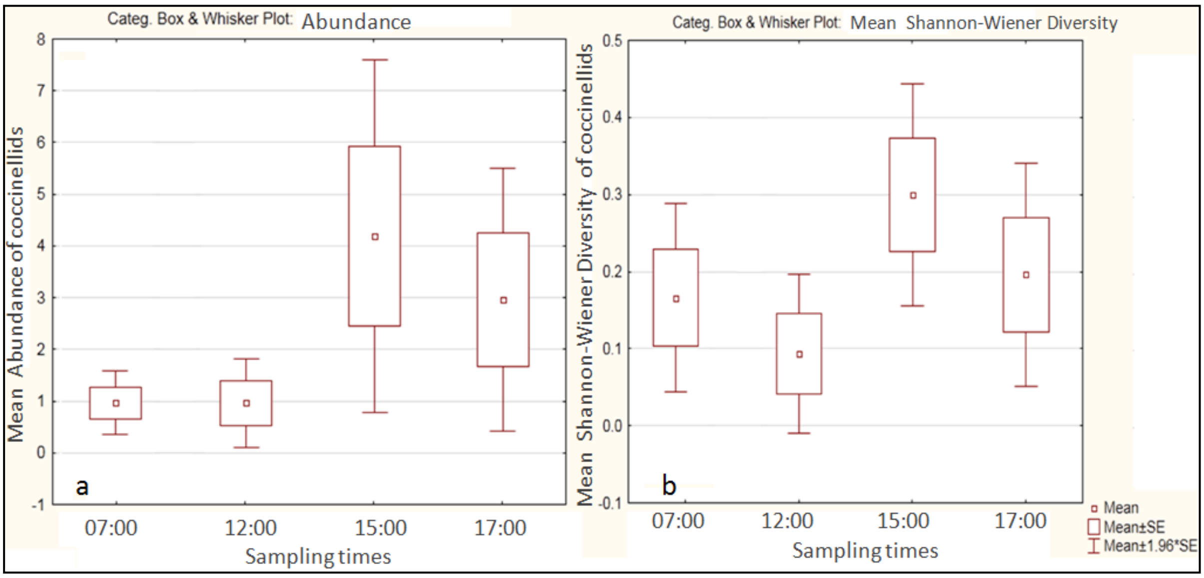

Figure 2.26: Mean abundance (a) and Shannon- Wiener Diversity Index (b) values for coccinellids within the untransformed grasslands at different sampling times.

2.4.6.3.4 Praying mantids

There were no significant differences in praying mantid diversity and abundance within the untransformed grasslands during the different sampling times (Table 2.19, Figure 2.27).

Table 2.19: Mixed Model analysis for differences in praying mantids richness and diversity index values between different sampling periods (07:00, 12:00, 15:00 and 17:00) within the untransformed grasslands.

	Index					
	Abundance		**Species Richness**		**Shannon-Wiener Diversity Index**	
Factor	F-value	P-value	F-value	P-value	F-value	P-value
Time	0.324	0.808	0.376	0.772	0.994	0.406

Significant p values at p<0.05

The highest abundance for the praying mantids were recorded during the late afternoon sampling time (17:00) with an average of 2.25 individuals per sample while the midday (12:00) sampling time yielded the least number of praying mantids with an average of one individual per sample plot (Figure 2.27). The lowest Shannon-Wiener diversity index value was during the 17:00 sampling time and the highest Shannon-Wiener diversity index value was observed to be at 07:00 and 15:00 .

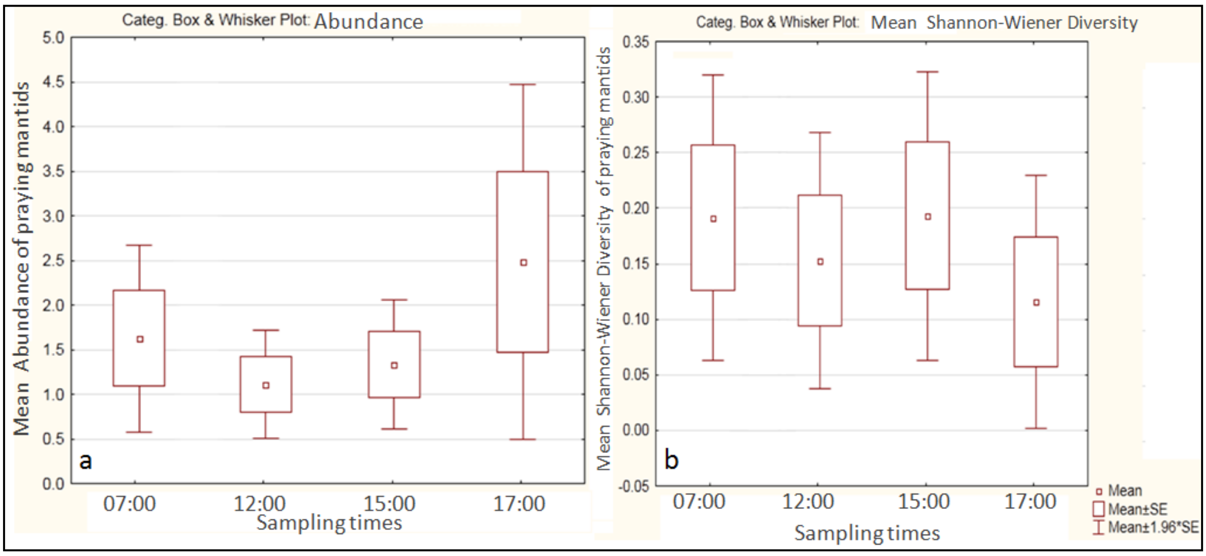

Figure 2.27: Mean abundance (a) and Shannon- Wiener Diversity Index (b) values for praying mantids within the untransformed grasslands at different sampling times.

2.5 Discussion

2.5.1 Time Overall

No statistically significant differences were found between the different sampling periods in this study. This was also reported by a study conducted by Wade *et al.* (2006) for the beat sheet method used in cotton fields with regards to 35 arthropod taxa. Some species of arthropods mentioned in the latter study did, however, show significant differences between sampling periods for arthropods such as ants (Hymenoptera: Formicidae) and a beetle species, *Rhyzobius lophanthae* (Blaisdell) (Coleoptera: Coccinellidae). Hence, the results of this maize field study should be further investigated to determine if individual predator species differed significantly between the different sampling periods, as was the case for some species reported by Wade *et al.* (2006).

Other factors beyond that of sampling time might have influenced the predatory species of this study. For example, the spider species *Achaearanea veruculata* (Urquhart) (Araneae: Theridiidae) was reported by Wade *et al.* (2006) to have had significantly different mean abundances which was related to the different sampling methods utilised as well as days passed after the crops were planted. The abundance of this spider increased from an average of 0.17 individuals per meter row to 2.41 individuals between the 34th and 60th day after planting had taken place. This could indicate the possibility that the developmental stage of crops influence predatory arthropod abundances. Saksa *et al*. (2013) did state the importance of the incorporation of temperature bias into arthropod sampling as it does influence the effectiveness of collection methods such as pitfall traps. Even though the previous study did not include sweep net sampling, it could have influence the abundances of the predatory arthropods in this maize field study. Despite the lack of significant differences it can be stated that the results indicate that predatory arthropods can be collected by sweep net in maize agro-ecosystems with success at any time of day.

2.5.1.1 Lacewings

Although no significant statistical differences were observed for the abundance of lacewings at the different sampling times, their abundancece did increase at 17:00. This result corresponds with the steady increase in lacewing abundance reported by Keulder and Van den Berg (2013) towards the late afternoon at which time lacewings tend to be more active, especially around 16:00. The results reported by Keulder and van den Berg (2013) showed that despite their more active behaviour around 16:00, this study showed that they were even active in the agro-ecosystem at 07:00.

2.5.1.2 Spiders

Spider diversity and abundance was quite uniform during the different sampling times. The only slight increase was in diversity at 15:00 in the afternoon. This slight increase may be a result of more favourable environmental conditions that is more applicable to the requirements of most spider species.

As suggested by Caprio *et al.* (2015) spiders are more influenced by habitat variables thus indicating that climatic fluctuations are not as essential to the survival of these arthropods as the habitat they occupy. This is possibly as a result of the strategies and mechanisms which they use to catch prey with. Spider feeding strategies are diverse and range from active hunting to building webs of different shapes and sizes (Dippenaar-Schoeman, 2014).

It should be noted that due to the fact that spiders are the only wingless predators in this study, their mobility and accordingly their ability to move from one area to another or flee in case of a disturbance, might influence the sample collections and could have made spiders appear more abundant than other predators.

2.5.1.3 Coccinellids

The diversity of coccinellids increased from 07:00 to 15:00 which might indicate that their activity increases at higher temperatures. This could indicate the possibility that coccinellid species prefer higher temperatures which could be possible as all the predators of this study are ectothermic (Matthews and Matthews, 1978).

It has been stated that coccinellids are influenced by daily fluctuations in light intensity, temperature and humidity (Honek, 2012)**.** This could be the reason for the variability in the presence and diversity of coccinellids at different times during the day. Landscape scale factors that influence the community of coccinellids is prey abundance, host plants, microclimatic factors and also the matrix of the surrounding landscape (Honek, 2012). The matrix of surrounding landscape can be seen as a source for the population which is being investigated.

2.5.1.4 Praying Mantids

The mantids were most diverse at 15:00 while they were most abundant in the morning (07:00) and late afternoon at 17:00. The higher abundance at 07:00 and 17:00 might be explained by the dominating species *Episiscopus chalybeus.* This could be a nocturnal species becoming active late in the afternoon and ending its period of activity at around 07:00 in the morning. It should be noted that due to the lack of knowledge of praying mantid ecology, especially that of South African species, explanations provided regarding the differences in their presence during the sampling times is only speculative.

2.5.2 Predator diversity during the day in the different zones

2.5.2.1 Maize fields

Predacious arthropod abundance was highest in maize fields in the early mornings (07:00) and the highest diversity was recorded in the afternoon (17:00), these times however did not significantly differ. This could be due to an increase in predator presence in the surrounding landscape which might allow for a spill over effect into the maize field or perhaps, the environmental circumstances in maize fields are less harsh later in the afternoon.

A recent study indicated that the distribution of two different coccinelid species was correlated with crop maturity, aphid density, planting date, but not nearby vegetation (Musser and Shelton, 2003). This could indicate factors influencing the presence of the selected predator arthropods. Coccinellids were the most abundant predators in the maize fields within this study. The structural and habitat quality in the maize fields could result in one species such as the coccinellid (*Hippodamia variegata*) to dominate in the maize field. Even though *Hippodamia* species are predators they are known to feed on maize pollen, thus indicating that perhaps due to the lack of prey the other predators were displaced but this coccinellid species was able to persist (Obrist *et al.*, 2006).

2.5.2.2 Field Margin

As the results indicated the time of the day at which the highest abundance of predacious arthropods were collected was also the time of day at which the lowest diversity of predacious arthropods were collected in the field margin. The data indicated that the spiders that were the most abundant in the field margin which was due to one species that completely dominated at this time, leading to a reduction in the diversity collected with sweep net sampling. The dominant spider species (*Misumenops rubrodecoratus)* is reported to be easily collected with the use of sweep nets (Dippenaar-Schoeman, 2014), which could have led to their dominance in this study.

The field margin being the most abundant and diverse section of the agro-ecosystem during the different sampling times with regards to the predacious arthropods of this study was influenced by an increase number of praying mantid species, *Episcopus chalybeus*, which was the dominating praying mantid species in this section of the agro-ecosystem. *Episcopus chalybeus* is dominant in the morning and late afternoon which, could suggest that they might have nocturnal behaviour. This could contradict Feng *et al.* (2012) who reported that sweep netting which is mostly done during the day does not always succeed in sampling nocturnal species.

2.5.2.3 Untransformed grasslands

Results show an increase in the numbers of coccinellids sampled in the untransformed grasslands at midday. The differences in dominating species between the field margin and the untransformed grasslands could indicate that the field margin due to its locality and prevalence to higher disturbance levels and even pesticide drift (Kleijn & Snoeijing, 1997) influences the presences of some arthropod predators, more than the sampling times.

2.5.3 Predators during sampling times within different zones of the agro-ecosystem

2.5.3.1 Lacewings

It is has been indicated that lacewings showed increased activity at night and tend to rest in their natural habitats during the day (Canard and Semeria, 1984). Thus the results of this study indicate that their activities during the day might have been as part of their escape flight behaviour due to a disturbance such as ploughing or pesticide applications (Canard and Semeria, 1984). These patterns of activity are short and is only used to get to a safe new refuge away from a disturbance (Canard and Semeria, 1984). This indicates that lacewings might be resting in the field margin during the day which could thus be seen as the habitat which they prefer especially due to their preference for low vegetation (Canard and Semeria, 1984). The field margin is an ideal locality as it is between the untransformed grassland which possibly has a higher habitat varability while the maize field are abundant in pest species (which increases food resources of these predators) and have good structure for egg laying. Newly hatched lacewing juveniles are therefore in an ideal environment to survive due to the high availibility of food resources.

2.5.3.2 Spiders

The field margin and untransformed grasslands were the most abundant and diverse habitats with regards to the spiders of this study, especially the field margin which was particularly abundant and diverse from midday to late afternoon. This might be due to the habitat variability such as rocks, crevices and plant structural diversity that can increase the ease of building webs. The vegetation density might also increase in the success of catching prey in these habitats, as webs might be less visible than in more open habitats such as maize fields. A recent study found that an increase in web-building spider richness was correlated to vegetation diversity and plant coverage (Diehl *et al.*, 2013).

2.5.3.2 Coccinellids

Coccinellids were the most abundant in maize fields during early morning sampling, while their abundance increased in the surrounding landscapes at midday, especially in the untransformed grasslands. In each community at least two to four species of coccinellids can dominate an environment (Honek, 2012). *Hippodamia variegata* is one of the most common species to dominate habitats that include vegetation structure which has a high abundance of herbaceous strands (Honek, 2012). This species did dominate the maize field but the surrounding landscape was dominated by *Cheilomenes lunata*.

2.5.3.4 Praying mantids

Praying mantids seem to be more diverse in the field margin, especially at 07:00. This could indicate that the vegetation structure is more preferable to the requirements of different species of praying mantids. It was noted in a study that vegetation height influences hunting capabilities of praying mantid species (Ilyse *et al.*, 1983). This could be the case within the field margin however, further studies are required to confirm this phenomenon.

2.6 Conclusions

Results indicated that there was no optimal sweep net sampling time for the selected predatory arthropods, thus resulting in a rejection of the hypothesis which stated that: If arthropods are susceptible to small changes in temperature, a greater abundance or diversity of predator arthropods could be collected at a certain time of day as the temperature changes at different times during the day, thereby possibly influencing the abundance and diversity of predacious arthropods. It seems however that the habitat in which predacious arthropods are being collected has a greater influence on collection success. Despite the lack of statistical significance, the most predators were collected at 17:00 and the least at 07:00. This study indicates that sweep net sampling in maize agro-ecosystems can be effectively done for predatory arthropods at any time of day, if it presides within the subsequent optimal seasons and maize developmental stage.

2.7 References

BAARS, M.A. (1979) Patterns of movement of radioactive carabid beetles. *Oecologia*, 44,125–140.

BATTELLE, B. A. 2013. What the clock tells the eye: Lessons from an ancient arthropod. *Integrative and Comparative Biology*, 53(1):144-153.

CANARD, M. & SEMERIA, Y. 1984. Biology of Chrysopidae. Boston: Junk Publishers.

CAPRIO, E., NERVO, B., ISAIA, M., ALLEGRO, G. & ROLANDO, A. 2015. Organic versus conventional systems in viticulture: Comparative effects on spiders and carabids in vineyards and adjacent forests. *Agricultural Systems*, 135:60-69.

CHIDAWANYIKA, F., MUDAVANHU, P. & NYAMUKONDIWA, C. 2012. Biologically based methods for pest management in agriculture under changing climates: Challenges and future directions. *Insects*, 3:1171-1189.

COOPER, N.W., THOMAS, M.A., GARFINKEL, M.B., SCHNEIDER, K.L & MARRA, P.P. 2012. Comparing the precision, accuracy, and efficiency of branch clipping and sweep netting for sampling arthropods in two Jamaican forest types. *Journal of Field Ornithology*, 83(4):381-390.

DIEHL, E., MADER, V.L, WOLTER, V. & BIRKHOFER, K. 2013. Management intensity and vegetation complexity affect web-building spiders and their prey. *Oecologia*, 173:579-589.

DIPPENAAR-SCHOEMAN, A. 2014. Field guide to the spiders of South Africa. Pretoria: LAPA Publishers.

DIVELY, G.P. 2005. Impact of transgenic VIP3A x Cry1Ab lepidopteran-resistant field corn on the non-target arthropod community. *Environmental Entomology.* 34: 1267-1291.

DOXON, E.D., DAVIS, C.A. & FUHLENDORF, S.D. 2011. Comparison of two methods for sampling invertebrates: Vacuum and sweep-net sampling. *Journal of Field Ornithology*, 82(1):60-67.

DUGATKIN, L.A. 2009. Principles of Animal Behaviour. New York: W.W Norton & Company.

ECKERT, J., SCHUPHAN,I., HOTHORN, L.A. & GATHMANN, A. 2006. Arthropods on maize ears for detecting impacts of Bt maize on non-target organisms. *Environmental. Entomology*, 35: 554-560.

ELLIS, S.M. & STEYN, H S. 2003. Practical significance (effect sizes) versus or in combination with statistical significance (p-values). *Management Dynamics*, 12(4):51-53.

FENG, J., XUE, D., SANG, W., AXMACHER, J.C. & ZOU, Y. 2012. A comparison of terrestrial arthropod sampling methods. *Journal of Resources and Ecology*, 3(2):174-182.

FLOWERS, R.W., SALOM, S.M., KOK, L.T. & MULLINS, D.E. 2007. Behaviour and daily activity patterns of specialist and generalist predators of the Hemlock Woolly Adelgid, *Adelges tsugae*. *Journal of Insect Science*, 7(44). Available online: http://www.insectscience.org/7.44.

HAMMER, Ø., HARPER, D.A.T. & RYAN, P.D. 2001. PAST: Paleontological statistics software package for education and data analysis. *Paleontologia Electronica,* 4(1):9.

HATTEN, T.D., BOSQUE-PEREZ, N.A., JOHNSON-MAYNARD, J. & EIGENBRODE, S.D. (2007) Tillage differentially affects the capture rate of pitfall traps for three species of carabid beetles. *Entomologia Experimentalis et Applicata*, 124:177–187.

HONEK, A. 2012. Distribution and Habitats. *In*: *Ecology and behaviour of ladybird beetles (Coccinellidae).* Oxford: Blackwell Publishing. 110-140 p.

ILYSE, H., RATHET, L. & HURD, L.E. 1983. Ecological relationships of three co-occurring mantids, *Tenodera sinensis* (Saussure), *T. angustipennis* (Saussure), and *Mantis religiosa* (Linnaeus). *American Midland Naturalist*, 110(2):240-248.

KEULDER, R. & VAN DEN BERG, J. 2013. Patterns of lacewing (Neuroptera: Chrysopidae) flight activity, flight height and spatial distribution of eggs on maize plants. *African Entomology*, 21(1):95-102.

KLEIJN, D. & SNOEIJING, G.I.J. 1997. Field boundary vegetation and the effects of agrochemical drift: botanical change caused by low levels of herbicide and fertilizer. *Journal of Applied Ecology*, 34:1413-1425.

MATTHEWS, R.W. & MATTHEWS, J.R. 1978. Insect Behaviour. Georgia: John Wiley & Sons.

MUCINA, L. & RUTHERFORD M.C. 2006. The vegetation of South Africa, Lesotho and Swaziland. *Strelitzia* 19. Pretoria: SANBI.

MUSSER, F.R. & SHELTON, A.M. 2003. Factors altering the temporal and within-plant distribution of Coccinellids in corn and their impact on potential intra-guild predation. *Environmental Entomology*, 32(3):575-583.

OBRIST, L.B., DUTTON, A., ALBAJES, R. & BIGLER, F. 2006. Exposure of arthropod predators to Cry1Ab toxin in Bt maize fields. *Ecological Entomology*, 31:143-154.

SAKSA, P., VAN DER WERF, W., HEMERIK, L., LUFF, M.L., HATTEN, T.D. & HONEK, A. 2013. Temperature effects on pitfall catches of epigeal arthropods: a model and method for bias correction. *Applied Ecology*, 50:181-189.

SAUNDERS, D.S. 1976. The biological clock of insects. *Science America*, 234:114-121.

SAUNDERS, D.S. 2009. Circadian rhythms and the evolution of photoperiodic timing in insects. *Physiological Entomology*, 34:301-308.

SCHIRMER, A.E., PRETE, F.R., MANTES, E.S., URDIALES, A.F. & BOGUE, W. 2014. Circadian rhythms affect electroretinogram, compound eye colour, striking behaviour and locomotion of the praying mantis *Hierodula patellifera*. *The Journal of Experimental Biology*, 217:2853-3861.

SCHMITT, A., SCHUSTER, M. & BARTH, F.G. 1990. Daily locomotor activity patterns in three species of Cupiennius (Araneae, Ctenidae): The males are the wandering spiders. *The Journal of Arachnology*, 18(3):249-255.

STATSOFT INC. 2013. Statistica, version 11. Tulsa: StatSoft.

TRUTER, J., VAN HAMBURG, H. & VAN DEN BERG, J. 2014. Comparative diversity of arthropods on Bt maize and non Bt maize in two different cropping systems in South Africa.. *Environmental Entomology*, 43(1):197-208.

VICKERMAN, G.P & SUNDERLAND, K.D. 1975. Arthropods in cereal crops: nocturnal activity, vertical distribution and aphid predation. *Journal of Applied Ecology*, 12(3):755-766.

WADE, M.R., SCHOLZ, B.C.G., LLOYD, R.J., CLEARY, A.J., FRANZMANN, B.A. & ZALUCKI, M.P. 2006. Temporal variation in arthropod sampling effectiveness: the case for using the beat sheet method in cotton. *Entomologia Experimentalis et Applicata*, 120:139-153.

XUE, D., ZOU, Y., FENG, J., SANG, W. & AXMACHER, J. 2012. A comparison of terrestrial arthropod sampling methods. *Journal of Resources and Ecology*, 3(2):174-182.

Chapter 3: Diversity of predacious arthropods in maize agro-ecosystems

3.1 Abstract

Biologically diverse ecosystems are threatened by disturbances such as agricultural activities and especially mono-cropping systems that reduce biodiversity. Due to the increased demand for organically produced food and adverse effects related to the use of chemical pesticides, conservation bio-control is becoming increasingly important for pest control. However, current knowledge is lacking regarding the diversity and abundance of arthropod predators in crop fields and their movement patterns within the agricultural environment. In this study we selected four groups of predaceous arthropods, namely lacewings (Chrysopidae), praying mantids (Mantodea), spiders (Araneae) and ladybirds (Coccinellidiae) and assessed their diversity and species assemblages in and around agricultural fields in the extensively cultivated Highveld grasslands of South Africa. We sampled maize fields along a disturbance gradient i.e. maize fields, field margins and untransformed grasslands. Our results indicated that field margins are the most diverse and abundant in terms of predacious arthropod diversity along a maize field-field margin-untransformed grassland gradient. Field margins could therefore play an important role in the maintenance of predacious arthropod diversity and could play an important role in integrated pest management strategies.

3.2 Introduction

Agriculture, although important for food security, impacts on the biodiversity of the landscape in which it resides. Associated agricultural activities that may influence biodiversity include tillage, cropping system, grazing as well as insecticide and fertilizer application (McLauglin and Mineau, 1995). The wide-scale planting of genetically modified crops also raises concern due to the possible non-target effects it might have on natural biota witin the agro-ecosystem (Pilcher *et al.,* 1997; Lundren and Wiedenmann, 2004).

Even though there has been an increase in the importance of conserving natural biota, small organisms such as arthropods are mostly neglected (Pimentel *et al.*, 1992). This neglect may be ascribed to the lack of knowledge on the importance of these arthropods and the ecosystem services they provide in agricultural systems. Arthropod diversity is strongly impacted by activities inside crop fields. A recent study indicated that there is a significant decrease not only in plant diversity but also arthropod diversity within maize fields as opposed to the surrounding natural grasslands (Botha *et al.,* 2015). It is suggested that arthropods are not only affected by the plant diversity of the surrounding agricultural areas but also by the biome in which the agricultural activities, especially a maize field, reside (Botha *et al.,* 2015). Truter *et al.* (2014) also indicated that maize fields in the grassland biome of South Africa are diverse and abundant habitats for arthropods. In the latter study over two seasons a total of 8771 individual arthropods consisting of 288 morpho species in 20 different arthropod orders were collected on maize plants. This diversity and functionality is a valuable resource due to the possible ecosystems services that it provides. Its contribution is however influenced by antroplogical activities such as agriculture of which the impacts are largely unknown.

Components of biodiversity, such as those described above, can be managed and the ecosystem services it provides can be exploited to the benefit of the farming system. A recent study indicated that a crop edge or field margin of the grass *Panicum maximum* (Jacq) could act as a reservior for predacious arthropods such as spiders whom prey upon the eggs of the maize stem borer *Chilo partellus* (Swinhoe, 1885) (Koij *et al.*, 2007). During 2005, in an attempt to increase biodiversity in agro-ecosystems, a new landscape feature i.e. a grassy strip just beyond the field margin of crops, was introduced in France to increase biodiversity (Ernoult *et al.*, 2013). Results showed that biodiversity of these strips was not higher than that in field margins or hedgegrows but that these areas acted as an animal and arthropod refuge in the agro-ecosystem (Ernoult *et al.*, 2013). A study has also indicated that the biodiversity in agro-ecosystems are dependant on the biodiversity of the surrounding landscapes, especially that of the above ground arthropods (Deuli *et al.*, 1999). This indicates the value of conservation within the agro-landscape which could contribute to the development of more sustainable and less chemical driven agricultural production systems.

The development of pest resistance to insecticides and certain genetically modified (GM) crops (Devos *et al.*, 2013; Naizi *et al.*, 2013) can be seen as an incentive for a "greener" approach to a sustainable agro-ecosystem in which the role of natural arthropod enemies are key. An example of possibilities beyond pesticides and GM crops is that of habitat manipulation (Midega *et al.*, 2008). Midega *et al. (*2008) reported that certain habitat manipulation strategies result in effective push-pull systems, which lead to an increase in spider diversity within maize fields. This habitat manipulation stratgegy included intercropping with Napier grass (*Pennisetum purpureum (*Schumach. 1827*)*) and also a perennial legume (*Desmodium uncinatum (*Jacq.*)*). This is an effective method to increase numbers of predatory spiders to suppress pest populations in maize fields (Midega *et al.*, 2008).

Truter *et al.* (2014), Botha *et al.* (2015), Midega *et al.* (2008) and Bonhof *et al.* (1997) indicated that large numbers of predators occur in maize fields . In Kenya, Midega *et al.* (2008) recorded 78 species of spiders in push-pull systems with desomodium whiel (*Desmodium uncinatum (*Jacq.*)*). Bonhof *et al.* (1997) reported on the importance of spiders and coccinellids as predators of eggs and larval stages of maize stem borers.

3.3 Aim and hypothesis.

Limited information exists on arthropod diversity in and around maize fields in South Africa. The aim of this chapter was to quantify the effect of different disturbance intensities on the species diversity and assemblages of selected predacious arthropods groups in an agro-ecosystem (maize field, field margin and untransformed grasslands).

Hypothesis: If the maize field has the highest intensity of disturbance due to frequent anthropological activities within the agro-ecosystem, the surrounding semi natural landscape is expected to have a higher abundance and diversity of predacious arthropods than the maize field.

3.4 Materials and Methods

Samples were collected in maize farming regions in the North-West province which is situated in the Highveld Grassland biome. Sampling sites were on maize farms near Ventersdorp (S26°27'58.4"; E26°55'45.1"), Potchefstroom (S26°36'34.5"; E27°02'08.6") and Rooipoortjie (15 km SW of Potchefstroom). A total of 62 different localities were sampled over a 3-year period (2013-2015).

Each sample area included a disturbance gradient inside of the maize agro-ecosystem. This gradient went from the highly disturbed maize field to the almost undisturbed untransformed grassland. The intermediately disturbed field margin or crop edge was also included (Figure 3.1). The field margin or crop edge is characterised by an intermediate level of mostly anthropological disturbances such as fences and path ways.

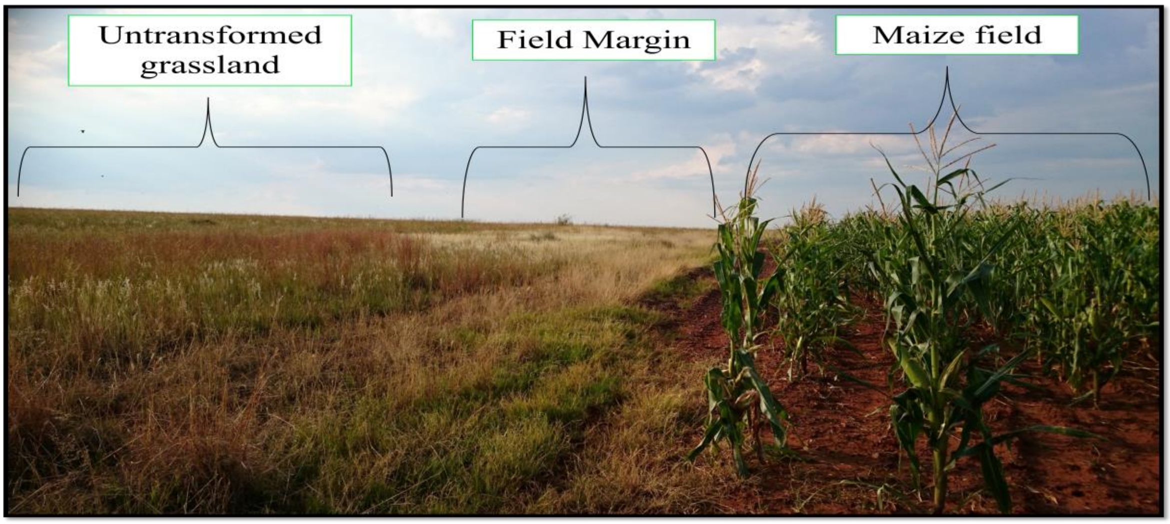

Figure 3.1: Disturbance gradient of the maize agro-ecosystem. Disturbance increasing from left to right.

Sweep net sampling was done on 10x10 m plots. Plots were 50 m apart, and a 50 m between zones (Figure 3.2; Appendix 1). Three (100 m^2) samples were taken in each of the following zones along the disturbance gradient: maize field, field margin and untransformed grassland. Nine samples (three per zone i.e maize field, field margin and untransformed grassland) were therefore taken per locality of which there were 62. A total of 186 samples were collected in each of the three zones (maize field, field margin and untransformed grassland) of the agro-ecosystem. Collectively a total of 558 samples were collected during the three years of sampling, which includes the three zones as well as the three replicates in each zone of the agro-ecosystem.

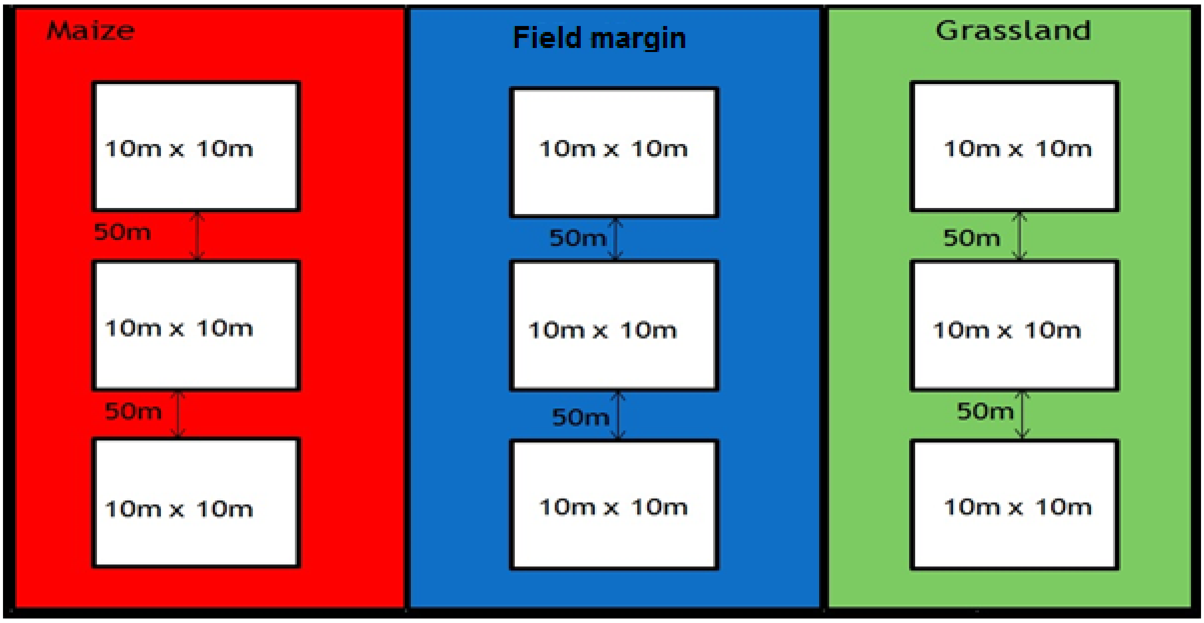

Figure 3.2: Sampling layout for arthropod collections in the three zones at each of the sampling sites.

Each plot was demarcated as a square by means of a pre-measured rope. Each plot in the grassland area was sampled systematically by sweeping from side to side. Inside the maize fields the sweeping was conducted with an up and down motion. Each sample consisted of 30 sweeps, regardless of motion.

The samples were then placed into marked bags and refrigerated before the collected arthropods were identified to species level. For identification purposes each bag was emptied into a bowl with 70 % ethanol, after which all lacewings, praying mantids, coccinellids and spiders of each plot were transferred into separate containers with 70% ethanol. Abundances and species richness were determined for each predator group per sample plot for all the localities and different zones of the agro-ecosystem. Identification of morpho-species was done in such a manner that a specific morpho-species would be recognisable over all study plots. A template of specimen was created and used to make sure that for example morpho-species 1 was the same specimen throughout all study plots. All specimen were then identified to species level at the Biosystematic Division of the Agricultural Research Council (ARC) in Pretoria, Gauteng. Mantid species were identified by using the reference collection at the ARC as well as a more recently updated collection at the Ditsong Museum in Pretoria.

3.5 Data Analyses

The data did not exhibit normality/normal distribution due to variation in and size of the predator populations in the different agro-ecosystem zones, and did not display constant variance. Shapiro-Wilk and Levene's tests, as well as Q-Q and Box-plots confirmed that data did not satisfy the assumptions required to analyse variance.

Linear mixed models (LMM), statistical modelling that uses Bayesian inference, is employed when data is not distributed normally, including data generated by counts, allowing for a variety of correlation patterns to be modelled. Population numbers of one zone were related to the population numbers of other zones of the agro-ecosystem, as individuals may switch between zones due to hunting or feeding strategies. Data were also collected from fixed points over time. This created a dependence of data (covariance), which is corrected for with LMM. Multiple measurements per plot per zone were performed over three seasons, i.e. sampling was replicated over time (2013-2015) and across populations and therefore correlated data arose in the statistical analyses.

Significant differences between population means were acquired by applying pairwise comparisons on estimated marginal means (EMM). EMM represent the transformed means of populations/samples of unequal size, enabling the comparison of sample means across different treatments, revealing significant differences between means. EMM represent a statistical prediction of population means and are values derived from linear mixed models applied to data with a non-normal distribution. By employing Sidak adjustments, multiple population and predator group means could be accurately compared and differences across habitats (maize field, field margin, untransformed grassland) determined. The effect size was also determined by using the LMM's and is independent of sample size and is a measure of practical significance. A minimal practical significance (practical implication) is $d > 0.2$, for a significant effect size $d = 0.5$, and a large practical significance is when $d > 0.8$ (Ellis and Steyn, 2003). IBM SPSS version 21 was used for all statistical analyses.

Mean abundance graphs as well as box plots indicating the variation within the data were compiled using Statistica software version 12 (Statsoft Inc. 2013). Abundance data was used to determine the diversity of each of the predator groups. Analysis considered the Shannon-Wiener Diversity Index, Margalef Species Richness Index, Pielou's Evenness and also Simpson’s Evenness and was done with the use of the Primer 6 Software (Clarke and Gorley, 2006). Species composition and turnover (beta diversity) were visualised with NMDS (Non-Metric Dimensional Scaling) graphs in Primer. Similarity Percentage (SIMPER) analysis to indicate species responsible for grouping within the data as indicated by the NMDS graphs. These analyses were performed in PAST version 2.15 which was released in 2012 (Hammer *et al.,* 2001).

3.6 Results

3.6.1 Overall predator diversity

The overall abundance of predacious arthropods in the agro-ecosystem over three years was as follows: 1002 spiders (41 species), 561 lacewings (2 species), 980 coccinellids (8 species) and 589 praying mantids (8 species). The numbers of individuals of each predator group collected in the different zones are provided in Figure 3.3.

The maize field was the least diverse and included only 21 spider species in comparison to the surrounding landscape which had 36 species. The praying mantids in the maize field consisted of only four species as opposed to the eight found in both the field margin and untransformed grasslands.

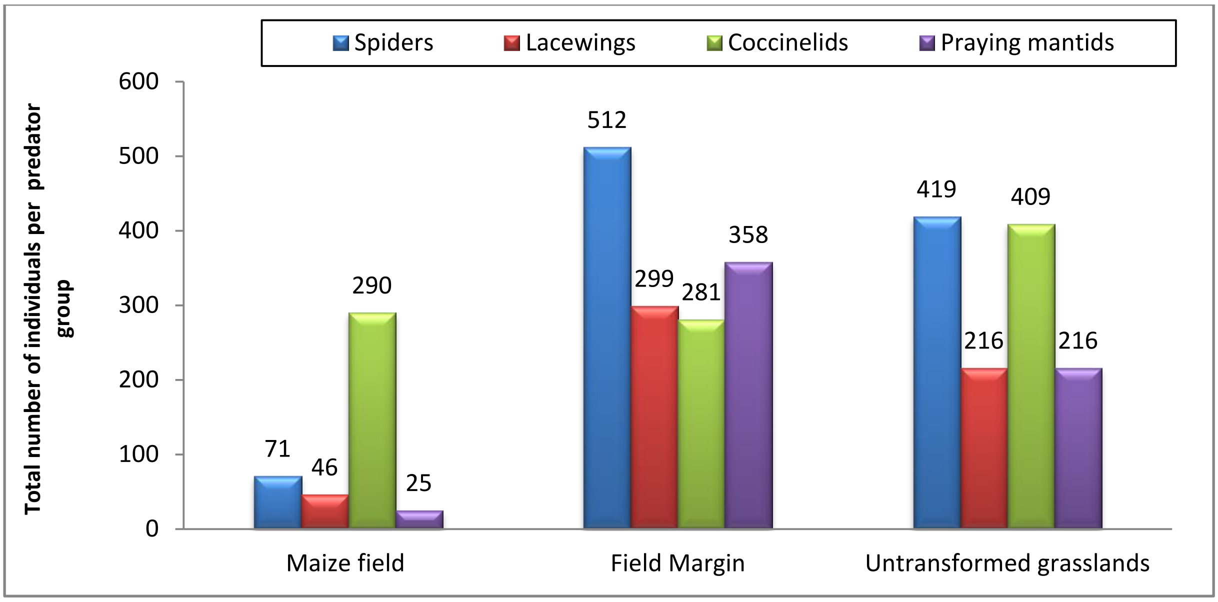

Figure 3.3: Predacious arthropod abundance along a disturbance gradient from inside the maize field to the untransformed grassland.

3.6.2 Predator community composition and species assemblages

Species assemblages differed between maize fields and surrounding landscape (Figure 3.4). The field margin and untransformed grassland however did not differ with regards to the species assemblage of the four predator groups.

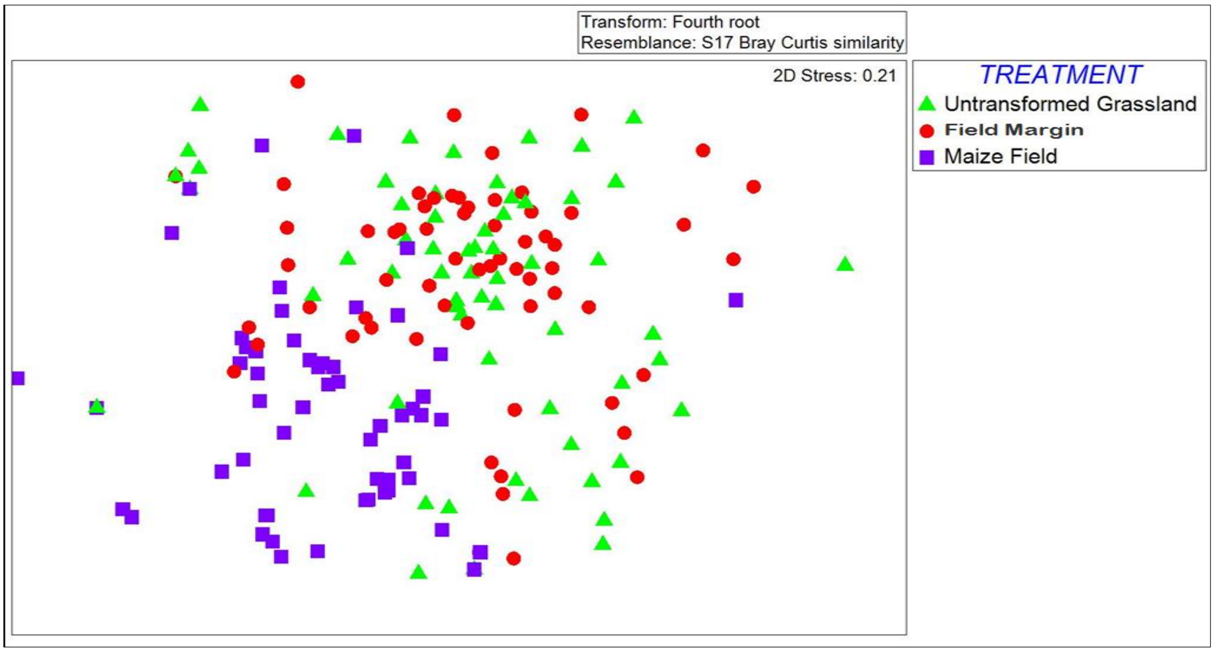

Figure 3.4: Species assemblage of predacious arthropods collected inside maize fields, field margins and untransformed grasslands.

The SIMPER analysis indicated that the species responsible for the dissimilarity between the maize fields and the surrounding landscapes (overall dissimilarity of 86.59%) were the lacewing *Chrysoperla congrua* (14.72%), the coccinellids *Cheilomenes lunata* (11.23%) and *Hippodamia variegata* (10.45%) and the spider species *Misumenops rubrodecoratus* (9.45%). These species were responsible collectively for 46% of the overall dissimilarity between the maize field and the surrounding vegetation.

3.6.3 Overall predator abundance and diversity

Results indicated that there was a significant difference between the different zones of the agro-ecosystem with regards to the Species Richness and the Margalef Species Richness Index. The remaining diversity indices did not differ significantly overall with regards to predacious arthropods in the maize agro-ecosystem (Table 3.1).

The LMM analysis indicated that the significant differences with regards to the diversity were as a result of the maize field which is lower in predator diversity. There were no differences between the surrounding landscapes i.e. field margin and untransformed grassland with regards to the selected predator arthropod diversity (Table 3.1).

There was no high practical significance, therefore no high practical implications, for any of the treatments, the medium implications were however detected for the species richness, Shannon-Wiener Diversity index and the Margalef Species Richness Index for the treatments between the maize field and the two surrounding agricultural zones for the predators collected during this study (Table 3.1).

Table 3.1: Linear Mixed Model analysis for differences and practical significance in arthropod richness and diversity index values between different zones within agro-ecosystem.

Zones	**Species richness**	**Abundance**	**Shannon-Wiener Diversity Index**	**Margalef Species Richness Index**	**Pielou's Evenness**	**Simpson's Diversity Index**
Field margin x Maize field	0.025*(b)	0.344(a)	0.110(b)	0.034*(b)	0.985(a)	0.638(a)

Field margin x Untransformed grasslands	0.990 (a)	0.988(a)	0.993(a)	0.999(a)	0.812(a)	0.934(a)
Maize field x Untransformed grasslands	0.049*(b)	0.526(a)	0.063(b)	0.047*(b)	0.610(a)	0.319(a)
P-value	0.015*	0.274	0.041*	0.180	0.520	0.281

P-values (P> 0.05) of statistical significance indicated by (*). Effect size and level of practical impact indicated as >0.2 = a (low), >0.5=b (medium), >0.8=c (high).

3.6.4 Individual predator groups in the agro-ecosystem

3.6.4.1 Lacewings

A significant difference in the abundance of lacewings between the maize field and the untransformed grassland was observed (P=0.004) but not between the abundance of lacewings in the field margin and untransformed grasslands (P=0.074) (Figure 3.5). The effect size was medium for the practical significance between the maize field and the field margin (>0.5) but was much higher for the maize field and the untransformed grassland (>0.8). This indicates that lacewings prefer the untransformed grasslands above the maize field itself, but that they are also present in the field margin in significantly greater abundance than in maize fields.

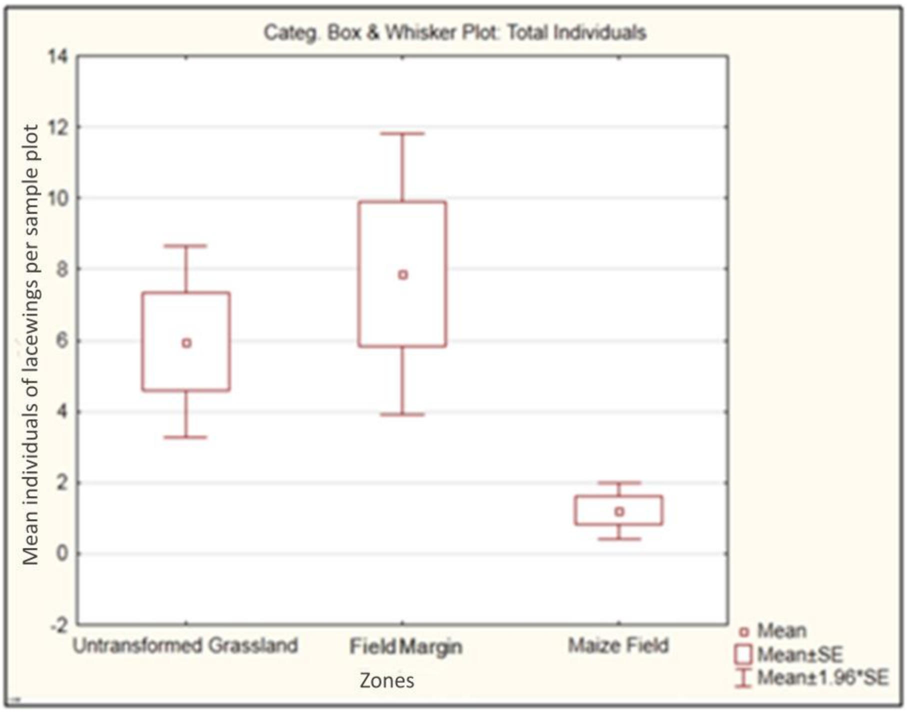

Figure 3.5: Lacewing abundance in the agro-ecosystem during three seasons of collections.

3.6.4.2 Coccinellids

The difference detected with regards to the coccinellids was in terms of species richness (P=0.013) and the Shannon-Wiener Diversity index (P=0.043).

The differences for these two indices were between the maize field and the field margin as well as the maize field and the untransformed grassland (Figure 3.6).

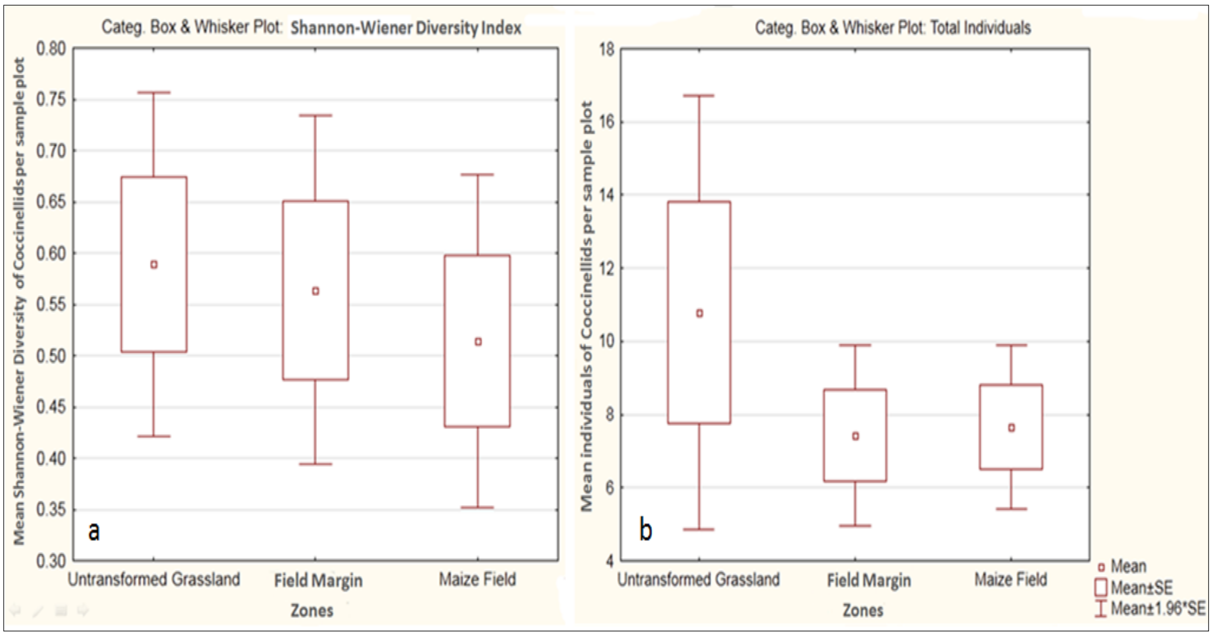

Figure 3.6: (a) Mean Shannon-Wiener diversity per sample plot for the Coccinellids within the different zones of the agro-ecosystem, (b) indicates the mean abundance of the Coccinellids per sample plot.

The untransformed grasslands had the highest species richness and diversity (Figure 3.6). The practical implications related to the effect size for these differences was the highest for the species richness (>0.8) between the maize field and field margin.

3.6.4.3 Spiders

Spiders were most abundant and diverse within the field margin in the agro-ecosystem (Figure 3.7).

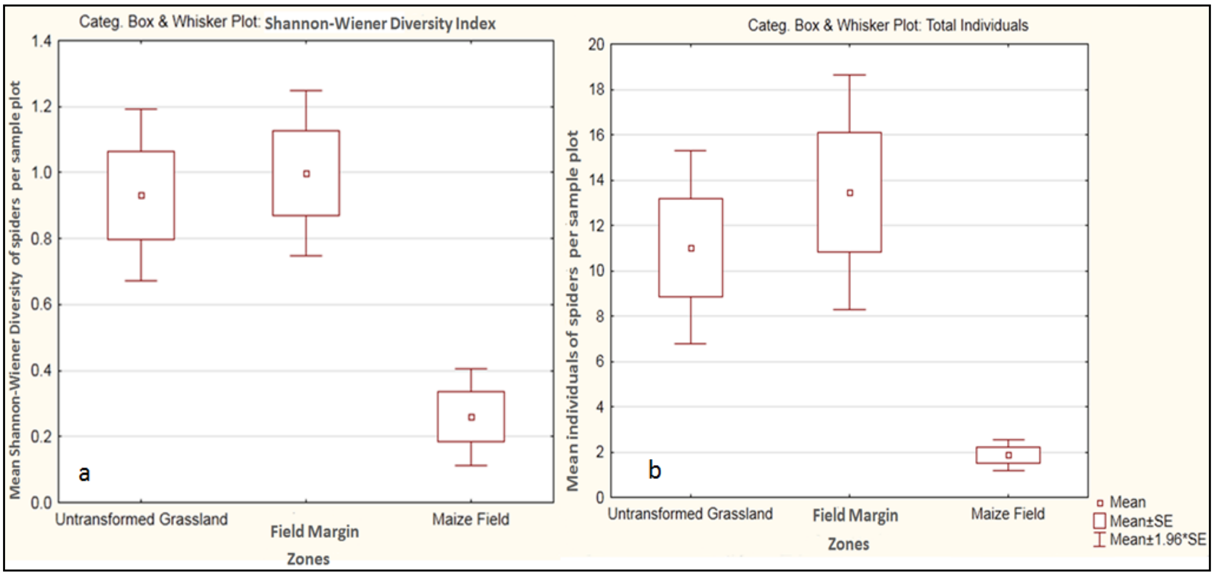

Figure 3.7: (a) Mean Shannon-Wiener diversity per sample plot for the spiders within the different zones of the agro-ecosystem, (b) indicates the mean abundance of the spiders per sample plot.

Statistically significant differences were observed for all the criteria of diversity and abundance with regards to the spiders in the agro-ecosystem (Table 3.2). The differences were due to the differences between the maize field and the field margin and the maize field and untransformed grasslands (Table 3.2). All indices indicated that the effect size between the maize field and field margin was larger than that of the maize field and the untransformed grasslands and thus has a higher practical implication. Pielou's Evenness and Simpsons Diversity index did suggest high practical significance for the maize field and field margin as well as the maize field and untransformed grasslands (Table 3.2).

Table 3.2: Linear Mixed Model analysis for differences and practical significance in spider abundance and diversity indices between the different zones within the agro-ecosystem.

Statistical Analysis	**Species richness**	**Abundance**	**Shannon-Wiener Diversity Index**	**Margalef Species Richness Index**	**Pielou's Evenness**	**Simpson's Diversity Index**
Field margin x Maize field	0.000*(c)	0.001*(c)	0.002*(c)	0.005*(c)	0.011*(c)	0.009*(c)
Maize field x Untransformed grasslands	0.035*(b)	0.149(b)	0.031*(b)	0.018*(b)	0.006*(c)	0.008*(c)

P-value	0.001*	0.001*	0.002*	0.003*	0.003*	0.003*

P-values (P> 0.05) of statistical significance indicated by (*). Effect size and level of practical impact indicated as >0.2 = a (low), >0.5=b (medium), >0.8=c (high).

3.6.4.4 Praying mantids

Significant differences were observed for the Species richness and Shannon-Wiener Diversity Index (Table 3.3). The differences were between the maize field and field margin. The effect sizes differed as the species richness had a high practical significance whilst the Shannon-Wiener Diversity Index only had a medium practical significance (Table 3.3).

Table 3.3: Linear Mixed Model analysis for differences and practical significance in praying mantids abundance and diversity indices between the different zones within the agro-ecosystem.

Statistical Analysis	**Species richness**	**Abundance**	**Shannon-Wiener Diversity Index**	**Margalef Species Richness Index**	**Pielou's Evenness**	**Simpson's Diversity Index**
Field margin x Maize field	0.011*(c)	0.077(b)	0.038*(b)	0.065(b)	0.151(b)	0.080(b)
Maize field x Untransformed grasslands	0.228(a)	0.284(b)	0.640(a)	0.491(a)	0.900(a)	0.694(a)
P-value	0.013*	0.071	0.043*	0.071	0.139	0.085

P-values (P> 0.05) of statistical significance indicated by (*). Effect size and level of practical impact indicated as >0.2 = a (low), >0.5=b (medium), >0.8=c (high).

Based on these results it seems that the field margin has higher species richness but the abundance was not definitive between the semi-natural landscape surrounding the maize field, in which the abundance was lower (Figure 3.8).

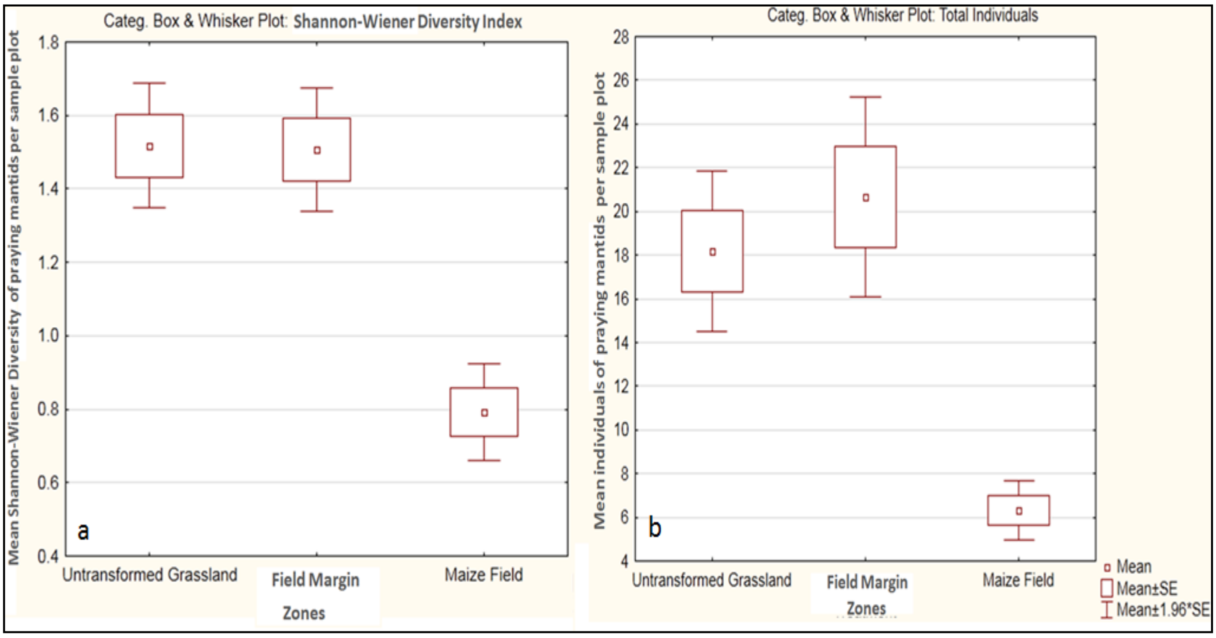

Figure 3.8: (a) Mean Shannon-Wiener diversity per sample plot for the praying mantids within the different zones of the agro-ecosystem, (b) indicates the mean abundance of the praying mantids per sample plot.

It should be noted that the dominating species in the maize field and the untransformed grassland was *Galepsus* sp. (Mantodea: Mantidae) while the dominating species in the field margin was *Episcopus chalybeus* (Mantodea: Mantidae).

3.7 Discussion

3.7.1 Overall predator diversity and species assemblage in the maize agro-ecosystem

The high abundance of predators in the agro-ecosystem indicated that they may play an important role as biological control agents. Native predator species and landscape composition are key aspects in biological control (Grez *et al.*, 2013). In a system with high abundance where native coccinellid species are active, aphids have been reported to be better controlled than in systems where predator abundance was low with fewer native coccinellid species (Grez *et al.*, 2013).

Due to the difference in species assemblage between the maize field and the adjacent agricultural landscape it can be deduced that maize fields have unique species assemblages. This was expected since the vegetation type is quite different from the adjacent fields. A recent study indicated that arthropod assemblages are primarily depended on vegetation structure, instead of surrounding matrices (Torma *et al.*, 2014). Spider's assemblages were found to be influenced by the landscape surrounding agricultural activities in the sandy grasslands of Hungary (Torma *et al.*, 2014).

Landscape composition has an effect on the abundance of pest species as well as their natural enemies. Agricultural areas with larger surrounding semi-natural areas such as field margins and untransformed grasslands have lower pest abundance due to the pest control services provided by natural enemies of pests (Veres *et al.*, 2013). Management practices as well as vegetation structure (Cilliers and Bredenkamp, 1999; Debano, 2006) and diversity could be the reason that the maize fields in this study have a lower predator presence. However, it is difficult to develop beneficial landscape management strategies since different species and species groups require different vegetation types or land cover types (Fahrig *et al.*, 2015). Heterogeneity within agricultural landscapes contributes to biodiversity and it can be expected that monocultures decrease diversity of predator arthropods as was observed in this study. However, it has also been indicated that crop field size spatially relates to internal habitat which can also influences the biodiversity of agro-ecosystems (Fahrig *et al.*, 2015).

3.7.2. Predator diversity and abundance in the maize agro-ecosystem

The significant difference found for lacewings in the agro-ecosystem indicated that the highest difference was for their abundance between the maize field and the untransformed grassland. Lacewings were thus more abundant in the untransformed grasslands than the field margin or the maize field, yet their abundance was still higher for the field margin than the maize field which was the least abundant habitat in the maize agro-ecosystem. This is a common phenomenon since lacewings are known for invading crops from surrounding areas (Stelz and Devetak, 1999). The species composition of Neuroptera in crops is depended on the species that are found within the surrounding agricultural landscape matrix. The reason for their preferred method of reinvasion of crops is due to the ability of the crop field to drastically change through human activities such as pesticide application and harvesting, which changes crops from a habitat with abundant resources into an essential desert with limited resources (Stelz and Devetak, 1999).

The coccinellids had the highest significant difference with regards to species richness between the field margin and the maize field (P=0.013) Therefore it can be deduced that coccinellids seem to be more diverse in the field margin but are abundant in the crops as well as in the untransformed grasslands. Coccinellids are quite mobile between different zones of agro-ecosystems and are influenced by pesticides, yet they have been known to suppress early season aphid densities (Obrycki *et al.*, 2009). The presence of coccinellids inside maize fields as well as the surrounding vegetation suggests that they are not as sensitive to disturbances as the other predator groups evaluated in this study, making them ideal for biocontrol systems.

Spiders were more abundant and diverse in the field margin than in the maize field. The highest significance was for the species richness between the field margin and the maize field (P=0.000), while the lowest significance, with regards to the species richness was between the untransformed grassland and the maize field (P=0.035). Therefore they seem to be, to a lesser extent, diverse and abundant in the untransformed grasslands. Habitat management has in the past been used as a strategy to utilize spiders as bio-control agents. An increase in population size of spiders in cereal fields was noted when a small area of grassland was incorporated into the landscape of a cereal agro-ecosystem (Landis *et al.*, 2000). Grasslands surrounding maize fields might offer a better habitat structure than maize fields themselves. Habitat manipulation such as digging holes (5-10 cm in diameter) in crop fields has been reported to increase spider density in crop fields, indicating their ability to adapt if habitat structures are favourable (Alderweireldt, 1994).

Visual observation of spider diversity in agro-ecosystems of India indicated that the dominating species were either web building spiders (orb and funnel web) or jumping spiders (Wilson *et al.*, 2014). In the study done by Wilson *et al*. (2014) a total of 1338 individuals comprising of 14 species were observed and the latter indicated that crop age has a significant influence on spider diversity and abundance, with crops being between 50-60 days old had higher spider presence than other crop ages. The dominating species of the above mentioned study is not in accordance with this study. The dominating species in maize fields of this study was a cobweb building spider (Theridiidae: *Enoplognatha molesta*) and a free living plant dweller (Thomisidae: *Misumenops rubrodecoratus*) and had very low abundances of 17 and 29 individuals respectively. This could indicate the importance of natural to semi natural vegetation that surrounds crops especially in the grasslands of South Africa. However it could also show the importance of different data collection methods.

The highest significant difference of the praying mantids resulted from the species richness (P=0.011) between maize fields and the field margin. The lowest significant difference was found between the maize field and the field margin with regards to the Shannon-Wiener Diversity Index (P=0.640). Praying mantids had higher species richness in the field margin which could indicate that the vegetation structure was more suitable for the different species of praying mantids in this study. The fact that two species dominated the mantid community structure in this study could be an indication that the field margin had a different vegetation structure which could influence the mantids. It was noted in a study that vegetation height influences hunting capabilities of praying mantid species (Ilyse *et al.*, 1983). This might be a factor which influenced the praying mantids, resulting in two different dominating species in the semi-natural habitats. This is however contradicted by a study that indicated that plant diversity and composition did not differ significantly between the field margin and untransformed grasslands (Botha *et al.,* 2015). Accordingly it could perhaps rather be an effect of inter-species competition that gives rise to the differences in species assemblage between different zones (Dugatkin, 2009). Praying mantids have been noted to prey on cutworm larvae (*Spodoptera litura*) (Fabricius, 1775), a pest in maize and other crops (Carasi *et al.*, 2014).This could indicate that different mantid species may have different prey preferences. The mantid species in the cutworm study was not identified to species level. Praying mantids are also known to have patch distributions which is dictated by the micro-climates (Holwell *et al.*, 2007), thus one species might have a different tolerance than another which could have contributed to the two different dominating species found in this study's results.

Ultimately it seems that the practice of mono-culture cropping decreases predator diversity and abundance in maize fields since the surrounding vegetation in the agro-ecosystem is more diverse and abundant with regard to the above mentioned predator species. This is in contrast to a previous study that found a high diversity of arthropods in genetically modified maize as well as non-Bt maize (Truter *et al.*, 2014), which could indicates that there is no restriction with regards to possible food resources for the predators in agro-ecosystems.

The findings of this study corroborates a recent study done on the ant communities of Colombia, in which different cropping systems (maize, soy and rice) where investigated as well as other disturbances such as oil palm plantations and rubber plantations (Sanabria *et al.*, 2014). Ant communities, abundance and richness were negatively influenced most, by the activities of agriculture (Sanabria *et al.*, 2014). It was suggested that this is due to the frequency of disturbances and vegetation types that are found in these agro-ecosystems (Sanabria *et al.*, 2014).

Different spider species have also indicated to have different responses towards the same organic cereal crop fields (Birkhofer *et al.*, 2014). It was found that spider species that live under stones or in soil crevices were more abundant in grassy field margins than other spider species that dominated in the organic cereal fields (Birkhofer *et al.*, 2014). This indicates the importance of habitat variability and also functional distinctness of species. Different species require different environmental elements to survive.

Arthropod diversity and abundance was investigated in field margins surrounding maize fields in Portugal and indicated distinct communities between spring and autumn (Simao *et al.*, 2015). The diversity and abundance were higher during spring which consisted mostly of Hymenoptera and Coleoptera species. However the autumn sampling was also diverse with regards to the ground dwelling arthropods (Simao *et al.*, 2015). This indicates the importance of a vegetation refuge in proximity to maize fields during different seasons.

Although it was not determined in this study, vegetation cover, especially the diversity thereof, seems to have a larger effect on predacious arthropod species than the different disturbance intensities found in the maize agro-ecosystem. A study has indicated that an increase in plant diversity does result in an increase in arthropod diversity, however, the herbivorous arthropods are more strongly correlated with the diversity of predacious and parasitic arthropods than with plant diversity (Siemann *et al.,* 1998). However, the disturbance intensities found in agro-ecosystems could have an influence on the plant species found in the different parts of the agro-ecosystem. An example of this phenomenon was reported by the study of bush encroachment due to grazing done by Blaum *et al. (*2009). This study indicated that ground dwelling arthropods specifically, ants and dung beetles, increased in diversity and abundance as the density of scrubs increased (Blaum *et al.,* 2009). Wan *et al. (*2014) reported that when the ground was covered with the herbaceous perennial plant *Trifolium repens* (L.) in peach orchards, the beneficial arthropod abundance and species richness increased by 107%, thus indicating the importance of vegetation cover to arthropod species. The implementation of no tillage and ground cover such as mulch is a part of a new strategy to increase biodiversity of soil micro-organisms as well as arthropods in agriculture to increase the sustainability of agro-ecosystem and is known as conservation agriculture (Dumanski, 2006).

Predatory arthropods are subjected to the natural enemy hypothesis that dictates that predators will be more abundant in areas where prey is most abundant (Finch and Collier, 2000). However, this study indicated that although it is thought that prey (also defined as pests in cropping systems) is abundant in maize fields, if the vegetation structure and diversity is not sufficient, predatory arthropods might not follow the abundance of prey. Coccinellids; however seem to be the exception to this theory.

3.8 Conclusions

Maize fields have a lower diversity and abundance of predators as well as a different species assemblage than that of the surrounding landscape. This decreased diversity could be ascribed to increased anthropological activity but more likely to the lack of different types of vegetation cover.

The hypothesis of this chapter stated that: If the maize field has the highest intensity of disturbance due to frequent anthropological activities within the agro-ecosystem, the surrounding semi-natural landscape is expected to have a higher abundance and diversity of predacious arthropods than the maize field and as thus this hypothesis is accepted.

The field margin, which had a higher disturbance than the untransformed grassland, did have an abundance of the selected predator species. The predatory species in this study however did not distinguish between these two surrounding habitat types. The field margin tended to have a higher diversity of spiders, coccinellids and praying mantids while the lacewings were more abundant in the untransformed grasslands. This study indicated that each species reacts differently to disturbances in their environment and that they adapt accordingly.

3.9 Reference list

ALDERWEIRELDT, M. 1994. Habitat manipulations increasing spider densities in agroecosystems: possibilities for biological control? *Journal of Applied Entomology*, 118:10-16.

BIRKHOFER, K., WOLTERS, V. & DIEKOTTER, T. 2014. Grassy margins along organically managed cereal fields foster trait diversity and taxonomic distinctness of arthropod communities. *Insect Conservation and Diversity*, 7:274-287.

BLAUM, N., SEYMOUR, C., ROSSMANITH, E., SCHWAGER, M. & JELTSCH, F. 2009. Changes in arthropod diversity along a land use driven gradient of shrub cover in savanna rangelands: Identification of suitable indicators. *Biodiversity and Conservation*, 18:1187-1199.

BONHOF, M.J, OVERHOLT, W.A, VAN HUIS, A. & POLASZEK, A. 1997. Natural enemies of cereal stemborers in East Africa: A review. *Insect Science Application*, 17(1):19-35.

BOTHA, M., SIEBERT, S.J., VAN DEN BERG, J., MALIBA, B.G. & ELLIS, S.M. 2015. Plant and arthropod diversity patterns of maize agro-ecosystems in two grassy biomes of South Africa. *Biodiversity and Conservation,* 24:1791-1824.

CARASI, R.C., TELAN, I.F. & PERA, B.V. 2014. Bioecology of common cutworm (*S. Litura*) of Mulberry. *International Journal of Scientific and Research Publications*, 4(4):1-8.

CILLIERS, S.S. & BREDENKAMP, G.J. 1999. Ruderal and degraded natural vegetation on vacant lots in the Potchefstroom Municipal Area, North West Province, South Africa. *South African Journal of Botany*, 65(2):163-173.

CLARKE, K.R., GORLEY, R.N., 2006. PRIMER v6: User Manual/Tutorial. PRIMER-E, Plymouth, 192pp.

DEBANO, S.J. 2006. Effects of livestock grazing on aboveground insect communities in semi-arid grasslands of southeastern Arizona. *Biodiversity and Conservation*, 15:2547-2564.

DEULI, P., OBRIST, M.K. & SCHMATZ, D.R. 1999. Biodiversity evaluation in agricultural landscapes: above-ground insects. Agriculture, Ecosystems and Environment, 74:33-64..

DEVOS, Y., MEIHLS, L. N., KISS., J. & HIBBARD, B.E. 2013. Resistance evolution to the first generation of genetically modified Diabrotica-active Bt-maize events by western corn rootworm: management and monitoring considerations. *Transgenic Resistance*, 22:269-299.

DIPPENAAR-SCHOEMAN, A. 2014. Field guide to the spiders of South Africa. Pretoria: LAPA Publishers (Pty) Ltd.

DUGATKIN, L.A. 2009. Principles of Animal Behaviour. New York: W.W Norton & Company.

DUMANSKI, J., R. PEIRETTI, J. BENETIS, D. MCGARRY, & C. PIERI. 2006. The paradigm of conservation agriculture. *Proceedings of World Association of Soil and Water Conservation*, 1(7):58-64.

ELLIS, S.M & STEYN, H.S. 2003. Practical significance (effect sizes) versus or in combination with statistical significance (p-values). *Management Dynamics*, 12(4):51-53.

ERNOULT, A., VIALETTE, A., BUTET, A., MICHEL, N., RANTIER, Y., JAMBON, O. & BUREL, F. 2013. Grassy strips in their landscape context, their role as new habitat for biodiversity. *Agriculture, Ecosystems and Environment*, 166:15-27.

FAHRIG, L., GIRARD, J., DURO, D., PASHER, J., SMITH, A., JAVOREK, S., KING, D., LINDSAY, K. F., MITCHELL, S. & TISCHENDORF, L. 2015. Farmlands with smaller crop fields have higher within-field biodiversity. *Agriculture, Ecosystems and Environment*, 200:219-234.

FINCH, S. & COLLIER, R.H. 2000. Host-plant selection by insects - a theory based on 'appropriate/inappropriate landings' by pest insects of cruciferous plants. *Entomologia Experimentalis et Applicata* 96: 91-102.

GREZ, A.A., ZAVIEZO, T. & GARDINER, M.M. 2013. Local predator composition and landscape affects biological control of aphids in alfalfa fields. *Biological Control*, 76:1-9.

HAMMER, Ø., HARPER, D.A.T. & RYAN, P.D. 2001. PAST: Paleontological statistics software package for education and data analysis. *Paleontologia Electronica,* 4(1):9.

HOLWELL, G.I., BARRY, K.L. & HERBERSTEIN, M.E. 2007. Mate location, antennal morphology, and ecology in two praying mantids (Insecta: Mantodea). *Biological Journal of the Linnean Society*, 91:307-313.

ILYSE, H., RATHET, L. & HURD, L.E. 1983. Ecological relationships of three co-occurring mantids, *Tenodera sinensis* (Saussure), *T.angustipennis* (Saussure), and *Mantis religiosa* (Linnaeus). *American Midland Naturalist*, 110(2):240-248.

KOIJ, S., KHAN, Z.R. & MIDEGA, C.A.O. 2007. Field boundaries of *Panicum maximum* as a reservoir for predators and a sink for *Chilo partellus*. *Journal of Applied Entomology*, 131(3):186-196.

LANDIS, D.A., WRATTEN, S.D. & GURR, G.M. 2000. Habitat management to conserve natural enemies of arthropod pests in agriculture. *Annual Review of Entomology*, 45:175-200.

LUNDREN, J.G. & WIEDENMANN, R.N. 2004. Nutritional suitability of corn pollen for the predator *Coleomegilla maculata* (Coleoptera: Coccinellidae). *Journal of Insect Physiology*, 50:567-575.

MCLAUGLIN, A. & MINEAU, P. 1995. The impact of agricultural practices on biodiversity. *Agriculture, Ecosystems and Environment*, 55:201-212.

MIDEGA, C.A.O., KHAN, Z.R., VAN DEN BERG, J., OGOL, C.K.P.O., DIPPENAAR-SCHOEMAN, A.S., PICKETT, J.A. & WADHAMS, L.J. 2008. Response of ground-dwelling arthropods to a 'push–pull' habitat management system: spiders as an indicator group. *Journal of Applied Entomology*, 132:248-254.

NAIZI, I.A.K., RAFIQUE, A., RAUF, S., DA SILVA, J.A.T. & AFZAL, M. 2013. Simultaneous selection for stem borer resistance and forage related traits in maize (*Zea mays* ssp. mays L.) x teosinte (*Zea mays* ssp.mexicana L.) derived populations. *Crop Protection*, 57:27-34.

OBRYCKI, J.J., HARWOOD, J.D., KRING, T.J. & O'NIEL, R.J. 2009. Aphidophagy by Coccinellidae: Application of biological control in agroecosystems. *Biological Control*, 51:244-254.

PILCHER, C.D., OBRYCKI, J.J., RICE, M.E. & LEWIS, L.C. 1997. Preimaginal development, survival, and field abundance of insect predators on transgenic *Bacillus thuringiensis* corn. *Environmental Entomology*, 26:446-454.

PIMENTEL, D., STACHOW, U., TAKACS, D.A., BRUBAKER, H.W., DUMAS, A.R., MEANEY, J.J., O'NEIL, J.A.S., ONSI, D.E. & CORZILIUS, D.B. 1992. Conserving Biological Diversity in Agricultural/Forestry Systems. *BioScience*, 42(5):354-362.

SANABRIA, C., LAVELLE, P. & FONTE, S. J. 2014. Ants as indicators of soil-based ecosystem services in agroecosystems of the Colombian Llanos. *Applied Soil Ecology*, 84:24-30.

SIEMANN, E., TILMAN, D., HAARSTAD, J. & RITCHIE, M. 1998. Experimental tests of the dependence of arthropod diversity on plant diversity. *The American Naturalist*, 152(5):738-750.

SIMAO, F.C.P., CARRETERO, M. A., DO AMARAL, M.J.A, SOARES, A.M.V.M. & MATEOS, E. 2015. Composition and seasonal variation of epigeic arthropods in field margins of NW Portugal. *Turkish Journal of Zoology*, 39:404-411.

STATSOFT INC. 2013. Statistica, version 11. Tulsa: StatSoft.

STELZ, M. & DEVETAK, D. 1999. Neuroptera in agricultural ecosystems. *Agriculture, Ecosystem and Environment*, 74:305-321.

TORMA, A., GALLE, R. & BOZSO, M. 2014. Effects of habitat and landscape characteristics on the arthropod assemblages (Araneae, Orthoptera, Heteroptera) of sand grassland remnants in Southern Hungary. *Agriculture, Ecosystems and Environment*, 196:42-50.

TRUTER, J., VAN HAMBURG, H. & VAN DEN BERG, J. 2014. Comparative diversity of arthropods on Bt maize and non Bt maize in two different cropping systems in South Africa. *Environmental Entomology*, 43(1):197-208.

VERES, A., PETIT, S., CONORD, C. & LAVIGNE, C. 2013. Does landscape composition affect pest abundance and their control by natural enemies? A review. *Agriculture, Ecosystems and Environment,* 166:110-117.

WAN, N., GU, X., JI, X., JIANG, J., WU, J. & LI, B. 2014. Ecological engineering of ground cover vegetation enhances the diversity and stability of peach orchard canopy arthropod communities. *Ecological Engineering*, 70:157-182.

WILSON, J.J., RAMAKRISHNAM, P.M. & SEVARKODIYONE, S.P. 2014. A study on the age-wise and seasonal distribution of spider fauna in the paddy and maize field. (Athikulam, Virudhunagar District, Tamil Nadu, India). *World Journal of Zoology*, 9(1):13-18.

4. CHAPTER 4: EFFECT OF AGRICULTURE AND URBANIZATION ON PREDACIOUS ARTHROPOD DIVERSITY OF HIGHVELD GRASSLANDS

4.1 Abstract

Biodiversity provides vital ecosystem services and more diverse ecosystems are known to be more stable and resilient in the face of disturbance. Predacious arthropods provide a valuable ecosystem service to control pest numbers and we hypothesised that anthropological activities, which result in land-use change and habitat fragmentation, deplete this species pool. This study generated baseline data to monitor the effects of anthropological activities on predator diversity and abundance in urban- and agro-ecosystems. Our aim was to quantify the changes in predacious arthropod species richness and diversity across disturbance gradients with the use of various diversity indices. We examined the effect of urbanization and agriculture on the diversity patterns of Chrysopidae, Mantodea, Aranea and Coccinellidiae. Study sites were located in the Grassland Biome and included agricultural (maize field-field margin-untransformed grassland) and urban (ruderal-fragmented-untransformed grassland) gradients. Our results indicated that urban areas had a similar species richness of arthropods than agro-ecosystems but predator species were more abundant in urban areas. With an increase in distance away from disturbance, in both the agricultural and urban environments, an increase was noticed in diversity and abundance. The maize fields, however, are where predators were the least diverse and abundant. Analyses also suggested that urban sprawl is not likely to deplete the species pool as overall predator diversity increases with increasing size of the urban area. The economic state of urban areas seems to have a larger influence on predator abundance and richness than previously thought.

4.2 Introduction

The more biological diverse an ecosystem is the more stable and resilient it will be in the face of disturbance (Begon *et al*. 2006). Calcagno *et al*. (2011) stated that “although species may appear functionally redundant when one function is considered under one set of environmental conditions, many species are needed to maintain multiple functions at multiple times and places in a changing world”. The way in which the world changes is mostly due to some form of anthropological activities that causes disturbances which results in declining biodiversity. Such activities can be anything from climate change, habitat destruction to agriculture and urbanization (Miller and Spoolman, 2012). These activities change the natural state of the environment in such a way that it influences ecosystems and their functions. Deutsch *et al*. (2005) stated that the only way towards a sustainable future is by the conservation of biodiversity.

4.2.1 Agriculture as a disturbance

South Africa considers agriculture as an important economic asset in the country's strife towards development (DAFF, 1998). Farming is essentially a business where the aim is to get maximum yields from minimum inputs. Crop pests threaten the maximum yield a farmer can expect to achieve since pesticides need to be applied to mitigate the effects of these arthropods (Collins and Qualset, 1999). Numerous studies have illustrated the effect of agricultural land-use on species richness/diversity to be positive. According to studies done by Burel *et al*. (1998), the intensification of agriculture does not always lead to a decrease in species richness. Another concern with regards to agricultural activities is regarding the unknown adverse effect of genetically modified maize on non-target arthropods especially that of natural enemies (Pilcher, 1997; Lundren and Wiedenmann, 2002; Lundren and Wiedenmann, 2004).

Agriculture not only introduces pesticides, herbicide and genetically modified monocultures into the environment it also causes topsoil erosion, salinization, aquifer depletion, loss of genetic diversity of wild crop strains and land degradation (Miller and Spoolman, 2012). Pesticides not only endanger biota of the agro-ecosystem, but the runoff thereof has been known to cause adverse effects such as decreasing fish populations and contaminating drinking water with its residues (Miller and Spoolman, 2012).

Furthermore, it has been stated that agricultural practices are a threat to the diversity and abundance of arthropods and could possibly induce a bottom-up effect into the thropic structure of the food chain in intensively cultivated areas (Marshall and Thomas, 1999). This suggests that agricultural practices are in need of revision to manage the treats that it poses to the natural environment.

Examples of a practical improvements for agricultural practices were suggested by Holland *et al.* (2005), Blackshaw and Vernon (2006) (cited by Grez *et al,* (2010:111). Surrounding semi- natural areas are vital in agro ecosystems as they act as refuges or reservoirs for biodiversity for both pest and beneficial arthropods such as predacious spider and beetle species (Kanya *et al.,* 2004).

The use of beneficial organisms to reduce the effects of pests in the agro-ecosystems is commonly referred to as biological control or integrated pest management (IPM) (Deutch *et al.,* 2005). Natural enemies are an integral part of this venture and should thus be investigated with regards to their ressilence towards disturbances. Climate change is a looming long term threat to all ecosystems and could alter IPM systems, making sustainable or less threating agricultural practices increasingly difficult. Therefore, baseline studies, such as this, will become increasingly important in the nearby future.

4.2.2 Urbanization as a disturbance

Anthropological activities that are responsible for habitat loss may vary (McKinney, 2002) but urbanization is responsible for some of the largest local extinction rates as well as the displacement of indigenous species (Blair and Launer, 1997). Studies done by Kowarik (1995) and Blair and Launer (1997) suggest that towards the core of urban areas there is an increase in invasive species which displaces and decreases indigenous species numbers.

An effect that has been discovered, resulting from urbanization, is that habitat loss is not reversed easily, when compared to other disturbances such as agriculture and forestry. This is ascribed to urban areas rarely being abandoned, which decreases the chances of nature reclaiming these areas. Urban areas not only persist disturbance effects, but expand these through urban sprawl (Stein *et al*., 2000).

Urban sprawl is influenced by population growth, economic growth, industrialisation, politics and also transportation (Bhatta, 2010). Urban sprawl has both positive and negative qualities. The positive qualities include better basic services, higher quality of life and lower unemployment percentages. However, the negative impacts include increased traffic, depletion of local resources and changing the physical environment, all which destroy the natural biota and causing more habitat fragmentation (Bhatta, 2010). The species that are influenced by urbanization are also susceptible to other activities such as farming and recreation.

The species assemblage and community composition is quite similar between agricultural and urban areas (Sattler *et al*., 2011). Sattler *et al*. (2011) suggested that agricultural sites might act as source populations for urban arthropod communities. These authors also suggested that some species might be vanishing from intensified agricultural activities and their new survival strategy could be by finding new habitats in urban areas (Sattler *et al.*, 2011).

The aim of this study was to better understand the effects that urbanization and agriculture have on predatory arthropods, as they are an important functional group within any ecosystem. Predacious arthropods could play a role in IPM systems and if they are using urban areas as a new survival strategy (Sattler *et al.,* 2011), these areas should then be seen for their importance in conserving predacious arthropod diversity. This has practical implications such as implementing corridors to decrease the distance between urbanized areas.

4.2.3 Aim, objectives and hypothesis

The aim of this chapter was to compare predacious arthropod diversity of grasslands between settlements of different size and to compare predacious arthropod diversity of urban areas with agro-ecosystems and untransformed grassland that have different disturbance intensities.

The spesific objectives of this chapter were to:

(1) quantify the effect of different disturbance intensities on the species diversity and assemblages of selected predacious arthropod groups in an urban ecosystem (ruderal, fragmented grassland and untransformed grasslands); and

(2) compare the effect of disturbance intensities and patch size on the abundance, diversity and species assemblages of selected predacious arthropods in agricultural and urban ecosystems.

The hypotheses of this chapter are:

(1) If arthropods are influenced by disturbances, and the intensity of the disturbance affects their diversity, arthropods in the urban ruderal areas will be less diverse and abundant than in the fragmented grassland areas where the intensity of the disturbances are lower.

(2) If the disturbance intensity of maize fields is higher than that found in urban areas due to a higher frequency of disturbances, the diversity and abundance of the selected predacious arthropods will be lower in the agro-ecosystem than the urban environment.

(3) If the selected predators adhere to the Island biogeography theory, which suggest that a larger island would be more diverse, it can be expected that patches in the urban environment that are larger would have a higher abundance and diversity of the selected predacious arthropods.

(4) If the size of a settlement influences the intensity or frequency of disturbance within that urban environment, it would be expected that larger settlements would be less diverse and abundant with regards to the predacious arthropods present in the green spaces of larger settlements.

4.3. Study area

This study consisted of three urban settlements, which contained seven ruderal and fragmented grassland sites. Each of these seven sites consisted of three replicates, which were sampled twice in 2015. A total of 252 samples were collected in the urban environment which consisted of 63 ruderal as well as fragmented grassland areas, sampled twice resulting in 126 samples each. Each settlement contributed 42 arthropod samples to this study. The agricultural sites included a maize agro-ecosystem which consisted of three zones i.e. maize field, field margin and untransformed grasslands. In each zone sampling consisted of three replicates at four different maize agro-ecosystems, which were sampled twice in 2015. A total of 288 samples were collected in the agricultural environment. Each zone contributed 48 arthropod samples which were sampled twice, thus amounting to 96 samples per zone.

4.3.1 Urban sites

Urban sites were chosen from accessible green spaces in Vanderbijlpark, Potchefstroom and Ventersdorp.

These settlements were chosen based on their population size. With an increase in population size it is seen that there is an increase in the urbanization effect which includes construction of houses, industries, infrastructure and transport. A higher population size also results in more waste products produced and more pollution due to an increase in transport, (Miller and Spoolman, 2012).

In each settlement two types of green spaces were chosen, namely fragmented grasslands and ruderal areas. All these settlements were located within the Highveld grassland of South Africa and are surrounded by an agricultural hub.

Fragmented grasslands are green spaces within the towns that have not been transformed by any direct anthropological disturbance, but reside within an urban area, are fragmented, mowed and polluted.

Some (ones that still remained) of the sites used in this study were also used in a study by Cilliers *et al.* (1999) which looked spesifically at the vegetation of fragmented grasslands as well as their importance to conservation. Fragmented grasslands as defined in these studies differed from other urban sites due to the intensity, type and frequence of the antropological disturbance.

Fragmented grasslands are considered to have undergone an intermediate intensity of disturbance. Fragmented grasslands are characterized by the grass *Themeda triandra* (Forssk), which is a species that prefers areas that are relatively stable as it is sensitive to disturbances (Leistner, 2000).

Ruderal areas are also green spaces within the selected towns that have been transformed by anthropological activities beyond that of just urbanization. In South Africa not much is known about ruderal vegetation in urban areas (Cilliers and Bredenkamp, 1999), especially with regards to their predacious arthropod communities.

Ruderal vegetation is known for their ability to progress regardless of intense disturbance. According to Grime (1979) these are plant species that withstand natural disasters such as floods and windstorms. These areas in the urban environments are characterized by a quite high level of disturbance such as soil disturbance, dumping, pathways, construction and mowing (Cilliers and Bredenkamp, 1999). These areas are typical where construction or some form of building or anthropological activity has taken place, but was then discarded and the area was left to be re-colonised.

Settlements that were selected were located on the Highveld and in the Grassland Biome. Settlements differed in population size because it reflected the level of urbanization.

4.3.1.1 Vanderbijlpark

Vanderbijlpark is situated on the banks of the Vaal River that not only separates it from the nearby settlements Vaalpark and Sasolburg, but it is also the border between the south of the Gauteng province, where it is located, and the northern Free State province.

Vanderbijlpark has about 95 840 inhabitants and encapsulates an area of 177.84 km² (Statistics South Africa, 2011). About 60% of the inhabitants are employed in factories (Statistics South Africa, 2011).

Urban areas in Vanderbijlpark were identified according to the above mentioned criteria that define fragmented grasslands and ruderal areas (Table 4.1).

Table 4.1: GPS coordinates, street names as well as temperatures recorded during the two sample times for sampling in Vanderbijlpark.

Ruderal Sites in Vanderbijlpark						**Fragmented Grassland Sites in Vanderbijlpark**				
	GPS Coordinates of urban sites	Street Name of urban sites	1st Sample collection (Temp)	2nd Sample collection (Temp)	Size of urban patch (m^2)	GPS Coordinates of urban sites	Street Name of urban sites	1st Sample collection (Temp)	2nd Sample collection (Temp)	Size of urban patch (m^2)
1	S 26°43”.619’ E 27°52”.476’	Hendrick van Eck	31°C	28°C	44 903	S 26°44”.248’ E 27°51”.293’	Louis Trichard	30°C	25°C	87 408
2	S 26°40”.840’ E 27°48”.936’	Hans Merensky	27°C	29°C	42 923	S 26°41”.397’ E 27°51”.172’	Wolmarans	29°C		38 659
3	S 26°43”.167’ E 27°48”.979’	William Nicol	27°C	27°C	33 543	S 26°43”.633’ E 27°51”.472’	Andrew Young	32°C	27°C	24 577
4	S 26°42”.028’ E 27°51”.054’	Wolmarans	29°C	28°C	27 118	S 26°40”.375’ E 27°48”.528’	Delfos	28°C	29°C	94 058
5	S 26°40”.446’ E 27°49”.798’	Delfos	29°C	28°C	369 960	S 26°44”.003’ E 27°52”.081’	Andries Potgieter	34°C	27°C	733 253
6	S 26°43”.885’ E 27°51”.567’	Oranje river	32°C	27°C	16 903	S 26°43”.306’ E 27°51”.271’	Japie Greyling	25°C	27°C	80 277
7	S 26°43”.448’ E 27°49”.070’	Vaal drive	24°C	24°C	10 543	S 26°44”.532’ E 27°50”.696’	Frikkie Meyer	24°C	24°C	46 187
2A	Alternative site used for the fragmented grassland area of Wolmarans street.					S 26°40”.397’ E 27°50”.017’	Delfos Boulevard		28°C	135 866

A satellite image of Vanderbijlpark is included to show the dispersal of the used green spaces and also to show the layout of factories and residential areas (Figure 4.1).

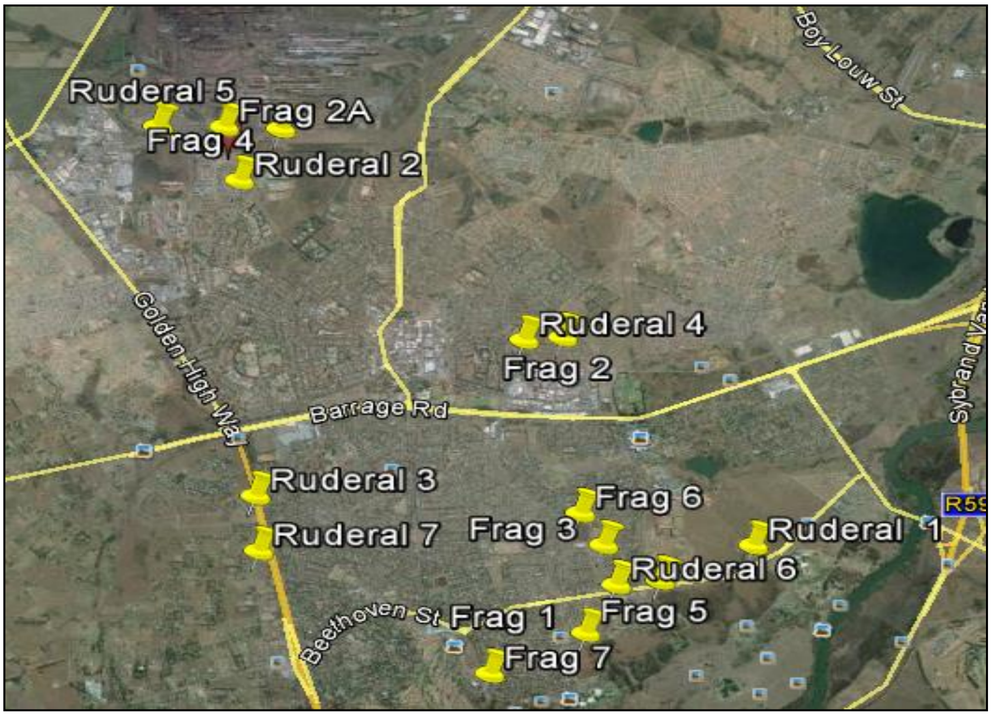

Figure 4.1: Satellite image depicting urban green spaces and sampling points in Vanderbijlpark.

4.3.1.2 Potchefstroom

Potchefstroom was founded in 1838 for farming purposes, but today it is an academic settlement due to the presence of North-West University. It is located in the North-West province, approximately 90 km west of Vanderbijlpark.

Potchefstroom has 43 448 inhabitants and is about 162.44km^2 (Statistics South Africa, 2011). It is situated on the banks of the Mooi river, which runs through the settlement. Potchefstroom is also important for industry and agriculture (Jenkins and Balkema, 1971).

Table 4.2: GPS coordinates, street names as well as temperatures recorded during the two sample times for sampling in Potchefstroom.

Ruderal Sites in Potchefstroom						**Fragmented Grassland Sites in Potchefstroom**				
Nr.	GPS Coordinates of urban sites	Street Name of urban sites	1st Sample collection (Temp)	2nd Sample collection (Temp)	Size of urban patch (m^2)	GPS Coordinates of urban sites	Street Name of urban sites	1st Sample collection (Temp)	2nd Sample collection (Temp)	Size of urban patch (m^2)
1	S 26°41”.756’ E 27°04”.803’	Rissik	25°C	28°C	10 730	S 26°40”.459’ E 27°06”.950’	MC Roode	28°C	24°C	17 856
2	S 26°42”.243’ E 27°04”.942’	Fontein	26°C	28°C	60 907	S 26°44”.328’ E 27°05”.005’	Viljoen	27°C	21°C	7 022
3	S 26°43”.334’ E 27°07”.509’	Kwartel	28°C	22°C	10 302	S 26°44”.585’ E 27°05”.566’	Louw	31°C	21°C	15 604
4	S 26°43”.014’ E 27°07”.280’	Steyn	27°C	23°C	33 353	S 26°41”.224’ E 27°07”.042’	MC Roode	28°C	24°C	87 849

5	S 26°41".475' E 27°04".215'	Deppe	23°C	27°C	43 556	S 26°41".842' E 27°06".187'	Langen-hoven	27°C	26°C	24 986
6	S 26°44".226' E 27°05".727'	Eland	27°C	21°C	37 372	S 26°40".613' E 27°04".518'	Tiger-moth	21°C	26°C	55 888
7	S 26°42".526' E 27°04".146'	Russel	27°C	27°C	4 666	S 26°39".713' E 27°06".851'	R501	27°C	25°C	123 919

A satellite image of Potchefstroom is included, to show the dispersal of the used green spaces and also to show the layout of factories and residential areas (Figure 4.2).

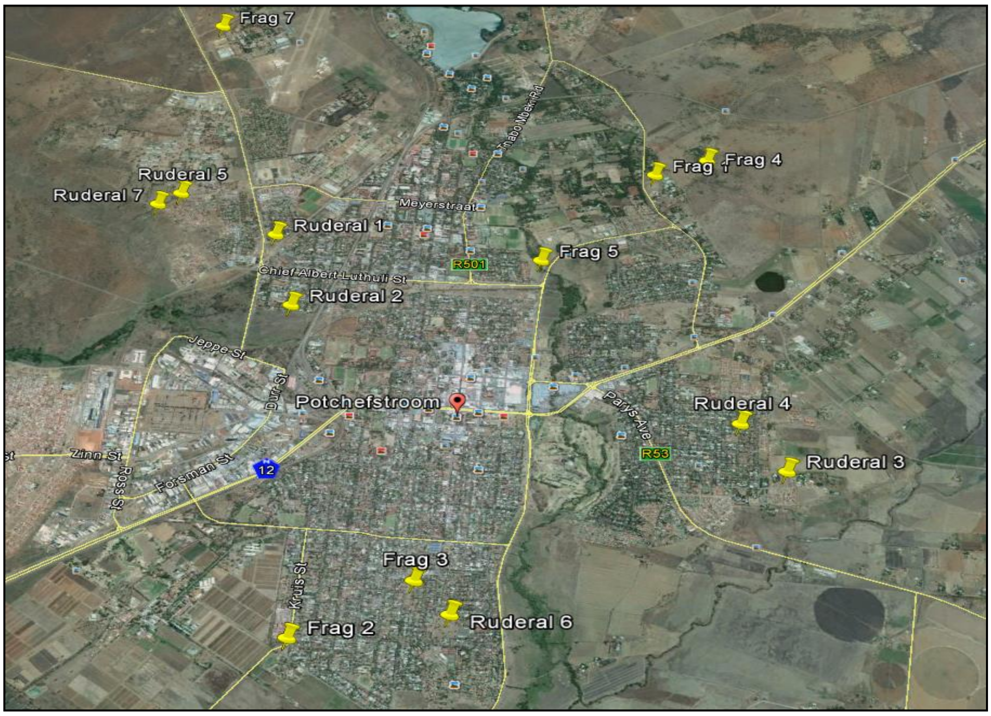

Figure 4.2: The satellite image depicting urban green spaces and sampling point in Potchefstroom.

4.3.1.3 Ventersdorp

Ventersdorp was established in 1866 around the Dutch reformed church and is most famous for one of its inhabitants Eugène Terre'Blanche, who was the founder of the Afrikaner Weerstandsbeweging (North-West Tourism, 2011). Ventersdorp is located in the North-West province and is located about 45 km north of Potchefstroom.

Ventersdorp has 22 073 inhabitants and encapsulates a surface area of 57.47 km^2 (Statistics South Africa, 2011). The river running through this settlement is called the Skoonspruit. Ventersdorp's main economic sectors are agriculture and manufacturing and services.

Table 4.3: GPS coordinates, street names as well as temperatures recorded during the two sample times for sampling in Ventersdorp.

Ruderal Sites in Ventersdorp						**Fragmented Grassland Sites in Ventersdorp**				
Nr	GPS Coordinates of urban sites	Street Name of urban sites	1st Sample collection (Temp)	2nd Sample collection (Temp)	Size of urban patch (m2)	GPS Coordinates of urban sites	Street Name of urban sites	1st Sample collection (Temp)	2nd Sample collection (Temp)	Size of urban patch (m2)
1	S 26°18”.173’ E26°49”.856’	Cheyne	28°C	26°C	3 078	S26°18”.494’ E26°48”.789’	Eland	30°C	24°C	38 287
2	S 26°18’’.798’ E 26°49’’.587’	Roth & Paarl	27°C	26°C	2 160	S26°18”.490’ E26°49”.577’	R30	23°C	23°C	6 017
3	S 26°19’’.256’ E 26°49’’.239’	Berg	30°C	27°C	2 563	S26°18”.274’ E26°49”.539’	R53	20°C	21°C	16 926
4	S 26°19’’.315’ E 26°49’’.417’	Mark	28°C	29°C	17 062	S26°18”.514’ E26°49”.084’	Steenbok	27°C	24°C	16 508
5	S 26°18’’.932’ E 26°49’’.118’	Cochrane	32°C	26°C	5 048	S26°18”.115’ E26°48”.868’	Eland	27°C	23°C	135 988
6	S 26°18’’.355’ E 26°49’’.604’	Graaf & Koekemoer	28°C	30°C	33 536	S26°18”.658’ E26°48”.942’	Mark	30°C	26°C	60 185
7	S 26°18’’.694’ E 26°49’’.342’	Anemay & Visser	27°C	27°C	5 638	S26°19”.360’ E26°49”.308’	Berg & van Riebeek	29°C	28°C	14 257

A satellite image of Ventersdorp is provided below to show the dispersal of the used green spaces and also to show the layout of factories and residential areas (Figure 4.3).

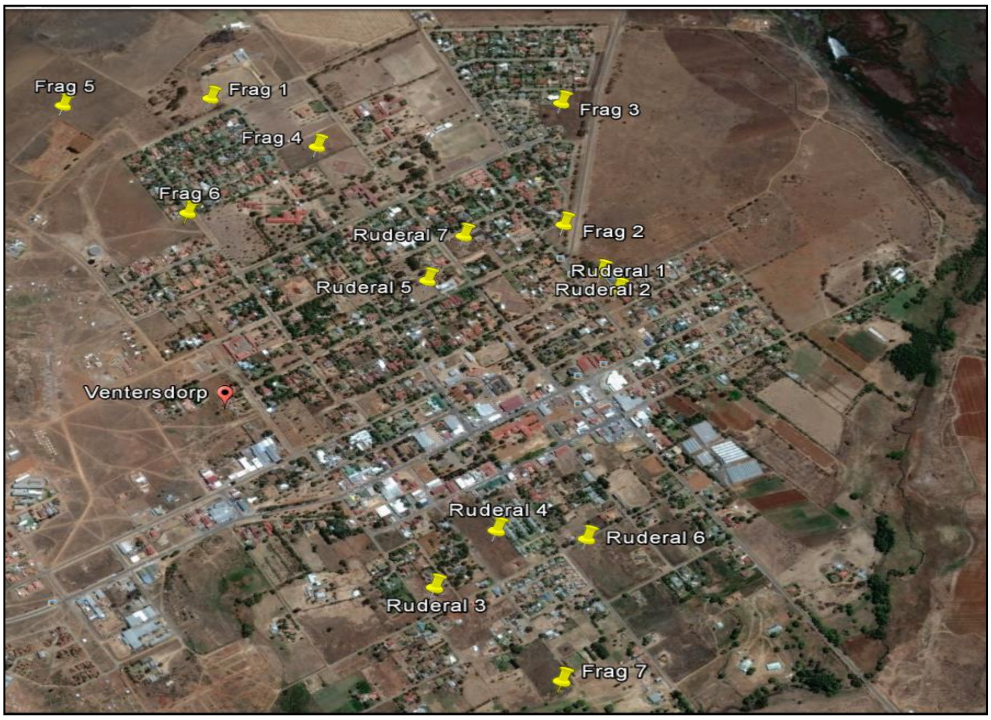

Figure 4.3: The satellite image depicting urban green spaces and sampling point in Ventersdorp.

4.3.2 Agricultural sites

The agricultural sites chosen to compare with the urban areas were situated on farms in the same region of the Highveld in the Grassland Biome. Sampling was conducted during the part of the growing season when maize was in the soft-dough stage of development, two to four weeks after flowering of the maize.

The agro-ecosystem included maize fields with adjacent field margins and untransformed grassland. The field margin is characterised by different degrees of anthropological disturbances such as roads, fences and power lines.

Figure 4.4: Layout of the agricultural sampling area.

Table 4.4: The GPS coordinates of the localities as well as temperature and dates that the agricultural sites were sampled. Agricultural areas were sampled during the 2015 maize season.

GPS Coordinates of Agricultural sites	Maize Field Sites		Field Margin Sites		Untransformed grasslands	
	1st Sample collection (Temp)	2nd Sample collection (Temp)	1st Sample collection (Temp)	2nd Sample collection (Temp)	1st Sample collection (Temp)	2nd Sample collection (Temp)
S26°27'58" E26°55'45"	18°C	19 °C	33°C	30°C	28°C	30°C
S26°28'15.6" E26°55'33.2"	19°C	22°C	30°C	28°C	28°C	31°C
S26°28'028" E26°56'.231"	22°C	19°C	28°C	31°C	27°C	33°C

4.3.3 Disturbance Intensities

Each of the areas sampled was subjected to a different intensity of disturbances (Table 4.5) due to the human impact upon it. The type of anthropological disturbance as well as the frequency with which each disturbance occurs on each sample area also differs (Table 4.5).

Table 4.5: Comparison between sample sites and their disturbance intensity.

Sample area	**Intensity of disturbance**	**Ecosystem**	**Photo**
Maize field	High - very frequent disturbances - plowing, insecticides	Agricultural	
Ruderal areas	High - very frequent disturbances - dumping construction	Urban	
Field Margin	Medium - intermediate frequency disturbances - pathway , insecticide drift	Agricultural	
Fragmented Grasslands	Medium - intermediate frequency disturbances - pathway, urban environment	Urban	
Untransformed Grasslands	Low/ none - limited frequency of disturbance - pathway	Agricultural	

According to Debano (2006) some insect communities are negatively influenced by grazing and therefore can be seen as a disturbance that influences the intensity of disturbance of the treatments sampled in this study. Management practices and the type of grazing as well as fire was found to have different effects on arthropod diversity (Fay, 2003) and should thus be noted (Table 4.6). The effects due to these disturbances ranged from decreases in diversity and abundance as a result of fire to increases in arthropod abundance and diversity as a result of grazing (Fay, 2003).

Table 4.6: Summary of disturbances located within sample areas. (Frequency of disturbance is rated in the table below with a score between 1 to 4, with 1 being never, 2 - sometimes, 3- often and 4 very often).

		Sample areas				
		Maize field	Field Margin	Untransformed Grassland	Ruderal areas	Fragmented grasslands
Disturbances	Grazing	1	2	3	1	2
	Fires	2	2	3	3	4
	Mowing	1	2	2	3	4
	Dumping	1	1	1	4	2
	Pesticides	4	3	1	1	1
	Air Pollution	2	2	2	4	4
	Plowing	4	2	1	1	1

4.4 Arthropod sampling

All sites, especially in urban areas were chosen due to the grasslands being of optimal height to create suitable habitats for predacious arthropods. These grasslands were not subjected to mowing for the duration of this study.

Arthropod samples were collected by means of sweep nets. The minimum area required for a sample collection was 200x150 m. Each sample plot was 100 m^2. Three replicates were taken at each of the sites. Therefore sampling sites had to be large enough to encapsulate the total area required for the sample collections. Each sample area thus had to be large enough to include three $100m^2$ samples as well as give a relative distance of about 50m between each sample to discourage disturbances of previous samples influencing the second and third replicates.

Each plot was demarcated as a square by means of a pre-measured rope. Each plot in the grassland area was sampled systematically by sweeping from side to side, each sample plot required about 10 sweeps with the side to side motion whilst walking 10m in a straight line. In the maize fields the sweeping was conducted up and down, each sample required individual maize plants to be sweeped and thus required about 12 sweeps whilst walking 10m in a straight line. This however was depended on the distance between the maize plants which was determined by the farmers. Temperature for all urban environments was recorded by a digital thermostat (Exo Terra Digital Thermometer). The samples were then placed into marked bags and refrigerated.

For identification purposes each bag was emptied into a bowl with 70 % ethanol and only the predator species in question, i.e. lacewings, praying mantids, coccinellids and spiders of each plot were then transferred into separate containers with 70% ethanol. Collection bottles were marked with different coloured adhesives to distinguish between the four predator groups. All specimen were identified to species level with the help of the Biosystematic Division of the Agricultural Research Council (ARC) in Pretoria, Gauteng. Mantid species were identified through comparison with the ARC's collection as well as a more recently updated collection at the Ditsong museum in Pretoria.

4.4.1 Data Analyses

The data did not exhibit normality/normal distribution due to variation in and size of the predator populations in the different agro-ecosystem zones, and did not display constant variance. Shapiro-Wilk and Levene’s tests, as well as Q-Q and Box-plots confirmed that data did not satisfy the assumptions required to analyse variance.

Linear mixed models (LMM), statistical modelling that uses Bayesian inference, is employed when data is not distributed normally, including data generated by counts, allowing for a variety of correlation patterns to be modelled. Population numbers of one zone was related to the population numbers of other zones of the agro-ecosystem as well as the urban environment, as individuals may switch between urban green space types or patches in due to biological necessities such as feeding strategies. Data were also collected from fixed points over time. This created a dependence of data (covariance), which is corrected for with LMM. Multiple measurements per plot per zone or patch in urban areas were performed over one season, i.e. sampling was replicated twice in 2015 and across populations and therefore correlated data arose in the statistical analyses.

Significant differences between population means were acquired by applying pairwise comparisons on estimated marginal means (EMM). EMM represent the transformed means of populations/samples of unequal size, enabling the comparison of sample means across different treatments, revealing significant differences between means. EMM represent a statistical prediction of population means and are values derived from linear mixed models applied to data with a non-normal distribution. By employing Sidak adjustments, multiple population and predator group means could be accurately compared and differences across habitats within two ecosystems the agro-ecosystem (maize field, field margin, untransformed grassland) and urban environment (ruderal areas and fragmented grasslands) determined. The LMM's were also used to determine statistical significance between three urban settlements of different sizes with regards to the two urban green space types. The effect size was also determined by using the LMM's and is independent of sample size and is a measure of practical significance. A minimal practical significance (practical implication) is $d > 0.2$, for a significant effect size $d = 0.5$, and a large practical significance is when $d > 0.8$ (Ellis and Steyn, 2003). IBM SPSS version 21 was used for all statistical analyses.

Satelite images and patch size were determined using Google Earth software and measurement tools. Measurements were taken in square meters to ease further analysis of the correlations between patch sizes and arthropod diversity. Corellations were done with LMM analysis.

Mean abundance graphs as well as box plots indicating the variation within the data were compiled using Statistica software version 12 (Statsoft Inc. 2013). Abundance data was used to determine the diversity of each of the predator groups. Analyses considered the Shannon-Wiener Diversity Index, Margalef Species Richness Index, Pielou's Evenness and also Simpson’s Evenness and was done with the use of the Primer 6 Software (Clarke and Gorley, 2006). Species composition and turnover (beta diversity) was visualised with NMDS (Non-Metric Dimensional Scaling) graphs in Primer. Similarity Percentage (SIMPER) analysis to indicate species responsible for grouping within the data as indicated by the NMDS graphs, these analyses were performed in PAST version 2.15 which was released in 2012 (Hammer *et al.,* 2001). Data was subjected to the Linear Mixed Model Analysis to ultimately determine whether any significant variation existed between the previously mentioned six criteria across different land-uses for the selected predacious arthropods.

4.5 Results and Discussion

4.5.1 Urban Ecosystem

4.5.1.1 Overall Predator Diversity

The urban green spaces were abundant and diverse with regards to predators with 830 spider individuals (Figure 4.5) belonging to 30 different species, 26 lacewing individuals which represented two species. A total of 391 Coccinellids and 74 praying mantids and both of these predators represented seven different species (Figure 4.6). The lacewings yielded only two species and therefore only their abundance is applicable for the analysis that follows.

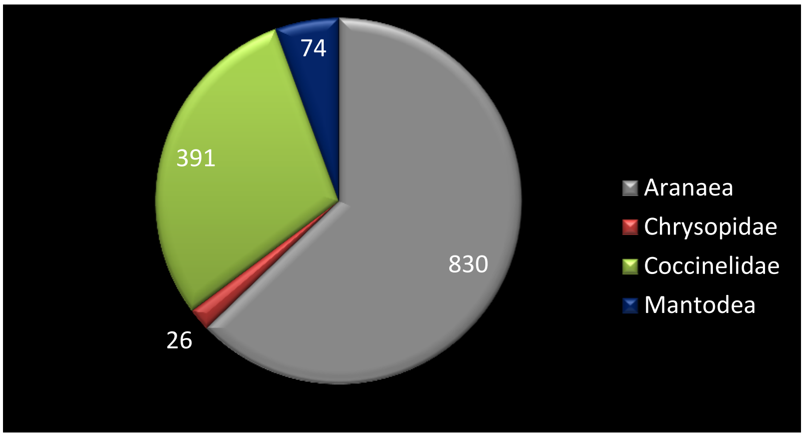

Figure 4.5: Number of total individuals of predacious arthropod groups in an urban green space.

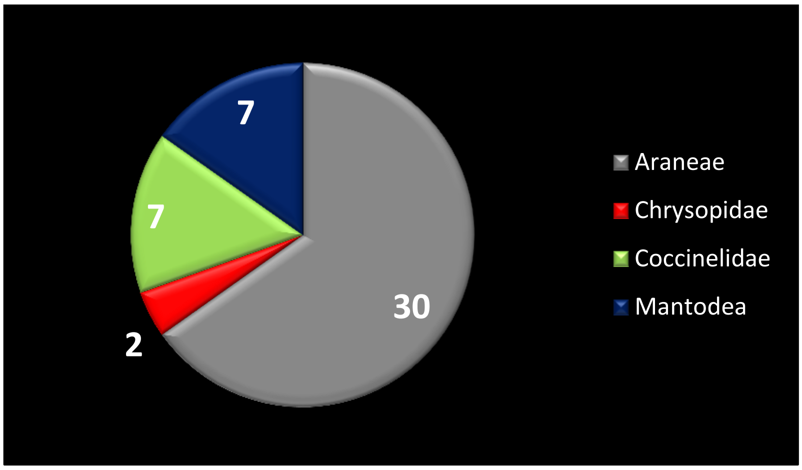

Figure 4.6: Species richness of four predacious arthropod groups in urban green spaces.

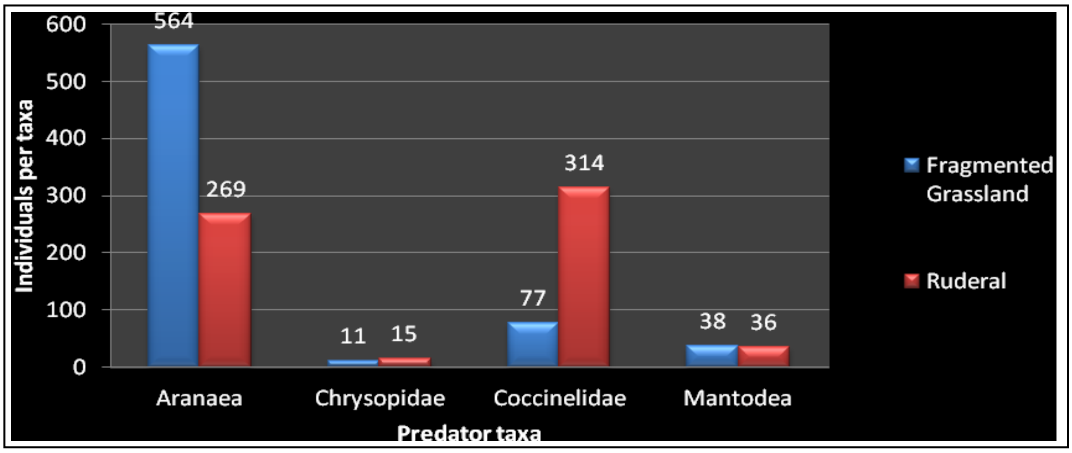

Figure 4.7: Predacious arthropod abundance within urban green spaces.

A total of 690 predatory arthropod individuals were collected in the fragmented grasslands while 634 individuals collected in the ruderal areas of the urban environments (Figure 4.7). Therefore the fragmented grassland which has a lower disturbance intensity had an overall higher abundance of predator individuals. This was the case for three of the four predator taxa i.e. spiders, lacewings and praying mantids. The coccinellids were the only predator taxa that were more abundant in the ruderal areas which had the higher disturbance intensity.

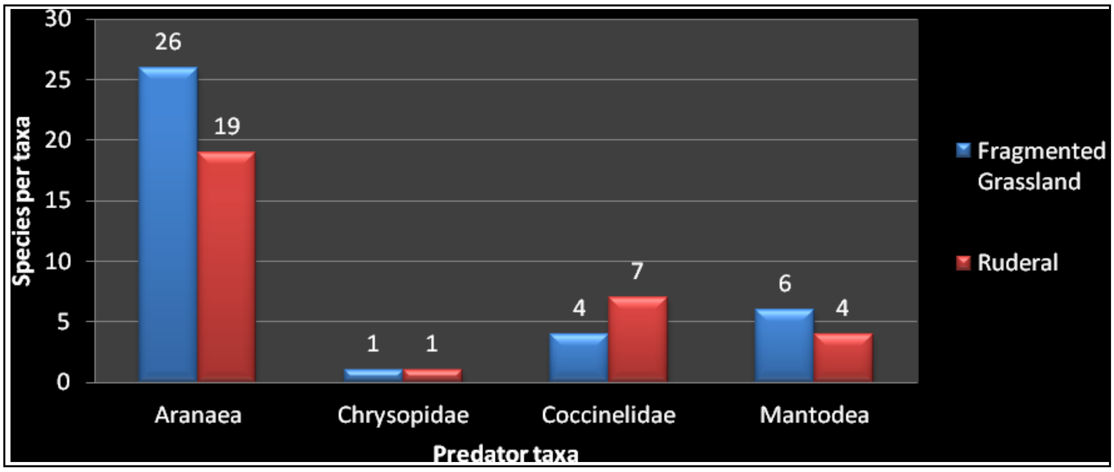

Figure 4.8: Predacious arthropod species richness within urban green spaces.

Species richness between the two green spaces showed that there were no major differences, but the spiders were less diverse within the more highly disturbed ruderal areas (Figure 4.8). Coccinellids consisted of more species in the ruderal areas than fragmented grasslands which has a higher species richness of the remaining predator taxa.

Spiders and praying mantids tended to be more abundant in the larger settlement of Vanderbijlpark than that of the small settlement of Ventersdorp, however the trend is not shared by lacewings and Coccinellids (Figure 4.9).

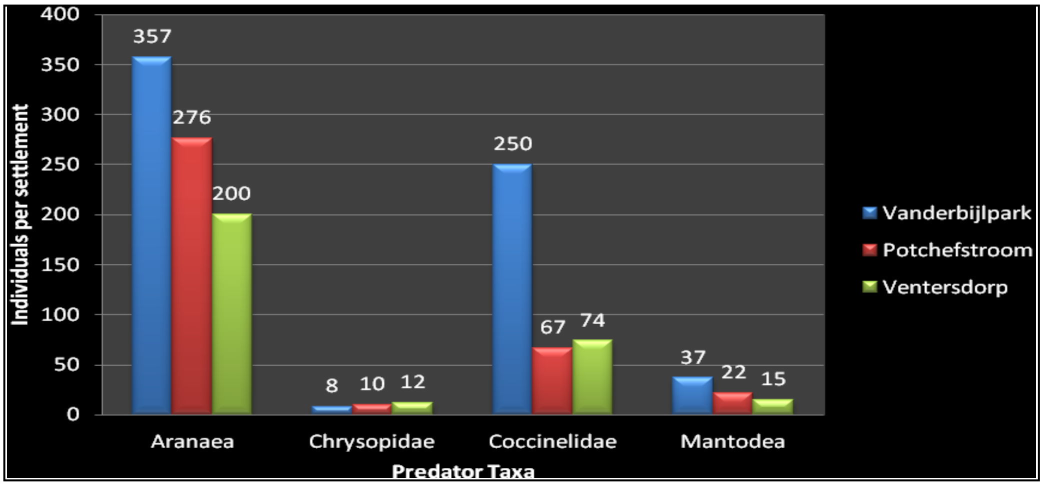

Figure 4.9: Predacious arthropod total individuals per urban settlement.

The trend is also not shared when the types of green spaces are separated (Figure 14). The praying mantids had a relatively equal abundance for both the large and medium settlement but the smallest settlement indicated their abundance to be somewhat higher in the less disturbed fragmented grasslands (Figure 4.10). Furthermore, the spider and lacewing abundances were higher in the fragmented grassland despite the size of the settlements.

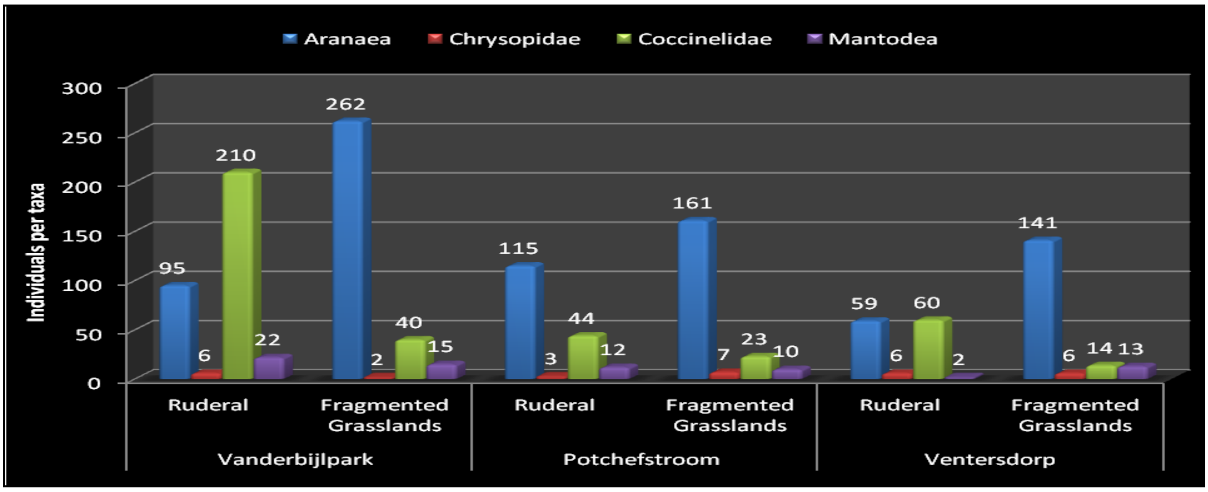

Figure 4.10: Abundance of predator taxa within green spaces of different settlements.

Refer to Table 4.7 for predator species names and occurrences within the urban environment.

Table 4.7. Predacious arthropod species recorded for the urban ecosystem. Species recorded from a single settlement is indicated with * and from only one land-use type with #.

Order	**Family**	**Species**	**Ruderal**			**Fragmented grasslands**		
			Vent	**VDBP**	**Potch**	**Vent**	**VDBP**	**Potch**
Araneae	Eutichuridae	*Cheiracanthium* sp.	4	1	2	5	5	2
Araneae	Theridiidae	*Enoplognatha* sp. 1	14	11	38	49	15	8
Araneae	Salticidae	*Festucula lawrencei*	12	15	30	45	119	81
Araneae	Araneidae	*Hypsosinga* sp. 3*#	-	-	-	2	-	-
Araneae	Araneidae	*Larinia natalensis*	-	1	-	1	2	1
Araneae	Theridiidae	*Latrodectus renivulvatus*	3	1	1	-	1	2
Araneae	Thomisidae	*Misumenops rubrodecoratus #*	1	3	-	-	-	3
Araneae	Thomisidae	*Monaeses paradoxus *#*	-	-	-	1	-	-
Araneae	Thomisidae	*Monaeses pustulosus *#*	-	-	-	1	-	-
Araneae	Araneidae	*Nemoscolus* sp.	-	-	-	1	3	-
Araneae	Araneidae	*Neoscona subfusca*	-	7	4	-	8	4
Araneae	Oxyopidae	*Oxyopes hoggi #*	-	-	-	4	2	-
Araneae	Oxyopidae	*Oxyopes longispinosus*	2	-	3	3	-	-
Araneae	Oxyopidae	*Oxyopes schenkeli*	14	27	21	23	44	22
Araneae	Araneidae	*Pararaneus* sp. 1 #	-	-	-	2	1	-
Araneae	Salticidae	*Pellenes bulawayoensis*	3	8	5	-	12	3
Araneae	Oxyopidae	*Peucetia striata*	1	15	4	-	27	21
Araneae	Thomisidae	*Runcinia flavida*	-	-	4	1	7	3
Araneae	Thomisidae	*Stiphropus bisigillatus*	2	3	4	-	10	5
Araneae	Philodromidae	*Thanatus atlanticus*	2	-	-	-	2	2
Araneae	Araneidae	*Thanatus atlanticus *#*	-	-	-	-	1	-
Araneae	Theridiidae	*Theridion pictum*	-	-	1	1	-	4
Araneae	Salticidae	*Thyene semiargentea *#*	-	-	-	2	-	-
Araneae	Salticidae	*Thyene thyenioides*	-	2	1	-	1	-
Araneae	Salticidae	*Thyenula* sp. 1 *#	-	-	1	-	-	-

Araneae	Philodromidae	*Tibellus hollidayi*	1	1	1	-	2	-
Neuroptera	Chrysopidae	*Chrysoperla congrua*	6	6	3	6	2	3
Coleoptera	Coccinellidae	*Buleae anceps*	7	11	10	7	12	11
Coleoptera	Coccinellidae	*Chelomenes lunata*	39	80	18	6	18	9
Coleoptera	Coccinelidae	*Exochomus flavipes*	1	23	12	-	1	3
Coleoptera	Coccinelidae	*Harmonia axyridis #*	2	3	-	-	-	-
Coleoptera	Coccinelidae	*Hippodamia variegata #*	6	74	6	-	-	-
Coleoptera	Coccinelidae	*Lioadalia flaromaculata*	3	17	-	1	9	-
Mantodea	Coccinelidae	*Micraspis cf. comma #*	2	2	-	-	-	-
Mantodea	Mantidae	*Compsothspis natalica #*	-	-	-	1	1	-
Mantodea	Mantidae	*Entella transvaalica *#*	-	2	-	-	-	-
Mantodea	Mantidae	*Epsiscopus chalybeus #*	-	-	-	-	13	9
Mantodea	Mantidae	*Galepsus* sp. #	-	17	11	-	-	1
Mantodea	Mantidae	*Harpogomantis discolour *#*	-	3	-	-	-	-
Mantodea	Mantidae	*Pyrgomantis rhodesica #*	-	-	-	1	1	-
Mantodea	Mantidae	*Tenodera iringana*	2	-	1	11	-	-

4.5.1.2 Community composition and assemblages within urban green space

4.5.1.2.1 Overall community composition

Arthropod species assemblages are difficult to assess with the use of NMDS graphs, as it only takes a snapshot of the community composition and assemblages. This is problematic as arthropods are quite mobile. However, increased data collection can lead to a snapshot of their community composition and species assemblage within a certain environment.

Predacious arthropods revealed separate but related clusters of community composition for the two urban green space types (Figure 4.11). The overlap in similarity suggests that some species assemblages are shared between green spaces. Both green spaces have a high beta diversity, also known as species turnover.

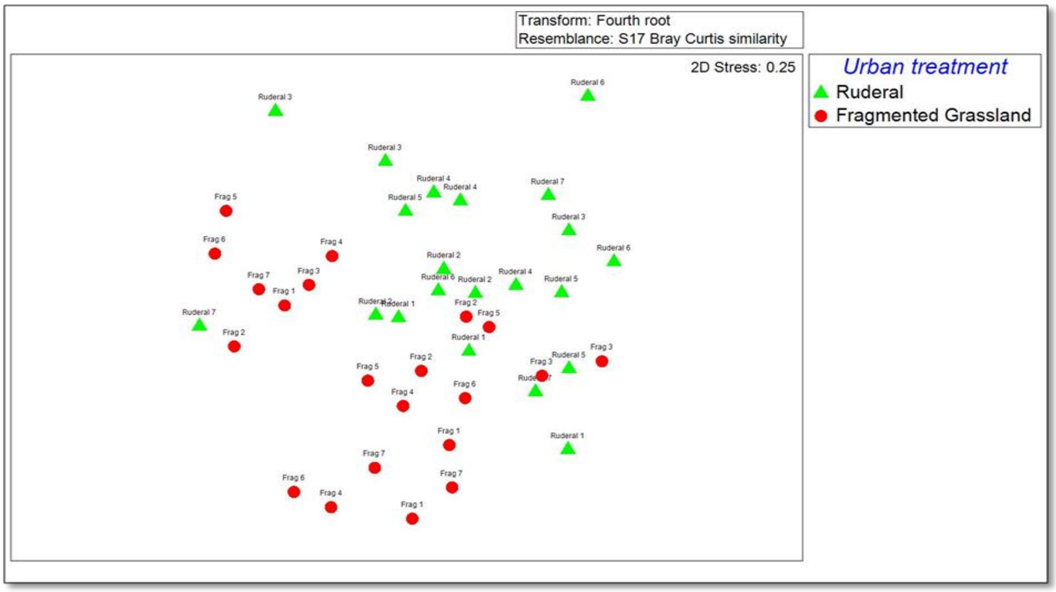

Figure 4.11: Predator communities in relation to green space types within an urban environments

The mean overall dissimilarity between the green space types is 72.19% as calculated by the Simper analysis within the PAST software. The dissimilarity within the data (Table 4.8) can be attributed to three Aranaea species and one Coccinellidae species (Figure 4.12).

Table 4.8 - Summary of species responsible for cumulative contributions between two urban green space types. All species that cumulatively contributed to ± 50% dissimilarity was presented.

Predator Species	Average dissimilarity (%)	Contribution (%)	Cumulative (%)
1. *Tibellus hollidayi* (Araneae)	15.7	21.7	21.7
2. Cheilomenes lunata (Coccinellidae)	9.6	13.3	35
3. Misumenops rubrodecoratus (Araneae)	6.5	9.1	44.1
4. Runcinia flavida (Araneae)	5.8	8.1	52.2

Figure 4.12 - Predator species responsible for dissimilarity <50% cumulatively.

4.5.1.2.2 Community composition of three different settlements.

The community composition across the urban settlements showed Vanderbijlpark and Ventersdorp had separate clusters (Figure 4.13) which indicated that the community composition and assemblage for these two settlements were not the same. The medium sized settlement (Potchefstoom) is, however, scattered between these two indicating that the community composition and assemblage is somewhat intermediate.

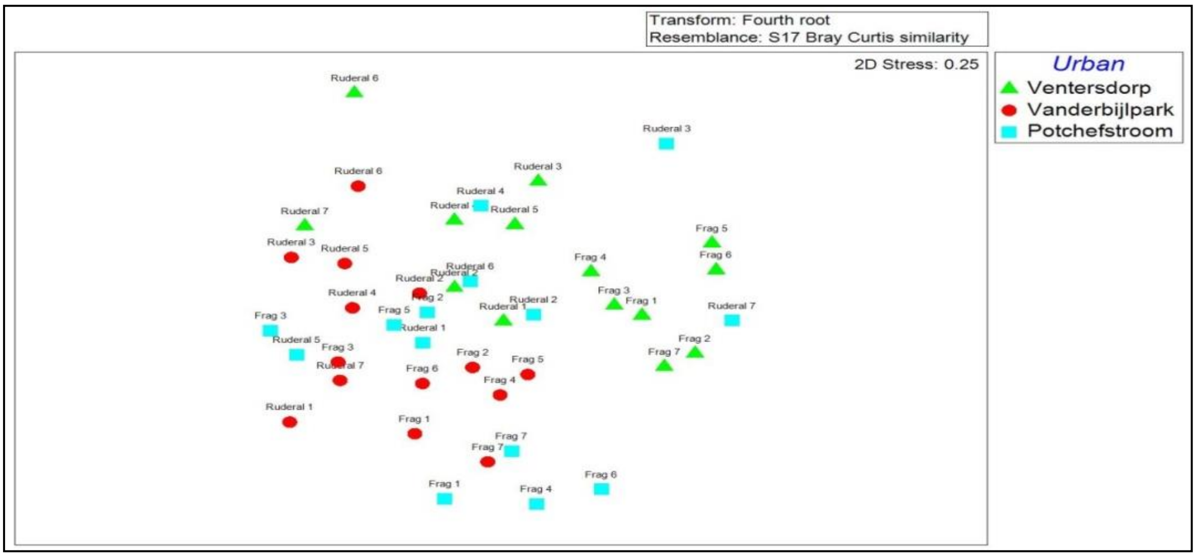

Figure 4.13: Species assemblage of urban green spaces within the urban settings.

Vanderbijlpark exhibits lower beta diversity when compared to that of Potchefstroom and Ventersdorp.

Overall dissimilarity between settlements was 71.19% according to the SIMPER analysis that was done with the PAST software. The predator species responsible for the difference was the above mentioned three Araneae species and one Coccinellidae species (Table 4.7).

4.5.1.2.3 Community composition and species assemblage of the urban green space type individually i.e. ruderal and fragmented grasslands

When the types of urban green spaces are investigated separately, the snapshot image of the predacious arthropods community composition and assemblage changes. Figure 4.14 indicates that Potchefstroom's predacious arthropods tend to have a similar community composition and assemblage as that of Vanderbijlpark. Ventersdorp, the smallest settlement, seem to differ from the larger settlements. This indicates a difference in arthropod species assemblage. However, Potchefstroom does seem to have higher beta diversity than Vanderbijlpark.

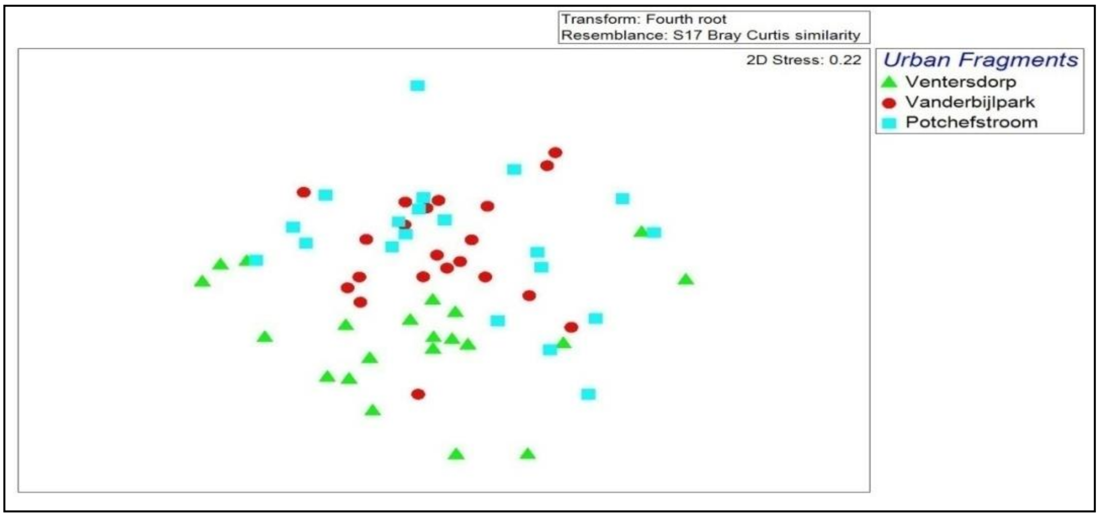

Figure 4.14: Predacious arthropod community composition and assemblage of the urban green space type - Fragmented grassland across the urban gradient.

The overall average dissimilarity along the urban gradient with regards to urban fragmented grasslands is 70.81% according to the SIMPER analysis that was done with the PAST software. In this case the species responsible for 50% of the dissimilarity is two Coccinellidae species as well as two Araneae species. The coccinellid species contributing to the dissimilarity is *Cheilomenes lunata* (18.12%), *Hippodamia variegata* (9.93%), while the spider species are *Tibellus hollidayi* (9.68%) and *Misumenops rubrodecoratus* (9.39%).

The highly disturbed ruderal areas (waste ground often associated with rubbish) and fragmented grasslands (undeveloped areas) of the three settlements where data was collected showed no identifiable clusters (Figure 4.15). This suggests that there are no dissimilarity with regards to the community composition and assemblage of the selected arthropod species within these urban green space types.

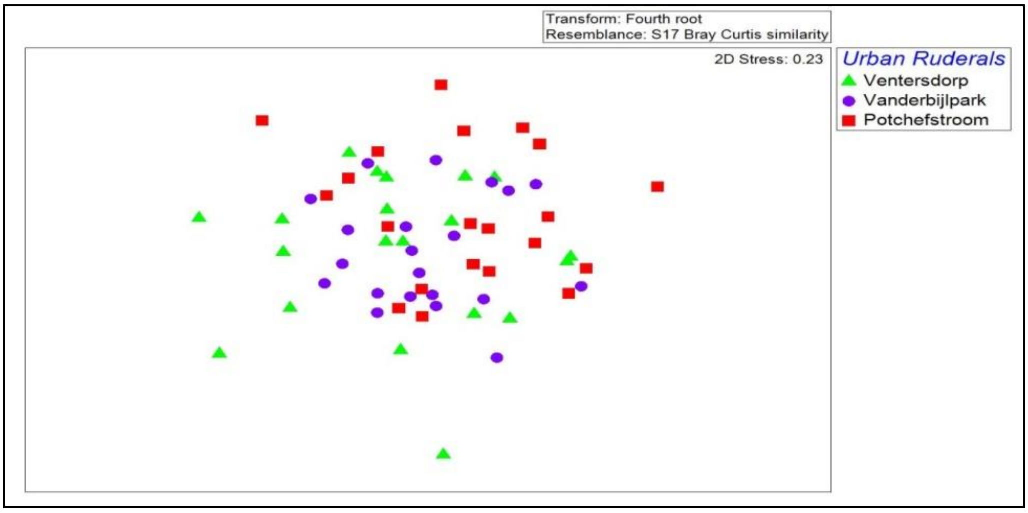

Figure 4.15: Predacious arthropod community composition and assemblage of the urban green space type - ruderal areas across the urban gradient.

The arthropod species assemblage between the three settlements overlaped but it is noted that Ventersdorp could have a higher beta diversity as data points are further apart in comparison to the other settlements. The outliers exhibit different states that are characterised by the following species: *Italoshrysa similis* (lacewing), *Synema decens* (spider), *Larinia bifida* (spider) and *Hemipusa capensis* (praying mantis) which are typically not found in urban environments.

4.5.1.2.4 Community composition and assemblage of individual settlements

The largest and meduim settlement's community composition and assemblage of the selected predators are different with regards to the fragmented grasslands and the ruderal areas, and can be seen by the visible clusters on Figure 4.16. The smallest settlement seemed to have no distinct differences in community composition and assemblage as there was no identifiable clusters for either of the two types of urban green space (Figure 4.16).

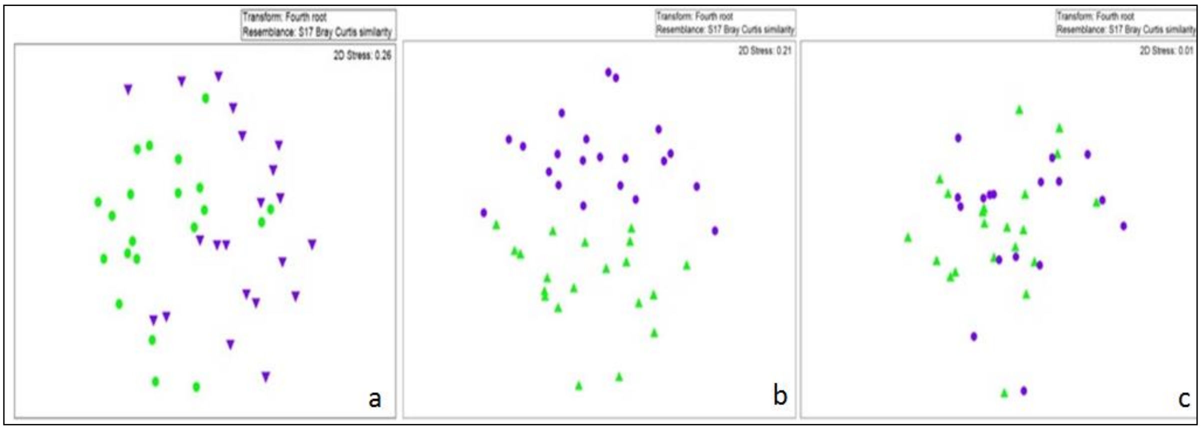

Figure 4.16: Community composition and assemblage for Vanderbijlpark (a), Potchefstroom (b) and Ventersdorp (c) Green symbols indicate the fragmented grassland treatment while the purple symbols indicate the ruderal treatments.

The dissimilarity in communities (cumulatively ± 50%) within Vanderbijlpark is due to three spider species and one coccinellid species, the same species as the overall predator community composition and assemblage i.e. *Tibellus hollidayi* (24.7%), *Misumenops rubrodecoratus* (13%)*, Cheilomenes lunata* (7.6%) *and Runcinia flavida* (7.5%). The arthropods responsible for the differences in communities of Potchefstroom are: Aranea: *Tibellus hollidayi* (24%) and *Runcinia flavida* (6.9%) and Coccinellidae: *Cheilomenes lunata* (15.1%) *and Hippodamia variegata* (8.7%)*.*

4.5.1.3 Statistical differences between the two types of urban green spaces for all urban environments

4.5.1.3.1 Overall diversity between the two types of urban areas

Statistical differences were found between the urban green spaces with regards to the Pielou's Eveness and the Simpson's diversity Index (Table 4.9). No statistical differences were found with regards to the diversity indices or the abundance (Table 4.9; Figure 4.17). The only medium effect size resulted from the fragmented grasslands ruderal areas was with regards to the Pielou's Eveness which therefore has a higher level practical implication than the other diversity indices.

Table 4.9: Linear Mixed Model analysis indicating statistical significant differences between the fragmented grasslands and ruderal areas for different abundance and diversity indices across all the urban settlements.

Urban green spaces		Species Richness per sample plot	Total Individuals per sample plot.	Margalef's species richness	Pielou's Eveness	Shannon-Wiener Diversity Index	Simpson's Diversity Index
Fragmented grasslands VS. Ruderal areas	Sig. (p)	0,195	0,551	0,756	0,011*	0,824	0,056*
	Effect size (d)	0,21*	0,09*	0,06*	0,47**	0,04*	0,34*

indicates P-values (P> 0.05) of statistical significance. effect size (d) (d>0.2 =low impact, d>0.5 = medium impact** and d>0.8= high impact ***)

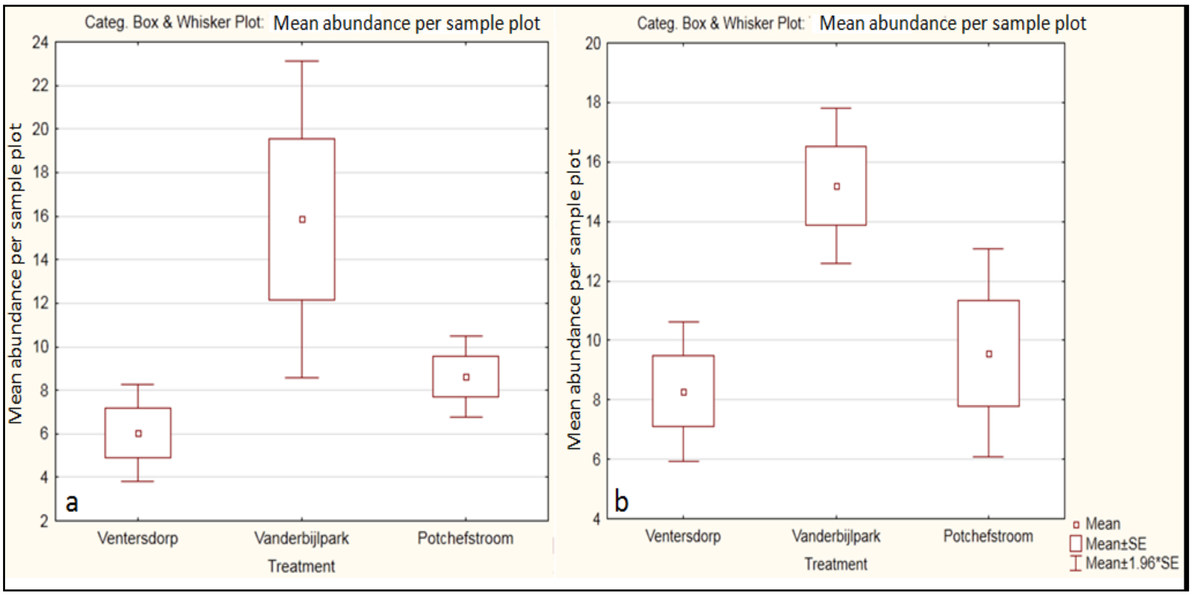

Figure 4.17: Data variation for the two types of urban green spaces between the three settlements of different sizes. (a) Mean abundance of predacious arthropods in ruderal areas (b) Mean abundance of predacious arthropods in fragmented grasslands.

4.5.1.4 Statistical differences for the three urban settlements (both urban green spaces types)

Statistical differences were found between the Vanderbijlpark and Ventersdorp with regards to Species Richness and Abundance (Table 4.10). The abundance, Shannon-Wiener Diversity Index and the Species Richness indicated a medium level of practical implication between the urban green spaces of Vanderbijlpark and Potchefstroom. This was also the case with regards to the Shannon-Wiener Diversity Index between green spaces of Vanderbijlpark and Ventersdorp. A high practical significance was found for the abundance between Vanderbijlpark and Potchefstroom and Ventersdorp, respectively. With regards to the Species Richness of Vanderbijlpark and Ventersdorp a high practical significance was found (Table 4.10).

Table 4.10: Linear Mixed Model analysis indicating statistical significant differences between urban green spaces of all the settlements (Vanderbijlpark=Vdbp, Potchefstroom=Potch & Ventersdorp=Venters).

Urban settlements		**Species Richness**	**Total Indivi-duals**	**Margalef's Diversity Index**	**Pielou's Eveness**	**Shannon-Wiener Diversity Index**	**Simpsons Diversity Index**
Vdbp X Potch	Sig.(p)	0.16	0.10	0.66	0.99	0.30	0.87
	Effect size (d)	0.63**	0.72**	0.30*	0.08	0.46**	0.18
Vdbp X Venters	Sig.(p)	0.04*	0.03*	0.50	0.77	0.07	0.91
	Effect size (d)	0.83***	0.93***	0.37*	0.22*	0.68**	0.16
Potch X Venters	Sig.(p)	0.89	0.88	0.99	0.93	0.81	1.00
	Effect size (d)	0.20*	0.22*	0.07	0.14	0.22*	0.02

indicates P-values (P> 0.05) of statistical significance. effect size (d) (d>0.2 =low impact, d>0.5 = medium impact** and d>0.8= high impact ***)

4.5.1.5 Correlation between urban green space patch sizes

According to Begon *et al.* (2006) the most important part of patch dynamics is migration. Without migration a patch becomes a closed entity with a limited lifespan. Factors that influence the migration of species to patches are the mobility of the species in question, size and shape of the patches, quality of the patches and the distance between the patches.

It was required to ensure that all of the results regarding the urban settlements and the predacious arthropods were not influenced by patch size therefore, correlations were done. A summary of the patch sizes for each of the urban green spaces used is given in Table 4.11.

Table 4.11: A Summary of urban green space size across the investigated urban settlements. Size of each urban patch is given in square meters.

Patch size (m^2) for each of the green spaces in the different settlements.						
Nr.	Vanderbijlpark		Potchefstroom		Ventersdorp	
	Fragmented Grasslands	Ruderal areas	Fragmented Grasslands	Ruderal areas	Fragmented Grasslands	Ruderal areas
1	87 408	44 903	17 856	10 730	38 287	3078
2	38 659	42 923	7022	60 907	6017	2160
3	24 577	33 543	15 604	10 302	16 926	2563
4	94 058	27 118	87 849	33353	16 508	17062
5	733 253	369 960	24 986	43556	135 988	5048
6	80 277	16 903	55 888	37372	60 185	33 536
7	46 187	10 543	123 919	4666	14 257	5638

The correlation coefficients for each of the predator species as well as the overall predator abundances and diversity was investigated across the entire urban environment for all the settlement and no significant correlations were found (Table 4.12). The closest correlation between urban patch size and the predacious arthropods was for the total individuals per plot (r = 0.379), which is to be expected as a larger patch would allow for a greater variety of habitats as well as less intense competition between these predators for prey. However, this is a very weak positive correlation and can easily be disregarded as the correlation coefficient is quite lower than the halfway threshold (r>0.5).

It should be noted that lacewings were only tested with the total individuals per plot criteria as lacewings are not very diverse and only consisted of two species during the duration of the study. They were however, abundant.

Table 4.12: Correlation coefficients for patch size and the overall urban environment that includes the fragmented grasslands and the ruderal areas.

Overall urban environment					
	Overall Predator diversity	Aranea: Spiders	Chrysopidea: Lacewings	Mantodea: Praying Mantids	Coccinelidae: Ladybirds
Species Richness per sample plot	0,262	0,240	n/a	0,315	-0,110
Total Individuals	0,379	0,314	-0,083	0,268	-0,053

per sample plot					
Margalef's Species Richness Index	0,091	0,141	n/a	0,041	-0,204
Pielou's Evenness	-0,188	-0,097	n/a	0,033	-0,233
Shannon-Wiener Diversity Index	0,080	0,147	n/a	0,051	-0,103
Simpson's Diversity Index	-0,076	0,030	n/a	0,037	-0,200

These results are verified by the scatter plots which visually represent the correlation coefficients between patch size, Species richness per plot, total individuals per plot and the Shannon-Wiener Diversity Index (Figure 4.18).

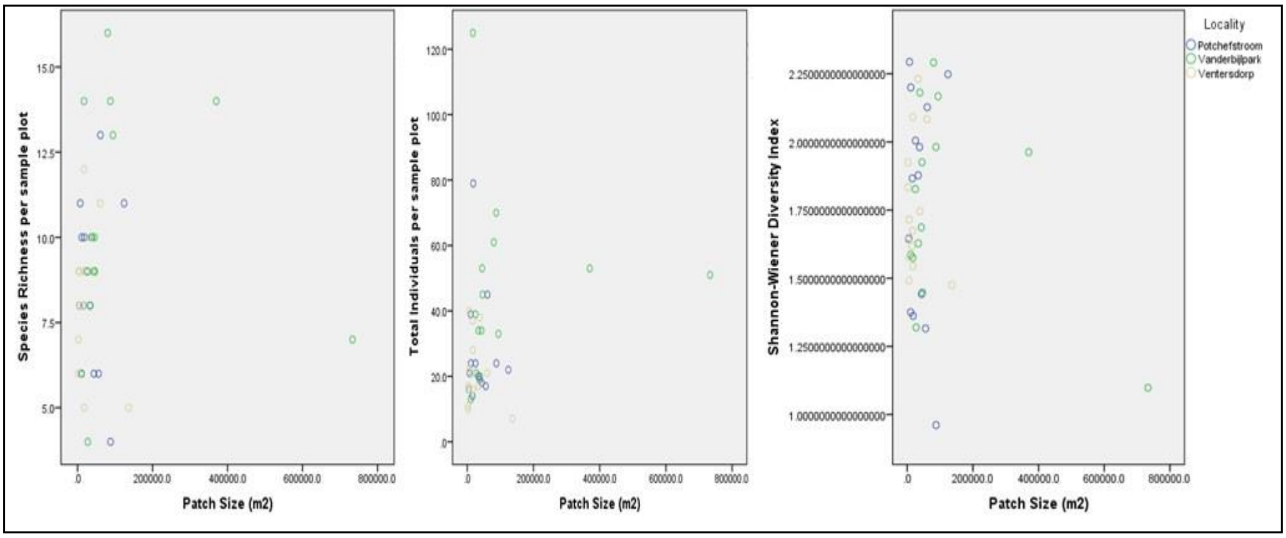

Figure 4.18: Overall urban green space patch size showing no correlations towards abundance or diversity.

When the urban green spaces were separated and investigated individually i.e. fragmented grasslands (Tabel 4.12) and ruderal areas (Tabel 4.13) for correlations , none were found for the fragmented grasslands. This indicates that the size of the patch that was sampled had no effect on the abundance or diversity of the predacious arthropods (Figure 4.19). This was despite some the patches that were clearly outliers within the data.

Table 4.13: Correlation coefficients for patch size the fragmented grasslands.

Fragmented Grasslands					
	Overall Predator diversity	Aranea: Spiders	Chrysopidea: Lacewings	Mantodea: Praying Mantids	Coccinellidae: Ladybirds
Species Richness per sample plot	0,020	-0,036	n/a	0,315	-0,256
Total Individuals per	0,150	0,111	-0,195	0,268	-0,173

sample plot					
Margalef's Species Richness Index	-0,136	0,084	n/a	0,041	-0,336
Pielou's Evenness	-0,086	0,060	n/a	0,033	-0,309
Shannon-Wiener Diversity Index	-0,099	0,058	n/a	0,051	-0,269
Simpson's Diversity Index	-0,083	0,088	n/a	0,037	-0,332

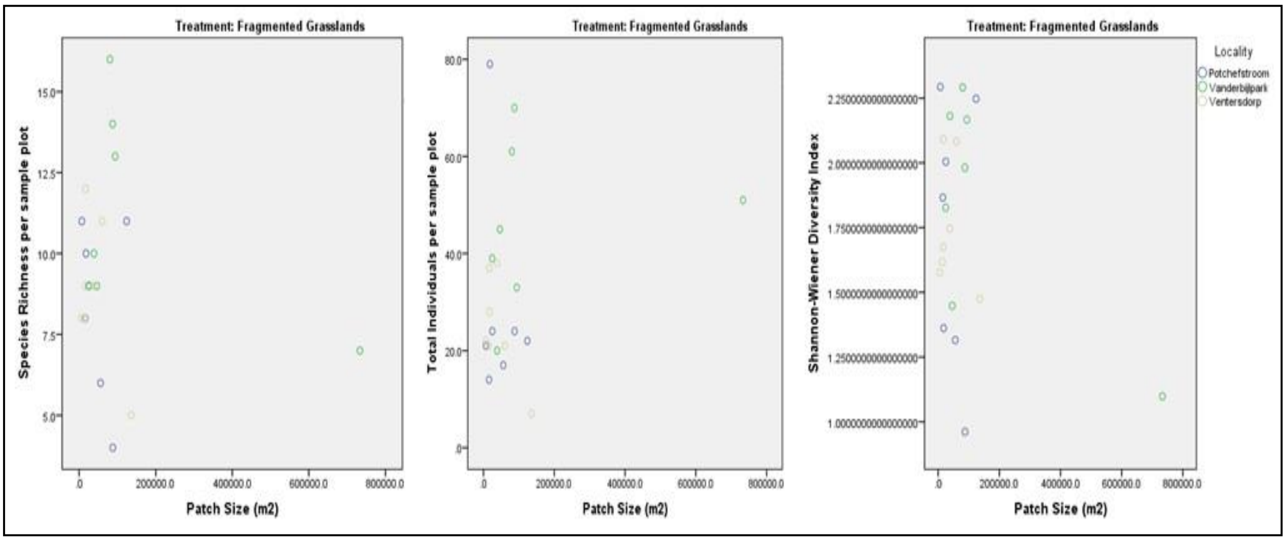

Figure 4.19: Patch size correlation for fragmented grasslands between, species richness per plot, total individuals per plot and the Shannon-Wiener Diversity Index.

The ruderal areas did show a minimal positive correlation (r =0.510) between patch size and the abundance (total individuals per plot) for one predator; the coccinellids (Table 4.14).

Table 4.14: Correlation coefficients for patch size and the ruderal areas.

Ruderal areas					
	Overall Predator diversity	Aranea: Spiders	Chrysopidea: Lacewings	Mantodea: Praying Mantids	Coccinellidae: Ladybirds
Species Richness per sample plot	0,415	0,283	n/a	0,232	0,289
Total Individuals per sample plot	0,596	0,234	0,040	0,207	0,510
Margalef's Species Richness Index	0,178	0,086	n/a	0,008	0,064
Pielou's Evenness	-0,171	0,088	n/a	0,008	0,032
Shannon-Wiener Diversity Index	0,273	0,172	n/a	0,046	0,248
Simpson's Diversity Index	-0,097	0,128	n/a	0,008	0,063

The scatterplots of correlation coefficient for patch size and, species richness per plot, total individuals per plot and the Shannon-Wiener Diversity Index shows the weak positive correlation (Figure 4.20).

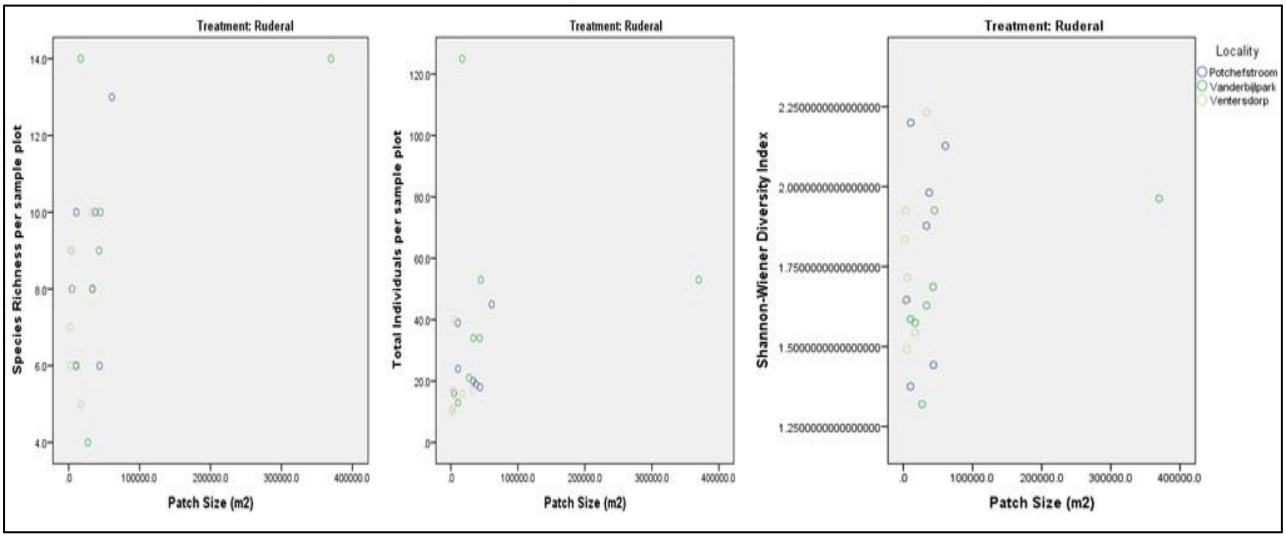

Figure 4.20: Patch size correlation for ruderal between the following criteria of abundance and diversity: Species richness per plot, total individuals per plot and the Shannon-Wiener Diversity Index.

4.5.2 Agricultural ecosystem

4.5.2.1 Overall Predator diversity

The diversity of predators in the agro-ecosystem differs between the maize field which has a high anthropological disturbance effect and the surrounding landscape.

A total of 299 spiders, 95 lacewings, 249 coccinellids and 197 praying mantids were collected during sample collection in the agro-ecosystem, which included the maize fields, field margin and untransformed grasslands. The highest abundance of predator individuals were collected in the untransformed grasslands (395 individuals) and the field margin (354 individuals) (Figure 4.21).

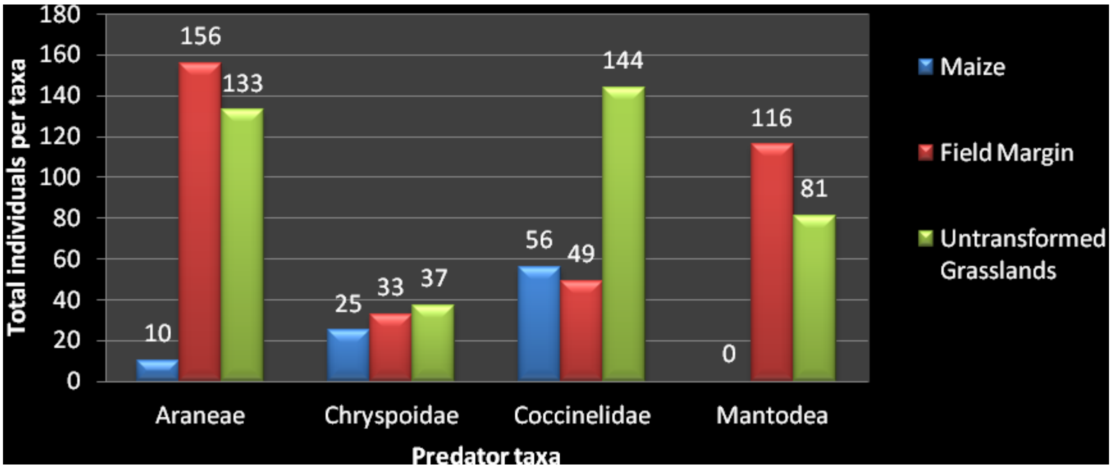

Figure 4.21: Overall predator abundance collected within the agro -ecosystem.

The spiders collected formed part of a total of 15 different species, lacewings only two species, coccinellid six species and the praying mantids eight species of which most were found in the field margin of the agro-ecosystem (Figure 4.22).

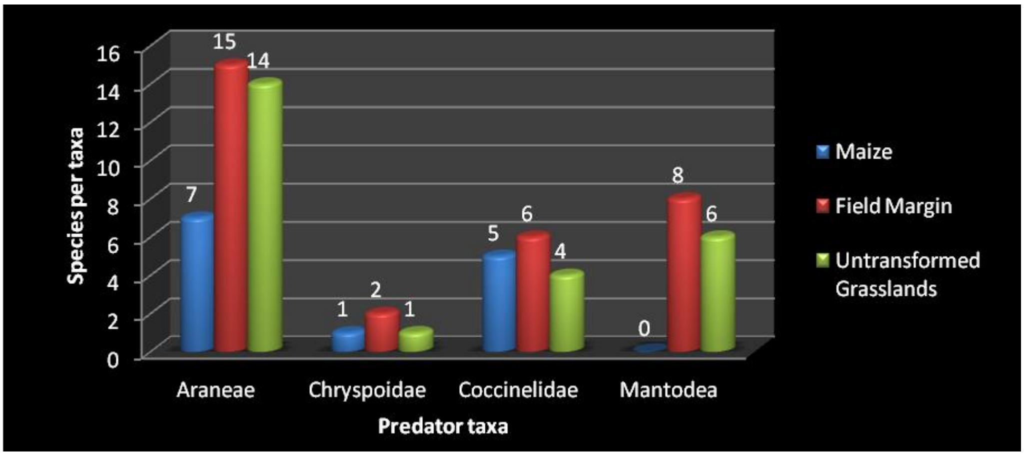

Figure 4.22: Overall predator richness collected within the agro -ecosystem.

This raw data indicates that these predators prefer the surrounding agricultural areas instead of the maize fields.

4.5.2.2 Community composition and assemblages within the agro-ecosystem

The community composition and assemblage within the agro-ecosystem of the selected predators revealed no visible community clusters which indicate that the composition and assemblage of the predator communities did not differ within the agro-ecosystem (Figure 4.23). Even though the beta diversity of the surrounding landscapes seems to be somewhat higher it is obscured by the outliers in the data of all the treatments and therefore the beta diversity is the same for all treatments. It should be noted however that the maize field is clustering in a minimalistic way, but due to the mobility of these predators this cluster could become hidden.

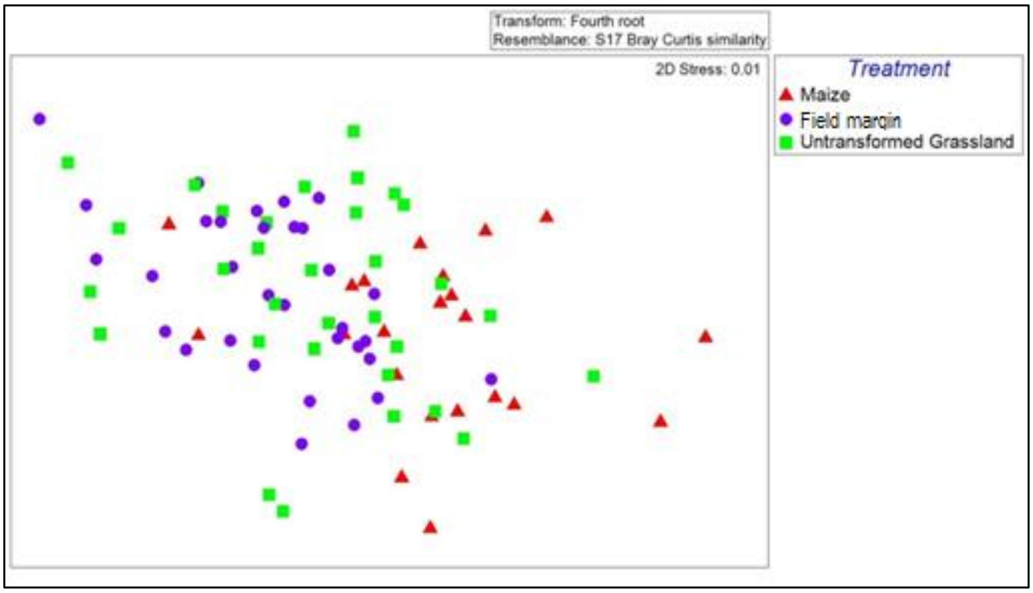

Figure 4.23: Predacious arthropod community composition and assemblage of the maize field and surrounding landscapes.

Dissimilarity within the agro-ecosystem was determined by SIMPER analysis and indicated an overall dissimilarity of 88.47%.The contributing species were different than the species responsible for the dissimilarity within the urban ecosystem. The species that cumulatively contributed to ±50% of the dissimilarity in the agro-ecosystem were: *Misumenops rubrodecoratus* (spider) 16.6%, *Chrysoperla congrua* (lacewing) 9.7%, *Cheilomenes lunata* 9.5% (Coccinellidae) and *Hippodamia variegata* 8.8% (Coccinellidae).

4.5.2.3 Statistical differences within the agro-ecosystem

Statistical significant differences were found for all abundance and diversity indices (Table 4.16). All of the significant differences were with regards to the field margin and untransformed grasslands and the maize field (Table 4.15; Figure 4.25). The effect size that indicates the practical significance of statistical differences are high for all criteria with regards to the maize field which differs from the field margin and untransformed grasslands, respectively. The effect size of the abundance of predators had the smallest practical significance but it is still regarded as high according to Ellis & Steyn (2003).

Table 4.15: Linear Mixed Model analysis indicating statistical significant differences between the zones within the agro-ecosystem.

Treatment		**Diversity Indices**					
		Species Richness	Total Individuals	Margalef's Species Richness Index	Pielou's Eveness	Shannon-Wiener Diversity Index	Simpson's Diversity Index
Field margin VS. Maize field	Sig.(p)	0.00*	0.00*	0.00*	0.00*	0.00*	0.00*
	Effect size(d)	1.33(c)	0.85(c)	1.46	1.08(c)	1.47(c)	1.22(c)
Field margin VS. Un-transformed grasslands	Sig.(p)	1.00	0.92	0.98	0.91	0.99	0.92
	Effect size(d)	0.05	0.13	0.07	0.14	0.06	0.13
Maize field VS. Un-transformed grasslands	Sig.(p)	0.00*	0.00*	0.00*	0.00*	0.00*	0.00*
	Effect size(d)	1.38(c)	0.98(c)	1.53(c)	1.22(c)	1.54(c)	1.36(c)

The P-values (P> 0.05) indicates a value of statistical significance and is indicated by (*). The effect size (d) is indicated as follow: (d>0.2 =low impact (a), d>0.5 = medium impact (b) and d>0.8= high impact(c).

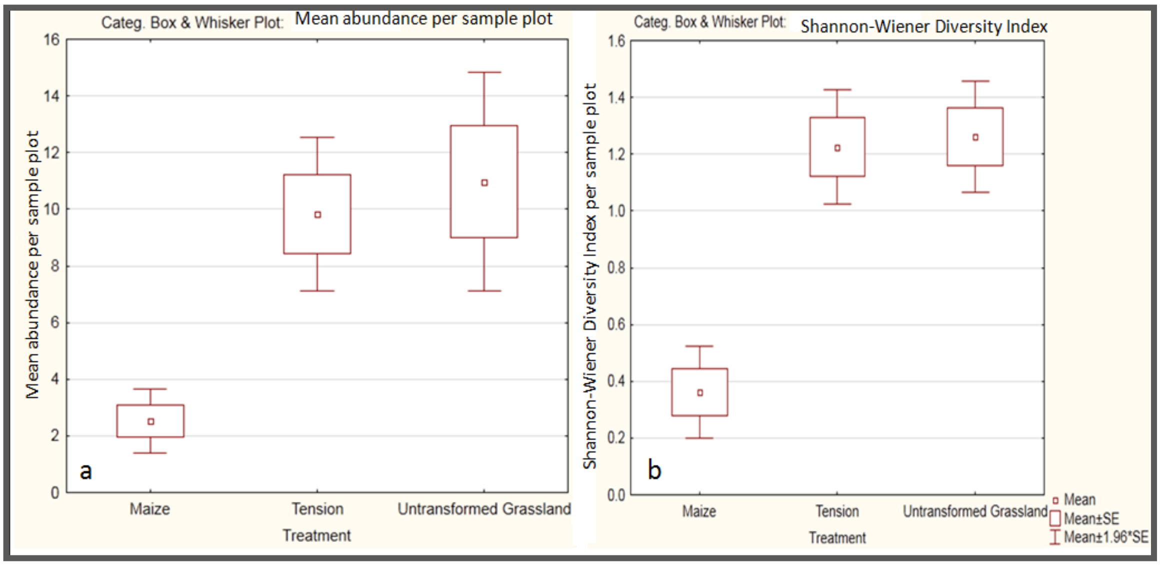

Figure 4.24: Predator presence within the different zones of the agro ecosystem with regards to the (a): Total individuals (Abundance) per sample plot. (b) Shannon-Wiener Diversity index.

4.6.3 Comparing the agro & urban ecosystems

4.6.3.1 Overall predator diversity and abundance between the two types of disturbances

A total of 1123 spider individuals belonging to 53 species, 126 lacewings of two species, 642 coccinellid of six species and 270 mantids of eight species were collected (Figure 4.25). The most spider and coccinellid individuals were collected in the urban green spaces, whilst the lacewings and mantids were more abundant in the agricultural system.

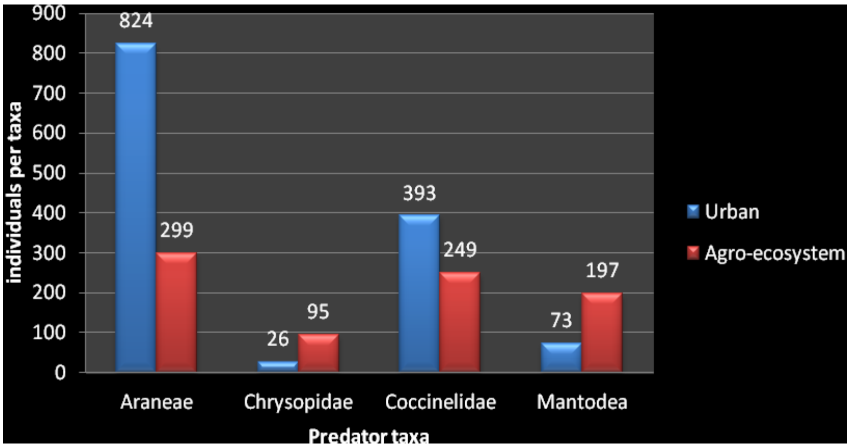

Figure 4.25: Individuals (abundance) per taxa collected in the agro- and urban ecosystems.

Species richness was overall higher for urban areas than agro-ecosystems. The agro-ecosystem however has a quite rich variety of predacious arthropods as it only had about 4 species less than the urban area (Figure 4.26).

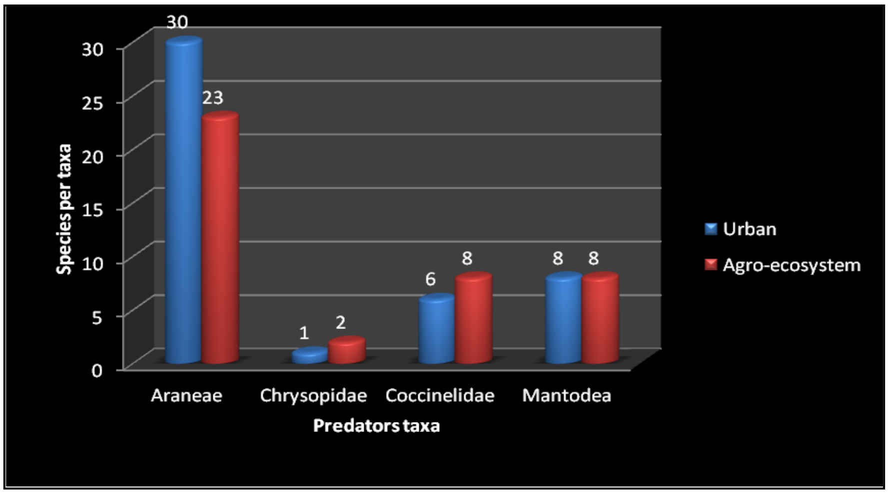

Figure 4.26: Species richness within each taxon of the predators in both disturbances.

4.6.3.2 Community composition and assemblages between the two types of disturbances

The two disturbances clearly influence the predator arthropod communities as clear clusters can be identified for the agro-ecosystem and the urban environment (Figure 4.27).

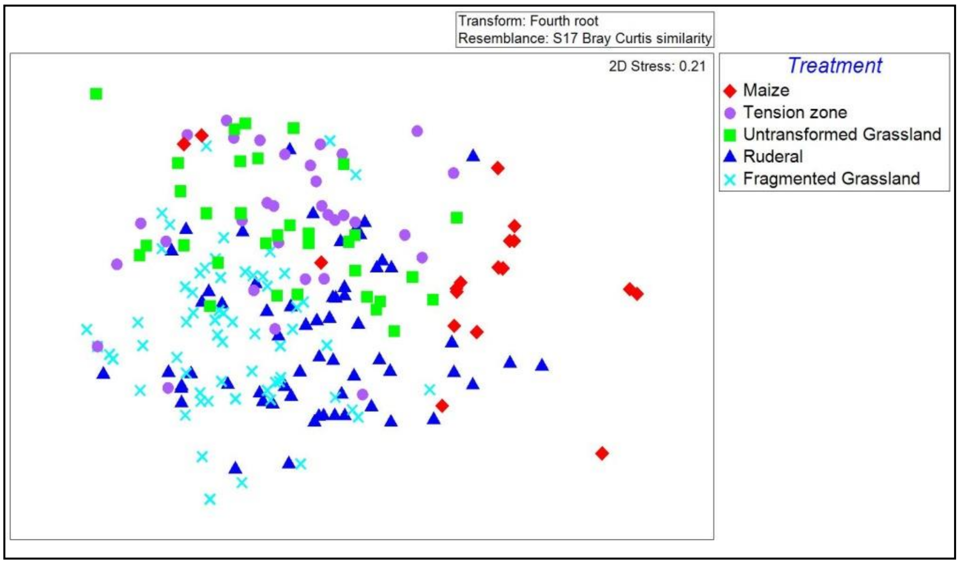

Figure 4.27: Community composition and assemblages for all predacious arthropods along the disturbance gradient which included the urban and ruderal areas.

The dissimilarity between these two disturbances can be attributed to three spider species and two coccinellid species (Table 4.16).

Table 4.16 - Summary of species responsible for cumulative contributions between the two disturbances. All species contributing cumulatively to ± 50% was presented.

Predator Species	**Average dissimilarity (%)**	**Contribution (%)**	**Cumulative (%)**
1. *Tibellus holidayi* (Araneae)	12.4	14.47	14.47
2.Misumenops rubrodecoratus (Araneae)	10.15	11.85	26.32
3. Cheilomenes lunata (Coccinellidae)	9.532	11.12	37.44
4.Runcinia flavida (Araneae)	7.128	8.317	45.76
5. Hippodamia variegata (Coccinellidae)	5.108	5.96	51.72

Of the species that are responsible for the dissimilarity between the urban and agro-ecosystem, the dominating species in the urban ecosystem contributed about 26.5%, and only consists of only two of these species. This could indicate the level of dominance by species such as *Misumenops rubrodecoratus* (Araneae) and *Cheilomenes lunata* (Coccinellidae).

4.6.3.3 Statistical differences between the two types of disturbances

Statistical significant differences were found between the maize field and the ruderal areas (with the same disturbance intensity) with regards to all diversity indices except for the species per sample plot (richness) as well as total individuals per sample plot (abundance) (Table 4.17; Figure 4.28). A similar situation was seen for the maize field and fragmented grasslands that differed significantly for all indices except predator abundance. As previously seen the maize field also significantly differed from the field margin and untransformed grassland with regards to all the abundance and diversity indices (Figure 4.28).

Table 4.17: Linear Mixed Model analysis indicating statistical significant differences between the different disturbance intensities with regards to the urban and agro-ecosystem. Disturbances indicated as Maize field= MF, Field margin= FM, Untransformed grasslands= UG, Fragmented grasslands= FG and Ruderal areas= RA.

		Species Richness	**Abundance**	**Margalef Diversity Index**	**Pielou's Even-ness**	**Shannon-Wiener Diversity Index**	**Simpson's Diversity Index**
FM & MF	Sig. (p)	0.00*	0.01*	0.00*	0.00*	0.00*	0.00*
FM & UG	Sig. (p)	1.00	1.00	1.00	1.00	1.00	1.00
FM & FG	Sig. (p)	1.00	1.00	1.00	0.91	0.99	0.87
FM & RA	Sig. (p)	0.98	1.00	0.98	1.00	0.97	1.00
MF & UG	Sig. (p)	0.00*	0.00*	0.00*	0.00*	0.00*	0.00*
MF & FG	Sig. (p)	0.02*	0.39	0.00*	0.00*	0.00*	0.00*
MF & RA	Sig. (p)	0.08	0.55	0.00*	0.00*	0.00*	0.00*
RA & UG	Sig. (p)	0.98	1.00	0.98	1.00	0.98	1.00
RA & FG	Sig. (p)	0.90	1.00	1.00	0.19	1.00	1.00
FG & UG	Sig. (p)	1.00	1.00	1.00	0.60	0.99	0.64

*indicates P-values (P> 0.05) of statistical significance.

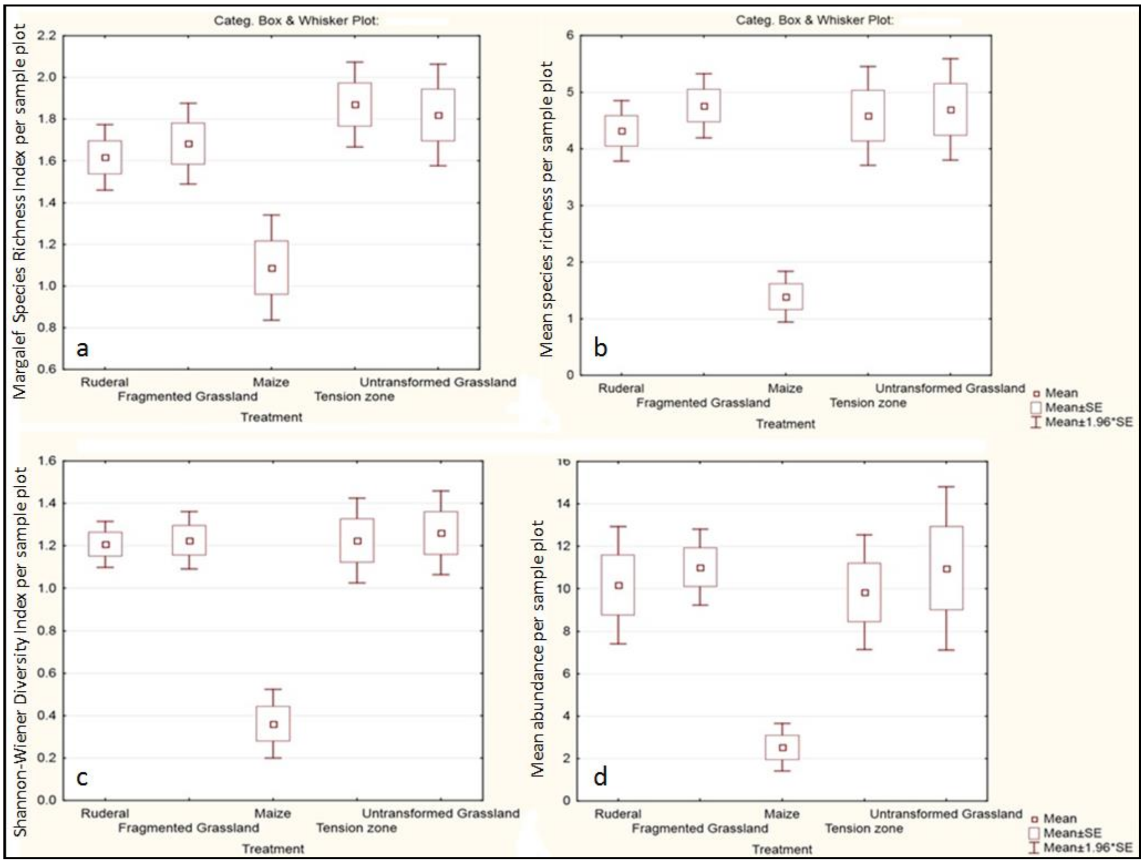

Figure 4.28: Statistical difference in all the criteria between maize field and all other disturbance intensities. (a) Margalef Species Richness Index. (b) Species Richness per sample plot. (c) Shannon-Wiener Diversity Index. (d) Mean Abundance (total individuals) per sample plot.

The field margin that has an intermediate disturbance intensity indicated to be closer with regards to diversity and evenness to the untransformed grasslands (minimal disturbance intensity) and was not significantly less diverse than the two urban green spaces. This could suggest that the field margin is not as disturbed as expected.

The same dominating spider species (*Misumenops rubrodecoratus)* was found for all disturbances within the agro-ecosystem i.e. maize field (4 specimens), field margin (156 specimens) and untransformed grasslands (45 specimens). The dominating coccinellid species (*Cheilomenes lunata)* was the same for the field margin (29 specimen) and untransformed grassland (81 specimen) but not for the maize field. The species dominating in the maize field was *Hippodamia variegata* (39 individuals). The dominating praying mantid species differed for the field margin (48 individuals of the species *Pyrogomantis rhodesica*) and the untransformed grasslands (81 individuals of the species *Galepsus* sp.). The most abundant species of lacewing collected did not differ between the different zones in the agro-ecosystem.

Ruderal areas were more closely related to untransformed grasslands despite its higher disturbance intensity. This was the same for the ruderal areas and the fragmented grasslands but Pielou's Evenness indicated an insignificant but interesting p-value of 0.19. Thus a dominating species could be causing a lower evenness value.

Fragmented grasslands had a single spider species (*Tibellus hollidayi*) of which 245 were collected in comparison with a total of 57 in the ruderal areas. The ruderal areas had a single coccinellid species (*Cheilomenes lunata*) of which 137 were collected and only 33 of them were collected in the fragmented grasslands. The ruderal areas had a total of 304 spiders (18 species) and 316 coccinellid (7 species) while in the fragmented grasslands 564 spiders (25 species) and 74 coccinellids (4 species) were collected. The spider with a total of 63 individuals (*Misumenops rubrodecoratus*) that dominated in the ruderal areas was not the same as the one in the fragmented grasslands. The coccinellid dominating in both urban green spaces was however the same. Praying mantids and lacewings were very similar in both fragmented grasslands and ruderal areas. It should be noted that despite the similar number of individual praying mantids that were collected, the species dominating in the ruderal area was *Galepsus sp*. in contrast to the dominating species in the fragmented grasslands (*Epsiscopus chalybeus*). Based on these numbers ruderal areas could be seen as the more even urban green space. Predator species list that indicates species found in each environment is provided in Table 4.18.

Table 4.18: Predacious arthropod species recorded for all three ecosystems. Species recorded from a single ecosystem is indicated with * and from only one land-use with #.

Order	Family	Species	Urban ecosystem		Agro-ecosystem		Grassland ecosystem
			Ruderal	Fragmented grasslands	Maize field	Field margin	Untransformed grasslands
Araneae	Eutichuridae	*Cheiracanthium* sp.*	3	2	-	-	-
Araneae	Theridiidae	*Enoplognatha* sp. 1	5	3	1	7	3
Araneae	Salticidae	*Festucula lawrencei*	20	48	-	-	1
Araneae	Araneidae	*Hypsosinga* sp. 3	7	12	-	6	13
Araneae	Araneidae	*Larinia bifida*	-	-	-	6	11
Araneae	Araneidae	*Larinia natalensis*	-	6	-	2	4
Araneae	Theridiidae	*Latrodectus renivulvatus*	-	4	-	1	-
Araneae	Thomisidae	*Misumenops rubrodecoratus*	58	72	4	86	45
Araneae	Thomisidae	*Monaeses paradoxus*	16	15	-	2	1
Araneae	Thomisidae	*Monaeses pustulosus*	4	11	-	12	15
Araneae	Araneidae	*Nemoscolus* sp.*	1	5	-	-	-
Araneae	Araneidae	*Neoscona subfusca**	4	3	-	-	-
Araneae	Oxyopidae	*Oxyopes hoggi* *#	1	-	-	-	-
Araneae	Oxyopidae	*Oxyopes longispinosus*	-	2	1	-	-
Araneae	Oxyopidae	*Oxyopes schenkeli*	-	1	-	1	-
Araneae	Araneidae	*Pararaneus* sp. 1 *#	-	2	-	-	-
Araneae	Salticidae	*Pellenes bulawayoensis*	-	1	1	1	2
Araneae	Oxyopidae	*Peucetia striata* *	3	1	-	-	-
Araneae	Thomisidae	*Runcinia flavida*	62	89	-	10	14
Araneae	Thomisidae	*Stiphropus bisigillatus*	-	1	1	2	1

Araneae	Thomisidae	*Synema decens*#*	-	-	1	-	-
Araneae	Philodromidae	*Thanatus atlanticus*	-	3	-	1	1
Araneae	Araneidae	*Thanatus atlanticus**	2	4	-	-	-
Araneae	Theridiidae	*Theridion pictum**	11	12	-	-	-
Araneae	Salticidae	*Thyene semiargentea*	1	4	-	1	-
Araneae	Salticidae	*Thyene thyenioides**	5	3	-	-	-
Araneae	Salticidae	*Thyenula* sp. 1	9	15	-	-	4
Araneae	Philodromidae	*Tibellus hollidayi*	57	245	1	18	18
Neuroptera	Chrysopidae	*Chrysoperla congrua*	15	11	25	32	37
Neuroptera	Chrysopidae	*Italoshrysa similis*#*	-	-	-	1	-
Coleoptera	Coccinellidae	*Buleae anceps*	27	30	-	11	2
Coleoptera	Coccinellidae	*Chelomenes lunata*	136	33	9	29	81
Coleoptera	Coccinellidae	*Exochomus flavipes*	36	4	1	5	42
Coleoptera	Coccinellidae	*Harmonia axyridis*#*	5	-	-	-	-
Coleoptera	Coccinellidae	*Hippodamia variegata*	86	-	39	1	19
Coleoptera	Coccinellidae	*Lioadalia flaromaculata*	20	10	6	2	-
Coleoptera	Coccinellidae	*Micraspis cf. comma*	4	-	-	1	-
Mantodea	Mantodea	*Compsothspis natalica*	-	2	-	1	1
Mantodea	Mantodea	*Entella transvaalica*	2	-	-	3	1
Mantodea	Mantodea	*Epsiscopus chalybeus*	-	22	-	33	23
Mantodea	Mantodea	*Galepsus* sp.	28	1	-	27	41
Mantodea	Mantodea	*Harpagomantis discolour*	3	1	-	1	-
Mantodea	Mantodea	*Hemipusa capensis*#*	-	-	-	1	-
Mantodea	Mantodea	*Pyrgomantis rhodesica*	-	1	-	48	10
Mantodea	Mantodea	*Tendora Iringana*	3	11	-	2	5

4.7 Discussion

4.7.1 Urban ecosystem

4.7.1.1 Overall predator diversity

The selected predatory arthropod s of this study were diverse and abundant in the urban habitats invetsigated. The results yielded only one invasive species which belonged to the Coccinellidae and is known as *Harmonia axyridis.* Coccinellids had a higher species richness in ruderal areas than fragmented grasslands which can be expected as they were also more abundant in this urban green space type.

A possible explanation for the relatively high abundance of arthropod predators in the urban settlements could be the urban heat island effect. A study conducted in Phoenix Arizona showed that the urban heat island effect increased the night temperatures with up to 5°C and the daily temperature with 3.1°C (Baker *et al.*, 2002). These increases could have a huge impact on certain arthropod species. Sweency and Vannote (1978) stated that there is a correlation between temperature, body size and fecundity for hemi-metabolic aquatic arthropods. This is however subjective to each species basic requirements (Sweency and Vannote, 1978) , thus the urban heat island effect might contribute to optimal temperatures for some of the predacious arthropods included in this study which could resulted in higher fecundity thus explaining why there are more individuals of some taxa in a larger settlement with a higher temperature.

4.7.1.2 Community composition and assemblages within urban green space

4.7.1.2.1 Overall community composition

It should be noted that a study done by Kutschbach-Brohl *et al.* (2010) on fragmented grasslands at the J.F.K International airport indicated that spiders were one of the most abundant non insect predators recorded in their study and can thus possibly be responsible for the differences in the species assemblages in the two urban green space types of this study. Community differences can also be due to habitat fragmentation, plant productivity and management intensity (Burkman and Gardiner, 2014).

4.7.1.2.2 Community composition of three different settlements

Clusters for the species assemblages were found with regards to the two urban green space types i.e. fragmented grasslands and the ruderal areas. These differences in species assemblage could be attributed to different management practices (Van der Walt *et al.*, 2013) or age, shape and edge effect of the urban patches (Bolger *et al.*, 2000). Van der Walt *et al.* (2013) stated that the quality of the patches of fragmented grasslands has a lesser effect on the diversity and that management practices such as mowing have a greater effect on these urban green spaces, with regards to plant communities, which increases species diversity.

4.7.1.2.3 Species assemblage of urban green spaces within the urban settings

The difference in predatory arthropod species assemblage between the three different settlements based on their size indicated that the largest and the smallest settlements had different species assemblages. This could be due to the cycles of urbanization that are defined by Cohen (2006). It is stated that it is expected that large settlements grow faster but in reality their population size actually decreases once a city reaches a certain size, thus meaning a lesser impact of the urban disturbance could be expected. Vanderbijlpark exhibits lower beta diversity when compared to that of Potchefstroom and Ventersdorp. This could be ascribed to disturbance, in this case urbanization (which in turn includes pollution, fragmentation and habitat destruction), which in effect is expected to increase with the size of a settlement (Miller and Spoolman, 2012; Sarzynski, 2012).

Berry (2008) states that the size of a city and urban densities had a significant effect on the climatic and hydrological consequences of that environment, which can lead to further disturbance effects on arthropods, especially, as they are very sensitive to changes in climate. Species composition found within the two types of urban green spaces of Vanderbijlpark differed from Ventersdorp and to some extent from Potchefstroom. However, within Vanderbijlpark itself, the species composition was more or less the same, despite the different types of urban green spaces. This indicates that other factors could be influencing the species assemblage beyond the disturbance intensity. Bolger *et al.* (2000) found that the size, age and shape of the patches (elongated shape has a bigger edge effect than that of a round shaped patch) sampled has an effect on arthropods.

The arthropod species assemblage of Potchefstroom has higher beta diversity than Vanderbijlpark which could suggest that the effect of urbanization is not as severe in this medium settlement as it is in the larger settlement. In contradiction with this statement, Bettencourt and West (2010) found that larger settlements utilise their resources more efficiently and are more eco-friendly than smaller settlements. This could therefore suggest that a smaller settlement such as Ventersdorp could have had a lower disturbance intensity or that larger settlements such as Potchefstroom utilised their resources in such a way that it decreased the disturbance intensities within their ecosystems.

The arthropod species assemblage indicated that Ventersdorp could have a higher beta diversity which is expected as the effect of urbanization should be lower in this small settlement. The suggested higher beta diversity of Ventersdorp could be due to agricultural areas surrounding Ventersdorp which could be the source for this urban sink. The agricultural areas around the medium and large settlement could also be the source for these urban environments even though the beta diversity of these settlements is somewhat lower. This possible sink-source effect was described by Sattler *et al*. (2011).

The clustering in species assemblage with regards to the two types of urban green space could suggest the effect is as a result of the larger urbanised areas. However, according to Bhatta (2010) settlements with a higher economic growth could possibly have more disturbances as this leads to urban sprawl. Vanderbijlpark might be the largest with the theoretical largest disturbance affect; however, this could be disproved if a settlement like Potchefstroom for example has a greater economic growth rate with more urban sprawl which could result in more activities such as construction which could increase the utilisation of more vacant urban spaces.

4.7.1.2.4 Community composition and assemblage of individual settlements

Ventersdorp, which is a small settlement, had little to no effect on the predator arthropod community and so the surrounding agricultural areas might have a high source influence for this urban sink. Another possibility is that due to the size of this settlement, green spaces are in a closer proximity to each other thus easing movement for arthropods in this urban matrix. The ease in mobility for the predators of Ventersdorp could thus increase the source-sink effect, especially if the urban patches create a corridor through this town.

It should be noted that settlements can be seen as environments of their own and could thus be in different stages of ecological succession, depending on the last major disturbance as well as the intensity of this disturbance. The disturbance could be intermediate and thus be decreasing a single species from dominating. This could also influence the community composition and assemblages and cause a settlement to adhere to the intermediate disturbance hypothesis (Conelle, 1978).

4.7.1.3 Differences between the two types of urban green spaces for all urban environments

4.7.1.3.1 Overall diversity between the two types of urban areas.

Addressing the earlier differences found Vanderbijlpark has a significantly higher abundance than Ventersdorp. This could be due to many micro scale circumstances such as favourable habitat and resources, less predation or perhaps less inter- species competition (Burkman and Gardiner, 2014). Less competition could be a result of frequent disturbances discouraging one species from dominating but it could also cause one species to dominate as it could be the most resilient predator species (Dugatkin, 2009).

As there was only one criterion that differed significantly in the ruderal areas it indicates that these vacant lots either experience the same intensity of urbanization disturbance regardless of the size of the settlement or that these areas all have the same habitat specific disturbance intensity. It could be that the management practises in these areas are similar, which as stated Cilliers and Bredenkamp (1999) is a determining factor for plant communities.

However, due to the fact that Vanderbijlpark had the most individuals in both of its urban green spaces could indicate that this urban environment is unique. This might be due to higher temperatures which could be due to its size; this is known as the urban heat island effect (Miller and Spoolman, 2012). Arthropods as stated are easily influenced by temperature (Chidawanyika *et al*., 2012), thus higher temperatures could explain why this urban environment has a higher abundance (Gardiner *et al.*, 2013).

Meineke *et al.* (2013) found that the pest insect *Parthenolecanium quercifex* (Hempitera: Coccidae) had a higher abundance in urban areas that were hotter. Further investigation also indicated that these effects found were induced by temperature as these pests had a significantly higher abundance in controlled greenhouses. The urban heat island effect might thus lead to higher abundances of food resources and so increasing predator presence.

The urban green space types could have dominant species influencing the Pielou's evenness thus resulting in statistical difference. If a dominant species is present in these areas it could explain this phenomenon and could be an indication that some species are more resilient. The dominant species found in the urban areas were *Runcinia flavida (*Simon, 1881), *Tibellus hollidayi* (Lawrence, 1952) and *Cheilomenes lunata* (Fabricius, 1775) with a total of 151, 302 and 170 individuals collected in the urban environment respectively.

4.7.1.4 Differences across the urban settlements (both urban green spaces)

Vanderbijlpark is the largest of the three settlements which could have two implications: Higher possible urbanisation effect but also more urban vacant land availability which increases the ability of corridors to exist. As previously discussed the urban heat island effect (Baker *et al.*, 2002) could have played a role in the abundance of some of the species. This is due to an increased use of materials that retain heat such as asphalt, buildings and concrete which are more abundant in a larger settlement like Vanderbijlpark than in the two smaller settlements (Potchefstroom and Ventersdorp). However, factors such as the management practices of vacant lots (Cilliers and Bredenkamp, 1999) or the age, area and edge effect (Bolger *et al.*, 2000) could influence the predacious arthopod population dynamics.

The effect sizes show that size of the settlement could have a practical effect on the predator arthropods found in these environments and that urbanisation is not necessarily linear to size. However, it seems to be more complex and more factors are responsible for the variance between different urban sized settlements i.e age, area, edge effect and heat island effect.

4.7.1.5 Correlation between urban green space patch sizes

Results of this study suggested that larger patches do not indicate a higher diversity of predators. This is validated by a study done by Gibbs and Hochuli, (2002) who reported that smaller patches did not have a decrease in species richness per sampling unit. The difference in size of the patches did, however, contain different ground dwelling arthropod assemblages with regards to predators and parasitiods. Gibbs and Hochuli (2002) suggested that arthropods are influenced by the size of a patch but even more by the decreasing quality of the habitat that is a result of urbanisation which has a larger effect on arthropods, especially arthropods of the higher trophic levels.

The abundance of coccinellids in the ruderal areas of the urban environment is likely due to favourable habitat, refugee site, spawning locality or abundance of prey within the ruderal areas that attract more coccinellid individuals, instead of the patch size itself. A study done by Eyre *et al.* (2003) indicated that herbivorous beetles, especially weevils are influenced predominately by the site drainage and vegetation cover. Thus the high abundance of coccinellids in ruderal areas could be following the same factors as coccinellid could be more abundant in patches that have a high level of food resources (prey).

Ultimately, patch size does influence the species richness and abundance of many plant and animal species, but all the analysis done indicated that the predaceous arthropods of this study seem to be influenced more by contributing factors beyond that of patch size such as the drainage, shape and quality of vegetation cover in the urban patches.

4.7.2 Agricultural ecosystem

4.7.2.1 Overall predator diversity

Predacious arthropods seem to be more abundant in the surrounding agricultural landscape than in the crop fields themselves. Crops, in this case maize fields, have a higher intensity of disturbance as a result of a wide range of anthropological activities or management practices taking place in these environments. Crop fields also have a higher frequency of disturbance than the associated surrounding field margin and untransformed grasslands. A study done by Deuli *et al.* (1999) varified this phenomenon and suggest that the cultivated areas are not as diverse as surrounding landscapes due to the ease of moving between habitats for arthropods which might influence data collected instead of the management practices involved.

4.7.2.2 Community composition and assemblages within the agro-ecosystem

Schaffers *et al.* (2008) suggested that arthropod assemblages are predicted by the plant species composition instead of disturbance and management practices of cultivated areas for seven functional different arthropod species i.e ground dwelling spiders and beetles, grasshoppers, hoppers, hoverflies, bees and weevils. This could explain the lack of distinct assemblages of predator species, as well as their mobility as mentioned previously (Deuli *et al.*, 1999). Botha *et al.* (2015) found that cultivated maize field had a different plant species compostion than the surrounding landscapes of which the species composition was distinct between two biomes. Arthropods of this study seemed to follow the plant species composition as stated by Schaffers *et al.* (2008). No distinct assemblages were found between field margins and untransformed grasslands but with regards the maize field arthropods assemblages did differ. This was the same trend found by Botha *et al.* (2015) as well as artropod assemblages being different between the two biomes of their study.

4.7.1.2.4 Differences within the agro-ecosystem

The significant differences found for all diversity indices indicated that the surrounding landscapes are essential to the abundance and diversity of the selected predacious arthropods. Although the arthropod diversity and abundances between the field margin and untransformed grasslands do not statistically differ from each other, their disturbance intensity is different. The field margin has an intermediate amount of disturbance whilst the untransformed grassland is the closest to a "natural" field that exists. This assumption is in accordance with the findings of the intermediate disturbance hypothesis of Conelle (1978) which could be the reason that these two treatments (field margin and untransformed grasslands) did not differ significantly. This hypothesis would suggest that the field margin has an increased level of species richness and abundance due to frequent disturbances that discourage a dominating species. Rand *et al.* (2006) stated that because of the locality of the field margin (crop edge) a spill over of predators could be expected as it increases their profusion of prey by being able to forage in agricultural areas as well as field margins and also being able to flee from disturbances. Marshall (2002) stated that the field margin is an important refugee site for predatory arthropods and pollinators and also acts as a corridor that eases movement between intensely disturbed cultivated areas. This, however, as previously discussed is predominantly about the vegetation cover and species composition (Deuli *et al.*, 1999).

4.7.3 Comparing the agro- and urban ecosystems

4.7.3.1 Overall Predator diversity and abundance between the two types of disturbances

The number of spider and coccinellid individuals that were collected in the urban environment was higher than the number collected in the agro-ecosystem while the lacewings and mantids were more abundant in the agro-ecosystem than in the urban system. This could indicate that the predator taxa found in the urban environments adapted to different disturbances compared to those whom are found in agricultural environments. This can be expected as different species and possibly different predator groups fulfil different niches within different environments as well as the different habitat requirements that different species prefer for optimum conditions to increase survival and reproductive success (Calcagno *et al*., 2011).

4.7.3.2 Community Composition and Assemblages between the two types of disturbances.

The raw data suggested that urban areas are rich and abundant in predator arthropods, which might emphasise their importance for conservation. However, agricultural areas are important as the species found in urban areas differed from the species collected urban areas.

The difference in species assemblages between the urban and agricultural environments is verified by Sattler *et al.* (2011) that reported the same pattern for arthropods in the Swiss platau. The only treatment that seems to be quite different with regards to community composition and assemblage is the maize field. The urban two treatments, when compared to the tree treatments of the agro-ecosystem, seem to be similar despite the changes found in previous analysis. This suggests that different species are adaptable to different disturbance. Table 4.22 indicates all species that only occur in one of the ecosystems.

Spiders and coccinellids might be more adapted to the urban setting than the lacewings and praying mantids that were more abundant in the agro-ecosystem. This might suggest that these predators are more sensitive to disturbances, or that the plant community assemblages in agro-ecosystems are more habitable for these predators (Schaffers *et al.*, 2008). Spiders specifically are described as good indicators of ecological change and pollution as they are abundant in most habitats and are affected by the top-down effect of tropic levels (Cardoso *et al.*, 2011).

4.7.3.2 Differences between the two types of disturbances

The high disturbance intensity within maize fields suggests that for predator arthropods the disturbance of a maize field is more devastating than that of urban areas, as it was the only treatment to differ significantly for all the diversity indices. These results could verify Sattler *et al.* (2011) theory of the new survival strategy of adapting to urban environments as maize fields are clearly unfavourable. Predacious arthropods are most influenced by anthropological and disturbance activities associated with maize fields as the urban disturbance had little effect, despite the differences in species assemblages, which is likely due to differences in plant community compositions (Schaffers *et al.*, 2008).

This study also indicates that most species found in agricultural areas do occur in nearby urban areas suggesting that a sink-source situation is a possibility as Sattler *et al.*, (2011) suggested, however there were some species that only occurred in either the urban or the agro-ecosystem (Table 4.18).

The dominance of certain species indicated that there could be inter-species competition (Dugatkin, 2009). This is with regards to urban fragments where spiders and coccinellids dominate. However, inter-specific competition could explain why some species of praying mantids are more dominant in the field margin for example and another species in the untransformed grasslands. To quote Burkman and Gardiner (2014) " across rural–urban landscape gradients, natural enemy communities shift toward dominance by habitat generalists and disturbance tolerant species in urban areas compared to rural or natural communities".

In agricultural systems, however, Caprio *et al.* (2015) indicated that spider guilds are affected differently in conventional vineyards in comparison to organic vineyard thus indicating that generalist spider species are more diverse in organic vineyards and influenced by habitat variables. Specialist species, however, were more diverse in conventional vineyards because of their food resources. The dominating species in the fragmented urban grasslands (*Tibellus hollidayi (Lawrence, 1952))* is a free-living generalist active hunter that hides in dense plant cover when inactive while ruderal and agricultural dominating spider species *(Misumenops rubrodecoratus (Millot, 1942*)) is known as a plant dweller especially in crops systems where it prefers tall grasses, neither of these species are specialists or web builders (Dippenaar-Schoeman, 2014). Spider species assemblages are dependent on four factors in unmanaged grasslands as stated by Horváth *et al.* (2015): patch size, plant cover, litter cover and bare ground cover. All of these factors influence specialists and generalist spider species. This explains the decreased abundance in maize fields, for example plant cover might be high but the surrounding landscape has a greater variety and is denser with regards to tall grasses (plant cover) and litter cover.

4.8 Conclusion

Predacious arthropods had distinct community composition and species assemblages with regards to the urban environment i.e. fragmented grasslands and ruderal areas. Statistical differences indicated that the evenness of predacious arthropods differed between the two urban green types investigated in this study. This leads to the rejection of the hypothesis which stated: If arthropods are influenced by disturbances, and the intensity of the disturbance affects their diversity, arthropods in the urban ruderal areas will be less diverse and abundant than in the fragmented grasslands areas where the intensity of the disturbances is lower. Although there were distinct differences there was no evidence that the fragmented grasslands were more abundant or diverse with regards to the selected predacious arthropods.

The maize field was the least abundant and diverse treatment within the two ecosystems, indicating that it has the highest disturbance intensity. However, the species composition and assemblage still differs between the urban and agricultural settings. This results in the acceptance of the hypothesis which stated that: If the disturbance intensity of maize fields are higher than that found in urban areas due to a higher frequency of disturbances, the diversity and abundance of the selected predacious arthropods will be lower in the maize field within an agro-ecosystem than the urban environment. It seems that many other factors beyond just disturbance intensity plays a vital role in both ecosystems.

There was no correlation between the abundance or diversity of the selected arthropods and patch size and therefore the hypothesis which stated that: If the selected predators adhere to the Island biogeography theory, which suggest that a larger island would be more diverse, it can be expected that patches in the urban environment that are larger would have a higher abundance and diversity in the selected predacious arthropods, is rejected. The size of the patches did not influence the presence of the predacious arthropods in this study. This might be due to corridors making movement between patches easier and thus rendering the possibly effect of the size of the patches as having a lesser impact. Further investigation is required to definitively assess the reason for this outcome.

The size of the settlements indicated that between the largest and the smallest settlements there were differences in species richness per sample plot as well as the abundance, however no other differences were detected. This leads to a partial acceptance and rejection of the hypothesis which stated: If the size of a settlement influences the intensity or frequency of disturbance within that urban environment, it would be expected that larger settlements would be less diverse and abundant with regards to the predacious arthropods present in the green spaces of larger settlements. The largest settlement i.e. Vanderbijlpark had a higher species richness and abundance than that of the small settlement Ventersdorp thus indicating that the size of the settlement did not affect the predacious arthropods in the way that was expected, a possible reason could be the urban heat island effect. However, more research is required to definitively assess the reason for this outcome.

Urban and agricultural areas are diverse and abundant in predacious arthropods which suggests that the way towards a sustainable future is to include and conserve not only urban green spaces but also surrounding agricultural landscapes as these landscapes could very well be the source of urban predator arthropod populations or refuges for arthropod predators in a world dominated by anthropological disturbances activities.

4.9 References

BAKER, L.A., BRAZEL, A.J., SELOVER, N., MARTIN, C., MCINTYRE, N., STEINER, F.R., NELSON, A. & MUSACCHIO, L. 2002. Urbanization and warming of Phoenix (Arizona, USA): Impacts, feedbacks and mitigation. *Urban Ecosystems*, 6(3):183-203.

BEGON, M., TOWNSEND, C.R. & HARPER, J.L. 2006. Ecology: From individuals to ecosystems. Oxford: Blackwell Publishing Ltd.

BERRY, B.J.L. 2008. Urbanization. *In: Urban Ecology: An International persepective on the interaction between humans and nature,* New York: Springer. 25-48 p.

BETTENCOURT, L. & WEST, G. 2010. A unified theory of urban living. *Nature*, 467:912-913.

BHATTA, B. 2010. Analysis of urban growth and sprawl from remote sensing data. *Advances in Geographic Information Science*, 2:17-36.

BLACKSHAW, R.P. & VERNON, R.S. 2006. Spatiotemporal stability of two beetle populations in non-farmed habitats in an agricultural landscape. *Journal of Applied Ecology*, 43:680-689.

BLAIR, R.B. & LAUNER, A.E. 1997. Butterfly diversity and human land use: Species assemblages along an urban gradient. *Biological Conservation*, 80:113-125.

BOLGER, D.T., SUAREZ, A.V., CROOKS, K.R., MORRISON, S.A. & CASE, T.J. 2000. Arthropods in urban habitat fragments in southern California: Area, age and edge effect. *Ecological Applications*, 10(4):1230-1248.

BOTHA, M., SIEBERT, S.J., VAN DEN BERG, J., MALIBA, B.G. & ELLIS, S.M. 2015. Plant and arthropod diversity patterns of maize agro-ecosystems in two grassy biomes of South Africa. *Biodiversity and Conservation* 24:1797-1824.

BUREL, F., BAUDRY, J., BUTET, A., CLERGEAU, P., DELLETTRE, Y., LE COEUER, D., DUBS, F., MORVAN, N., PAILLAT, G., PETIT, S., TENIAL, C., BRUNEL, E. & LEFEUVRE, J. 1998. Comparative biodiversity along a gradient of agricultural landscapes. *Acta Oecologica*, 19:47-60.

BURKMAN, C.E. & GARDINER, M.M. 2014. Urban green space composition and landscape context influence natural enemy community composition and function. *Biological Control*, 75:58-67.

CALCAGNO, V., HECTOR, A., CONNOLLY, J., HARPOLE, S.W., REICH, P.B., SCHERER-LORENZEN, M., LOREAU, M., ZAVALETA, E.S., WILSEY, B.J., WEIGELT, A., VAN RUIJVEN, J., TILMAN, D., SCHMID, B. & ISBELL, F. 2011. High plant diversity is needed to maintain ecosystem services. *Nature*, 477:199-204.

CAPRIO, E., NERVO, B., ISAIA, M., ALLEGRO, G. & ROLANDO, A. 2015. Organic versus conventional systems in viticulture: Comparative effects on spiders and carabids in vineyards and adjacent forests. *Agricultural Systems*, 135:60-69.

CARDOSO, P., PEKAR, S., JOCQUE, R. & CODDINGTON, J.A. 2011. Global patterns of guild composition and functional diversity of spiders. *PLoS ONE*, 6(6):e21710.

CHIDAWANYIKA, F., MUDAVANHU, P. & NYAMUKONDIWA, C. 2012. Biologically based methods for pest management in agriculture under changing climates: Challenges and Future Directions. *Insects*, 3:1171-1189.

CILLIERS, S.S. & BREDENKAMP, G.J. 1999. Ruderal and degraded natural vegetation on vacant lots in the Potchefstroom Municipal Area, North West Province, South Africa. *South African Journal of Botany*, 65(2):163-173.

CILLIERS, S.S., VAN WYK, E. & BREDENKAMP, G.J. 1999. Urban nature conservation: vegetation of natural areas in the Potchefstroom municipal area, North West Province, South Africa. *Koedoe*, 42(1):1-30.

CLARKE, K.R., GORLEY, R.N., 2006. PRIMER v6: User Manual/Tutorial. PRIMER-E, Plymouth, 192pp.

COHEN, B. 2006. Urbanization in developing countries: Current trends, future projections, and key challenges for sustainability. *Technology in Society*, 28(1-2):63-80.

COLLINS, W.W. & QUALSET, C.O. 1999. Biodiversity in Agroecosystems. London: CRC Press LLC.

CONELLE, J.H. 1978. diversity in tropical rain forests and coral reefs. *Science*, 4335:1302-1310.

DEBANO, S.J. 2006. Effects of livestock grazing on aboveground insect communities in semi-arid grasslands of southeastern Arizona. *Biodiversity and Conservation*, 15:2547-2564.

DEPARTMENT OF AGRICULTURE, FORESTRY AND FISHERY (DAFF). 1998. Agricultural policy in South Africa. http://www.nda.agric.za/docs/Policy/policy98.htm Date of access: 25 September 2015.

DEULI, P., OBRIST, M.K. & SCHMATZ, D.R. 1999. Biodiversity evaluation in agricultural landscapes: above-ground insects. *Agriculture, Ecosystems and Environment*, 74:33-64.

DEUTSCH, B., PAULIAN, M., THIERRY, D. & CANARD, M. 2005. Quantifying biodiversity in ecosystems with green lacewing assemblages. *Agro Sustainable Development*, 25:337-343.

DIPPENAAR-SCHOEMAN, A. 2014. Field guide to the spiders of South Africa. Pretoria: LAPA Publishers (Pty) Ltd.

DUGATKIN, L.A. 2009. Principles of Animal Behaviour. New York: W.W Norton & Company.

ELLIS, S.M & STEYN, H.S. 2003. Practical significance (effect sizes) versus or in combination with statistical significance (p-values). *Management Dynamics*, 12(4):51-53.

EYRE, M.D., LUFF., M.L. & WOODWARD, J.C. 2003. Beetles (Coleoptera) on brownfield sites in England: An important conservation resource? *Journal of Insect Conservation*, 7:223-231.

FAY, P.A. 2003. Insect Diversity in Two Burned and Grazed Grasslands. *Environmental Entomology*, 32(5):1099-1104.

GARDINER, M.M., BURKMAN, C.E. & PRAJZNER, S.P. 2013. The value of urban vacant land to support arthropod biodiversity and ecosystem services. *Environmental Entomology*, 42(6):1123-1136.

GIBBS., H. & HOCHULI, D.F. 2002. Habitat fragmentation in an urban environment: large and small fragments support different arthropod assemblages. *Biological Conservation*, 106:91-100.

GREZ, A.A., TORRES, C., ZAVIEZO, T., LAVANDERO, B. & RAḾIREZ, M. 2010. Migration of coccinellids to alfalfa fields with varying adjacent vegetation in Central Chile. *Ciencia e Investigación Agraria*, 37(2):111-121.

GRIME, J. P. 1979. Plant Strategies and Vegetation Processes. Chichester: Wiley.

HAMMER, Ø., HARPER, D.A.T. & RYAN, P.D. 2001. PAST: Paleontological statistics software package for education and data analysis. *Paleontologia Electronica,* 4(1):9

HOLLAND, J.M., THOMAS, C.F.G., BIRKETT, T., SOUTHWAY, S., & OATEN, H. 2005. Farm-scale spatiotemporal dynamics of predatory beetles in arable crops. *Journal of Applied Ecology*, 42:1140-1152.

HORVÁTH, R., MAGURA, T., SZINETÁR, C., EICHARDT, J., KOVÁCS, E. & TÓTHMÉRÉSZ, B. 2015. In stable, unmanaged grasslands local factors are more important than landscape-level factors in shaping spider assemblages. *Agriculture, Ecosystems and Environment*, 208:106-113.

JENKINS, G & BALKEMA, A.A. 1971. The Story of Potchefstroom. Potchefstroom: Potchefstroom Herald.

KANYA, J.I., NGI-SONG, A.J., SÉTAMOU, M.F., OVERHOLT, W. & OCHORA, J. 2004. Diversity of alternative hosts of maize stemborers in Trans-Nzoia district of Kenya. *Environmental Biosafety Research*, 3:159-168.

KOWARIK, I. 1995. On the role of alien species in urban flora and vegetation. I*n: Plant Invasions—General Aspects and Special Problems.*, Amsterdam: SPB Academic. 85-100 p.

KUTSCHBACH-BROHL, L., WASHBURN, B.E., BERHARDT, G.E., CHIPMAN, R.B. & FRANCOEUR, L.C. 2010. Arthropods of a semi-natural grassland in an urban environment: the John F. Kennedy International Airport, New York. *Insect Conservation*, 14:347-358.

LEISTNER, O.A. 2000. Seed plants of Southern Africa: families and genera. Pretoria: Strelitzia 10.

LUNDREN, J.G. & WIEDENMANN, R.N. 2002. Coleoptera-specific Cry3Bb toxin from transgenetic corn pollen does not affect the fitness of a non-target species, *Coleomegilla maculat*a (Coleoptera: Coccinellidae). *Environmental Entomology*, 31:1213-1218.

LUNDREN, J.G. & WIEDENMANN, R.N. 2004. Nutritional suitability of corn pollen for the predator *Coleomegilla maculata* (Coleoptera: Coccinellidae). *Journal of Insect Physiology*, 50:567-575.

MARSHALL, E.J.P. 2002. Introducing field margin ecology in Europe. *Agriculture, Ecosystems and Environment*, 89:1-4.

MARSHALL, E.J., & THOMAS, C.F. 1999. Arthropod abundance and diversity in differently vegetated margins of arable fields. *Agriculture, Ecosystems and Environment*, 72:131-144.

MCKINNEY, M.L. 2002. Urbanization, Biodiversity, and Conservation. *BioScience*, 52(10):883-890.

MEINEKE, E.K., DUNN, R.R., SEXTON, J.O. & FRANK, S.D. 2013. Urban warming drives insect pest abundance on street trees. *PLOS ONE*, 8(3):1-7.

MILLER, T.G. & SPOOLMAN, S.E. 2012. Living in the environment. Toronto: Brooks/Cole Cengage learning.

NORTH WEST TOURISM. 2011. Ventersdorp. Tourism North West Date of access: August 2015.

PILCHER, C.D., OBRYCKI, J.J., RICE, M.E. & LEWIS, L.C. 1997. Preimaginal development, survival, and field abundance of insect predators on transgenic *Bacillus thuringiensis* corn. *Environmental Entomology*, 26:446-454.

RAFFERTY, J.P. 2014. Urban Sprawl. http://www.britannica.com/topic/urban-sprawl. Date of access: 14 Sept 2015.

RAND, T.A., TYLIANAKIS, J.M. & TSCHAMTKE, T. 2006. Spillover edge effects: The dispersal of agriculturally subsidized insect natural enemies into adjacent natural habitats. *Ecology Letters*, 9:603-614.

SARZYNSKI, A. 2012. Bigger Is Not Always Better: A Comparative Analysis of Cities and their Air Pollution Impact. *Urban studies*, 49:3121-3138.

SATTLER, T., OBRIST, M.K., DEULLI, P. & MORETTI, M. 2011. Urban arthropod communities: Added value or just a blend of surrounding biodiversity? *Landscape and Urban Planning*, 103:347-361.

SCHAFFERS, A.P., RAEMAKERS, I.P., SYKORA, K.V. & TERBRAAK, C.J.F. 2008. Arthropod assemblages are best predicted by plant species composition. *Ecology*, 89(3):782-794.

STATISTICS SOUTH AFRICA. 2011. Emfuleni Municipality. http://www.statssa.gov.za Date of access: August 2015.

STATSOFT INC. 2013. Statistica, version 11. Tulsa: StatSoft.

STEIN, B.A., KUTNER, L. & ADAMS, J. 2000. Precious Heritage. Oxford (United Kingdom): Oxford University Press.

SWEENCY, B.W. & VANNOTE, R.L. 1978. Size varation and the distribution of hemimetabolous aquatic insects: Two thermal equilibrium hypotheses. *Science*, 200:446-450.

VAN DER WALT, L., CILLIERS, S.S., KELLNER, K., DU TOIT, M.J. & TONGWAY, D. 2013. To what extent does urbanisation affect fragmented grassland functioning? *Journal of Environmental Management*, 151:517-530.

5. Future research possibilities

5.1 Chapter 2: Optimal sweep net sampling times

Future studies of this nature could include other methods of arthropod collection such as a D-vac (Dietrick vacuum) which is highly effective in sampling grass vegetation. This will allow for a better overview that is more inclusive than using the sweep net method only. This study could also be repeated during different seasons to establish whether differences in diversity are present during different season and whether this is related to temperature. The micro-climate of the different sections of the agro-ecosystem should be included as well as plant structure. This will define the differences in the different zones of the agro-ecosystem, which may aid in understanding the population dynamics of predacious arthropods as well as their behaviour and movement regimes.

5.2 Chapter 3: Diversity of predacious arthropods in maize agro-ecosystems

As three years of data was used in this chapter, recommendations to improve studies of this nature are that comparisons be made between maize fields and semi-natural areas of conventional and organic maize productions systems, as well as between genetically modified (GM) and non-GM maize varieties and surrounding natural vegetation in the grassland biome of South-Africa. Pitfall traps together with sweep net sampling can be used to provide an improved estimate of overall diversity. Plant surveys should also be included and should consider factors such as structure, density and height. This might give much needed insight into the results found in this study.

5.3 Effect of agriculture and urbanization on predacious arthropod diversity of Highveld grasslands.

Future recommendations for this study would be to increase the duration of the study to include at least two to three years of sample collection, as well as an increase the number of settlements to include two or three settlements of different sizes each. Different disturbances in settlements can be investigated as well as investigating the micro-climate of the city and if the city is subject to the urban heat island effect.

The patches within each settlement should be further investigated with regards to their shape, size, edge effects, plant community and age. This will aid in better understanding the biodiversity dynamics of the arthropods within these habitats. The viability of corridors should also be identified in each settlement and investigated and assessed for different groups with regards to the difficulty of using the identified corridor for moving between vacant plots. This could give insight to the differences between arthropod with wings and without for example.

6. Annexures

Appendix 6.1 - Materials and methods to collect arthropods

6.1.1. Detailed description of sweep nets used

Arthropod samples were collected with sweep nets. The sweep nets were made of aluminium rods and nylon nets (Figure 1).

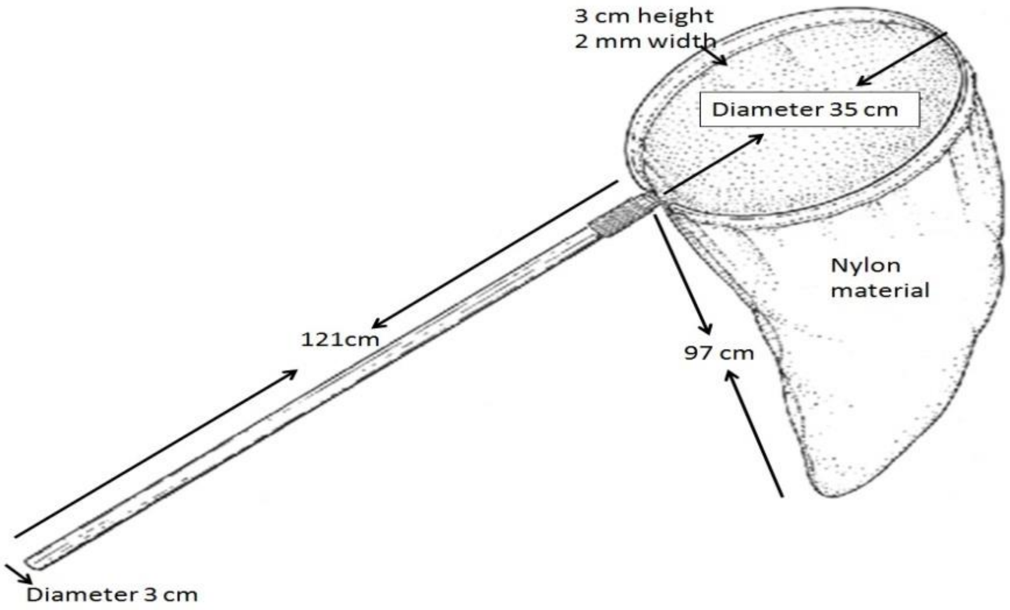

Figure 4.1: Schematic representation with measurements of sweep nets used to collect arthropods.

6.1.2. Adhesive stickers for sample bottles of different treatments

A method was adopted to allocate the same colour sticker per predator group throughout the study, while other colours were used for different treatments (Table 1).

Table 6.1: Example of the application of coloured adhesive stickers for data recording.

Predators	Colour Sticker	Treatments	Colour Sticker
Spider	🟡	Ruderal (Urban area)	🔴
Lacewings	🟣		
Ladybird	🟠	Maize field	🔵
Praying Mantids	🟢	07:00	🔴

Each treatment i.e. the different sampling times of Chapter 2 (07:00, 12:00, 15:00 and 17:00), the different zones of the agro-ecosystem of Chapter 3 (maize field, field margin and untransformed grassland), and the urban and agricultural environments of Chapter 4 (ruderal and fragmented grassland in urban areas), all had a different colour sticker to aid in recording the data effectively and efficiently (Figure 2).

Figure 6.2: Distinguishing colours of adhesive stickers of the collection bottles for different sampling times, treatments and predators.

Appendix 6.2: Predator arthropod species and number of individuals collected at different times during the day in the agro-ecosystem

Family:	**Species name:**	**Time of day**			
		07:00	**12:00**	**15:00**	**17:00**
Spiders: Araneae					
Oxyopidae	*Oxyopes longispinosus*	2	-	2	1
Salticidae	*Thyene semiargentea*	1	-	1	-
Araneidae	*Argiope australis*	-	1	-	-
Eutichuridae	*Cheiracanthium* sp.1	1	-	-	-
Theridiidae	*Enoplognatha molesta*	7	10	4	6
Theridiidae	*Enoplognatha* sp. 1	6	4	3	7
Salticidae	*Festucula lawrencei*	-	-	-	2
Salticidae	*Heliophanus debilis*	-	-	1	-
Araneidae	*Hypsosinga* sp. 1	19	15	14	13
Araneidae	*Larinia bifida*	9	11	3	5
Araneidae	*Larinia natalensis*	2	4	5	3
Theridiidae	*Latrodectus renivulvatus*	1	3	1	2
Thomisidae	*Misumenops rubrodecoratus*	81	58	63	81
Thomisidae	*Monaeses paradoxus*	2	3	2	4
Thomisidae	*Monaeses pustulosus*	18	13	16	7
Araneidae	*Nemoscolus* sp.	-	2	-	1
Araneidae	*Neoscona blondeli*	-	-	1	-
Araneidae	*Neoscona moreli*	-	1	-	-
Araneidae	*Neoscona subfusca*	4	4	2	-
Oxyopidae	*Oxyopes schenkeli*	-	-	-	-
Oxyopidae	*Oxyopes* sp. 1	4	8	7	-
Araneidae	*Pararaneus* sp.1	1	-	-	-
Salticidae	*Pellenes bulawayoensis*	3	6	2	1
Philodromidae	*Philodromus* sp.1	3	-	-	-
Salticidae	*Rhene konradi*	-	-	1	2
Thomisidae	*Runcinia erythrina*	-	-	1	-
Thomisidae	*Runcinia flavida*	26	16	19	20
Thomisidae	*Stiphropus bisigillatus*	2	-	2	1
Thomisidae	*Synema decens*	-	-	-	1
Philodromidae	*Thanatus atlanticus*	-	2	1	2
Araneidae	*Thanatus atlanticus*	1	1	1	1
Philodromidae	*Thanatus* sp.1	1	1	-	-
Theridiidae	*Theridion pictum*	1	-	-	1
Salticidae	*Thyene thyenoides*	-	4	2	2
Salticidae	*Thyenula* sp. 1	3	2	2	2
Philodromidae	*Tibellus hollidayi*	6	18	21	22
Lacewings: Chrysopidae					
Chrysopidae	*Chrysoperla congrua*	47	123	98	156
Chrysopidae	*Italochrysa similis*	-	4	1	-

Ladybirds: Coccinelidae					
Coccinellidae	*Buleae anceps*	8	24	17	2
Coccinellidae	*Cheilomenes lunata*	23	91	70	41
Coccinelidae	*Declivitata bohemani*	-	1	-	-
Coccinelidae	*Declivitata hamata pygmaea*	-	1	1	-
Coccinellidae	*Exochomus flavipes*	11	30	17	10
Coccinellidae	*Harmonia axyridis*	4	13	5	3
Coccinellidae	*Hippodamia variegata*	37	38	47	27
Coccinellidae	*Lioadalia flaromaculata*	5	2	5	8
Praying Mantis : Mantodea					
Mantodea	*Galepsus* sp.	57	48	41	56
Mantodea	*Compsothspis natalica*	2	3	1	-
Mantodea	*Entella transvaalica*	-	1	4	2
Mantodea	*Epsiscopus chalybeus*	65	19	30	68
Mantodea	*Harpogomantis discolour*	-	1	2	-
Mantodea	*Hemipusa capensis*	-	-	-	10
Mantodea	*Pyrogomantis rhodesica*	22	28	33	34
Mantodea	*Tenodera iringana*	7	-	2	2

Appendix 6.3 - Predatory arthropod species collected in the different sections of the agro-ecosystem throughout the duration of the study (2013-2015).

Arthropod Order / Family	Species name:	Agro-ecosystem sections			Presence of species per year		
		Maize field	Field margin	Untransformed grassland	2013	2014	2015
Araneae: Spiders							
Linyphiidae	*Agyneta habra*	1	5	2	8	-	-
Araneidae	*Araneus apricus*	-	-	-	-	-	-
Araneidae	*Argiope australis*	-	-	1	-	1	-
Eutichuridae	*Cheiracanthium* sp.1	-	-	1	-	1	-
Theridiidae	*Enoplognatha molesta*	17	14	13	11	33	-
Theridiidae	*Enoplognatha* sp. 2	1	12	9	1	-	21
Salticidae	*Festucula lawrencei*	-	-	2	-	-	2
Salticidae	*Heliophanus debilis*	-	4	3	6	1	-
Salticidae	*Heliophanus sp.*	-	4	1	5	-	-
Araneidae	*Hypsosinga lithyphantoides*	1	-	1	2	-	-
Araneidae	*Hypsosinga* sp.1	1	38	38	13	3	61
Araneidae	*Larinia bifida*	-	12	17	-	-	29
Araneidae	*Larinia natalensis*	-	8	8	-	-	16
Araneidae	*Larinia* sp.1	-	1	-	1	-	-
Theridiidae	*Latrodectus renivulvatus*	1	3	3	-	4	3
Thomisidae	*Misumenops rubrodecoratus*	29	189	120	43	10	285
Thomisidae	*Monaeses paradoxus*	-	9	8	1	1	15
Thomisidae	*Monaeses pustulosus*	1	21	37	-	-	59

-Araneidae	*Nemoscolus sp.*	-	3	2	-	-	5
Araneidae	*Neoscona blondeli*	-	1	-	-	1	-
Araneidae	*Neoscona moreli*	1	2	1	3	1	-
Araneidae	*Neoscona subfusca*	2	7	10	8	7	4
Oxyopidae	*Oxyopes longispinosus*	1	1	3	-	-	5
Oxyopidae	*Oxyopes schenkeli*	-	-	-	-	-	-
Oxyopidae	*Oxyopes* sp. 1	2	9	11	-	22	-
Araneidae	*Pararaneus* sp.1	1	3	-	1	3	-
Lycosidae	*Pardosa crassipalpis*	2	-	1	2	1	-
Salticidae	*Pellenes bulawayoensis*	2	21	12	23	3	9
Philodromidae	*Philodromus* sp.1	1	5	8	11	3	-
Araneidae	*Pycnacantha tribulus*	-	1	1	1	1	-
Salticidae	*Rhene konradi*	-	3	-	-	3	-
Thomisidae	*Runcinia affinis*	1	21	14	36	-	-
Thomisidae	*Runcinia erythrina*	-	1	1	1	1	-
Thomisidae	*Runcinia flavida*	-	48	37	-	-	85
Thomisidae	*Stiphropus bisigillatus*	1	3	1	-	-	5
Thomisidae	*Synema decens*	1	-	-	-	-	1
Araneidae	*Thanatus atlanticus*	2	2	3	3	-	4
Theridiidae	*Theridion pictum*	-	-	2	-	-	2
Thomisidae	*Thomisus schultzei*	-	-	-	-	-	-
Thomisidae	*Thomisus* sp.1	2	3	-	5	-	-
Salticidae	*Thyene semiargentea*	-	2	1	-	-	3
Salticidae	*Thyene thyenoides*	-	4	5	-	9	-
Salticidae	*Thyenula* sp.	-	7	7	5	-	9

Philodromidae	*Tibellus hollidayi*	3	45	39	5	3	79
Chrysopidae : Lacewings							
Chrysopidae	*Chrysoperla congrua*	45	287	179	6	299	206
Chrysopidae	*Italoshrysa similis*	1	12	33	41	2	3
Coccinellidae : Ladybirds							
Coccinellidae	*Buleae anceps*	19	33	35	30	21	36
Coccinellidae	*Chelomenes lunata*	51	149	193	117	74	202
Coccinellidae	*Exochomus flavipes*	6	16	73	7	1	87
Coccinellidae	*Harmonia axyridis*	32	10	28	37	33	-
Coccinellidae	*Hippodamia variegata*	182	59	45	116	59	111
Coccinellidae	*Lioadalia flaromaculata*	11	11	20	22	9	11
Coccinellidae	*Micraspis cf. comma*	-	3	8	10	-	1
Mantodea : Praying Mantids							
Mantodea	*Comsothspis natalica*	-	3	3	-	-	6
Mantodea	*Entella transvaalica*	1	8	3	4	-	8
Mantodea	*Epsiscopus chalybeus*	1	93	31	8	1	116
Mantodea	*Galepsus sp.*	19	104	109	4	30	198
Mantodea	*Harpagomantis discolour*	-	1	2	-	-	3
Mantodea	*Hemipusa capensis*	-	1	9	-	-	10
Mantodea	*Pyrgomantis rhodesica*	2	141	44	3	3	181
Mantodea	*Tendora iringana*	-	7	15	11	-	11