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Ant and spider diversity in a rural landscape of the Vhembe Biosphere, South Africa

By

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DECLARATION

I **EVANS VUSANI MAUDA,** hereby declares that the dissertation for the degree of Masters of Science at the University of Venda, hereby submitted by me, had not been previously submitted for a degree at this university or any other university, that it is my own work in design and execution and that all reference material contained has been duly acknowledged.

Signature.....

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DEDICATION

"I would like to dedicate my thesis to my beloved grandmother, you're the best"

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ABSTRACT

The greatest threat to biodiversity is the loss of habitat through landscape fragmentation and attrition. Land use changes are therefore among the most immediate drivers of species diversity. Urbanization and agriculture are the main drivers of habitat loss and transformation in the Savanna biomes of South Africa. Agricultural expansion and intensification in particular, takes place at the expense of biodiversity and will probably be the primary driver of biodiversity loss in this century. Arthropods show measurable behavioural responses to changing land mosaics at the smallest scale and heterogeneous environments are therefore predicted to support more complex and diverse biological assemblages. Ants are premier soil turners, channelers of energy and dominates the insect fauna, while spiders are a mega-diverse group that can regulate other invertebrate populations. This study aims to quantify the response of these two taxa in a rural-urban mosaic of a rapidly developing communal area. The study took place in and around two villages in the north-eastern corner of South Africa. Two replicates for each of the dominant land use categories, *viz*. urban settlements, dryland cultivation and cattle rangelands, were set out in each of the villages and sampled during the dry and wet seasons for a total of 2 villages × 3 land use categories × 2 seasons = 24 assemblages. Local scale variables measured included vertical and horizontal habitat structure as well as structural and chemical composition of the soil. The response of richness to local, landscape (land use) and regional (village) factors were modelled using generalized linear mixed models with Poisson error distributions whereas assemblage structure was analysed using both non-parametric (non-metric multidimensional scaling, ANOSIM, PERMANOVA and PERMDISP) as well as parametric (RDA) approaches. Ant richness was not affected by land use but local scale variables such as vertical vegetation structure (+) and leaf litter cover (+), although vegetation complexity at lower levels was negatively associated with ant richness. However, ant richness was largely shaped by regional and temporal processes invoking the importance of dispersal and historical processes. Spider species richness was mostly affected by land use and p value indicated that local conditions highlighting their landscape elements. Spider richness did not vary much between villages and across seasons and seems to be less dependent on context or history. There was a considerable amount of variation in spider richness that was not explained and this could be related to factors which were not measured in this study such as temperature, competition with e.g. ants (ants were very abundant in the settlements). For both ant and spider assemblages the constrained ordination explained 18 %

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of variation in these taxa. Three environmental variables (leaf litter cover, active carbon and rock cover) were important in explaining ant assemblage structure, while two (sand and leaf litter cover) were important for spider assemblage structure. This study highlights the importance of disturbance (land use activities) and leaf litter with the associated effects on ant and spider assemblages across the study area.

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CHAPTER 1: INTRODUCTION

Biodiversity in decline

Human populations are unlikely to stabilize within this century and the biggest increase is predicted for Africa with an estimated population of 4 billion people by 2100 (Gerland *et al.* 2014). Humans are now the greatest force in nature (Steffen *et al.* 2007). Larger human populations translates agriculture that would probably become more intensive and much more extensive is (Haberl *et al.*, 2005). Developing effective policies to slow the rate with which biodiversity is being depleted requires a better understanding of human pressures on biodiversity (Haberl et al., 2005).

Among the most immediate drivers of species diversity, land use changes are at the centre of the discussion (van der Putten *et al.*, 2000) and it is well established that habitat destruction is the leading cause of biodiversity loss, however land use is a multi-faceted notion that includes activities as diverse as clear-cutting or burning of original forest cover (Haberl *et al.*, 2005). Land use results in various changes in ecosystems on different spatial scales that include, *inter alia* changes in land cover and landscape structure, changes in productivity and biomass stocks. Anthropogenic activities such as urbanization and agro-ecosystems creates modified land (Magura *et al.*, 2013).

Agro-ecosystems

Agricultural expansion take place at the expense of biodiversity as natural habitat are cleared to make space for farmland (Shackelford *et al.*, 2015) and further agricultural intensification has led to a decrease in farmland biodiversity measured across many different taxa (Benton *et al.*, 2003). Intensive agricultural management has led to high levels of ecological degradation while less intensive managed agricultural areas enrich regional species diversity and could have a positive effect on beneficial arthropods by providing habitats after crop harvest or for hibernation (Duelli 1997). Goedmakers (1989) points out that changing agricultural land into conservation areas substantially increases the ecological potential of rural development. Cultivated land has the potential to be wildlife habitat, and a vital part of a wildlife-friendly matrix of agricultural and natural habitat (Shackelford *et al.*, 2015).

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Urbanisation

Urbanization is a driving force behind habitat destruction (Otoshi *et al.*, 2015). Habitat disturbance and transformation affect communities in many ways, e.g. altering the balance of competitive interactions i.e. effecting the process of competitive exclusion, or by clearing space for colonization of new organisms (Lach *et al.*, 2009).

The importance of natural fragments as refuges for most spiders in the landscape is emphasized (Gaigher *et al.*, 2014). Human settlement may provide habitat to some spider species but may also provide more predators of spiders (e.g. birds) (Underwood & Quinn, 2010). Green spaces and small-scale agricultural area may provide habitat and resources for arthropods within densely developed cities.

Although urbanization tends to negatively impact biodiversity, some characteristics of urban areas may promote biodiversity (Otoshi *et al.*, 2015). Human activities, such as urbanisation often creates as much more permanent loss of biodiversity unlike other forms of disturbance in the landscape. Otoshi *et al.*, (2015) emphasize that as human population increases, so has the number of people living in urban areas. Urban development results in the conversion of natural space to building and pavement resulting in fragmentation and habitat loss (Philpott *et al.*, 2014).

Mainstays of biodiversity and their services

Terrestrial arthropods

On a changing planet, the impacts of biodiversity loss must be carefully considered. The anthropogenic footprint on ecosystem and biodiversity is more notable now than at any other point in history (Del Toro *et al.*, 2012). Leaders have failed to incorporate ecosystem services benefits in decision making. Arthropods are ideal for studying ecological effects of urbanisation on the environment for several reasons. Insects reproduce relatively quickly and usually have high rates of birth, leading to them responding rather quickly to small-scale changes such as differences in soil or vegetation. Arthropods are abundant, allowing for large sample size and relatively easy to sample in comparison to large organisms (Otoshi *et al.*, 2015)

Ants often dominate insect assemblages and have considerable influence over soil formation (Lobry de Bruyn, 1999; Del Toro et al., 2012), invertebrate assemblage structure, seed predation and dispersal (Andersen, 1995; Munyai & Foord, 2012). Majer *et al.* (2007) mentioned that ant ecology is well understood and therefore they have an advantage over other insect taxa. Ants are also regarded as terrestrial engineers because of their ability to modify

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and regulate the distribution of resources available to other living organisms (Jones *et al.*, 1994). They form a staggeringly diverse and abundant group and constitute a substantial part of animal biomass (Hölldobler & Wilson, 1994) and have their finger on the pulse of ecosystem function (Andersen & Majer, 2004). Ants are one of the few insect groups that can be reliably identified to family by just about anyone, regardless of age or background (Alonso *et al.*, 2000). Ants are the 'premier soil turners, channelers of energy, and dominatrices of the insect fauna' (Hölldobler & Wilson, 1990; Folgarait, 1998). Indeed, it would be difficult to overstate the importance of ants in the functioning of terrestrial ecosystems.

Spiders are important in agroecosystems and contribute to the control of agricultural pest population (Wu *et al.*, 2008). Spiders are able to disperse long distances by wind (Horváth *et al.*, 2009). Spiders have been found in almost all environments and is a mega-diverse group with about 44 032 described species. They present relevant ecological function as regulators of other invertebrate populations (Flórez, 2000). As arthropod predators they are important components of natural and agricultural ecosystems, playing a role in structuring arthropod communities and also in natural pest control (Sunderland & Samu, 2000). Usually spiders are the most diverse and abundant predatory animal group in any habitat. Besides fragmentation spider assemblages are influenced by many aspects of land use and management like mowing regime and grazing intensity (Dennis *et al.*, 2001; Cattin *et al.*, 2003).

Ecosystem services

Future habitat degradation and climate change impacts are expected to have devastating consequences not only on the conservation status of biodiversity but on the loss of ecosystem services provided by biodiversity (Otoshi et al., 2015). Various living organisms provides one or more ecosystem services, which are defined as any service that benefits human society and supports well-being (Del Toro *et al.*, 2012). Various forms of ecosystem services provided by organisms are: (1) provisioning services, (2) regulating services, (3) supporting services and cultural services (Del Toro *et al.*, 2012).

Folgarait (1998) showed that ants play a role in mediating ecosystem functions and were recognized as having major ecological roles in most terrestrial ecosystem. Ants provide various services, *inter alia* indirect pollination facilitation and maintaining food chain robustness (Landis *et al.*, 2000; Wielgoss *et al.*, 2013), soil mixing, nutrient transport, and soil aeration (Graham *et al.*, 2009), decomposition, carbon flux, and provision of habitat (Del Toro *et al.*,2012). Ant community structure affect services and disservice for agriculture in complex

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ways, with species-rich and even communities promoting highest yield (Wielgoss *et al.,* 2013). Philpott *et al.* (2004), found that ant communities with high abundance may result in reduced top down effects on invertebrate fauna as compared to those communities with low diversity. Ground-dwelling arthropods are highly diverse and abundant in crop fields and semi-natural habitats (Finke & Snyder, 2010).

Similar to ants, spiders are important in agro-ecosystems and contribute to the control of agricultural pest populations. Nyffeler & Sunderland (2003) found that soil surface-dwelling spiders reduced close to 20-30 aphids (pest) m^{-2} per day in the agricultural field. Spiders rarely feed on predaceous arthropods and that enhances their appropriateness as biological control agent (Nylffeler & Sunderland, 2003). Potentially they are highly beneficial because of their efficient foraging behaviour enabling them to discover and seize smaller instars of various species of lepidopteran and heteropteran pests. Nylffer & Sunderland (2003) also point out that their value as biocontrol agents might be limited due to low density, infrequent feeding and involvement in intraguild predation.

Maintaining biodiversity

Habitat heterogeneity

Landscape heterogeneity is a major structuring agent of ecological assembles promoting alpha, beta diversity and ultimately contribution to overall higher global diversity (McClain & Barry, 2010). Arguably, the greatest threat to maintenance of insect diversity is loss of habitat through landscape fragmentation and attrition. Heterogeneous environments are predicted to support more complex and diverse biological assemblages (Tews *et al.,* 2004). They may also regulate diversity through patch variation in a suite of other well examined processes e.g. disturbance regimes (Roxburgh *et al.* 2004), and connectivity (Chase & Ryberg, 2004). The effect of disturbance on spatial heterogeneity accounts for higher species richness in moderately disturbed habitats (Graham *et al.,* 2009).

The loss of ecological heterogeneity at multiple spatial and temporal scales is a universal consequence of multivariate agricultural intensification (Benton et al. 2003). Changing farming practices suggested that there is a decrease in farmland biodiversity in Europe and North America on a farm scale while, comparison of organic and conventional farms often indicated an increase in arthropods biodiversity in organic farms (Feber *et al.*, 1998). Habitat heterogeneity is a useful multivariate measure of the intensity of farming and concentrating on

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particular farming practices, have an identifiable universal management objective of promoting heterogeneity.

Field heterogeneity in the farmed landscape is brought by non-cropped habitat, such as field margins. Individual insects show measurable behavioural response to changing landscape mosaic at the smallest scale (Sundays, 2005). The benefits of no-cropped habitats for different taxa also interact, for example plant biodiversity might be greater attracting herbivorous insects that in turn attract their own natural enemies (Benton *et al.*, 2003). Habitat management such as creation of new semi-natural habitats around or within fields enhance spider abundance and species diversity (Wu *et al.*, 2008). Maintenance of a naturally heterogeneous landscape is important in conserving a wide variety of invertebrates. Healthy populations usually requires the combination of large habitat size, patch quality and reduced patch isolation, for example, South African dragonflies need a variety of structural vegetation types (Samways, 2007).

Habitat connectivity

Arthropods are small and therefore affected by boundaries at distances beyond what human perceive as the vegetation boundary. Landscape elements boundaries are important features in management planning. The ideal situation is to reduce contrast between source areas and their surroundings to encourage movement through the differential landscape filter (Samways, 2007).

Land sparing

Small patches have a greater proportion of edge to interior than do large patches, the quality of the patch is decreased as the patch becomes smaller in size. This small sizes can lead to population loss (Samways, 2007). Outside large, good quality patches in general the transformed matrix may not necessarily be unsustainable for all species (New, 2005). The area surrounding a good patch can be viewed as a differential filter, favouring some species but not others and even a certain age sex, age or ecotype (Ingham & Samways, 1996).

Both metapopulation dynamics and island effects may be taking place. Habitat patches are usually variable in quality with large patches sometimes acting as metapopulation units in their own right and small patches functioning only as temporary or semi-permanent habitats (Sutcliffe *et al.*, 1997). Land sparing and activities in it becomes an important feature of landscape management, however land sparing is in need of much more development, and it is more complicated than just leaving parcels of land (Samways, 2007).

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Connect like patches of quality habitat

Corridors are continuous linear strips that connect habitats and improve the chance of survival of other isolated population. As insects are small and speciose a landscape feature that is beneficial for a large mammal will not necessarily be beneficial for a particular individual or particular insect species (Samways, 2007). Determining what is meant by connectivity for a species or landscape is a critical initial step in developing any conservation assessment for connectivity (Aune et al., 2011).

Corridors have multiple roles depending on the focal organisms at any particular time (Ingham & Samways, 1996). Corridors may be habitats that support certain species and where normal ecological interaction between plants and insects take place (Aune, *et al*., 2011), however, not all species or individuals can move along these corridors with equal ease (Samways, 2007). The network of corridors needs to be a source habitat, in turn it is only acceptable as a link between habitats when it is a movement corridor or stepping stones to a new patch or habitat (Samways, 2007). Improving the landscape for population dispersal goes a long way to facing the deadly anthropogenic cocktail and it can also encourage other threatening factors such as invasion, biocontrol agent pathogens and genetically modified organism (Samways, 2007).

The Vhembe Region

The Limpopo province is one of the nine provinces of the Republic of South Africa, constituting 10 percent of the total land area of 122 million hectares (Statistics South Africa, 2006). The area is situated in the dry savannah sub region, characterised by open grasslands with scattered trees and shrubs (Acocks, 1988). The province has a rapidly growing population and currently comprises 5.4 million people of the national population of almost 52 million (Statistics South Africa, 2011). Provincial growth and development strategies are centred on further promoting agriculture, mining and tourism, with agriculture comprising 15 percent of Provincial GDP and 20 percent of the workforce. A lack of understanding of the impacts of land use, land degradation and biodiversity loss on ecosystem services provision especially on vulnerable members of society. There is little research investigating the relationship between socio-economics drivers of change in rangeland vegetation and in Africa. There is no study that has been done on the response of ants and spiders to a rural landscape that include rural (settlement), agro-ecosystem and rangeland elements and knowing the biodiversity of such areas is crucial for future landscape changes and conservation planning in rural villages.

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Vhembe Biosphere Reserve (VBR)

UNESCO's Man and Biosphere programme declared the Vhembe region a biosphere reserve in 2009. The Vhembe Biosphere Reserve covers an area of about 30 000km^2 in the northern part of the Limpopo Province and supports 1.5 million human inhabitants. It is a globally recognised centre of endemism, biodiversity and cultural importance. At the core of the VBR is the Soutpansberg Mountain which contains between 2500 and 3000 species of plants that belong to 1000 genera, a higher generic richness than the Cape Floristic Kingdom[1].

Core, buffer and transition areas form the framework within which biosphere reserves are designed. No development is allowed in core areas while lower impact but sustainable land use are still permitted in the buffer zones. Currently the VBR lacks a coherent network of core and buffer area that accurately reflects the distribution of critical biodiversity and ecosystem services. Biodiversity has an increasing value in communities and even in agricultural policy, conservation practice and management of this landscapes is important. Within the agricultural landscape of Limpopo Province there is a need to understand and predict the combined effects of land use and climate change processes on rangeland vegetation, its unique biodiversity and rural livelihood.

Rural African landscapes should be an important focus if we are to conserve the biodiversity. No studies have been done in these rural landscapes and achieving what they actually have is important in decision of biodiversity-ecosystem management in such areas. Furthermore, if we are to operationalize the conservation of these groups we would have to identify the local and regional determinants of their diversity. Ecological studies of urban ants generally focus on investigating changes in species richness and species composition in different urban habitat types, urban habitat fragments of different size or age, or along urban to rural gradients (Lessard & Buddle, 2005). This study aims to develop that for two dominant invertebrate taxa in the region, ants and spiders, by quantifying the impact of the three dominant land use categories, *viz.* settlements, dryland cultivation and rangelands on their diversity and composition. To determine the role of local and regional factors in driving ants and spider assemblage patterns, and attempt to link the relative role of humans in these processes.

[1] www.vhembebiosphere.org/sarchi/

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CHAPTER 2: METHODOLOGY

2.1. Introduction

Limpopo province is located in the northern part of South Africa and shares borders across the Limpopo River with Botswana, Zimbabwe and Mozambique. The province has five districts, namely Capricorn, Mopani, Sekhukhune, Vhembe, and Waterberg, and it covers an area of 125 754 km^2, constituting 10.3% of South Africa's total land area (Statistics South Africa, 2011). The Vhembe district has the largest human population in the province with 1.3 million people, followed by Capricorn districts with approximately 1.2 million (Statistics South Africa, 2011). The Vhembe Biosphere Reserve [2] (Fig. 1) which was declared by UNESCO in May 2009 and launched on the 18th of August 2011 covers most of Limpopo province north of the Tropic of Capricorn.

2.2. Study Area and Site Selection

The study area is located in the Vhembe region of Limpopo province, on the plains south east of the Soutpansberg Mountains (Fig. 1). The area is mainly under tribal management with villages scattered across the landscape. The land use in and around villages mainly consists of dryland subsistence farming, rangeland for cattle and the rural settlements themselves. Two of the villages, Vyeboom (S 23.15345°, E 030.39459°) and Ndengeza (S 23. 30867°, E 030.40531^0) were selected for this study (Fig. 2 & 3). Two replicates were set out in each of the land use types in each of the two villages respectively, resulting in a total of 12 action sites (2 replicates x 3 land use type x 2 villages = 12) throughout the study area (Appendix A). Rangeland sites in both villages and one settlement site of Vyeboom were on Gravelotte Rocky Bushveld, a threatened endangered vegetation type (Mucina & Rutherford, 2006). The remaining actions sites were on Granite Lowveld.

[2] www.unesco.org

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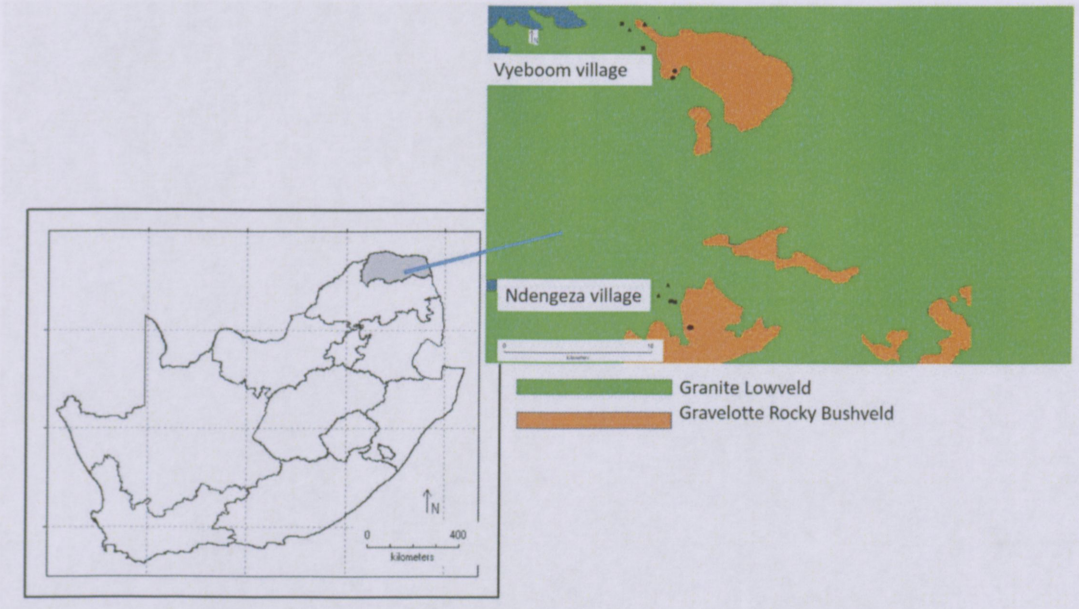


Figure 1. Location of all 12 replicates on the two selected villages projected onto vegetation map from Mucina & Rutherford (2006).

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Figure 2. Geographical location of all 6 replicates on 3 selected land use types at Vyeboom village (Google Earth image). VA1= Vyeboom Agro-ecosystem 1, VA2= Vyeboom Agro-ecosystem 2, VN1= Vyeboom Rangeland 1, VN2= Vyeboom Rangeland 2, VS1= Vyeboom Settlement 1, VS2= Vyeboom Settlement 2.

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Figure 3. Geographical location of all 6 replicates on 3 selected land use type at Ndengeza Village (Google Earth image). NA1= Ndengeza Agro-ecosystem 1, NA2= Ndengeza Agro-ecosystem 2, NN1= Ndengeza Rangeland 1, NN2= Ndengeza Rangeland 2, NS1= Ndengeza Settlement 1, NS2= Ndengeza Settlement 2.

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The landscape features tall shrubland with few trees to moderately dense low woodland on the deep sandy uplands with *Terminalia sericea, Combretum zeyheri* and *Combretum apiculatum* and a ground layer including *Pogonarthria squarrosa, Tricholaena monache* and *Eragrotis rigidior*.

The area is comprised of dense thicket to open savanna in the bottomlands with *Acacia nigrescens, Dichrostachys cinerea, Grewia bicolor* in the woody layer. At seep lines, where convex topography changes to concave, a dense fringe of *Terminalia sericea* occurs with *Eragrotis gummiflua* in the undergrowth (Mucina & Rutherford, 2006). The sites in Gravelotte Rocky Bushveld (NLa, NLb, VLa, VLb and VSb) is characterized by open deciduous to semi-deciduous woodland on rocky slopes and inselbergs, contrasting strongly with the surrounding plains (Figs. 2 & 3).

The soil groups of the area is mainly red-yellow well-drained soils lacking a strong texture contrast, with limited pedological development and rocky areas. Land types of the area is mainly Ib (rock with limited soils), Fa, Fb (soils with minimal development, usually shallow on hard or weathering rock, with or without intermittent diverse soils) and Ae (red massive or weakly structured soils with high base status) (Mucina & Rutherford, 2006).

2.2.1. Climate

The region experience summer rainfall with dry winters. The mean annual precipitation of about 450 mm on the eastern flats to about 900 mm near the escarpment in the west. The area is generally a frost-free region. The annual mean temperature of about 20.9 °C and a mean monthly maximum and minimum temperature ranges of 38 °C and 3.7 °C respectively. The area has an annual mean potential evaporation of 2007 mm and mean annual soil moisture stress of about 77% (Mucina & Rutherford, 2006).

2.2.2. Dominant vegetation

Dominant plants species include **Tall trees**: *Pterocarpus angolensis, Acacia nigrescens, Sclerocarya birrea subsp. caffra,* **Small Trees**: *Acacia nilotica, Albizia harveyi, Ficus stuhlmannii, Dombeya rotundifolia, Pterocarpus rotundifolius, Acacia exuvialis, Acacia gerrardii, Combretum apiculatum, Combretum imberbe, Combretum zeyheri, Peltophorum africanum, Terminalia sericea, Bolusanthus speciosus, Cassia abbreviata subsp. beareana.* **Tall Shrubs**: *Strychnos madagascariensis, Combretum hereroense, Euclea divinorum, Gardenia volkensii, Hibiscus micranthus, Tephrosia polystachya, Dichrostachys cinerea.* **Low**

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Shrubs: *Agathisanthemum bojeri, Barleria elegans, Abutilon austro-africanum, Aptosimum lineare, Clerodendrum ternatum, Commiphora africana, Gossypium herbaceum subsp. africanum, Pavonia burchellii.* **Woody Climbers**: *Sphedamnocarpus pruriens* subsp. *pruriens*, *Bauhinia galpinii.* **Graminoids**: *Brachiaria nigropedata, Digitaria eriantha subsp. eriantha, Eragrostis rigidior, Melinis repens, Aristida congesta, Alloteropsis semialata subsp. semialata, Andropogon schirensis, Bothriochloa bladhii, Monocymbium ceresiiforme, Pasplum scrobiculatum, Schizachyrium sanguineum, Themeda triandra.* **Herb**: *Achyranthes aspera, Cucumis africanus, Vahlia capensis subsp. vulgaris, Waltheria indica.* **Succulent Herbs:** *Orbea rogersii, Stapelia leendertziae, Stapelia gigantea* (Mucina & Rutherford, 2006).

2.2.3. Conservation status

Of the targeted conservation status of 19% of the Granite Lowveld vegetation, some 17% are statutorily conserved in the Kruger National Park and about the same amount of the vegetation is conserved in privately owned reserves. More than 20% are already transformed, mainly through cultivation and settlement development, leading to this vegetation classified as vulnerable. The Gravelotte Rocky Bushveld is classified as least threatened and has no statutory conservation areas. Some 7% of the targeted 19% conservation is conserved in private reserves. Conservation of this vegetation is promoted due to the land use of game and cattle farming and due to its low agronomic potential (Mucina & Rutherford, 2006).

2.3. Ant and Spider sampling

Sampling at each action site followed a protocol as set forth by Zhao *et al.* (2013), known as the Two Circled Method (TCM). The method requires the design of a grid of multiple pairs of pitfall traps with different distances between the pairs. The method presumes that the trapping area of a trap is circular and the density of ground dwelling arthropods at different trapping area is constant. Paired pitfall traps were set up at 0 m (only a single pitfall), 1.5 m, 2.0 m and 2.5 m apart with 15 m spacing between traps (Fig. 4) for a total of 35 pitfalls × 3 land use type × 2 villages = 420 pitfall traps per sampling event.

Each grid of pitfall traps were left open for five nights. Replicates within a land use type were more than 300 m apart to avoid pseudo-replication (Munyai & Foord, 2015). Pitfall traps (ø 62 mm) contained 50 ml propylene glycol solution which does not attract nor repel ants (Munyai & Foord, 2012). Pitfall traps were left open for five days in April/June 2014 (dry season) and January 2015 (wet season). Munyai & Foord (2015) have argued that leaving traps open for

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five days is sufficiently long to allow sampling of representative ants at a site and not sample excessive amount of ant specimens.

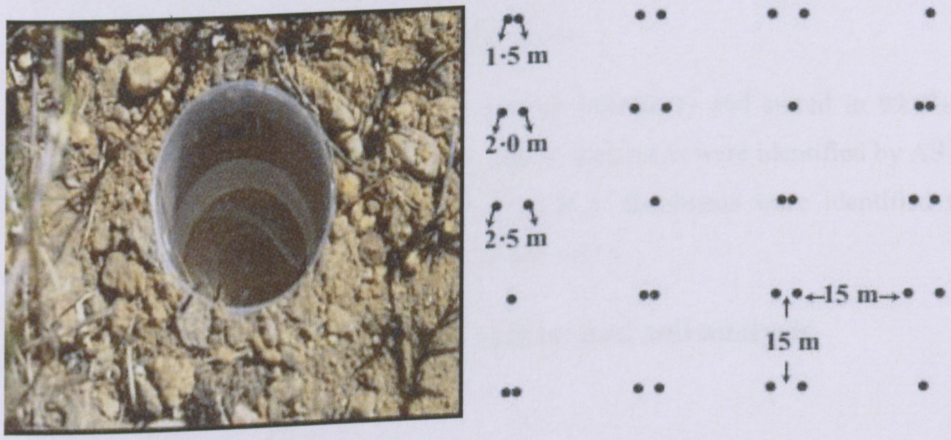


Figure 4. Illustration of the spatial arrangement of pitfall traps for the Two-circle method design and a pitfall with propylene glycol flushed to the ground.

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Ant samples were washed and sorted in the laboratory and stored in 99.9% ethanol to preserve specimens before identification. Ant specimens were identified to morpho-species and where possible to species level using Bolton (1994) and by comparison with the reference collection housed in the University of Venda, Zoology Department, South Africa. Antweb[3] and Antcat[4] were used to confirm new updates of valid names.

Spider samples were washed and sorted in the laboratory and stored in 99.9% ethanol to preserve specimens before identification. Spider specimens were identified by AS Dippenaar-Schoeman Agricultural Research Council (ARC). Specimens were identified to morpho-species and where possible to species level and sexed.

2.4. Temperature, Vegetation structure, and soil analysis

2.4.1. Vegetation structure

Vertical and horizontal distribution of plants were quantified following methods used by Botes *et al.* (2006) and Munyai & Foord (2012 & 2015). Plant height profiles were measured by placing a 1.5 m rod marked at 25 cm height intervals (0-25; 25-50; 50-75; 75-100; 100-125; 125-150 & 150+ cm (representing canopy cover) at four points on 1.5 m radius centred on each pitfall trap. The number of foliage hits at 25 cm intervals were recorded. The horizontal habitat structure was quantified by placing a 1 m^2 grid over each pitfall trap capturing an image and then estimating the percentage cover of four categories: exposed bare ground, vegetation, leaf litter and exposed rock (Fig. 5).

[3] www.antweb.org
[4] www.antcat.org

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Figure 5. Estimation of horizontal habitat structure with 1 m^2 grid and pitfall in the centre.

2.4.2. Soil analysis.

Soil samples were collected in June 2014 using a soil auger. A total of four soil sub samples were taken in each of the 12 replicates. Soil were sent to ZZ2 Laboratories (Mooketsi, South Africa) where it was analyzed for active carbon, Phosphate, Magnesium and Nitrogen (PMN) (µgN/gDM/week), clay (%), Silt (%) and Sand (%) (Appendix B).

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2.5. Data analysis

Except when stating otherwise, all data manipulation and analyses were done in the R statistical environment (R Core Team, 2014).Principle component analysis was performed on the vertical measures of habitat structure to account for co-linearity and reduce the number of predictor variables in the model. For vertical habitat structure, each of the seven intervals along the measuring rod was used as input variables. The PCA therefore not only accounts for the number of hits but also represents the location of the hits. The first two principal components accounted for 89% of the variation. Principal component axis one (PC1) is positively associated with habitats that are less complex and has more hits at higher intervals, with little vegetation cover whereas PC2 is positively related with sites that have considerable ground cover linked to the large number of hits at the lowest interval (< 25 cm) on the rod (Appendix C). Settlement sites in particular had limited vertical stratification and rangelands were vertically more complex (Appendix C).

Collinearity of the remaining predictors were assessed using the Pearson's product correlation coefficient. Variables that had a significant $r > 0.05$ were inspected and the variable that was biologically the most relevant retained. There was almost 100 percent correlation between the clay and silt content of the soil while phosphate, magnesium and nitrogen (PMN) was significantly correlated (Appendix D). There was a significant negative correlation between sand and clay. Only active carbon and sand were therefore retained for further analysis. Based on the correlogram of horizontal habitat structure only leaf litter and rock cover were the two variable that was retained for analysis (Appendix E).

For the purpose of species richness analysis, sample units were represented by replicates sampled during each survey, with a total of 6 sites × 2 replicates × 2 surveys = 24 assemblages. Species richness in this study is therefore equivalent to species density, i.e. the number of species detected per area sampled, or per spatially standardized survey effort (Lomolino, 2001). Coverage (Chao & Jost, 2012) was calculated for each of these assemblages. Coverage based rarefaction and extrapolation works on the principle that samples of equal completeness and not equal size should be compared between communities (Chao & Jost, 2012). Richness estimates between all three land use treatments was done using Chao 2.

Ant and spider richness are count data and their response to the explanatory variables were therefore modelled using Generalised Linear Mixed Models (GLMM) with Poisson distributions. Fixed factors in the model included soil characteristics and habitat structure

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(vertical and horizontal) as continuous variables and land use as a categorical variable. Season, (dry vs. wet), vegetation type, and sites which were nested within villages were included as random variables. This random structure took care of both spatial and temporal pseudo-replication. Coverage at each site was used to add prior weights to the observed species richness used in the fitting process. All explanatory variables were scaled and standardised to allow greater interpretability of the regression coefficients (Schielzeth, 2010)

Ant data for each survey were square root transformed for multivariate analysis. Ant assemblage structure was analyzed using non-metric multidimensional scaling (nMDS). Comparison of species composition and species turnover was done using PERMANOVA and PERMDISP for all three land use categories. Species responses to land use and the other predictor variables were modelled using redundancy analysis (RDA). Those predictors that best explained variation in species composition were identified using forward selection of environmental variables.

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CHAPTER 3: RESULTS

A total of 141 259 ant specimens composed of 126 species in 39 genera and 7 subfamilies were collected over the period of the study. The most diverse subfamilies were Myrmicinae with 76 species and 15 genera, followed by Formicinae with 29 species and six genera. Ponerinae had the second highest generic diversity with 10 genera and 13 species. Dorylinae had four species and the least diverse subfamilies were Amblyoponinae, and Pseudomyrmicinae, each with 1 species. The most specious genera were *Tetramorium* with 22 species, *Monomorium* (16 species), *Camponotus* (12 species). Nine species were restricted to Vyeboom village and two species to Ndengeza village.

Ant species richness differed significantly between land use types (Table 1) and pair-wise comparisons suggest that it is mainly the result of significantly lower richness in the agro-ecosystems as compared to the rangelands (Table 2). Ant species richness peaked during the wet season except in the agro-ecosystem. The rangelands had the highest species richness followed by the villages and agro-ecosystem (Fig. 6a). Sampling in all land use was complete and sampling coverage for all land use was almost 100 percent completeness (Fig. 6b, Table 1).

Table 1. Total observed ant species richness, mean observed richness (± standard deviation), species estimates and coverage for overall ant data during the period of the study (2014/15).

Land use	Total observed Richness	Mean (± sd)	Chao 2	Coverage (overall)
Cultivation	79	28.75 ± 1.9	97.21	0.99
Rangeland	101	40 (±2.24)	116.88	0.99
Settlement	90	34.75 (±2.08)	107.95	0.99

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Table 2. Pair-wise comparisons of ant richness in the rangeland, agro-ecosystem and settlement sites of the two villages using Tukey tests.

Comparison	Species richness
Land use	Z-values
Rangeland ↔ Agro-ecosystem	2.15*
Settlement ↔ Agro-ecosystem	0.78
Settlement ↔ Rangeland	-1.43

Significant level codes: 0 '***', 0.001 '**', 0.01 '*' , 0.05 '.', 0.1 '', >0.1

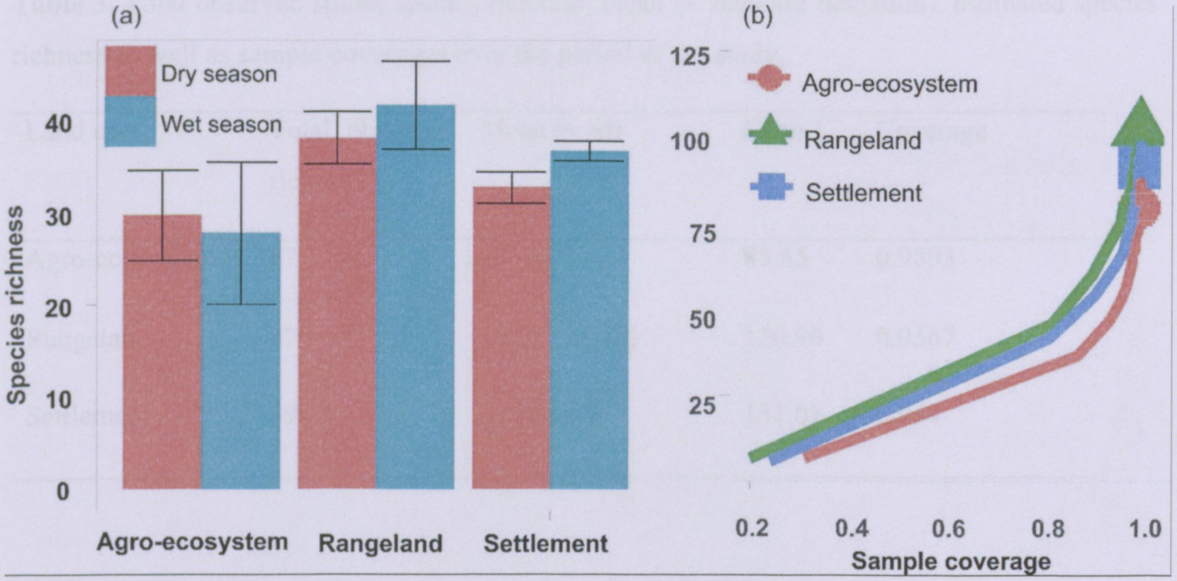


Figure 6. (a) Barplots of observed ant species richness ± SE in the three land use types for the dry and wet seasons, and (b) species richness as a function of sample coverage for each land use based interpolated and extrapolated samples.

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A total of 123 spider species representing 90 genera in 31 families were collected over the period of the study. The family Gnaphosidae and Salticidae had the highest generic richness with 13 genera each, followed by Lycosidae (10 genera) and Zodariidae (7 genera). The most speciose genera were *Zelotes* (8 species) and *Oxyopes* (5 species). Gnaphosidae had the highest number of species (22), followed by Lycosidae (18 species), Salticidae (14 species) and Zodariidae (9) and Linyphiidae (6 species), other families had a total number of species between 5 and 1.

The spider species richness differed significantly between land use types (Table 3) and pair-wise comparison suggest that it is mainly the result of significantly lower species richness in the settlements (Table 4). Similar to ants, spider richness peaked in the rangelands, followed by the agro-ecosystems, while settlements had the lowest richness (Fig. 7a, Table 3). However, the extrapolation indicated that villages have actually high species richness (Fig.7b, Table 3). Sampling coverage was almost complete in all land uses, with agro-ecosystem showing more complete coverage (Fig. 7b).

Table 3. Total observed spider species richness, mean (± standard deviation), estimated species richness as well as sample coverages over the period of the study.

Land use	Total observed richness	Mean (± sd)	Chao 1	Coverage
Agro-ecosystem	67	23 (±1.7)	83.65	0.9803
Rangeland	87	25.25 (±1.78)	120.96	0.9567
Settlement	59	17 (±1.46)	137.01	0.964

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Table 4. Pair-wise comparisons of spider richness in the rangeland, agro-ecosystem and settlements using Tukey tests.

Comparison	Species richness
Land use	Z-value
Rangeland ↔ Agro-ecosystem	0.92
Agro-ecosystem ↔ Settlement	-2.29 *
Rangeland ↔ Settlement	-3.24 **

Significant level codes: 0 '***', 0.001 '**', 0.01 '*' , 0.05 '.', 0.1 '',>0. 1

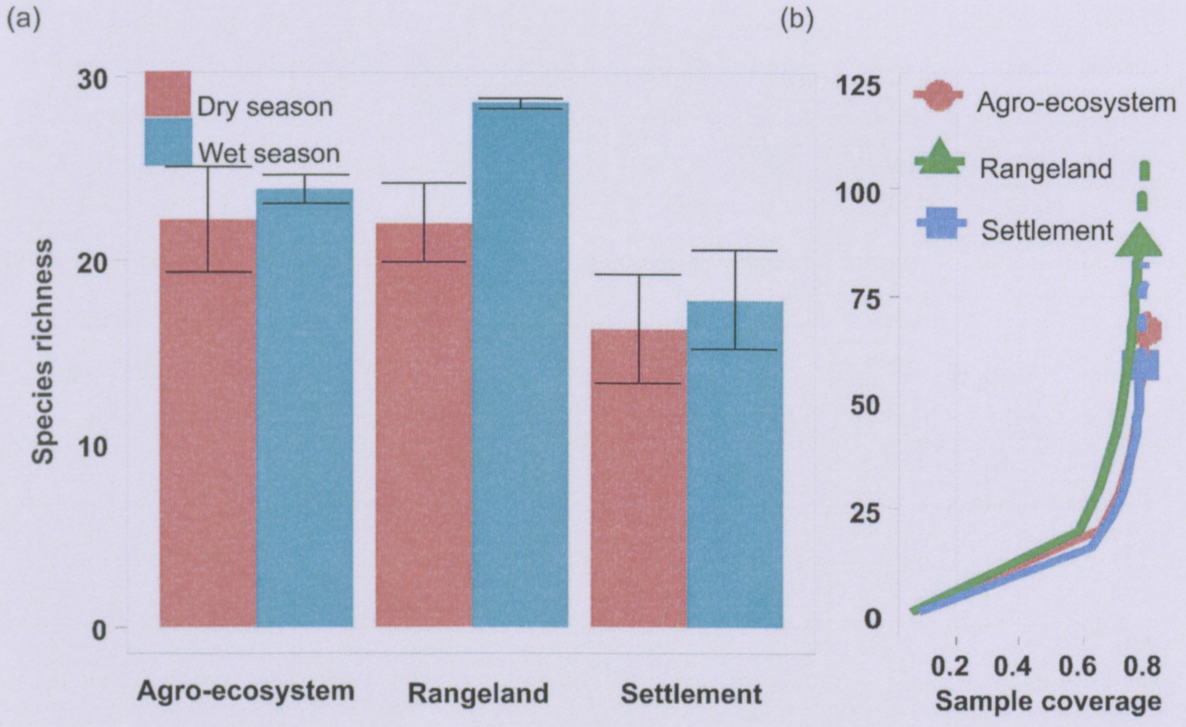


Figure 7. (a) Barplots of observed spider species richness ± SE in the three land use types for the dry and wet seasons and (b) spider species richness as a function of sample coverage for each land

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use based interpolated and extrapolated samples species richness estimates for rarefied and extrapolated samples with coverage and richness for each of the land use types.

Local scale drivers of ant and spider richness

The ten best models for ant richness, ranked according their AICc, are listed in Table 5. The best GLMM model explained 94% of the variation in ant richness, while 31% of this was explained by fixed factors ($R^2_m = 0.31$, Table 5). A considerable amount of variation was therefore accounted for by random factors (Village, replicate and season). Important fixed factors included leaf litter, and the two principle coordinates that represent vegetation structure (Table 6). Leaf litter cover was positively related to ant species richness (Fig. 8a), while richness decreased in sites that were vertically less complex (Fig. 8b) as well as in sites that were more hits at lowest stratum (0 - 25cm) (Fig. 8c).

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Table 5. Comparative and summary statistics for generalized linear mixed models explaining variation in ant richness across land use types. Predictors were leaf litter, vegetation, rock cover, PC1 and PC2. Listed are the degrees of freedom (d.f.), maximum log-likelihood (LL), Akaike's bias corrected information criterion (AICc) and it's change relative to the top ranked model (ΔAICc), the model probabilities (wAICc) and the marginal and conditional R^2s. Marginal R^2 (R^2_m) is the amount of variation explained by the fixed effects, conditional R^2 (R^2_c) is that explained by the fixed and random effects.

Model	d.f.	LL	AICc	ΔAICc	wAICc	R^2_m	R^2_c
~ Leaf litter - PC1 - PC2	8	-60.5	147.4	0.00	0.09	0.31	0.94
~ Leaf litter + PC1 - PC2 + Rock cover	9	-58.1	148.1	0.77	0.06	0.27	0.94
~ Land use + Leaf litter + PC1 + PC2	10	-54.9	148.2	0.85	0.06	0.25	0.95
~ Leaf litter + PC1 + Rock cover	8	-61.2	148.8	1.40	0.04	0.27	0.93
~ Land use + Leaf litter + PC1	9	-58.5	148.9	1.56	0.4	0.24	0.92
~ Leaf litter + PC1	7	-63.7	148.9	1.57	0.04	0.25	0.9
~ Land use + PC1 + Rock cover	9	-58.7	149.2	1.82	0.04	0.09	0.88
~ Leaf litter + PC1 + PC2 + vegetation	9	-58.7	149.2	1.88	0.03	0.09	0.88
~ Leaf litter + PC2	7	-63.9	149.3	1.9	0.03	0.21	0.9
~ Leaf litter + PC2 + Rock cover	8	-61.5	149.3	1.98	0.03	0.21	0.92

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Table 6. Test statistics (χ^2), degrees of freedom (D.f) and p values for type III Wald chisquare tests on the best model (top ranked model from Table 3). Explanatory variables included leaf litter, PC1 and PC2.

	χ^2	D.f	p
Leaf litter	8.41	1	>0.004 **
PC1	3.01	2	0.159
PC2	24.81	2	0.127

Significance level codes: 0 '***' 0.001 '**' 0.01 '*' 0.5 '.' 0.1 '' 1

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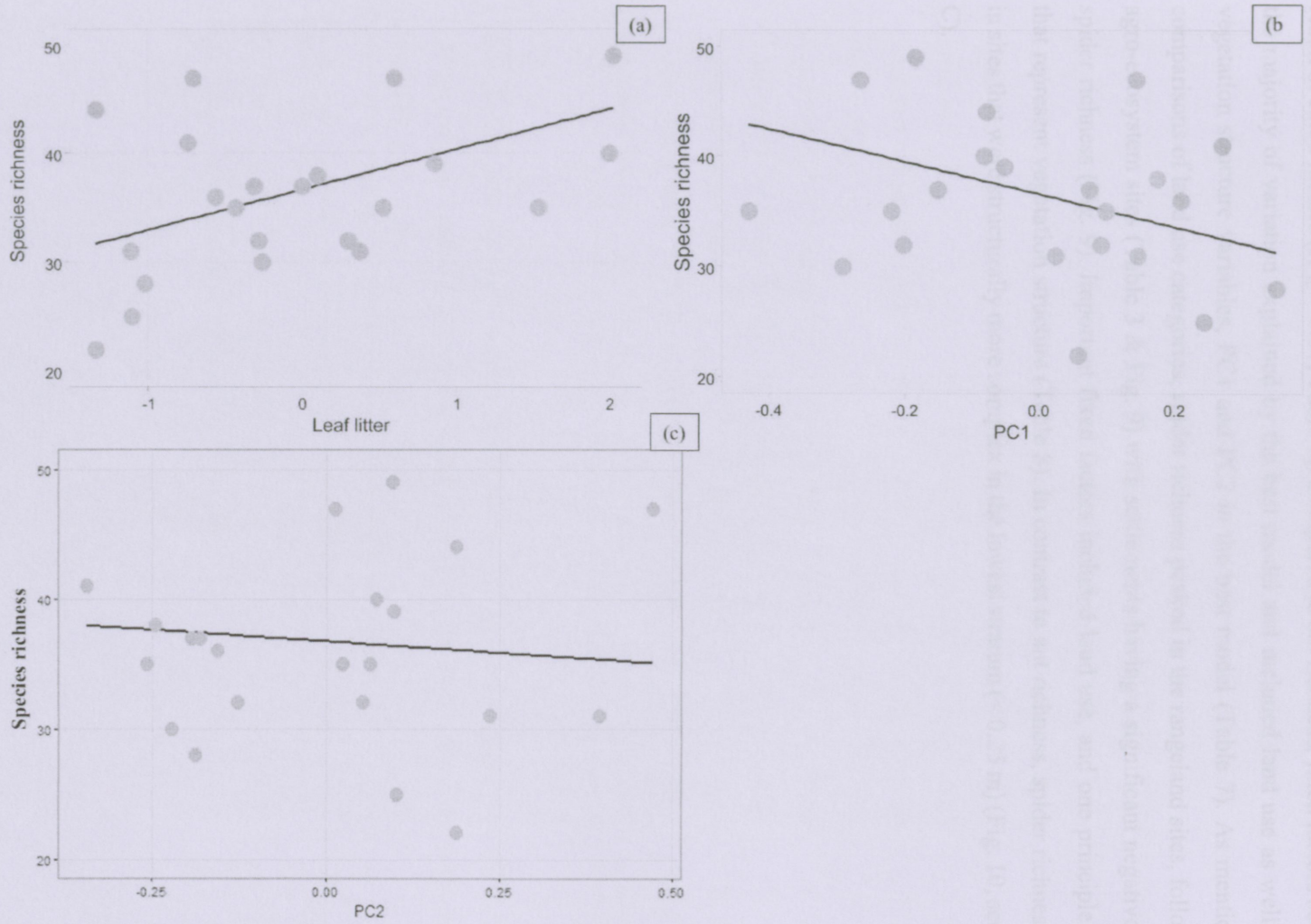


Figure 8. Plots showing the relationship between observed species richness and leaf litter (a), (b) PC1 and (c) PC2 during the two sampling seasons. Lines display model predictions.

The ten best models for spider richness, ranked according to their AICc is listed in Table 7. The best GLMM model explained 47% of the variation in spider species richness, 37% of this was the result of fixed factors ($R^2_m = 0.37$, Table 7). Compared to ant richness, fixed factors accounted for the majority of variation explained by the best model and included land use as well as vertical vegetation structure variables, PC1 and PC2 in the best model (Table 7). As mentioned in the comparisons of land use categories, spider richness peaked in the rangeland sites, followed by the agro-ecosystem sites (Table 3 & Fig. 9) with settlements having a significant negative impact on spider richness (Fig. 9). Important fixed factors included land use, and one principle coordinate that represent vegetation structure (Table 8). In contrast to ant richness, spider richness increased in sites that were structurally more complex in the lowest stratum (< 0.25 m) (Fig. 10, see Appendix C).

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Table 7. Comparative and summary statistics for generalized linear mixed models explaining variation in spider richness across land use types. Predictors were leaf litter, vegetation, rock cover, PC1 and PC2. Listed are the degrees of freedom (d.f.), maximum log-likelihood (LL), Akaike's bias corrected information criterion (AICc) and it's change relative to the top ranked model (ΔAICc), the model probabilities (wAICc) and the marginal and conditional R^2s. Marginal R^2 (R^2_m) is the amount of variation explained by the fixed effects, conditional R^2 (R^2_c) is that explained by the fixed and random effects.

Model	d.f.	LL	AICc	ΔAICc	wAICc	R^2_m	R^2_c
~ Land use - PC1 - PC2	9	-56.17	144.2	0.00	0.10	0.37	0.47
~ Land use - PC2	8	-59	144.3	0.06	0.10	0.38	0.48
~ Land use + PC1	8	-59.21	144.7	0.52	0.08	0.37	0.45
~ Land use	7	-62	145.5	1.32	0.05	0.37	0.48
~ Land use + Leaf litter + PC2	9	-57	145.9	1.69	0.04	0.3	0.5
~ PC1 + PC2	7	-62.3	146	1.83	0.04	0.23	0.55
~ Land use + PC1 + vegetation	9	-57.1	146.1	1.91	0.04	0.18	0.53
~ Land use + Leaf litter + PC1	9	-57.2	146.3	2.07	0.04	0.37	0.48
~ Land use + Leaf litter	8	-60	146.4	2.17	0.04	0.37	0.51
~ Land use + PC1 + PC2 + vegetation	10	-54.1	146.5	2.35	0.03	0.35	0.47

Table 8. Test statistics (χ^2), degrees of freedom (D.f) and p values from type III Wald chisquare tests on the best models (top ranked model from Table 7). Explanatory variables included land use and PC2.

	χ^2	D.f	p
Land use	7.2	2	>0.004 **
PC2	24.81	1	0.127

Significance level codes: 0 '***' 0.001 '**' 0.01 '*' 0.5 '.' 0.1 ' ' 1

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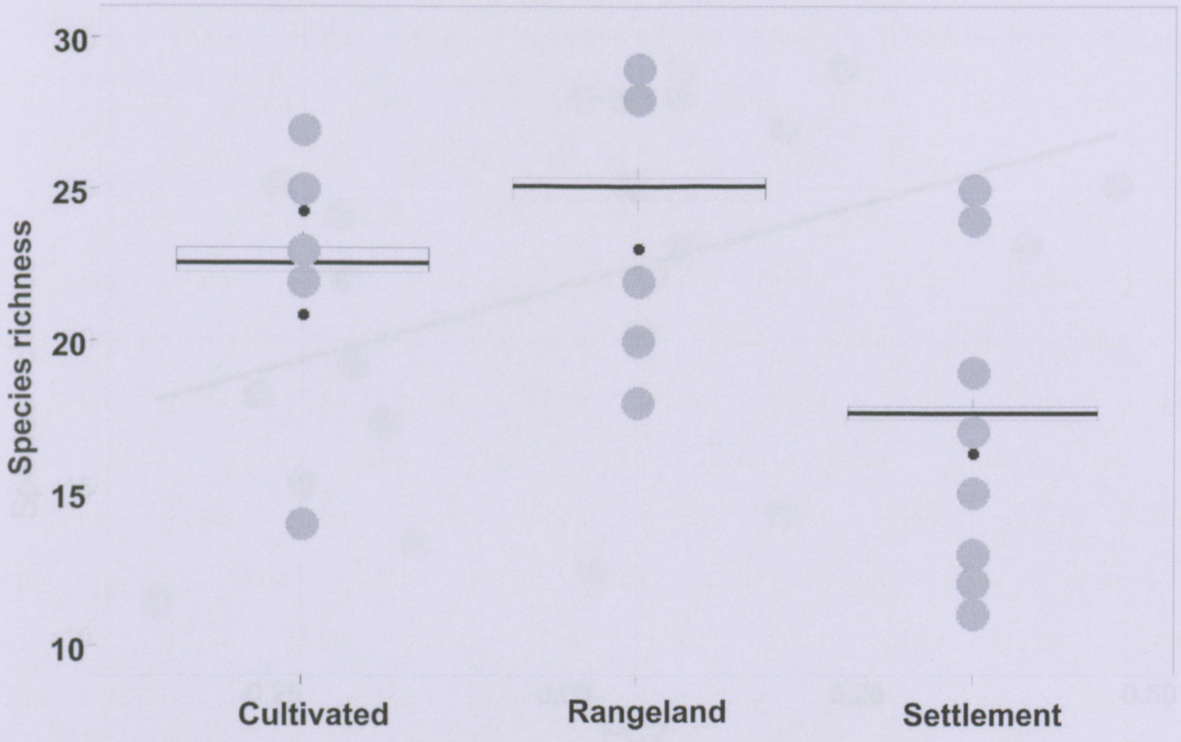


Figure 9. Box plots showing the median (line) and interquartile range (box) for the model, with outliers (black dots), summarizing the total species richness in each land use. Grey symbols are the observed richness.

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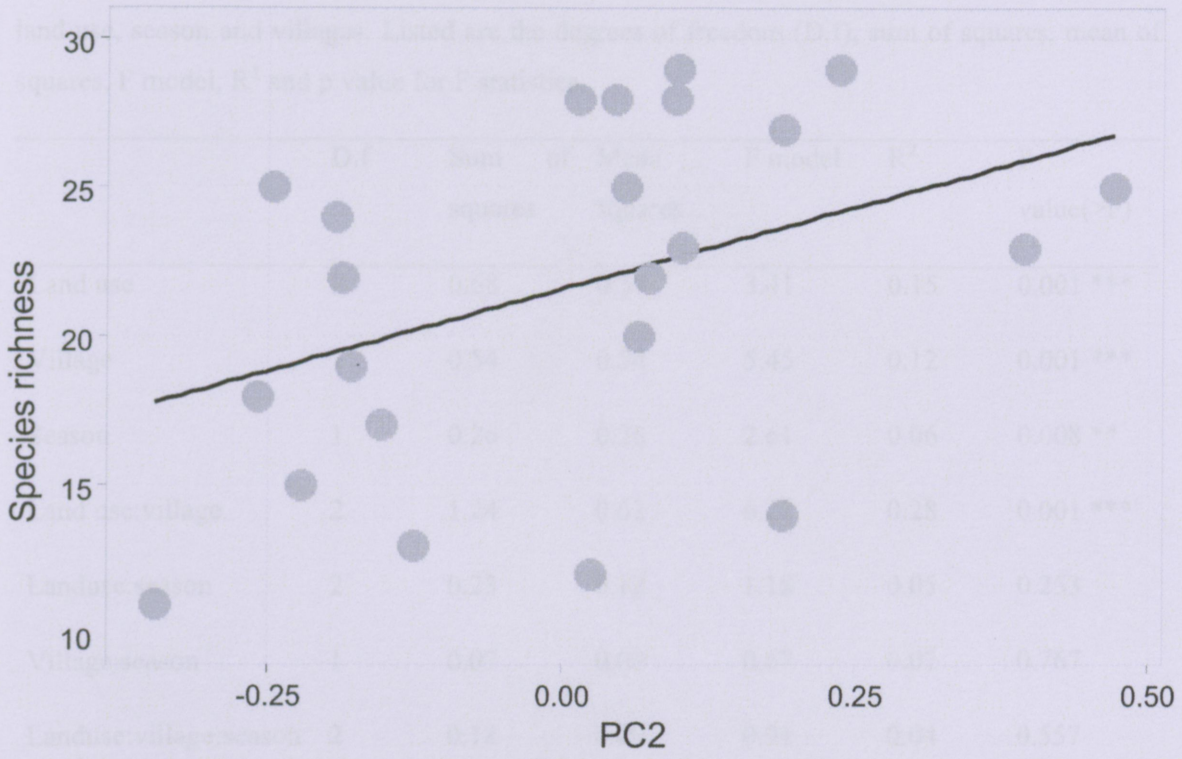


Figure 10. Plot showing the relationship between observed spider species richness and PC2. Grey circles represent observed richness while the line display model predictions.

Ant and spider assemblage structure

PERMANOVA of ant assemblages suggest that land use, season and village had significant impacts on structure (Table 9 & Fig. 11a&b). There was also an interaction between land use and village that explains largest amount of variation in assemblage structure. This suggests that the impact of land use differed between the two regions. This is mainly because of the considerable turnover within rangeland and agro-ecosystem sites of the Vyeboom region, while sites in settlement were very similar between regions (Fig. 11b). The latter probably points to filtering of ant assemblages in settlements.

There was some evidence for the assertion that land use have an impact on ant assemblage structure (ANOSIM global R = 0.13 p = 0.03). Pair-wise comparisons of land use types support the observation that it is mainly the agro-ecosystems that are distinct from the rest (Table 10).

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Table 9. Summary statistics for PERMANOVA explaining variation in ant assemblages across land use, season and villages. Listed are the degrees of freedom (D.f), sum of squares, mean of squares, F model, R^2 and p value for F statistics.

	D.f	Sum of squares	Mean squares	F model	R^2	P value(>F)
Land use	2	0.68	0.34	3.41	0.15	0.001 ***
Village	1	0.54	0.54	5.45	0.12	0.001 ***
Season	1	0.26	0.26	2.61	0.06	0.008 **
Land use:village	2	1.24	0.62	6.27	0.28	0.001 ***
Landuse:season	2	0.23	0.12	1.18	0.05	0.253
Village:season	1	0.07	0.07	0.67	0.02	0.767
Landuse:village:season	2	0.18	0.09	0.91	0.04	0.557

Significant codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

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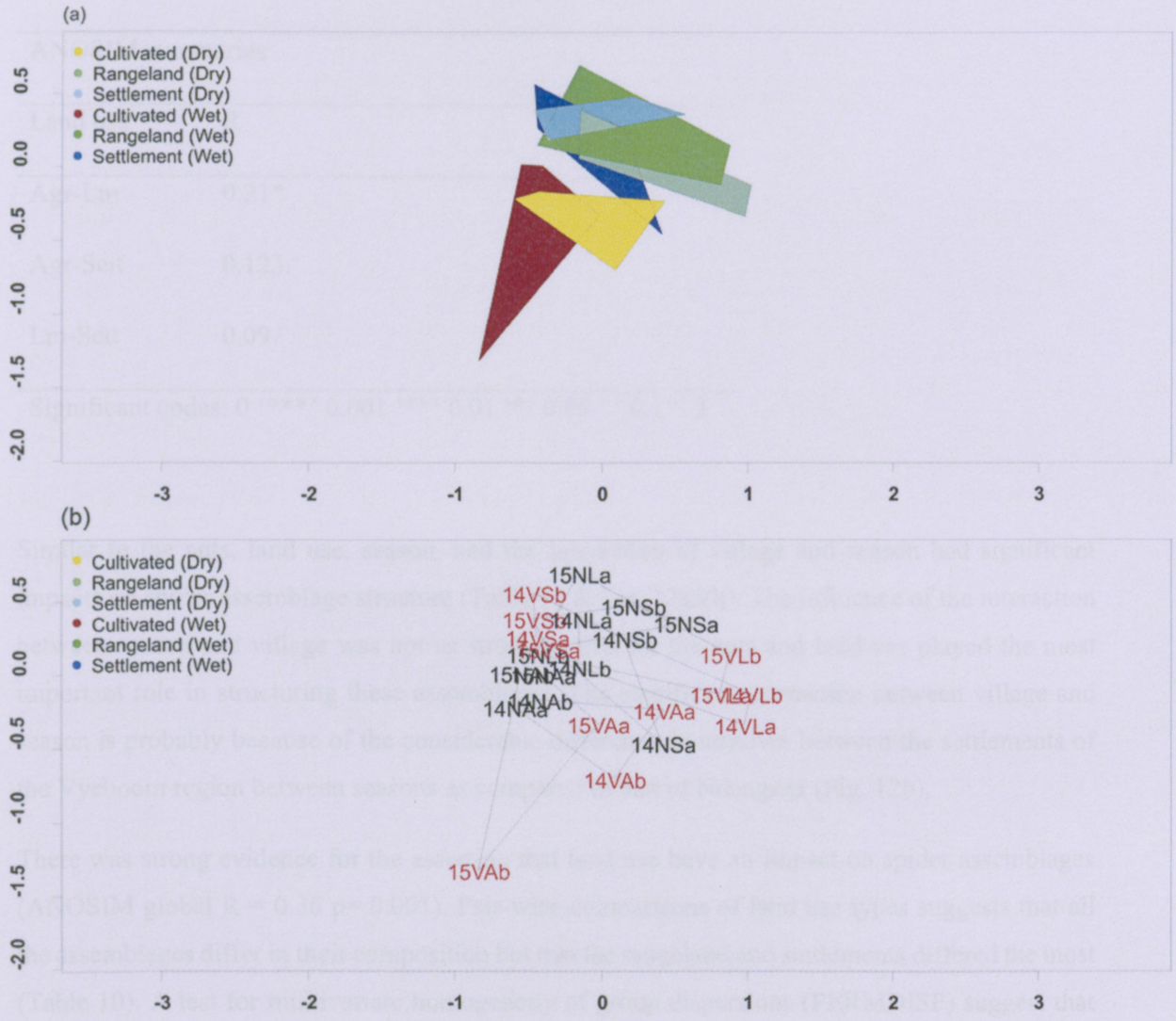


Figure 11. Non-metric multidimensional scaling of ant assemblage structure with minimum convex hulls around each of the three land use categories for each of the two seasons and villages.

A test for multivariate homogeneity of group dispersions (PERMDISP) suggest that ant turnover did not differ significantly between land use activities in both the dry season (2014) ($F_{2,9} = 1.14$ $P = 0.287$) and wet season (2015) ($F_{2,9} = 1.30$ $P = 0.5$).

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Table 10. Pair-wise comparison using ANOSIM of ant assemblages caught in the three land use types. Agr, Agro-ecosystem; Lm, Rangeland; Sett, Settlement.

ANOSIM summaries	
Land use	R
Agr-Lm	0.21*
Agr-Sett	0.123.
Lm-Sett	0.09

Significant codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Similar to the ants, land use, season, and the interaction of village and season had significant impacts on spider assemblage structure (Table 11 & Fig. 12a&b). The influence of the interaction between season and village was not as strong as that for the ants and land use played the most important role in structuring these assemblages. The significant interaction between village and season is probably because of the considerable difference in turnover between the settlements of the Vyeboom region between seasons as compared to that of Ndengeza (Fig. 12b).

There was strong evidence for the assertion that land use have an impact on spider assemblages (ANOSIM global R = 0.36 p= 0.001). Pair-wise comparisons of land use types suggests that all the assemblages differ in their composition but that the rangeland and settlements differed the most (Table 10). A test for multivariate homogeneity of group dispersions (PERMDISP) suggest that spider turnover showed no significant difference between land use activities during the dry season (2014) ($F_{2, 9} = 0.89$ $P = 0.54$) and wet season (2015) ($F_{2, 9} = 0.41$ $P = 0.74$).

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Table 11. Summary statistics for PERMANOVA explaining variation in spider assemblages across land use, season and villages. Listed are the degrees of freedom (D.f), sum of squares, mean of squares, F model, R^2 and p value for F statistics.

	D.f	Sum of squares	Mean squares	F model	R^2	P value(>F)
Land use	2	1.06	0.53	4.05	0.21	0.001 ***
Village	1	0.45	0.45	3.46	0.09	0.001 ***
Season	1	0.37	0.37	2.79	0.07	0.001 ***
Land use:village	2	0.65	0.33	2.49	0.13	0.001 ***
Landuse:season	2	0.36	0.18	1.37	0.07	0.107
Village:season	1	0.20	0.20	1.53	0.04	0.078 .
Landuse:village:season	2	1.00	0.15	1.12	0.06	0.301

Significant codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

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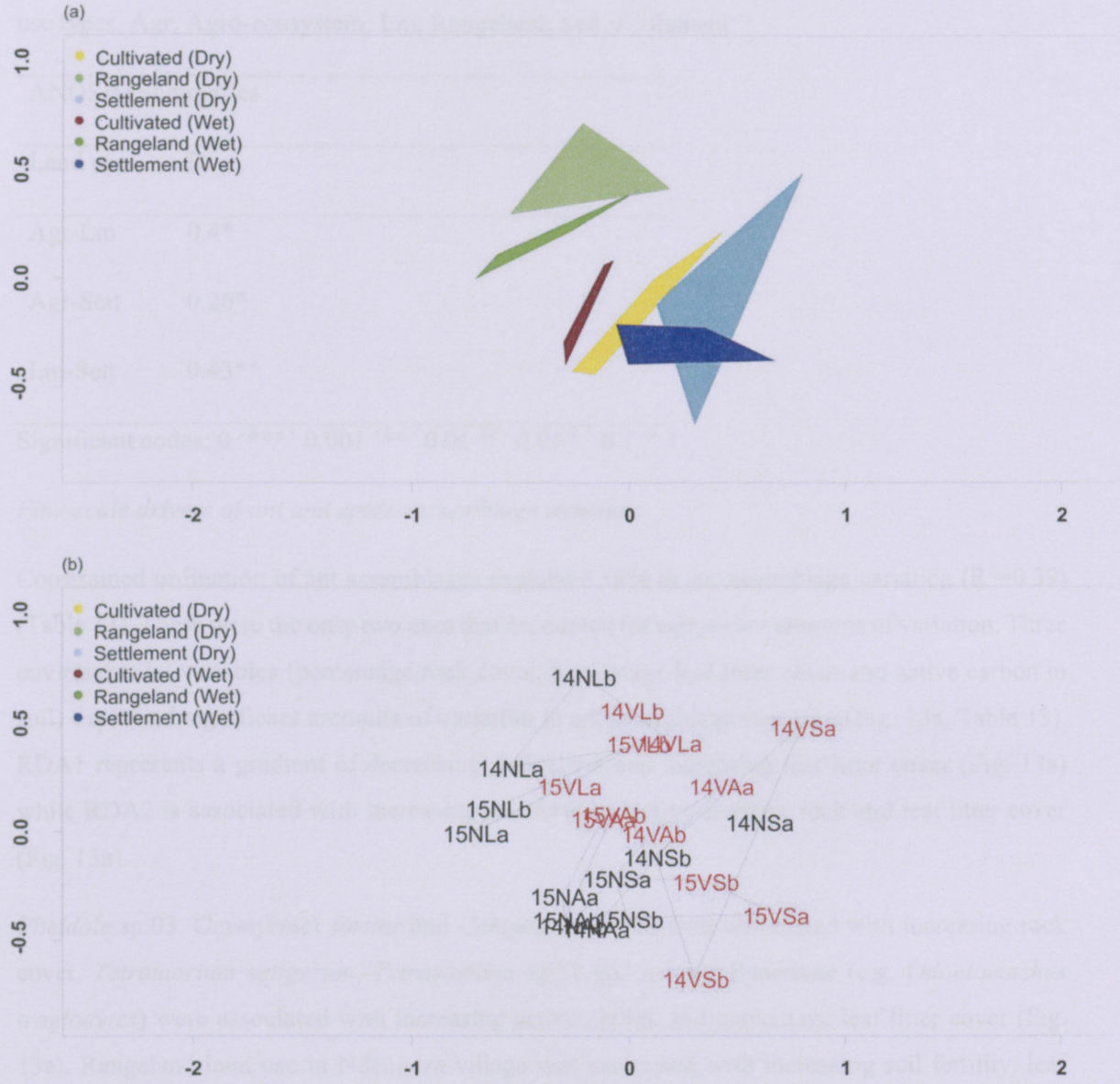


Figure 12. Non-metric multidimensional scaling of spider assemblages with minimum convex hull for each of the three land use categories and two seasons of sampling in the two villages.

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Table 12. Pair-wise comparisons using ANOSIM for spider assemblages caught in the three land use types. Agr, Agro-ecosystem; Lm, Rangeland; Sett, Settlement.

ANOSIM summaries	
Land use	R
Agr-Lm	0.4*
Agr-Sett	0.26*
Lm-Sett	0.43**

Significant codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Fine-scale drivers of ant and spider assemblage structure

Constrained ordination of ant assemblages explained 18% in ant assemblage variation (R^2=0.39) (Table 13). These were the only two axes that accounted for significant amounts of variation. Three environmental variables (percentage rock cover, percentage leaf litter cover and active carbon in soil) explained significant amounts of variation in ant assemblage structure (Fig. 13a, Table 13). RDA1 represents a gradient of decreasing rock cover and increasing leaf litter cover (Fig. 13a) while RDA2 is associated with increasing soil fertility (active carbon), rock and leaf litter cover (Fig. 13a).

Pheidole sp.03, *Ocymyrmex fortior* and *Camponotus* sp.18 were associated with increasing rock cover, *Tetramorium setigerum*, *Tetramorium* sp.31 and a large Ponerinae (e.g. *Odontomachus troglodytes*) were associated with increasing active carbon and percentage leaf litter cover (Fig. 13a). Rangeland land use in Ndengeza village was associated with increasing soil fertility, leaf litter cover and rock cover (Fig. 13b) while Vyeboom village rangeland was closely associated with decreasing leaf litter (Fig. 13b).

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Table 13. Environmental variables that explained significant amounts of variation in ant assemblage structure for both the dry and wet seasons.

Model summaries			
R^2	0.39		
R^2adj	0.18		
Variables	R^2Cum	AdjR^2Cum	*P* value
Percentage leaf litter	0.10	0.06	0.007**
Percentage rock cover	0.19	0.12	0.007**
Active Carbon in soil	0.26	0.15	0.035*

Significance code: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

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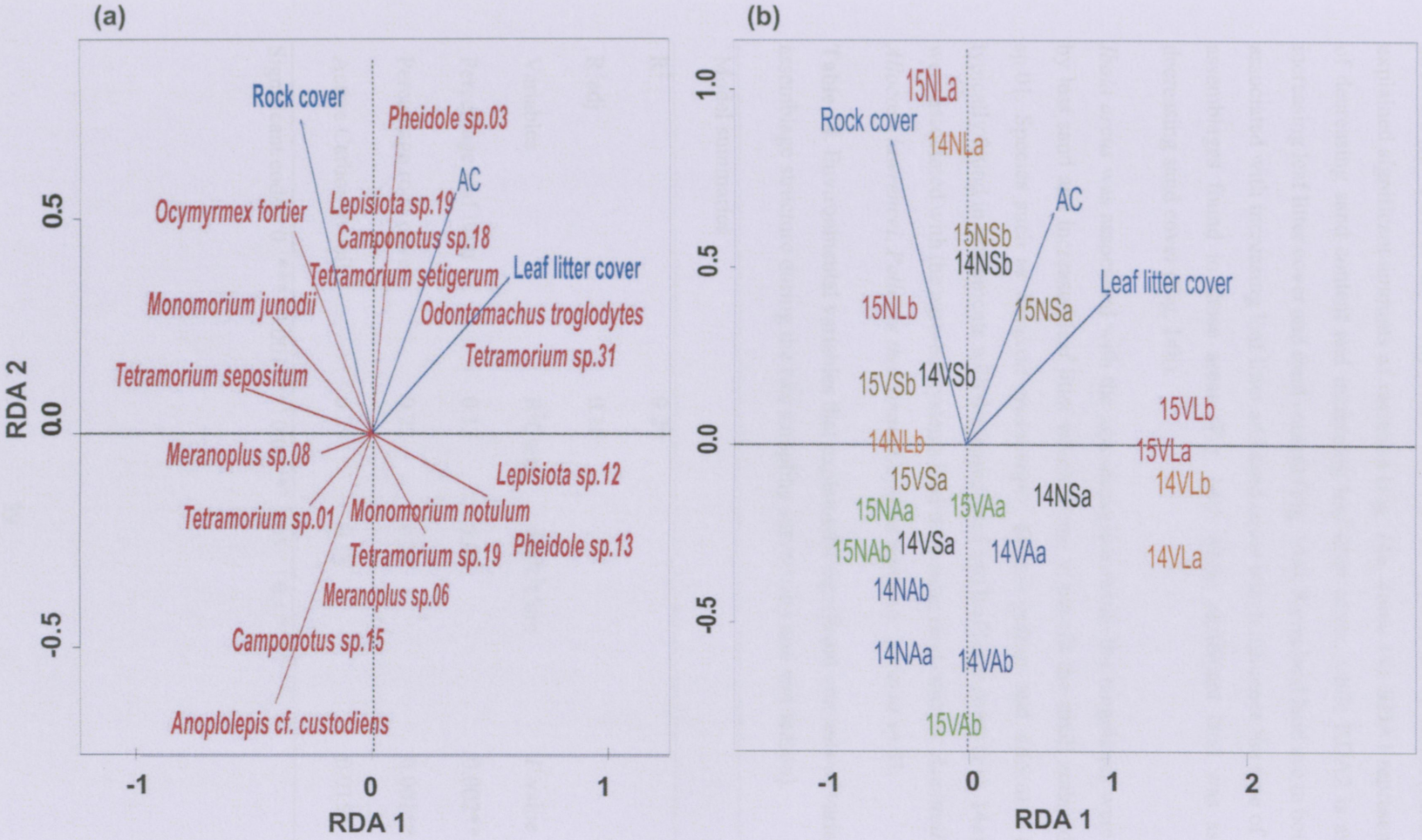


Figure 13. Biplot of first two axes of RDA ordination of ant assemblage structure of (a) species and (b) sites. Rock cover = percentage rock cover, AC = active carbon in soil, Leaf litter cover = percentage leaf litter cover. Agro-ecosystem: Blue (Dry season), Green (Wet season); Rangeland: Orange (Dry season), Red (Wet season); Settlement: Black (Dry season), Brown (Wet season).

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The constrained ordination of spider assemblages also explained 18% of the spider assemblage structure (R^2=0.29, R^2adj=0.18) (Table 14). Two environmental variables (sand and leaf litter) explained significant amounts of variation (Fig. 14a, Table 14). RDA1 represented a gradient of decreasing sand content and increasing leaf litter cover, while RDA2 is associated with increasing leaf litter cover and sand content (Fig. 14a). Rangeland land use in both villages was associated with increasing leaf litter and sand cover which influence the type of spider species assemblages found in those areas (Fig. 14b) while settlement land was associated with decreasing sand cover (Fig. 14b).

Ibala arcus was associated with the agro-ecosystem while the rangelands were characterized by less sand and increased leaf litter which seems to benefit the small zodariid, *Microdiores* sp.01. Species such as *Pardosa crassipalpis, Ocyale guttata* and *Allocossa leipoldti* were typically found in settlements with decreased sand and leaf litter cover (Fig. 14a). Four species were associated with the agro-ecosystems land increasing sand content: *Asemesthes ceresicola*, *Allocosa lawrencei*, *Pellenes tharianae*, *Oxyopes dumonti*, *Micaria* sp.01.

Table 14. Environmental variables that explained a significant amounts of variation on spider assemblage structure during the two sampling survey (dry and wet season).

Model summaries			
R^2	0.29		
R^2adj	0.18		
Variables	R^2Cum	AdjR^2Cum	*P* value
Percentage leaf litter	0.12	0.08	0.002**
Percentage rock cover	0.22	0.14	0.001**
Active Carbon in soil	0.29	0.18	0.015*

Significant codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

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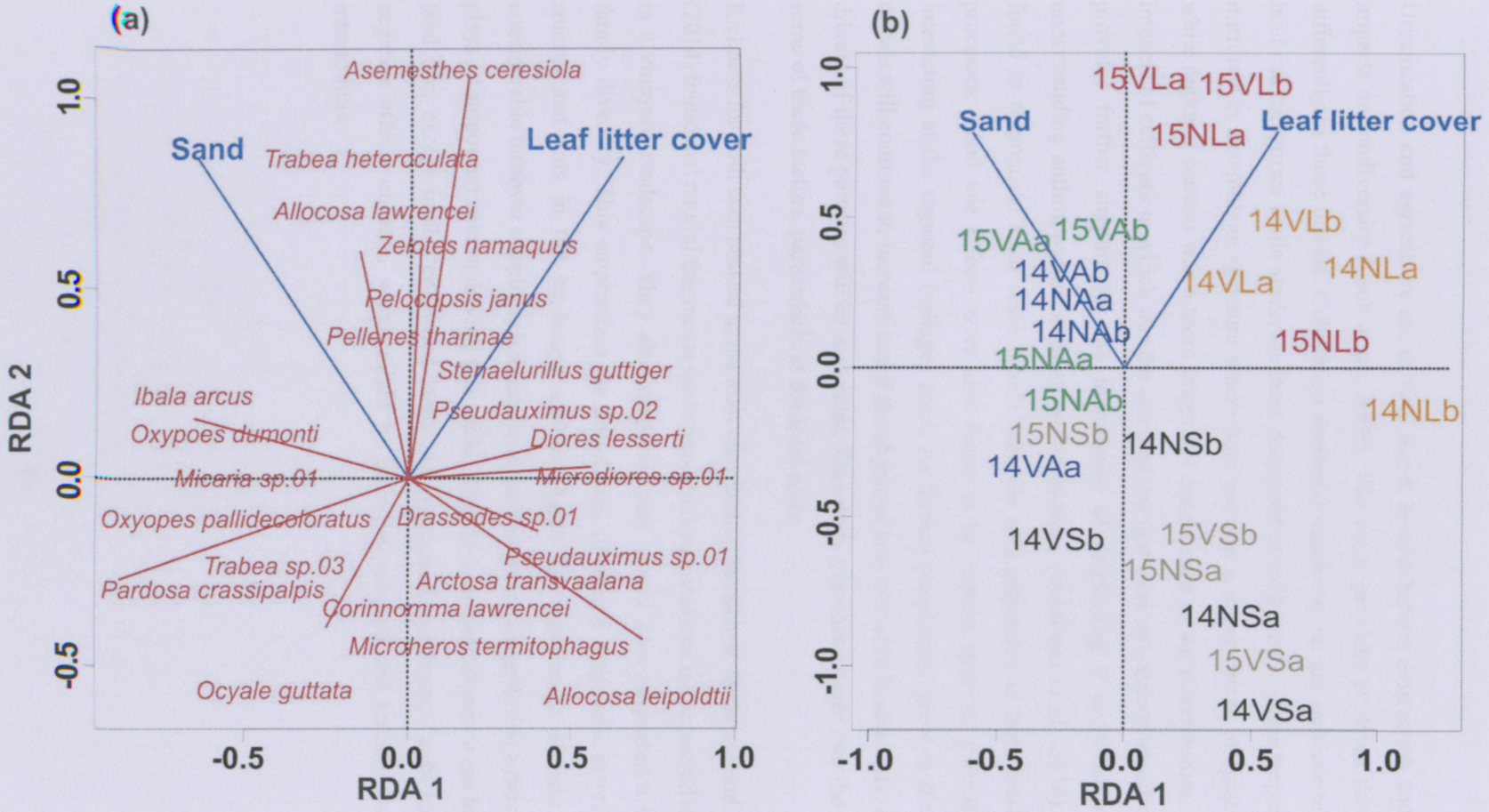


Figure 14. Biplot of first two axes of RDA ordination of spider assemblage structure of (a) species and (b) sites. Sand = percentage sand in soil, Leaf litter cover = percentage leaf litter cover. Agro-ecosystem: Blue (Dry season), Green (Wet season); Rangeland: Orange (Dry season), Red (Wet season); Settlement: Black (Dry season), Brown (Wet season).

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CHAPTER 4: DISCUSSION

Urbanization and agriculture are driving forces behind habitat destruction and has dramatic impacts on biodiversity (Lach *et al*., 2009). This study provides evidence that taxa respond differently to these threats. Cultivation impacted negatively on ant richness relative to other land use categories while spider richness decreased in settlements. These responses were also mirrored in assemblage structure where land use was a major driver of spider assemblages while regional context was a more important determinant of ant communities, except for the impacts of cultivation, which was the one land use that were not context specific for ants. This provides further support for the importance of employing a multi-taxon approach in understanding anthropogenic impacts on biodiversity (Yekwayo *et al*., 2016). Spiders were found to respond to local scale factors whereas ants responded to both local and regional processes. Land use effects were also found to be context specific, particularly for ants, interacting at the regional (village) level. As human populations grow in the region, these threats will continue to increase and if the objective is to conserve biodiversity, identifying the drivers of these patterns will be important. This study provides insight into the importance of some of these factors, particularly at the local scale.

Richness for both taxa peaked in the least transformed landscape, the rangeland. Gaigher *et al*. (2014) found that natural fragments made important contributions to the overall spider diversity in a vineyard landscape. They also suggested that natural sites supported a relatively high family diversity. This emphasizes the importance of natural fragments as refuges for most spiders and ants in the landscape and highlights the importance of land sparing. The considerable turnover of spiders between land use categories suggests that a mosaic of land use plays an important role in driving this spider diversity. In contrast ants were less affected by land use, except in the cultivated lands. As mentioned, cultivated lands had particularly negative effects on ants, which might be linked to reduced soil fertility and competitive interactions.

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Spiders

Spider species richness was mainly affected by land use and points to the response of spiders to local conditions to a greater extent than ants. They were more sensitive to landscape elements and heterogeneity than ants and seem to respond to these changes at finer temporal and spatial scales. This might be related to their superior dispersal abilities for example some spider species (e.g. Linyphiidae spiders) use ballooning and rappelling to disperse and they can disperse to long distances and short distances (Bonte *et al*., 2009). A far smaller proportion of variation in spider richness was explained by random factors as compared to ants. Spider richness therefore does not seem to vary much between the villages and across seasons and thus are far less dependent on context or history. There was a considerable amount of variation in spider richness that was not explained and this could be related to factors which were not measured, such as temperature, competition with ants which were very abundant in the settlements.

In a study of urban landscapes, Philpott *et al*. (2014) found that spider species richness was the highest in vacant lots (vacant for at least 9 - 15 years), followed by gardens and lowest in the forests (natural forest fragments within parks). Spider abundance increased in built up areas, and decreased in forb rich areas and taller shrubs. Philpott *et al*. (2014) also found that spider richness likewise increased in more built up areas and less open space and with fewer trees. Abundance and richness of arthropods was significantly dependant on the urban habitat type and patterns of abundance and richness differed depending on trophic and taxonomic group (Philpott *et al*., 2014).

Increased complexity as measured by increasing hits at the lowest stratum (0 – 25 cm) positively affected epigeal spider species richness. There is considerable evidence that vegetation structure and habitat diversity increases spider diversity (Wu *et al*., 2009; Pozzi *et al*., 1998). This seems to be true for other strata as well including the grass layer (Horvath *et al*., 2009) and tree layers Foord *et al*. (2008). Several studies also emphasise the importance of small-scale variation in habitat structure for spider diversity (Alaruikka *et al*., 2002; Foord *et al*., 2008; Dennis *et al*., 2001; Harris *et al*., 2003). Tscharntke *et al*. (2005) found that spider diversity in organic and conventional agriculture was supported by the increase in spider diversity with landscape complexity. Underwood & Quinn (2010) found that herbaceous vegetation and leaf litter was the most important factor explaining spider richness in an area. Underwood & Quinn (2010) suggest that spiders may often be responding to decreased prey,

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due to disturbance such as fire in the habitat. Otoshi *et al.* (2015) found that several ground cover variables such as mulch, bare soil, concrete and leaf litter were correlated with the increase in spider activity density and spider richness.

In this study 29% of variation in spider assemblages was mainly explained by two environmental variables (percentage sand in the soil and leaf litter cover). Land use did have an effect on spider assemblage structure. Sattler *et al.* (2010) found that management intensity and temperature variables measured at each site structured urban spider communities. Urban environments can be seen as a close knit mosaic of many different habitat types that is disturbed on a regular and frequent basis by the many human activities, thus inhibiting both the development and installation of spatially organized environmental variable and biotic processes (Sattler *et al.*, 2010).

Batáry *et al.* (2008) suggested that the high intensity management of grassland could result in fewer species, which may lead to poor assemblages dominated by a few r-selected species. Management intensity of grasslands at the local scale had no significant effect on spider assemblages structure (Batáry *et al.*, 2008). At local scale vegetation and leaf litter cover were the two most important variables that significantly affected the overall spider assemblage structure (Batáry *et al.*, 2008) which corresponds to the findings in this study where leaf litter cover and percentage sand in the soil was the most important variables. Sattler *et al.* (2010) reported that the loss of habitat complexity through increased management (frequency of lawn/meadow cuttings) was an important variable explaining spider community structure. In our study the amount of sand in the soil affected species assemblage structure and settlements affected spiders negatively.

Ants

In contrast to spiders where land use was important in affecting richness, ants were influenced by both local (leaf litter cover and vertical vegetation structure) and regional and temporal processes (Village and seasonality). This suggest species pool of the landscape from which local communities are drawn is important to ant diversity and that ant richness is also probably largely influenced by historical processes and limited by dispersal. The amount of leaf litter cover on the ground affected ant richness positively, this might be because litter on the ground provides micro habitats for ants above and beneath litter (Pacheco & Vasconcelos, 2012). An increase in complexity at lower vegetation strata had a negative effect on ant richness (Gibb & Parr, 2013).

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Uno *et al.* (2010) compared urban green spaces and found that ant richness in vacant lots (formerly residential properties vacant for at least 9 - 15 years) did not differ from other habitat types, but forests (natural forest fragments within parks) had a significantly greater richness than gardens (organically managed community gardens). There was general support for two main groups of factors that influenced species richness of ants in this urban spaces: landscape factors, and competitive interactions (Uno *et al.*, 2010).

The positive impact of increased leaf litter cover in this study might be because litter on the ground provides micro habitats for ants, particularly in these woodlands where leaf litter are often relatively rare (Munyai & Foord 2012). Ivanov & Keiper (2010) in their study found a negative response of some ant species to decreased leaf litter depth. Rocha-Ortega & Castaño-Meneses (2015) found that turnover of ground ants was correlated with the degree of soil water saturation and relative soil moisture and that the vegetation structure had a greater effect on the alpha diversity of ground and leaf litter ant species.

Munyai & Foord, (2012) in their ant epigaic study found that richness increased in habitats characterized by high proportions of bare ground. In our study increased complexity at lower vegetation strata had a negative effect on ant richness. Ossola *et al.* (2015) also found that habitat complexity was a major determinant of structure and diversity of ants with ant species richness that were high in low complexity habitats.

Ivanov & Keiper, (2010) sampled ants in urban landscapes and variably sized fragments of forested areas. Ants of these urban forests were habitat generalists, adapted to a greater range of environmental conditions and were able to persist in these often degraded environment (Ivanov & Keiper, 2010). In a recent study, Ossola *et al.* (2015) found that ant abundance was negatively correlated with increased litter cover in the three habitat complexity types namely: low-complexity parks (native and non-indigenous eucalyptus trees with a simplified understory), high-complexity parks (golf courses) and high complexity remnants (nature reserves).

Other local scale processes not measured could have been associated with these random factors and these include temperature, food availability, and soil moisture. One important aspect of ant ecology, competition, was not evaluated in this study. However, in a study in the Kruger National Park, Parr (2008) found that *Pheidole* and *Crematogaster* species, aggressively displace other ants from baits but were never displaced themselves. But one species, *Anoplolepis cf. custodiens* won all the encounters, except with *Crematogaster* and *Myrmicaria*

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natalensis was dominant in all encounters except with *Pheidole* and *Crematogaster*. Similar to our study, ant richness was affected negatively in sites where *Anoplolepis* cf. *custodiens* dominated. *Anoplolepis cf. custodiens* was the most abundant ant species found in both the settlement and agro-ecosystem. This species might be similar to *Tapinoma sessile* which increased its abundance with proximity to human structures (Toennisson *et al*., 2011). This could be because *Tapinoma sessile* displaces other ant species or that there is costs associated by increased disturbance regimes that only *T. sessil*e can tolerate disturbance (Salyer *et al*., 2014). Carpintero *et al*. (2003) found that ants in urban habitats tend to have large aggressive colonies, generalist, opportunistic species and competitive dominants.

A change in ant assemblage structure was correlated with increased leaf litter and active carbon content of the soil. The agro-ecosystem was dominated by sandy soil and shrubs (Fig. 14b). The structural complexity (soil texture, soil moisture) of the soil have been shown to influence the composition of ant assemblages in both natural and semi natural ecosystems (Ossola *et al*. (2015). Soil structure can also indirectly influence vegetation composition and soil hydrological properties which are important factors shaping ant assemblages (Ossola *et al*., 2015). Active carbon in the soil is linked to increased soil fertility and played an important role in structuring ant assemblages. Sanabria *et al*. (2014) found that increased soil fertility positively affected ant diversity, while rocks provide potential refuges after disturbance, e.g. burning (Underwood & Quinn, 2010).

The settlement and agro-ecosystem had decreased leaf litter cover and decreased rock cover which provide some explanation for the distinct assemblage structure of these to land use categories. Land use impacts on ants were contexts specific, i.e. varied between the two regions. Ślipiński *et al*. (2012) study which found that the structure of the ant assemblages in the four urban biotopes differed significantly. They also found that species composition recorded in green areas in housing estates and bordering streets constitutes subsamples of the assemblage recorded in urban wooded areas and parks. Byrne, (2007) indicated that in urban ecosystems land use management was the main factor shaping habitat complexity which in turn affected ants assemblages structures. Uno *et al*. (2010) suggested that competitive interactions may be a factor mediating diversity in urban areas.

The significant relationship between species assemblage and three environmental variables (percentage leaf litter, percentage rock cover and active carbon in soil) showed that ants preferred habitat complexity at high strata significantly impacted on species assemblages. The

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most common genera on these open sites were *Anoplolepis cf. custodiens*, *Camponotus* sp.15, *Meranoplus* sp.05, *Pheidole* sp.13 and *Lepisiota* sp.12. Ossola *et al.* (2015) indicated that habitat complexity can ultimately act as an environmental filter for species through their morphological traits, contributing to structure ant assemblages and evolution of ant species over longer period. Settlements in this study were dominated by polygamous (*Pheidole*, *Monomorium* and *Anoplolepis*), opportunistic (*Tetramorium* and *Meranoplus*) and competitive (*Pheidole* and *Crematogaster*) genera known in the region (Munyai & Foord, 2015).

This study is the first in Africa to report on the impacts of rural landscapes on ant and spider diversity in a matrix consisting of cultivation, settlements and rangeland. These rural landscapes dominate the African landscape and will in future become even more prevalent. The study area has a remarkably diverse invertebrate fauna partly as a result of the mosaic of disturbances (spiders) and partly because of historical and dispersal processes (ants). Cultivation has got particularly negative impacts on ant diversity and might be linked to filtering and competitive processes, while the decreased habitat complexity (richness) decreased litter cover and the fact that villages seems to be developed on more fertile soils (assemblage structure) affected spider diversity. These variables are important to consider if we are to operationalize the conservation of these taxa.

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Appendix A: Twelve action sites, their codes, coordinates and the vegetation types they occur in of the twelve replicates occurring at different vegetation types at the two villages from Mucina and Rutherford (2006).

Village	Land use	Replicate code	Latitude	Longitude	Vegetation types
Vyeboom	Settlement	VSa	-23.14	30.38	Granite Lowveld
Vyeboom	Settlement	VSb	-23.14	30.39	Gravelotte Rocky Bushveld
Vyeboom	Agro-ecosystem	VAa	-23.14	30.38	Granite Lowveld
Vyeboom	Agro-ecosystem	VAb	-23.15	30.39	Granite Lowveld
Vyeboom	Rangeland	VLa	-23.17	30.41	Gravelotte Rocky Bushveld
Vyeboom	Rangeland	VLb	-23.17	30.41	Gravelotte Rocky Bushveld
Ndengeza	Settlement	NSa	-23.31	30.40	Granite Lowveld
Ndengeza	Settlement	NSb	-23.30	30.40	Granite Lowveld
Ndengeza	Agro-ecosystem	NAa	-23.31	30.41	Granite Lowveld
Ndengeza	Agro-ecosystem	NAb	-23.31	30.41	Granite Lowveld
Ndengeza	Rangeland	NLa	-23.33	30.42	Gravelotte Rocky Bushveld
Ndengeza	Rangeland	NLb	-23.33	30.42	Gravelotte Rocky Bushveld

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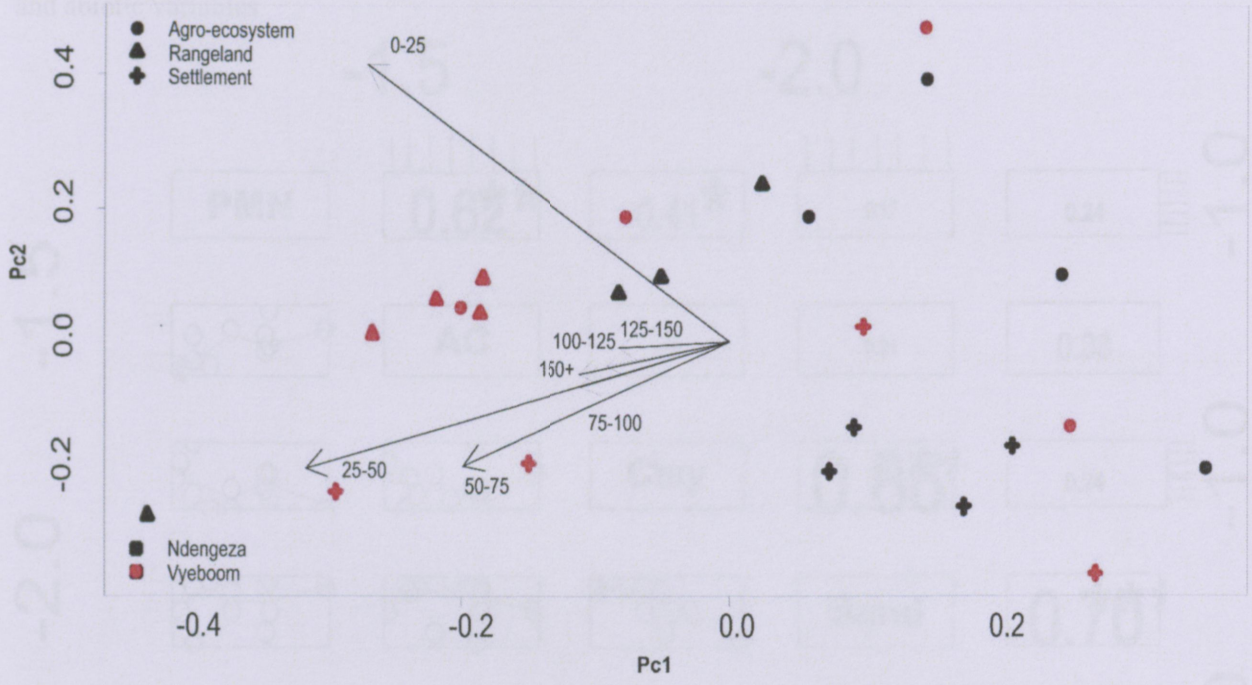
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Appendix B: Soil properties of all 12 replicates on the two action sites. Soil samples collected and analyzed once in June 2014.

Village	Replicate	Active Carbon (ppm)	PMN (µgN/gDM/ week)	Clay %	Silt %	Sand %
Ndengeza	NAa	333	0.8	20	7	73
Ndengeza	NAb	317	1	24	9	67
Ndengeza	NLa	386	5.8	18	7	75
Ndengeza	NLb	389	1.8	30	15	55
Ndengeza	NSa	436	6	22	12	66
Ndengeza	NSb	394	3.9	18	15	67
Vyeboom	VAa	332	2.7	12	10	78
Vyeboom	VAb	341	1.7	16	8	76
Vyeboom	VLa	368	6	14	8	78
Vyeboom	VLb	391	3.9	12	14	74
Vyeboom	VSa	355	5.8	21	20	59
Vyeboom	VSb	394	9	12	12	76

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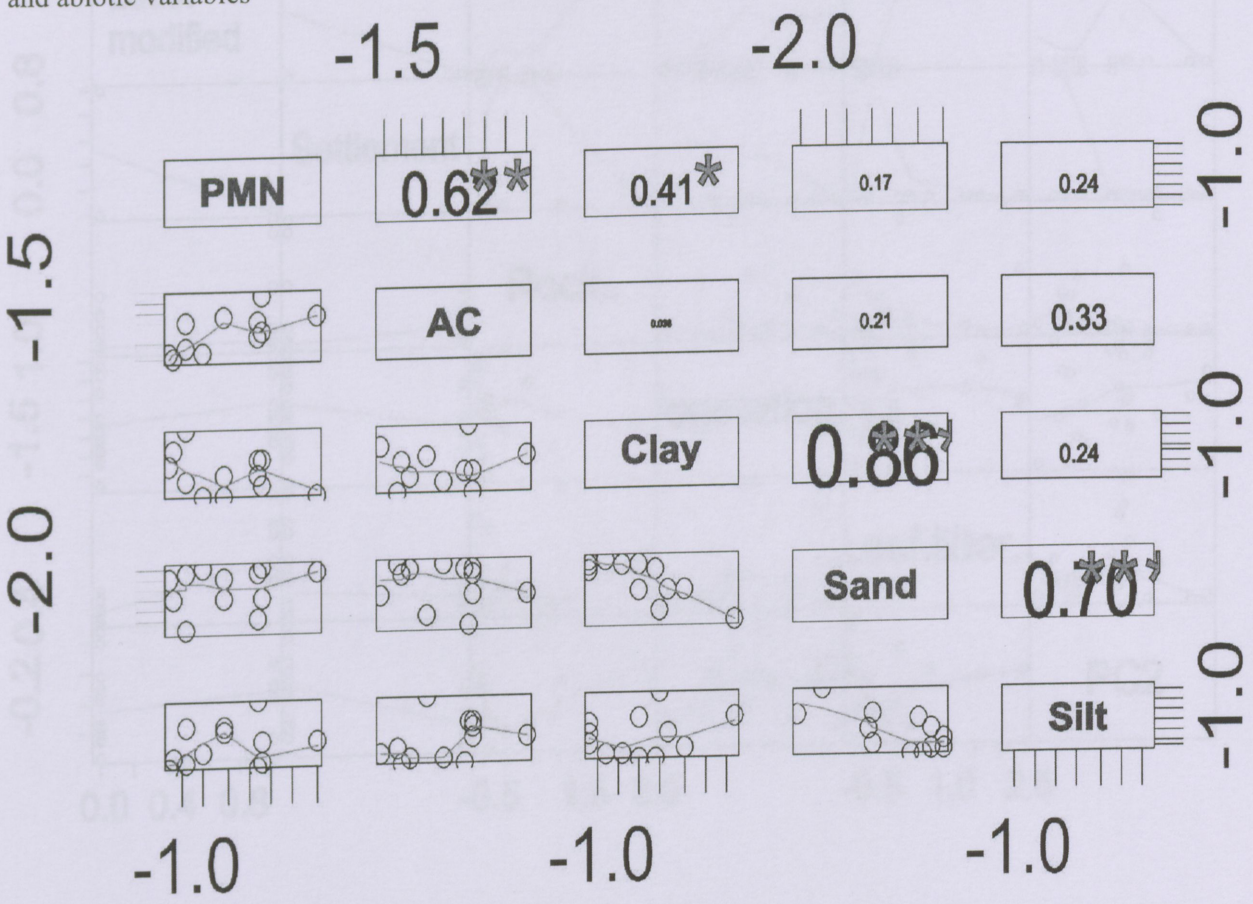
Appendix C: Principal Components Analysis (PCA) based on vertical distribution of vegetation among land use and villages during dry season (2014) and wet season (2015).



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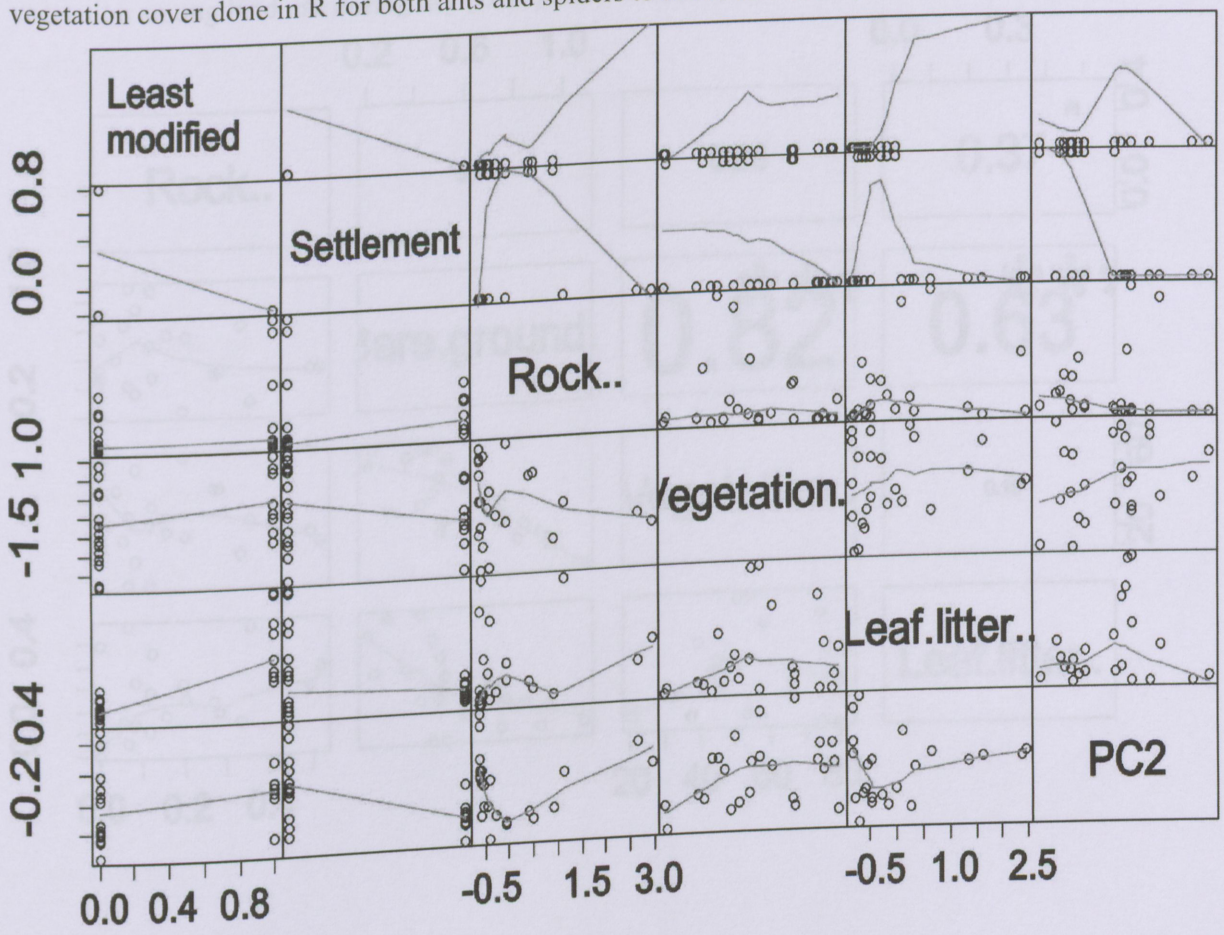
Appendix D: Pearson's product moment corrections done in R to avoid collinearity among 5 biotic and abiotic variables



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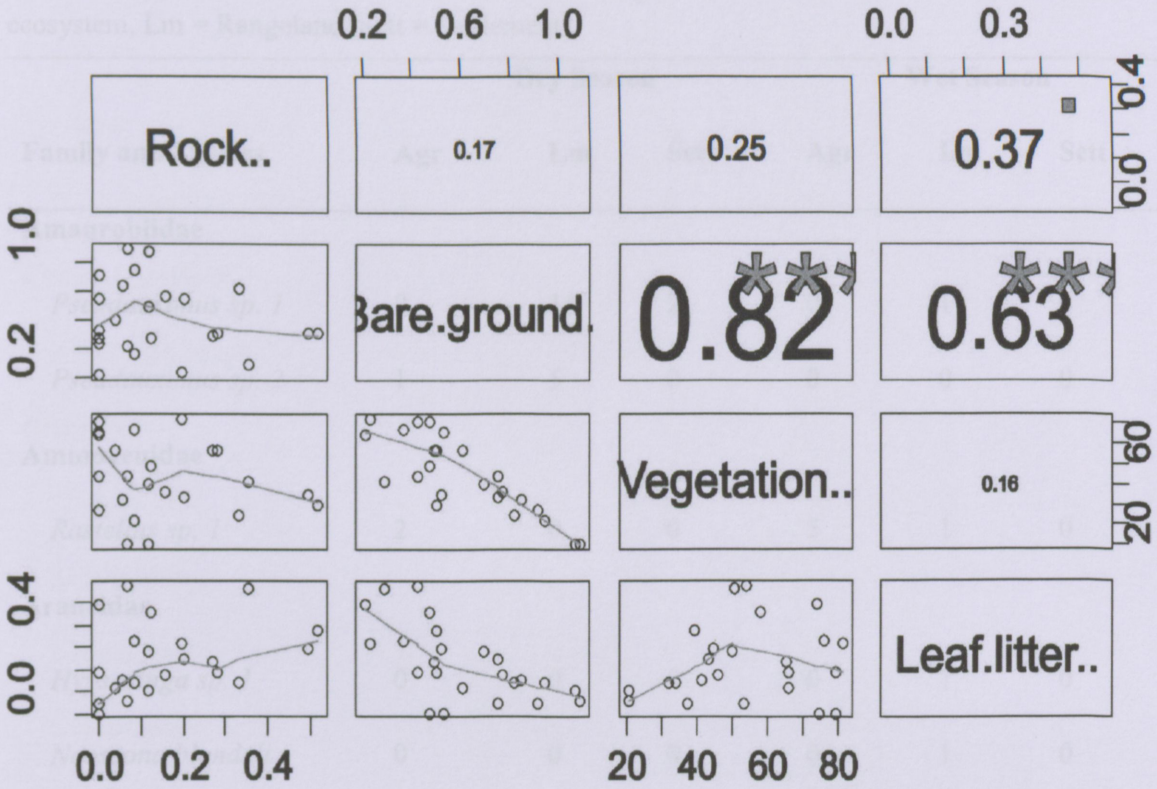
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Appendix E: Pearson's product moment correlation for land use, horizontal cover and vertical vegetation cover done in R for both ants and spiders to avoid collinearity



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Appendix F: Pearson's product moment correlation on horizontal vegetation distribution of vegetation among land use during dry (2014) and wet (2015) season of sampling.



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Appendix G: Family and spider species collected during the dry (2014) and wet (2015) season in different land use. The abundance of the species collected in each replicate is given. Agr = Agro-ecosystem, Lm = Rangeland, Sett = Settlement.

	Dry Season			Wet Season		
Family and Species	**Agr**	**Lm**	**Sett**	**Agr**	**Lm**	**Sett**
Amaurobiidae						
Pseudauximus sp. 1	0	14	2	0	1	0
Pseudauximus sp. 2	1	5	0	0	0	0
Ammoxenidae						
Rastellus sp. 1	2	0	0	5	1	0
Araneidae						
Hypsosinga sp. 1	0	0	0	0	1	0
Neoscona blondeli	0	0	0	0	1	0
Caponiidae						
Caponia chelifera	0	1	0	1	0	0
Clubionidae						
Clubiona sp.1	0	0	0	0	1	0
Corinnidae						
Cambalida dippenaarae	0	0	0	2	0	0
Copa flavoplumosa	0	0	0	1	2	0
Corinnomma lawrencei	24	1	3	13	6	8

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Corinnomma semiglabrum	0	0	0	0	5	0
Ctenidae						
Ctenus caligineus	0	2	0	0	0	0
Ctenus pulchriventris	0	1	1	0	1	0
Cyrtaucheniidae						
Ancylotrypha sp. 1	0	0	1	0	0	0
Dictynidae						
Dictyna sp. 1	1	0	0	0	0	1
Eutichuridae						
Cheiracanthium sp. 1	2	0	1	0	0	0
Cheiramiona sp. 1	0	0	0	0	1	0
Gnaphosidae						
Asemesthes ceresicola	12	18	1	29	37	4
Asemesthes pallidus	0	1	0	0	0	0
Camillina sp. 1	0	1	1	0	0	0
Drassodes sp. 1	3	2	4	1	0	0
Eilica lotzi	0	0	0	0	1	0
Heser nilicola	0	0	0	2	10	0
Ibala arcus	26	3	5	65	21	18
Leptodrassex sp. 1	0	0	0	1	0	0

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Micaria sp. 1	6	2	4	1	3	2
Nomisia varia	1	1	0	1	2	1
Scotophaeus relegatus	0	0	0	1	0	0
Setaphis subtilis	3	0	1	0	1	1
Trabea sp. 3	0	1	0	0	0	0
Xerophaeus bicavus	1	0	3	0	0	0
Zelotes bastardi	0	1	0	0	0	0
Zelotes caldarius	0	0	0	0	1	1
Zelotes imm	0	0	0	1	0	0
Zelotes namaquus	5	8	2	8	9	8
Zelotes scrutatus	1	0	1	2	0	0
Zelotes sp.	1	0	3	1	1	4
Zelotes tuckeri	2	2	0	2	0	0
Zelotes uquathus	0	0	0	0	0	2
Linyphiidae						
Limoneta sirimoni	0	0	0	0	1	0
Limoneta sirimonii	0	2	0	0	0	1
Mermessus fradeorum	0	0	1	0	1	0
Metaleptyphantes perexiguus	0	0	0	0	1	0

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Metaleptyphantes perexiguuss	5	3	3	5	0	7
Pelecopsis janus	3	5	1	0	3	0
Liocranidae						
Rhaeboctesis trinotatus	0	0	0	0	1	0
Lycosidae						
Allocosa lawrencei	10	9	3	9	6	0
Allocosa leipoldti	0	0	4	0	0	1
Allocosa leipoldtii	2	41	8	2	2	16
Arctosa brevispina	0	2	0	0	0	0
Arctosa transvaalana	0	0	0	16	1	7
Evippomma squamulatum	0	2	0	0	0	0
Hippasa australis	0	2	0	1	1	1
Hogna adjacens	1	2	0	0	0	0
Hogna sp. 1	0	0	0	0	0	1
Ocyale guttata	11	3	13	7	2	19
Pardosa crassipalpis	55	5	33	44	16	30
Pardosa sp.	1	0	0	0	0	0
Pardosa sp. 2	4	0	0	1	1	3
Proevippa albiventris	3	0	0	2	0	0
Trabea heteroculata	12	4	1	6	10	4

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Trabea purcelli	1	4	1	2	0	0
Trabea sp. 3	9	2	3	10	1	9
Zenonina albocaudata	0	3	0	2	4	0
Oecobiidae						
Oecobius navus	0	0	0	0	1	1
Oonopidae						
Gamasomorpha australis	2	0	0	4	2	0
Orsolobidae						
Azanialobus lawrencei	0	0	0	0	0	1
Oxyopidae						
Oxyopes decoloratus	0	0	1	0	0	0
Oxyopes bothai	2	0	0	1	1	0
Oxyopes dumonti	5	0	1	3	2	0
Oxyopes pallidecoloratus	7	1	3	6	0	2
Oxyopes vogelsangeri	0	3	1	1	2	0
Palpimanidae						
Diaphorocellus biplagiata	0	0	0	0	1	0
Diaphorocellus sp. 2	0	0	0	0	1	0
Palpimanus sp. 1	0	3	0	0	2	0
Philodromidae						
Hirriusa arenacea	8	0	1	1	0	0

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Philodromus sp. 1	0	1	0	0	0	0
Thanatus dorsilineatus	0	2	0	0	0	0
Pisauridae						
Afropisaura rothiformis	2	0	0	0	0	0
Maypacius bilineatus	2	0	0	0	0	0
Prodidomidae						
Prodidomus sp. 1	0	2	0	0	0	0
Theuma fusca	0	1	1	1	1	0
Theuma fusca sp. 1	0	0	0	1	0	0
Salticidae						
Baryphas ahenus	0	0	0	0	2	0
Euophrys sp. 1	0	0	0	0	3	0
Evarcha striolata	0	0	0	1	0	0
Hyllus argyrotoxus	0	0	0	0	3	0
Mexcala elegans	0	1	0	0	0	0
Microheros termitophagus	2	0	0	6	0	6
Myrmarachne foreli	1	0	0	1	0	0
Myrmarachne lulengana	0	0	1	0	0	0
Natta horizontalis	0	0	0	0	0	4
Nigorella hirsuta	0	0	1	1	2	1

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Pellenes tharinae	6	0	0	13	7	0
sp. 1 PHLEGRA?	0	0	0	0	1	0
Stenaelurillus guttiger	31	28	6	44	56	29
Thyene natali	0	0	0	1	0	0
Scorpiones						
(blank)	0	0	0	0	1	0
Scytodidae						
Scytodes caffra	0	1	1	0	1	0
Scytodes lawrencei	0	1	0	0	0	1
Selenopidae						
Selenops sp.1	0	0	0	0	1	0
Solpugidae sp. 1	0	0	0	0	0	5
sp. 1	0	1	1	15	12	14
sp. 2	0	0	0	6	0	0
sp. 2 zebra	0	0	0	0	1	0
Solifugae						
sp. 1	0	0	0	0	0	1
sp. 2	0	0	0	4	0	0
Solpugidae						
sp. 1	0	0	0	0	2	0

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Sparassidae						
Olios correvoni	0	0	0	2	0	0
Pseudomicrommata longipes	1	0	0	1	0	0
Theraphosidae						
Harpactirella sp.1	0	0	0	0	1	0
Theridiidae						
Euryopis episinoides	0	1	4	0	0	0
Euryopis sp. 1	0	0	2	0	0	1
Phycosoma sp. 1	0	2	0	0	3	0
Steatoda erigoniformis	0	1	1	3	2	0
Theridion purcelli	1	0	0	0	1	0
Thomisidae						
Heriaeus foordi	0	1	0	3	0	0
Pactactes sp. new	0	0	0	0	1	0
Synema decens	0	0	0	1	0	0
Xysticus sp. 1	0	0	0	0	0	1
Zodariidae						
Caesetius inflatus	0	0	0	0	1	0
Capheris sp. 1	0	0	0	0	3	0
Cydrela sp. 2	0	1	0	0	0	1

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Cydrela spinimana	0	2	0	0	0	0
Diores lesserti	1	3	5	0	5	2
Heradida sp. 1	3	3	2	7	3	14
Heradida sp. 2	3	0	1	1	1	4
Microdiores sp. 1	0	3	0	0	18	6
Ranops caprivi	5	1	1	6	4	6



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Appendix H: Subfamily and ant species collected during dry (2014) and wet (2015) season in different land use. The abundance of the species collected in each season and land use is given. Agr = Agro-ecosystem, Lm = Rangeland, Sett = Settlement.

	Dry Season			Wet Season		
Subfamily and species	**Agr**	**Lm**	**Sett**	**Agr**	**Lm**	**Sett**
Amblyoponinae						
Prionolpelta sp.01	0	0	0	0	0	1
Dolichoderinae						
Tapinoma sp.03	0	12	1	0	3	0
Technomyrmex pallipes	3	8	8	2	14	0
Dorylinae						
Aenictus rotundatus	0	0	2	0	2	5
Cerapachys sp.03	0	0	2	0	0	0
Cerapachys sp.04	0	0	0	0	0	1
Dorylus sp.04	3	0	1	0	0	1
Formicinae						
Anoplolepis cf. custodiens	87	52	73	114	92	80
Camponotus mayri	0	3	0	9	19	0
Camponotus sp.02 (rufoglaucus gp.)	1	4	0	6	16	0
Camponotus sp.04 (etiolipes gp.)	0	0	0	0	3	0
Camponotus sp.11 (cintellus gp.)	1	0	0	0	0	0
Camponotus sp.12 (cintellus gp.)	1	6	0	0	19	0
Camponotus sp.14 (cintellus gp.)	10	4	9	20	39	15
Camponotus sp.15	31	8	4	27	32	25
Camponotus sp.16 (niveosetosus gp.)	0	0	1	0	1	5
Camponotus sp.18 (maculatus gp.)	5	9	17	15	23	32
Camponotus sp.23	0	1	0	6	0	0
Camponotus sp.24	6	1	0	14	3	0
Camponotus sp.25	0	0	0	0	1	0
Lepisiota sp.01 (capensis gp.)	0	3	0	0	4	1
Lepisiota sp.12	10	43	4	13	71	2

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Lepisiota sp.13	0	0	2	0	1	0
Lepisiota sp.14	2	0	0	22	0	0
Lepisiota sp.16	0	12	6	9	5	1
Lepisiota sp.17 (spinosior gp.)	0	1	0	0	2	0
Lepisiota sp.18	0	15	0	0	24	1
Lepisiota sp.19	0	0	4	0	0	15
Lepisiota sp.20	0	0	0	0	6	0
Plagiolepis sp.04	0	1	4	0	0	1
Plagiolepis sp.05	0	5	0	0	0	0
Plagiolepsis sp.06	0	2	9	0	2	2
Polyrhachis shistacea	0	7	0	1	24	0
Tapinolepsis sp.01	35	13	9	37	19	23
Tapinolepsis sp.02	4	9	15	0	53	14
Tapinolepsis sp.03	1	0	2	39	1	7
Myrmicinae						
Calyptomyrmex sp.01	1	1	3	1	3	9
Cardiocondyla sp.05	6	4	45	9	23	41
Cardiocondyla sp.08	10	0	0	20	0	0
Cardiocondyla sp.09	0	0	3	0	0	2
Cardiocondyla sp.10	0	0	0	11	0	0
Carebara sp.02	11	1	1	1	2	0
Carebara sp.03	0	0	0	0	0	1
Cataulacus sp.03	0	0	2	0	0	7
Cataulacus wissmanni	0	0	0	0	1	0
Crematogaster sp.06 (castanea complex)	0	6	6	0	15	2
Crematogaster sp.07	0	31	0	0	34	0
Crematogaster sp.11	0	1	0	0	2	1
Meranoplus cf. glaber	3	0	0	22	8	8
Meranoplus inermis	0	0	1	0	4	3
Meranoplus magretti	1	0	0	14	0	0
Meranoplus sp.05	2	0	1	0	1	7

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Meranoplus sp.06	8	3	0	40	8	0
Meranoplus sp.07	0	0	1	0	1	0
Meranoplus sp.08	0	2	0	0	12	0
Meranoplus sp.09	0	1	0	0	4	0
Messor sp.02	13	0	6	8	0	4
Monomorium emeryi	22	1	51	28	6	28
Monomorium notulum	13	0	4	8	0	22
Monomorium cf. drapenum	1	1	1	32	8	9
Monomorium cf. junodi	64	46	78	86	72	97
Monomorium damarense	44	25	36	64	48	39
Monomorium sp.01	0	1	0	0	1	0
Monomorium sp.08	1	12	9	9	3	14
Monomorium sp.10 (salamonis gp.)	6	4	7	0	2	9
Monomorium sp.13	1	0	0	0	0	0
Monomorium sp.14	11	9	15	16	10	2
Monomorium sp.15	4	7	0	2	19	0
Monomorium sp.16	10	2	0	0	11	0
Monomorium sp.17	0	1	0	0	1	0
Monomorium sp.19	0	0	1	0	3	0
Monomorium sp.20	0	8	0	0	3	0
Monomorium sp.21	0	0	0	0	1	0
Myrmicaria natalensis	1	5	0	7	10	0
Ocymyrmex fortior	11	1	11	114	55	57
Ocymyrmex sp.04	1	0	0	5	0	0
Pheidole sp.01 (megacephala gp.)	84	72	55	100	52	40
Pheidole sp.03 (megacephala gp.)	4	8	26	8	16	71
Pheidole sp.06 (crassinoda gp.)	0	1	4	0	4	4
Pheidole sp.09 (crassinoda gp.)	0	20	3	0	11	2
Pheidole sp.13	7	37	29	5	36	0
Pheidole sp.14	1	13	2	11	10	23
Pheidole sp.15	5	58	6	3	29	8

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Rhaptomyrmex sp.03	0	1	0	0	1	0
Solenopsis sp.02	0	6	6	2	15	3
Solenopsis sp.04	7	7	7	0	5	0
Solenopsis sp.05	1	6	7	0	0	6
Solenopsis sp.06	0	1	0	0	0	0
Strumigenys sp.03	0	1	0	0	1	0
Strumigenys sp.04	0	0	1	0	0	2
Tetramorium baufra	6	9	15	3	14	4
Tetramorium cf. setigerum	2	7	6	4	11	5
Tetramorium notiale	0	0	2	0	1	1
Tetramorium sepositum	62	12	39	62	39	53
Tetramorium setuliferum	10	0	2	31	6	5
Tetramorium sp.01 (squaminode gp.)	24	24	18	30	26	10
Tetramorium sp.04 (gabonense gp.)	0	3	0	4	4	0
Tetramorium sp.07 (Weitzeckeri gp.)	0	2	4	0	0	0
Tetramorium sp.08 (simillimun gp.)	24	1	2	19	0	10
Tetramorium sp.13	0	1	0	0	0	0
Tetramorium sp.19 (simillimun gp.)	3	6	12	7	4	12
Tetramorium sp.24 (weitzeckeri gp.)	0	0	10	0	0	0
Tetramorium sp.25 (squaminode gp.)	0	7	11	0	14	30
Tetramorium sp.26 (simillimun gp.)	9	0	10	19	0	0
Tetramorium sp.27 (sereiceventre gp.)	4	1	15	30	0	33
Tetramorium sp.28	0	1	0	0	0	0
Tetramorium sp.31	16	1	0	13	6	3
Tetramorium sp.32	0	0	1	0	0	0
Tetramorium sp.33 (simillimun gp.)	1	0	11	0	0	1
Tetramorium sp.34	0	1	1	0	2	1
Tetramorium sp.35	0	0	0	0	0	4
Tetramorium sp.37	0	0	0	1	0	0
Ponerinae						
Anochectus cf. treagordhi	0	0	0	0	1	0

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Bothroponera sp.02	0	10	0	2	16	0
Hypoponera sp.02	0	0	0	0	1	0
Leptogenys sp.03	0	0	1	0	0	2
Megaponera sp.02	14	10	1	8	3	0
Mesoponera sp.02	0	2	4	0	4	0
Mesoponera sp.03	0	15	0	0	13	0
Mesoponera sp.04	0	0	9	0	0	7
Odontomachus troglodytes	7	14	0	17	10	1
Platythyrea lamellosa	0	0	0	1	0	0
Ponera sp.02	0	0	0	0	0	1
Pseudoponera sp.01	6	0	0	2	1	0
Pseudoponera sp.03	1	0	1	0	0	0
Pseudomyrmicinae						
Tetraponera sp.02	0	1	0	0	0	0

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