Micro scale heterogeneity of spiders (Arachnida: Araneae) in the Soutpanberg, South Africa: a comparative survey and inventory in representative habitats

by

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DECLARATION

I, **Ms Mafadza Maria**, declare that the dissertation hereby submitted to the University of Venda for the degree of Master of Science has not previously been submitted by me for the degree at this or any other university, that it is my own work in design and in execution, and that all material contained therein has been duly acknowledged.

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ABSTRACT

Coarse-scale studies that focus on species distributions and richness neglect heterogeneity that may be present at finer scales. Studies of arthropod assemblage structure at fine (1 × 1 km) scales are rare, but important, because these are the spatial levels at which real world applications are viable. The study investigates fine-scale variation in spider assemblages, comparing five representative vegetation types in the western Soutpansberg, Limpopo Province, South Africa. We assess these vegetation types in terms of their family and species composition, as well as levels of endemicity, relating these differences with vegetation structure. The study inventoried 297 species (9752 individuals), representing 158 genera, and 50 families in an area less than 450 ha were collected during the three sampling periods, namely May 2004 (late) Autumn, November 2004 (late) Spring and March 2005 (late) Summer as part of the South African National Survey of Arachnida database (SANSA).

Analysis of the results suggests spider abundance and richness was highest in November as compared to March in five habitat types with 34% of the spiders caught in spring being adults decreasing to 25% later in the season. Endemic taxa are associated with Tall Forest and, to a lesser extent, Woodland. The Woodland had the highest species diversity, and much of the variation observed in spider assemblage structure is explained by these two vegetation types. Inventory completeness was more than 70% for all the habitats except the Mosaic Grassland. Twenty three species had significant IndVals > 70 and majority of indicators were associated with the Woodland (13), followed by Tall Forest (8), which was dominated by web builders such as *Nephila pilipes*, *Leucauge decorata* and *Glenognatha* sp. The restricted distribution of taxa at the local scale, often overlooked at the broader regional scale, is confirmed by the observation that only 27.4% of the species were found in all the habitats, at a scale less than 2 km in extent. Based on vegetation structure variables that explained significant variation in spider assemblages, human influence through bush encroachment will result in a change of spider assemblages to that of Short Forest and Mosaic Woodland vegetation types, with implications for biodiversity maintenance and heterogeneity. Beating targeted the most species, followed by ground collecting and leaf litter sifting.

Key words: endemicity, South Africa, spider, SANSA, conservation, Soutpansberg

CHAPTER 1. INTRODUCTION

Biodiversity and its measurement

Biodiversity is the variety of all living things, *inter alia*, plants, animals and microorganisms, the genetic information they contain and the ecosystems they form (Keystone Center 1991). There are three levels at which biodiversity can be assessed: genetic (diversity within species (Groombridge 1992)), species (number of species), or community (diversity of communities) and the abiotic environment they interact with, forming ecosystems (Noss 1990; Pearson 1995). Biodiversity is therefore a multi-faceted concept requiring various approaches to estimation.

The 1992 Rio Earth Summit heralded the introduction of biodiversity to a much wider audience than just scientists and environmental activists, obligating countries that ratified Agenda 21 to develop strategic plans for the conservation and sustainable use of biodiversity (Magurran 2004). The Millenium Ecosystem Assessment had its roots in the Convention on Biodiversity (CB) and has further highlighted the magnitude of biodiversity loss due to human activities over the last fifty years, which is more rapid as compared to any other time in the history of man, 1000 times higher than the natural rate (Powledge 2006).

The CB defines biodiversity loss as "the long-term permanent qualitative or quantitative reduction in components and their potential to provide goods and services". The operating definition of biodiversity in this study will therefore be "the variety and abundance of species in a defined unit of area of study," taking into account Magurran's (2004) assertion that species richness is the common currency of biodiversity. The concept of biodiversity has been characterized as vague, with very little application in the real world. However, this can be corrected if biodiversity is considered an end in itself and measurable indicators of biodiversity status are monitored over time (Noss 1990); species richness is one such an indicator. This could be an efficient way to maintain overall biodiversity, particularly by protecting locations with a relatively large number of native species (Myers *et al*. 2000).

As signatories to the Convention of Biological Diversity[1] (CBD), South Africa is obliged to develop a strategic plan for the conservation and sustainable utilization of this unique biological heritage. The CBD has shown us the importance of good biodiversity data, and has contributed to renewed interest in specimen databases of natural history collections. The term "Biodiversity Informatics" is defined as the 'application of informatics to recorded and yet to be discovered information specifically about biodiversity, and the linking of this information with genomic, geospatial and other biological and non - biological datasets', and this term is now generally in use for this type of data.

Habitat heterogeneity and biodiversity

Researchers have revealed that eucalypt forest and woodlands provide three broad habitats for invertebrates: namely overstorey trees, understorey shrubs and grasses, and the ground layer of litter, woody debris and bare earth (Majer *et al*. 1997). Within these habitats the spatial scale at which invertebrate diversity occurs can be small, and individual plants or forest patches may support distinct invertebrate communities, reflecting differences in bark, foliage and litter characteristics (Barton *et al*. 2010; Doherty *et al*. 2000). At the single tree scale, the richness of invertebrate species is determined by the structural complexity of the plant, its biochemical defences against attack and its foliage nutrient levels. Hence invertebrate diversity is therefore likely to be greatest in heterogeneous habitats

[1]www.cdb.int

containing a variety of structural arrangements, ranging from open areas beneath over-storey trees to dense thickets of tall shrubs.

Although habitat heterogeneity is important in determining biological diversity (González Megías *et al*. 2007; MacArthur & MacArthur 1961), the strength of this relationship varies with spatial scale and depends on how organisms perceive their environment (Magurran 2004; Tews *et al*. 2004; Wiens 1989). For instance, highly mobile bird species will use large areas of the landscape for breeding and foraging (Manning *et al*. 2007), whereas some arthropod species may be spatially restricted as a result of specialization on particular food resources (Novonty & Basset 2005). Habitat complexity is seen as the heterogeneity in the arrangement of physical structure in the habitat surveyed (Lassau & Hochuli 2004).

A consequence of the habitat heterogeneity-diversity relationship is spatial patchiness in species richness, and this leads to variation in composition and ecological functions driven by species assemblages at different spatial scales (Barton *et al*. 2010). Beetle assemblages, however, can be influenced by habitat heterogeneity at much finer spatial scales at distances of hundreds of meters or less (Antvogel & Bonn 2001; Barton *et al*. 2009; Gibb *et al*. 2006; Hoffman & Wiens 2004; Janssen *et al*. 2009; Niemelä *et al*. 1996). Although the litter under trees is known to be a significant habitat for a rich diversity of beetles (e.g. Barton *et al*. 2009; Stork & Grimbacher 2006), there are very few studies that examine how subtle differences in leaf litter heterogeneity influence fine-scale beetle diversity patterns (see Koivula *et al*. 1999; Niemelä *et al*. 1992).

González Megías *et al*. (2007) reported that the diversity-habitat heterogeneity relationship is scale – variant. Furthermore, many diversity habitat descriptors are good predictors of species abundance and diversity. However, the relevance of each predictor changed with the scale. Spider abundance and diversity may be correlated with the specific vegetation characteristics, suggesting that the availability of habitats is important for spider colonisation and establishment (Jõgar *et al*. 2004).

The delineation of spatial heterogeneity in communities or assemblages at the local scale is an important step if a country's conservation goal is to retain as much of the region's biodiversity as possible. That is, information on the extent to which local biotas vary between and within habitat types is necessary for a better understanding of the underlying processes explaining local community patterns (Cornell 1993; Ricklefs 1987). Priority conservation areas identified at broad global or regional scales are often heterogeneous and cannot be translated into effective local conservation strategies without reference to local landscape and species distribution patterns. This is because local heterogeneity can be present across scales and may significantly complicate the development of effective regional conservation strategies (Flather *et al.* 1997; Rodrigues *et al.* 1999). The identification of fine-scale spatial heterogeneity in local biotas is therefore important, because: (1) such information will enhance area selection techniques (which are currently mostly conducted at coarse regional scales) to select units defined at the scale of land management units (Flather *et al.* 1997; Rodrigues *et al.* 1999; Wiens 1989), and (2) long-term conservation of the biota that constitutes these heterogeneous communities will be achieved (Rodrigues *et al*. 1999).

The study by Clough *et al*. (2005) indicates that landscape heterogeneity and the location within fields are found to be the major drivers of spider abundance and diversity. The study stated that species richness was correlated with landscape heterogeneity, and further indicate that landscapes less dominated by arable fields have more perennial habitats such as grassland, leys, woodland and roadside verges (Clough *et al*. 2005), hence are structurally and vegetationally more diverse and with more food resources. This makes such landscapes viable for a greater number of spider species (Clough *et al*. 2005).

Diversity and ecosystem services

An important aspect of biodiversity, that falls outside the scope of this study, is the qualitative components of biodiversity, i.e. ecological services that form the basis and rationale for conservation, because they embody the direct benefit humans derive from ecosystems and includes the generation of soils, maintenance of air, water quality, pest control, detoxification and decomposition of wastes, pollination and crop production, climate stabilization, prevention and mitigation of natural disasters and provision of food security (Gibbs 2004).

Studies in forest and woodland environments have shown that diversity of resources is often associated with a range of different structural components, including the layering of the canopy, decaying logs, hollow bearing trees, the presence of particular understorey species, and the availability of trees with different types of bark (Doherty *et al.* 2000). However, a forest or woodland with a variety of structural components is considered likely to have a variety of resources and species which utilize these resources (Brokaw & Lent 1999; Tanabe *et al.* 2001). Consequently, there is often a positive correlation between biodiversity and measures of the variety and/or complexity of arrangement of structural components within an ecosystem (MacNally *et al*. 2001).

Invertebrate conservation and diversity

Studies show that an overwhelming proportion of terrestrial biodiversity is contributed by invertebrates (Gaston 1991; Stork 2007), and the conservation of invertebrates is necessary if ecosystems are to be effectively managed for both biodiversity and ecological function (Leather *et al*. 2008; Samways 2007). For example, some rare endemic Florida dune spiders may persist on smaller habitat fragments than are required by vertebrates and could require special management regimes for persistence (Marshall & Rypstra1999). Site biodiversity estimates that do not consider invertebrates not only omit the greatest components of what they are attempting to measure, but also ignore groups that are very significant contributors to terrestrial ecosystem processes (Foord *et al.* 2011a).

Determining invertebrate diversity is particularly challenging because: (1) there is a high proportion of undescribed species; (2) a large percentage of specimens are juveniles (ca. 50%); (3) often no revisions or keys are available, making species-level identification time- consuming and, in taxa whose taxonomy is poorly known, impossible (Cardoso *et al.* 2011).

Invertebrate conservation and diversity pose a significant challenge to planners and managers (Engelbrecht 2010), and in spite of the central role that insects and arachnids play in terrestrial biodiversity, they still remain peripheral to decision-making processes. The reality is that, for Africa in particular, there are very few conservation organizations that have both the resources and expertise to include invertebrates as part of their monitoring and management initiatives (however, see South African River Health Programme[2]).

Estimating species richness

Monitoring and conservation evaluation programs rely heavily on estimates of diversity (Margules 1986), which are often most conveniently assessed as species richness derived from species inventories. Even species richness, the simplest version of biodiversity, is fraught with problems of measurement (Colwell & Coddington 1994) and often under-represents hyper - diverse taxa (Kremen *et al.*

[2] www.dwa.gov.za/iwqs/r

1993). Species richness is a useful measure for conservation and land use planning, environmental monitoring, and natural resource management (Coddington *et al.* 1996). Although species richness is only one criterion used by land managers and conservationists, and only one component of diversity, it is the most easily measured and crucial because of the global loss of diversity (Coddington *et al.* 1996).

Estimating species richness can be expressed as the actual number of species present in a given area. Many field studies carried out in community ecology use it as a basic step and a matter of crucial concern when dealing with the conservation and management of biodiversity (Colwell & Coddington 1994). Species richness can furthermore be estimated in three ways: (1) extrapolation of species accumulation or species-area curves, (2) by using the shape of the species abundance distribution to deduce total species richness, and (3) by using a nonparametric estimator (Chazdon *et al.* 1998; Colwell & Coddington 1994).

Determinants of spider diversity

Herbivory and spider diversity

It is likely that direct effects of herbivores on vegetation will result in an indirect influence on spider diversity (e.g. Haddad *et al*. 2010; Warui *et al*. 2005). Findings by Jonsson *et al*. (2010) indicate that large mammalian herbivores (notably elephants) had significant negative effects on total invertebrate abundance, while medium-sized mammalian herbivores affected the abundance and richness of beetles and grasshoppers negatively.

The level of herbivore impact seems to have an influence on vegetation cover and woody plant species richness that, as a result, together drive spider community composition, richness and abundance (Chari 2011). The study showed that the higher the herbivore impact is, the greater the difference in spider communities on and off termite mounds, as termitaria are impacted differently from the adjacent woodland matrices (Chari 2011).

Studies have also indicated that the presence or absence of spider species may be related to the subtle changes in the vegetation structure as a result of grazing (Jansen *et al*. 2013). Churchill & Ludwig (2004) mention that livestock grazing changes habitat structure and inhibits ecological succession in the Australian savannah. Dennis (2003) indicated that arthropod diversity can be altered by livestock grazing through changes in plant community composition and soil physical properties in Scotland. In Africa, intensive grazing by domestic livestock has also been shown to alter savannah vegetation cover (Warui *et al*. 2005). Grazing by domestic livestock has also been shown to significantly reduce the diversity of spider fauna (Warui *et al*. 2005).

Dennis *et al*. (2001) suggested that there was higher species richness and abundance of spiders in pastures grazed only by sheep than by both sheep and cattle. On the other hand, Abensperg-Traun *et al*. (1996) also recorded spider diversity as being strongly influenced by sheep grazing in Australia, stating that intense grazing had an adverse effect on the primitive spider sub-order Mygalomorphae, although moderate grazing favoured wolf spiders (Family: Lycosidae) and those species belonging to the Idiopidae, known as brown trapdoor spiders. Churchill (1998) further recorded the Zodariidae as being strongly affected by grazing in the Mitchell Grasslands of the Northern Territory of Australia.

Bush encroachment

Bush encroachment is a process whereby the density of woody plants e.g. trees and shrubs, increases in an area (Tainton 1999). Brown & Archer (1999) indicated that the demand above-ground or below-ground competitive capacity of grasses subjected to grazing is as a result of bush encroachment. According to Ward (2005), bush encroachment is the suppression of palatable grasses and herbs by encroaching woody species that are often unpalatable to domestic livestock. Studies show that there may be a decrease in endemic browsers owing to the replacement by cattle, and this promotes bush encroachment (Richter & Meyer 2001). It is evident that the elimination of keystone herbivores and introduction of goats, and suppression of fires as a result of afforestation (Hahn 2006), may lead to bush encroachment.

Alien invasive species

Robertson *et al*. (2003) ranked *Chromolaena odorata* as the second worst alien weed species in the savannah biome of South Africa after *Lantana camara*. L. *Chromolaena odorata* reduces vegetation heterogeneity in grasslands, savannah and forests (Goodall & Zachariades 2002). Some invasive species may also be considered to be ecological engineers, as they modify the ground surface micro-environment in encroached areas (Pétillon *et al*. 2005).

Mgobozi *et al*. (2008) found significant impacts of *C. odorata*, which has resulted in: (1) a reduction of numerically dominant spiders, (2) changes in spider assemblage patterns, species richness and diversity, (3) abundance in patches of *C. odorata* and (4) the decrease of habitat heterogeneity in savanna in KwaZulu-Natal (South Africa). However, Robertson *et al*. (2011) indicated that *Opuntia strica* (Cactaceae) did not have a major impact on spider assemblages, species richness or species density.

Habitat and vegetation structure

Downie *et al.* (1995) indicated that the variation found in spider assemblages was attributable to an increase in vegetation density throughout all levels of the profile. A change in vegetation structure resulting from alien plant invasion has been cited as one of the principal causes of changes in arthropod assemblages (Coetzee *et al.* 2007; Standish 2004). Furthermore, research findings have shown that spider distribution patterns were effectively influenced by habitat structure, which can be altered by land use practices (reviews by Turnbull 1973; Uetz 1991; Wise 1993).

Conservation of South African spiders

Recent developments in South Africa have added weight to the advocacy of the inclusion of spiders in conservation assessments and red listing of species (Dippenaar-Schoeman *et al.* 2010). However, the inclusion of spiders in conservation planning will depend on the amount of compiled information on their distribution and conservation status available to conservation professionals.

For conservation purposes and applied problems that focus on large areas, species density is probably of more interest because it measures the number of species within a specified area (Gotelli & Colwell 2001). Disturbance or management regimes that affect abundance might have to be considered in choosing among such areas (Gotelli & Colwell 2001). The only spiders currently considered to be of conservation importance are the baboon spiders, *Harpactira* spp. and *Harpactirella* spp. and both taxa are relatively common under rocks. Also worth noting was the unusual *Stasimopes* sp. (trapdoor spider), of which only males were collected (Haddad & Dippenaar-Schoeman 2009).

Conserving and enhancing landscape diversity appears to be one of the keys to conserving spider biodiversity in agro-ecosystems (Clough *et al.* 2005). The establishment of networks of protected areas for conservation is an obligation placed on parties to the CBD[3], the Ramsar Conservation on Wetlands, the Bern Convention on the conservation of European wildlife and natural habitats (Rodrigues *et al*. 1999). Conservation strategies should also take into consideration the role that invertebrates play in ecosystem functioning, and not only focus on plants and vertebrates (Haddad & Dippenaar-Schoeman 2009).

Protected species are organisms that are of such high conservation value or national importance that they require national protection. Species listed in this category will include, among others, species listed in terms of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) (Dippenaar-Schoeman *et al*. 2015. Presently, all of the species of the baboon spider genera Ceratogyrus, Harpactira and Pterinochilus (Theraphosidae) are listed on the SANBI website (http://www. speciesstatus.sanbi.org) as protected species, as identified by the National Environmental Management: Biodiversity Act 10 of 2004 (NEMBA). All the spiders of South Africa are presently being assessed for their conservation status using the World Conservation Union (IUCN) criteria for red-listing of species. This is done with the support of the SANBI Threatened Species Programme (Dippenaar-Schoeman *et al*. 2015.

The importance of spiders in ecosystems

Although the Araneae constitutes an abundant and highly successful group of invertebrates, little is known of their diversity within most ecosystems in South Africa (Foord *et al*. 2011a). As many invertebrate taxa are undescribed, our

[3] www.cdb.int

knowledge of the geographical ranges of these animals is virtually non-existent (Foord *et al*. 2011a).

Spiders (Arachnida: Araneae) can be considered a critical component of terrestrial biodiversity, with > than 45 000 species described globally (World Spider Catalog 2015). Coddington *et al*. (1996) and New (1999) proposed that spiders are a group that show potential as biodiversity indicators, as they have characteristics required for efficient indicators (i.e. they are diverse, abundant, easily sampled, functionally important and reflect changes in the environment). Spiders fulfil a critical function as faunal components of terrestrial ecosystems, as all species are predators and play an important role in the natural control of populations of terrestrial arthropods, particularly insects (Cardoso *et al*. 2011). There has been a recent increase in the application of spiders as bio-indicators to measure ecological disturbance and pollution, which has raised the profile of the group and its importance as a surrogate taxon (Cardoso *et al*. 2011; Marc *et al*. 1999).

As an ubiquitous component of invertebrate assemblages and generalist predators, their importance as natural control agents of pest insects and mites in agro-ecosystems has also received attention during the last few decades (Dippenaar-Schoeman *et al*. 2013; Marc *et al*. 1999; Riechert 1999; Symondson *et al*. 2002). Ground dwelling spiders may be important in transferring energy directly from the below-ground detritus food web to the above-ground terrestrial food webs of familiar birds, reptiles, amphibians, and mammals (Johnston 2000).

Studies of spider diversity in the Savanna Biome of Africa

Savanna is one of the world's major biomes and covers approximately half of Africa,'s land surface (Scholes & Walker 1993), occupying extensive areas between the equatorial forests and deserts. It is characterised by a well-developed grassy layer with a prominent woody layer of trees and/or shrubs that may be evergreen and/or deciduous. Although the main vegetation types are trees and grasses, the ecology of the Savanna is a complex interaction between the woody and herbaceous plants, giving it a unique character (Scholes & Walker 1993). The major delimiting factors are the effects of fire and rainfall, with the latter varying from 235 to 1 000 mm per year with frost 0-120 days/year (Cowling *et al*. 2004). Several surveys on spiders in the African Savanna have been undertaken during the last thirty years, of which the majority of published studies were carried out in South Africa (Dippenaar-Schoeman & Jocqué 1997; Foord *et al*. 2011b).

In South Africa, the Savanna Biome is the largest biome and occupies over one third of the country's surface area (Low & Rebelo 1996; Scholes & Archer 1997). It is especially well developed in the Northern Cape, North West, Limpopo, Mpumalanga and parts of KwaZulu-Natal and Eastern Cape provinces. Although the Savanna Biome has the largest number of records of any of the South African biomes, the spiders of this vegetation type are still poorly known (Foord *et al*. 2011b).

Table 1.1 Spider diversity recorded from protected areas in South Africa.

Conservancy	Fam.	Spp.	Reference
Blouberg Nature Reserve & Soutpansberg Conservancy	44	291	Muelelwa *et al*. (2010)
De Hoop Nature Reserve	52	252	Haddad & Dippenaar-Schoeman (2009)
Erfenis Dam	18	108	Fourie *et al*. (2013)
Erfenis Dam	33	120	Haddad *et al*. (2015)
Hluhluwe - iMfolozi Park	30	106	Mgobozi *et al*. (2008)
Karoo National Park	38	116	Dippenaar-Schoeman *et al*. (1999)
Kruger National Park	40	152	Dippenaar-Schoeman & Leroy (2003)
Lajuma Farm	49	367	Present study & Foord *et al*. (2002)
Makalali Game Reserve	38	268	Whitmore *et al*. (2001)
Mountain Zebra National Park	34	76	Dippenaar-Schoeman (2006)
Ndumo Game reserve	46	431	Haddad *et al*. (2006)
Polokwane Nature Reserve	39	275	Dippenaar-Schoeman *et al*. (2008)
Ophathe Game Reserve	47	268	Haddad & Dippenaar-Schoeman (2015)
Roodeplaat Dam Nature Reserve	27	110	Dippenaar-Schoeman *et al*. (1989)
Swartberg Nature Reserve	45	186	Dippenaar-Schoeman *et al*. (2005)
Skukuza region of Kruger National Park		128	Robertson *et al*. (2011)
Sovenga Hill		76	Modiba *et al*. (2005)
Tembe Elephant Park	36	251	Haddad *et al*. (2010)

Presently 71 spider families, 471 genera and 2170 species are known from South Africa, representing approximately 4.8% of the world fauna (Dippenaar- Schoeman *et al*. 2015). Of the 2028 spider species, 1286 (59.3%) are endemic to the country (Dippenaar-Schoeman *et al*. 2015).

The South African National Survey of Arachnida

The South African National Collection of Arachnida (excluding the Acari) (NCA) was established in 1976 under the Biosystematics Programme at the Agricultural Research Council (ARC)-Plant Protection Research Institute, and the digitization of the arachnid specimens in the NCA began in 1991. The South African National Survey of Arachnida (SANSA) was launched in 1997. SANSA is an umbrella project dedicated to unify biodiversity research on spiders in South Africa, and is coordinated by a team at the ARC in collaboration with the South African National Biosystematics Institute (SANBI). SANSA runs on a national basis in collaboration with other institutions with an interest in arachnid fauna. The aims are to describe and document the spider fauna for conservation assessments and sustainable use. It addresses aspects including the following: surveys; online biodiversity informatica; awareness through road-shows, talks and lecture series; media releases; product development and an online virtual museum; capacity building through the training of post-graduate students, as well as in-house training; and an electronic newsletter (Dippenaar-Schoeman *et al*. 2010).

The main activities of SANSA consist of consolidating all data into a relational database; to do gap analyses regularly and determine where to undertake surveys in areas not yet sampled; to identify and database all the sampled material; to undertake taxonomic research to describe and revise species; to make people aware of the arachnids; and to develop infrastructure and build capacity through support to young researchers busy with post-graduate qualifications based on Arachnida research. All these activities contribute to an increase in our knowledge about the group and provide the basis for the compilation of data for conservation assessments and preparation of Red Data lists (Dippenaar-Schoeman *et al*. 2010).

SANBI came on board for the project's second phase (2006-2010) in partnership with the ARC. During the 16-year project an attempt was made to consolidate all the available data on South African spiders into one database, to determine the distribution ranges of species in South Africa. The information gathered is organized in a relational database (>80 000 entries) collating data from 22 institutions. As part of SANSA, a number of projects are underway to determine the diversity of the arachnid fauna of South Africa, which includes an inventory of the spider fauna of the different floral biomes. These projects recently culminated in the production of the *First Atlas of the Spiders of South Africa* (FASSA) (Dippenaar-Schoeman *et al*. 2010).

The third phase of SANSA started in 2011 and several review articles were published and bioinformatics actions are planned, such as Red Listing of species, a handbook series for all the biomes, publication of the atlas, and description of new species. The last decade has seen an exponential growth in the knowledge of the group in South Africa, but there are certainly many more species that still have to be discovered and described (Dippenaar-Schoeman *et al*. 2015).

Due to the large number of specimens sampled that still need to be processed, the Steering Committee recommended that an annotated checklist must be prepared of the spiders of South Africa (about 2000 species), which will indicate those species that need to be assessed for Red Listing at a later stage, and which will provide a preliminary broad statement about the conservation value of each species (Dippenaar-Schoeman *et al*. 2010).

The Soutpansberg Mountains

The Soutpansberg Mountains form an inselberg in northern-eastern South Africa. It contains an intricate mosaic of habitats and microclimates as a result of its complex topography, geographical position and macro-climate (Berger *et al.* 2003). The Soutpansberg is a major centre of plant endemism and biodiversity, and has the highest plant generic and family level diversity among the 18 recognized Centres of Plant Endemism (CPEs) for southern Africa (Van Wyk & Smith 2001). Recent botanical studies conducted in the Soutpansberg estimate a total of 3 000 plant species, including 1 066 genera (the world-renowned Cape Floral Kingdom has 1 000 genera) (Hahn 2002). The region also contains: (1) 33% of South Africa's reptiles, 116 species in total, equalling that of the entire Kruger National Park (Berger *et al.* 2003); (2) 60% of South Africa's mammal species, which represents more mammal species per unit area than seven of the eight most diverse biodiversity hotspots of the world (Berger *et al.* 2003); and (3) 75% of South Africa's avifauna (Berger *et al.* 2003).

The very high level of biological diversity of the Soutpansberg gives it a high long-term priority for conservation (Van Wyk & Smith 2001). In this current study, bush encroachment at the farm Lajuma might be mainly as a result of reduced numbers of mega-herbivores. Spider assemblages are likely to change should bush encroachment continue in the current study. Further bush encroachment would most certainly result in the loss of heterogeneity at this small scale and homogenization of spider assemblages, as well as the potential loss of endemic taxa.

Recent arachnid surveys in the sourveld on the northern slopes have also resulted in the description of two endemic taxa, the spider *Tyrotama soutpansbergensis* (Foord & Dippenaar-Schoeman 2005) and a scorpion, *Hadogenes soutpansbergensis*

(Prendini 2006). The endemic corinnid genus *Vendaphaea* (Haddad 2009) and the endemic zodariid species, *Australutica africana* (Jocqué 2008) were predominantly found in Mosaic Grasslands and Woodlands, respectively (Appendix 1).

Vhembe Biosphere Reserve

Biosphere reserves are areas of terrestrial and coastal ecosystems promoting solutions to reconcile the conservation of biodiversity with its sustainable use. Biosphere reserves serve in some ways as 'living laboratories' for testing out and demonstrating integrated management of land, water and biodiversity. Collectively, biosphere reserves form a global network, the World Network of Biosphere Reserves (WNBR). Within this network, exchanges of information, experience and personnel are facilitated. There are over 500 biosphere reserves in over 100 countries[4].

The origin of Biosphere Reserves goes back to the "Biosphere Conference" organized by UNESCO in 1968. This was the first intergovernmental conference examining how to reconcile the conservation and use of natural resources, thereby foreshadowing the present-day notion of sustainable development. This Conference resulted in the launching of the UNESCO "Man and the Biosphere" (MAB) Programme in 1970. One of the original MAB projects consisted of establishing a coordinated World Network of sites representing the main ecosystems of the planet, in which genetic resources would be protected, and where research on ecosystems, as well as monitoring and training work, could be carried out. These sites were named "Biosphere Reserves", in reference to the MAB programme itself. The functions of Biosphere Reserves are namely: (1) a conservation function – to contribute to the conservation of landscapes, ecosystems, species and genetic

[4] www.unesco.org

variation; (2) a development function – to foster economic and human development which is socio-culturally and ecologically sustainable; and (3) a logistic function - to provide support for research, monitoring, education and information exchange related to local, national and global issues of conservation and development[5].

The biosphere reserve concept can be used as a framework to guide and reinforce projects to enhance people's livelihoods and ensure environmental sustainability. UNESCO's recognition can serve to highlight and reward such individual efforts. The designation of a site as a biosphere reserve can raise awareness among local people, citizens and government authorities on environmental and developmental issues. In addition, it can help to attract additional funding from different sources. At the national level, biosphere reserves can serve as pilot sites or 'learning places' to explore and demonstrate approaches to conservation and sustainable development, providing lessons which can be applied elsewhere. Furthermore, they are a concrete means for countries to implement Agenda 21, the Convention on Biological Diversity (for example the Ecosystem Approach), many Millennium Development Goals (for example on environmental sustainability), and the UN Decade of Education for Sustainable Development. In the case of large natural areas which straddle national boundaries, transboundary biosphere reserves and transfrontier parks can be established jointly by the countries concerned, testifying to long-term cooperative efforts.

The Vhembe Biosphere Reserve (VBR) was the sixth South African Biosphere Reserve to be established and the third in Limpopo Province. The VBR aims to conserve the area's uniquely biodiverse environment, while simultaneously supporting and promoting much needed sustainable development[6].

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[6] www.vhembebiosphe.org

Aims and Objectives

The Soutpansberg Mountain forms the focal point of the Vhembe Biosphere Reserve (VBR). Determining conservation priorities for spiders, namely core areas and buffer zones as set out in biosphere reserve planning, is not a viable option at present because of a paucity of data. Here we attempt to use information on spiders to estimate the relative conservation importance of the dominant vegetation types in the western Soutpansberg. This area is climatically and topographically the most varied part of the Soutpansberg, and recent *ad hoc* collections recorded 47 spider families (74% of the known families from South Africa) in an area less than 450 ha (Foord et al. 2002).

The aims of this study are therefore (1) to determine the spider diversity found in different habitat types on the southern slopes of the western Soutpansberg; (2) to determine small scale heterogeneity in spider assemblages by measuring variation of spiders within and between habitat types; and (3) to determine the influence of habitat structure on spider diversity and assemblage structure. This quantitative small-scale survey therefore seeks to determine the degree of habitat- associated heterogeneity and endemism of spider assemblages in major plant communities, and to determine the species richness values of spider assemblages for the different major plant communities.

A quantitative small-scale survey was therefore initiated, which afforded the following opportunities: (1) contribute to SANSA's database by inventorying spiders and measuring species richness, abundance and endemicity in the major vegetation types of the Soutpansberg; (2) evaluating inventory completeness of this study; (3) determine whether spider assemblage structure differed between the major vegetation types at a fine scale, and how restricted taxa are to these habitats; (4) establishing which, and to what extent, vegetation structure variables are related to these differences; and (5) investigate the complementarity of various sampling techniques.

CHAPTER 2. MATERIALS AND METHODS

Study area

Field work in this study was conducted in the western parts of the Soutpansberg mountain range, situated near the northern border of South Africa with Zimbabwe (Fig. 2.1). The mountain forms a geological unit with the Makgabeng plateau and Blouberg Mountain to the west as well as the sandstone ridges that extend into the Limpopo Valley and Zimbabwe. It is the main geographical feature of the Soutpansberg Centre of Endemism (SCE) and contains several floristic elements as a result of its geographical location and varied climatic conditions (Van Wyk & Smith 2001). The Soutpansberg mountain range affects the rainfall distribution, resulting in an east-west rainfall gradient (annual rainfall exceeds 2000 mm at Entabeni in the east and decreases to 390 mm in some areas in the west (Berger *et al*. 2003).

The sampling sites were situated on the farm Lajuma, 25 kilometres east of the town Vivo and 45 kilometres west of the town Makhado (Louis Trichardt) (Fig. 2.2). It has a surface area of 430 hectares. The highest point of the mountain range, Letjume (1 747m a.s.l), is on Lajuma and ca. 800-900 meters above the surrounding plains (Gaigher 2006). Annual rainfall at Lajuma is 730 mm, varying considerably from year to year (Gaigher 2006). Commercial farming in the Western Soutpansberg has resulted in vegetation that mainly consists of thickets and scrubland vegetation, with patches of grass and fragmented groundwater forests below cliffs. Higher altitudes are characterised by areas with an interesting mosaic vegetation type that consists of short grassland and "islands" of closed woody cover, from here on referred to mosaic woodland (MW), respectively. Each "island" has at least one *Odontotermes* termite mound (Gaigher 2006).

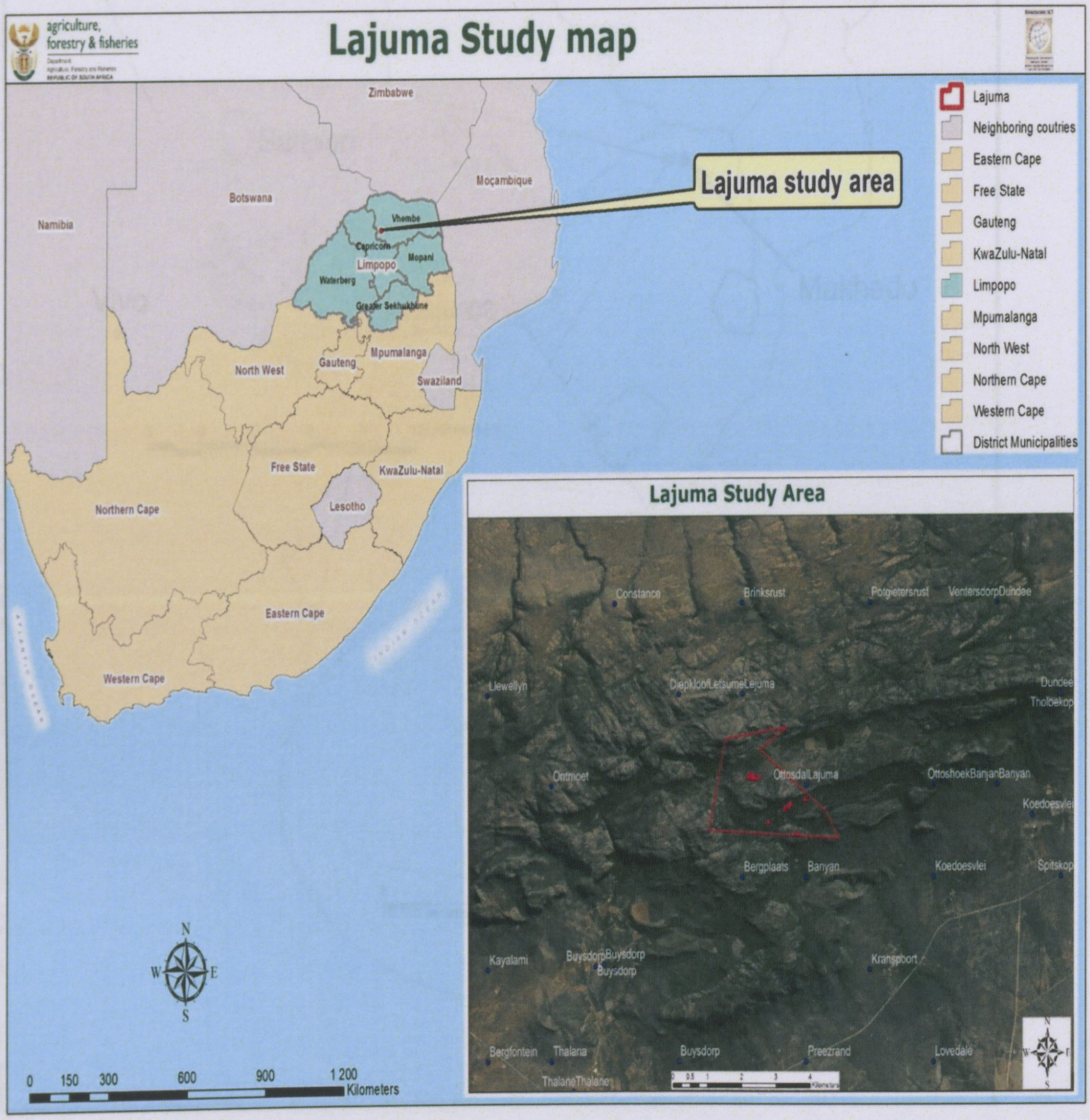

Figure 2.1 Location of study site.

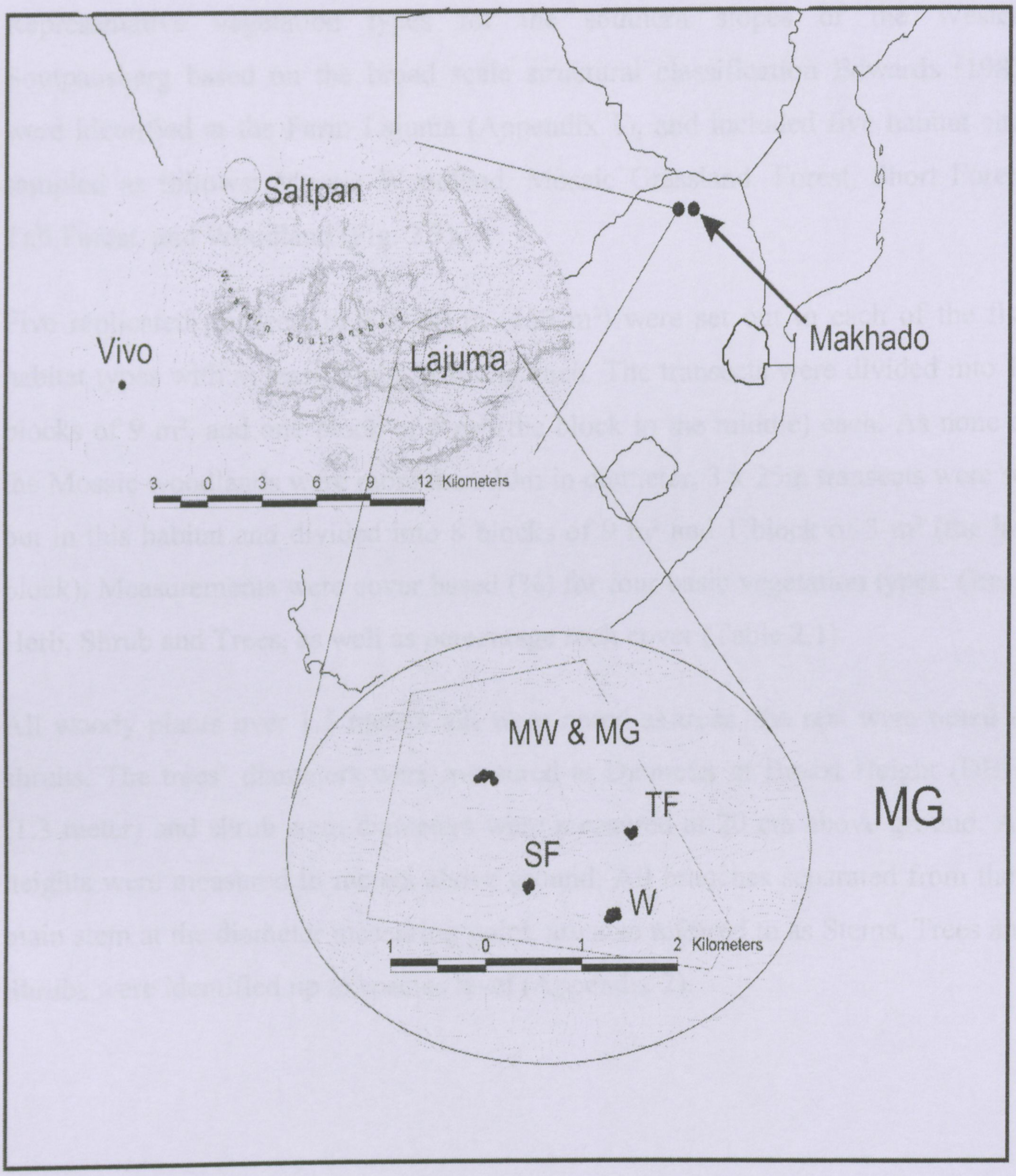

Figure 2.2 Farm Lajuma with locations of transects, i.e. Mosaic Woodland (MW) and Mosaic Grassland (MG), is situated in the Woodland Mosaic vegetative type, Short Forest (SF), Woodland (W), Tall Forest (TF).

Sampling and analytical procedures

Representative vegetation types for the southern slopes of the Western Soutpansberg based on the broad scale structural classification Edwards (1983) were identified at the Farm Lajuma (Appendix 1), and included five habitat sites sampled as follows: Mosaic Woodland, Mosaic Grassland, Forest, Short Forest, Tall Forest, and Woodland (Fig. 2.3).

Five replicated transects of 3 × 50 m (150 m^2) were set out in each of the five habitat types with at least 20m separating each. The transects were divided into 16 blocks of 9 m^2, and one block of 6 m^2 (the block in the middle) each. As none of the Mosaic woodlands were more than 30m in diameter, 3 x 25m transects were set out in this habitat and divided into 8 blocks of 9 m^2 and 1 block of 3 m^2 (the last block). Measurements were cover based (%) for four basic vegetation types: Grass, Herb, Shrub and Trees, as well as percentage rock cover (Table 2.1).

All woody plants over 1.5 meters tall were noted as trees, the rest were noted as shrubs. The trees' diameters were measured at Diameter at Breast Height (DBH) (1.3 meter) and shrub stem diameters were measured at 20 cm above ground. All heights were measured in meters above ground. All branches separated from their main stem at the diameter measuring point, are also referred to as Stems, Trees and Shrubs were identified up to species level (Appendix 2).

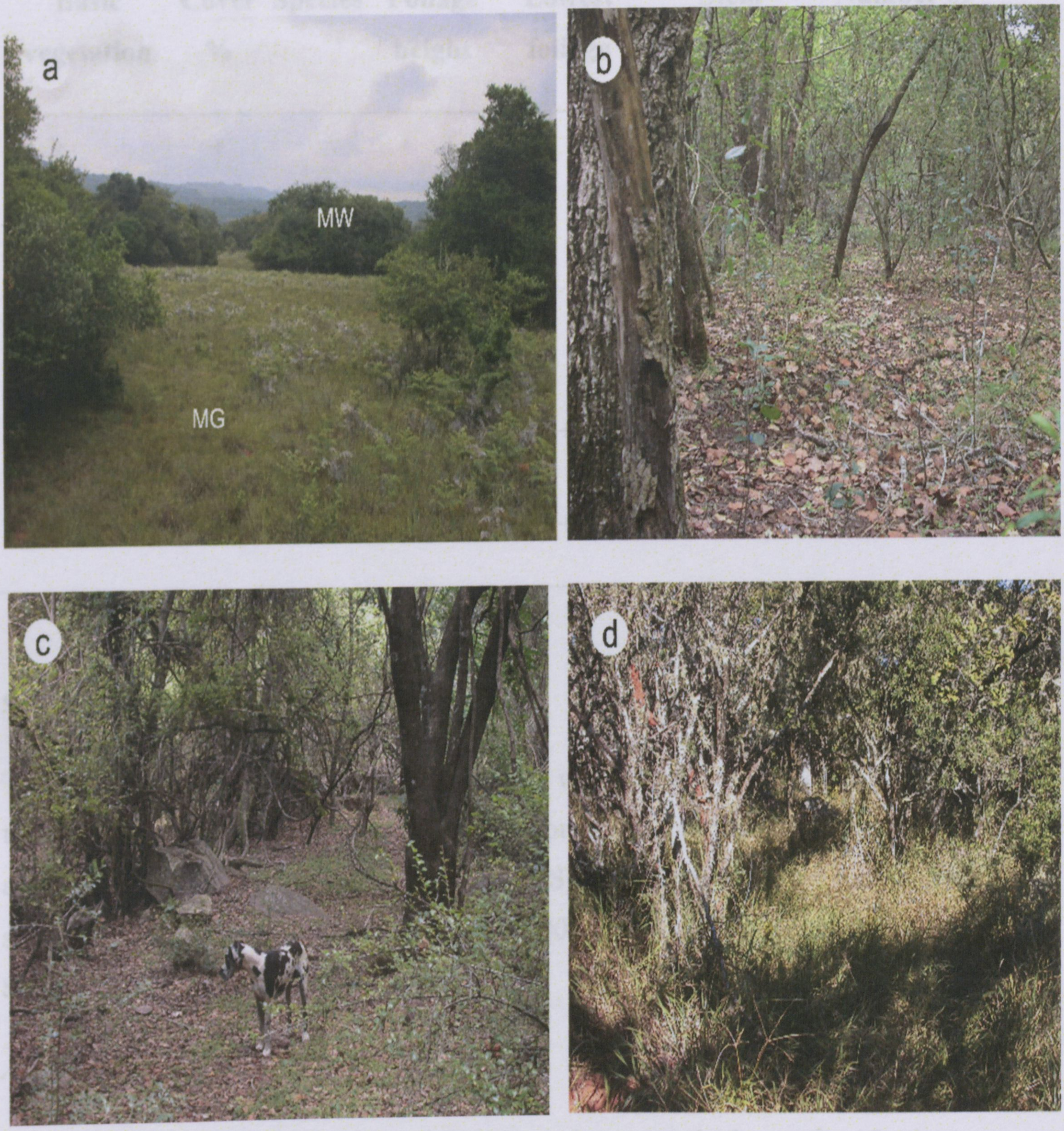

Figure. 2.3 The five habitats sampled in the Western Soutpansberg (Farm Lajuma), Limpopo Province: (a) Mosaic Woodland (MW) and Mosaic Grassland (MG), (b) Short Forest, (c) Tall Forest, (d) Woodland.

Table 2.1 Vegetation measurements taken at each transect.

Basic vegetation	Cover %	Species	Foliage height	Lowest foliage	Stem Diameter cm	Number of stems
Rock	+					
Grasses	+	+	+			
Herbs	+	+	+			
Shrubs	+	+	+	+	+	+
Trees	+	+	+	+	+	+

Spider sampling

The following five sampling techniques were used at each of the sites, subject to the availability of substrate, to ensure a thorough representation of the spider assemblages occurring in each habitat type. Spiders were collected during three sampling periods, namely Autumn (May 2004), Spring (November 2004) and Summer (March 2005).

Active Searching

Active searching, comprising three 1 × 1 m quadrats, searched once for 15 minutes in three of the five transects and during the three sampling periods. The ground, shrubs, rocks, logs, bark and stones were thoroughly searched for spiders.

Specimens were collected using either the hand to- jar technique or an aspirator sampler.

Beating

Beating was done by randomly selecting four trees, all different species, once within each transect and during the three sampling periods. Trees were beaten by firmly striking the tree branches with a stick (>1.5 kg) 20 times each. A white beating sheet was held below branches during beating (Fig. 2.4). Spiders were collected from the sheet with an aspirator.

Figure 2.4 An example of tree beating.

Sweep netting

A sweep net, 0.6 m in diameter and with a 1.2 m long, handle were swept through the grass and herb layer (Fig. 2.5). Each sweep covered an arc of approximately 1.5 m through the vegetation on every alternate step (Southwood 1978). Sweep-netting involved walking through the herb layer swinging a sweep net through the understorey vegetation for a standard number of times (Coddington *et al*. 1996).

Two samples, comprising 20 sweeps each, were done at each transect once and during the three sampling periods. Samples were emptied into a plastic container and specimens were killed adding 70% alcohol and sorted at a later stage, separating insects and spiders.

Figure 2.5 An example of sweep netting.

Pitfall traps

A transect of five unbaited pitfall traps, ten metres apart and flush with the surface of the ground, were inserted at each transect (Fig. 2.6). Traps had a diameter of 9.5 cm and contained a 50 ml solution of 3 parts 70% ethyl alcohol and 1 part glycerol (Samways *et al.* 1996). The ethyl alcohol acts as a preserving agent and the glycerol prevents the ethyl alcohol from evaporating. The drift fence technique was used to increase capture of spiders moving from one location to another. The samples were emptied on a weekly basis and pitfalls remained active for 14 days during each of the three sampling periods.

Pitfalls are, however, prone to damage by animals (Warui *et al.* 2004). The pitfall-trapping method has been widely used for surveys of ground dwelling spiders and other invertebrates (Coddington *et al.* 1991; Green 1999; New 1999; Russell-Smith 1981; Russell-Smith *et al.* 1987; Van der Merwe *et al.* 1996). The merits of this cost-effective method include continuous sampling effort (including diurnal and nocturnal in all weather conditions) and yielding of a more accurate estimate of actual species richness in a community (Uetz & Unzicker 1976).

Figure 2.6 An example of pitfall traps.

Leaf litter

Leaf litter, in three (1m × 1 m) randomly selected quadrates per transect, was sifted through a sieve with a 5mm x 5mm mesh size, onto a white sheet (Fig. 2.7) and was done during the three sampling periods. Specimens were collected with an aspirator sampler and preserved in alcohol.

Samples were kept separate for analysis. All samples, except pitfall trapping, were collected between 9h00 and 15h00 with cloud cover less than 50%. All specimens collected were transferred to 70% ethyl alcohol. Spiders were sorted up to family level. Species level identifications were done by A.S. Dippenaar-Schoeman at the Agricultural Research Centre (ARC) Plant Protection Research Institute in Pretoria. Guild classification of spiders is provided in Table 2.2 (Cardoso *et al*. 2011).

Figure 2.7 An example of leaf litter sifting

Table 2.2 Guild classification of spiders (Cardoso *et al.* 2011).

Guilds	Abbreviation	Guild explanation
Ground wandering spiders (GW)		
Free living	FGW	free-living spiders running on the soil surface when active
Burrow living	BGW	living in burrows
Rock living	RGW	living permanently or semi-permanently on or under rocks.
Sand living	SGW	living beneath the soil surface when not active
Plant wandering spiders (PW)		
Grass living	GPW	found mainly on grass
Flower living	FPW	found mainly on flowers
Foliage living	FPW	found mainly on the foliage of shrubs, herbs or thickets
Web-builders (WB)		
Orb-web	OWB	webs consist of a frame with mooring and bridge lines that anchor the web and radial signal threads arranged like the ribs of an umbrella, converging onto the centre of the web with circular spiral threads.
Funnel-web	FWB	webs made over soil surface with a funnel-shaped retreat
Gumfoot-web	GWB	three-dimensional webs consisting of a central area with or without a retreat. The upper part comprises mooring, signal and catch threads and a lower part with mooring, catch threads studded with sticky droplets attached to substrate.
Retreat-web	RWB	silk threads radiating from retreat used to catch prey, usually made with cribellate silk.
Sheet-web	SWB	webs that usually consist of an upper sheet with mooring, signal and catch threads
Space-web	SPWB	webs that fill open space and are usually attached with mooring threads to different substrates.

Data analysis

Species accumulation curves plot the cumulative number of species (s) recorded as a function of sampling effort (n) (Colwell & Coddington 1994). The number of individuals collected, or surrogate measures such as the cumulative number of samples or sampling time (Colwell & Coddington 1994), can be referred to as effort. Species accumulation curves illustrate the rate at which new species are found, but these curves do not directly reveal the total species richness unless sampling has been exhaustive.

Representivity of the spider assemblages sampled was assessed based on two sample-based rarefaction species richness estimators, Incidence-based Coverage Estimator (ICE) (Chazdon *et al*. 1998) and Michaelis Menten (MM) (Colwell & Coddington 1994), that were compared with the actual number of species observed (Sobs). The two estimators and Sobs tend to converge when samples are representative of the spider species present. The Coleman method of individual-based rarefaction was rarefied to compare species richness between sites (Coleman *et al*. 1982).

Spiders can be grouped into functional groups based on information of their habitat preferences and predatory methods (Cardoso *et al*. 2011). Habitat differences could potentially be reflected in these life history strategies (Whitmore *et al*. 2002). In this study, three major functional groups were identified: plant wanderers (PW), ground wanderers (GW) and web dwellers (WB). Based on this alternative classification of spiders, as well as their taxonomic classification, diversity (Hill's diversity number, N1; Hill 1973), richness, abundance and evenness (Hill's ratio, E5), of spider assemblages were calculated.

Differences between spider assemblage structure and associated heterogeneity were analysed by summing the number of individuals of each species trapped over the course of the study period, for each sampling site and sampling technique. Spider assemblage structure was then analysed with PRIMER v6.1.6. (Clarke & Warwick 2001). All species abundances were square root transformed prior to analysis to down-weight common species (Clarke & Warwick 1994). Complementarity of sites was assessed visually by representing these relationships based on group averaging cluster analysis of the Bray - Curtis (Bray & Curtis 1957) similarity matrices. The term "cluster analysis" encompasses a number of different classification algorithms (Faith 1991). It is a useful data reduction technique, helpful in identifying patterns and grouping of objects. Non-metric multi-dimensional scaling (MDS) was used to display the relationships between the sites in an ordination analysis with the ranked Bray - Curtis similarities as the underlying matrix.

To investigate if spider assemblages differed between and within the vegetation types, analysis of similarity (ANOSIM), using PRIMER v6.1.6. (Clarke & Warwick 2001) was used to establish the significance of differences. This is a non-parametric permutation procedure applied to rank similarity matrices underlying sample ordinations (Clarke & Warwick 1994), in which a significant global R-statistic of close to one indicates distinct differences between the assemblages/ habitats compared.

The contribution of species to habitat similarity (within habitats) or dissimilarity (between habitats) was done in SIMPER (Clarke 1993) after square root transforming the data. The consistency of a species contribution to either similarity of a site or dissimilarities between sites are a function of both its contribution to similarity/dissimilarity between sites and the consistency over sites, and therefore inversely related to its standard variation. Vegetation variables for each site were then analysed using PRIMER's BEST Procedure, with the species and vegetation matrices as the input (Clarke & Warwick 1994).

CHAPTER 3. RESULTS

In total, 297 spider species (9752 individuals), representing 158 genera, and 43 families were collected during the three sampling periods (Appendix 3). The abundance distribution between six most abundant families, namely Lycosidae (1396 records), Theridiidae (1105), Thomisidae (800), Salticidae (800), Linyphiidae (741), and Clubionidae (657), is summarized in Fig. 3.1. The most diverse family was Thomisidae, with 39 species, followed by Theridiidae with 36 species (Table 3.1). Spiders were also most abundant in the Tall Forest followed by the Woodland (Table 3.1). The most diverse genus was *Theridion,* with 15 species, followed by *Oxyopes* with 11 species.

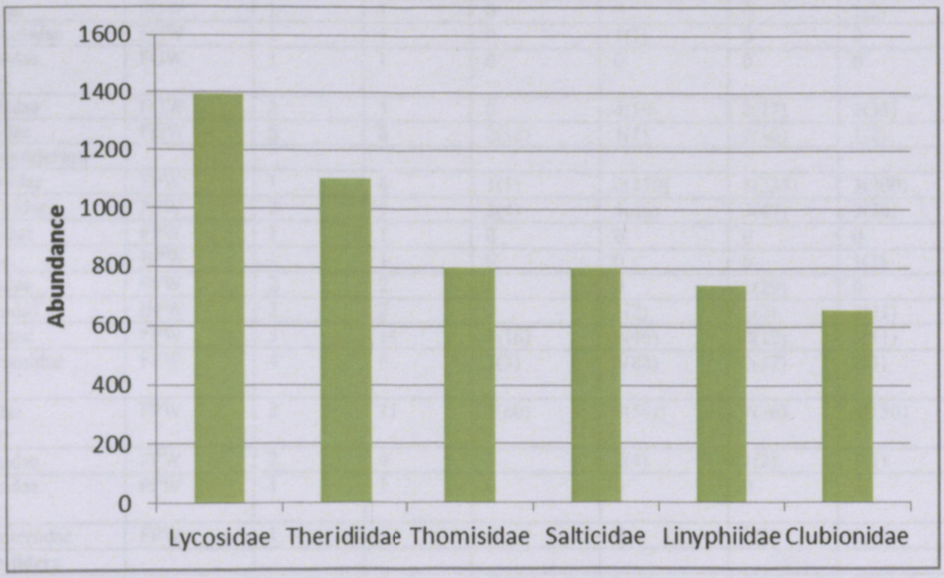

Figure 3.1 Abundance distribution of the six most abundant families caught over the period of the study.

Table 3.1 Summary table of genera and species densities calculated. The density of genera is given for the whole assemblage, whereas species densities are given for the whole assemblage and for the habitats (abundances are given in parentheses). Mosaic Grassland (MG), Mosaic Woodland (MW), Short Forest (SF), Tall Forest (TF), and Woodland (W).

	Functional Group	Genera	Species	Mosaic grassland	Mosaic woodland	Short forest	Tall forest	Wood-land
Ground wanderers								
Caponiidae	FGW	1	1	0	1(2)	1(2)	1(1)	1(2)
Corinnidae	FGW	13	18	1(8)	7(73)	6(36)	5(21)	5(84)
Ctenidae	FGW	2	3	0	2(21)	2(25)	2(22)	3(51)
Cyrtaucheniidae	BGW	2	2	1(4)	0	0	2(4)	1(1)
Gnaphosidae	FGW	10	14	6(13)	11(86)	8(25)	7(12)	12(202)
Lycosidae	FGW	5	11	5(9)	6(452)	9(450)	5(337)	8(149)
Oonopidae	FGW	1	1	0	1(5)	1(26)	1(10)	1(31)
Orsolobidae	FGW	1	1	0	1(1)	0	0	0
Palpimanidae	FGW	1	1	1(1)	1(6)	1(18)	1(8)	1(10)
Philodromidae (in part)	FGW	1	1	1(1)	0	0	0	1(2)
Prodidomidae	FGW	1	1	0	0	0	1(1)	1(2)
Salticidae (in part)	FGW	19	27	4(16)	3(10)	8(49)	4(51)	8(61)
Scytodidae	FGW	1	4	0	0	3(21)	2(114)	2(5)
Selenopidae	FGW	2	3	1(1)	0	2(14)	2(246)	2(3)
Sicariidae	FGW	1	1	0	0	0	1(2)	0
Theraphosidae	FGW	1	1	0	1(1)	0	0	0
Thomisidae (in part)	FGW	1	1	0	0	0	0	1(2)
Trachelidae	FGW	2	5	0	4(19)	2(17)	2(35)	5(18)
Zodariidae	FGW	5	8	2(11)	1(1)	2(16)	1(2)	5(22)
Plant wanderers								
Clubionidae	FPW	1	4	1(1)	4(150)	3(235)	3(200)	3(71)
Eutichuridae	FPW	2	7	2(4)	4(46)	3(47)	2(26)	5(213)
Hersiliidae	FPW	1	1	0	0	0	0	1(6)
Migidae	BPW	1	1	0	0	0	1(1)	0
Mimetidae	FPW	2	2	0	0	1(29)	0	2(3)
Nemesiidae	BPW	2	2	0	1(2)	1(2)	2(31)	1(4)
Oxyopidae	FPW	3	14	8(16)	9(46)	8(19)	3(11)	10(166)
Philodromidae (in part)	FPW	4	6	2(3)	4(82)	2(17)	2(3)	6(140)
Salticidae (in part)	FPW	2	11	7(60)	9(59))	7(76)	7(150)	11(50)
Sparassidae	FPW	2	4	0	2(4)	1(2)	1(1)	3(12)
Thomisidae (in part)	FPW	1	1	0	0	0	0	1(2)
Trochanteriidae	FPW	1	1	0	0	1(3)	0	1(1)
Web builders								
Agelenidae	FWB	1	2	2(14)	2(18)	0	0	2(8)
Amaurobiidae	RWB	1	1	0	0	0	1(7)	1(1)
Araneidae	OWB	18	28	13(65)	18(212)	17(127)	15(108)	14(101)
Deinopidae	OWB	2	2	1(1)	1(11)	1(16)	1(17)	1(8)
Dictynidae	RWB	1	1	0	1(1)	1(1)	1(6)	1(1)
Eresidae	RWB	2	1	0	1(3)	1(3)	1(3)	1(5)

Hahniidae	SHWB	1	1	0	1(2)	0	0	1(1)
Linyphiidae	SHWB	2	7	2(4)	5(66)	4(197)	4(70)	5(409)
Nephiliidae	OWB	2	2	0	1(1)	1(2)	2(82)	1(3)
Pholcidae	SPWB	2	4	1(2)	2(7)	3(15)	4(170)	3(27)
Phyxelididae	RWB	1	1	0	0	1(5)	0	0
Pisauridae	FWB	1	1	1(3)	1(1)	1(8)	1(1)	1(6)
Segestriidae	RWB	1	1	1(1)	1(1)	1(1)	1(1)	1(2)
Tetragnathidae	OWB	6	7	1(1)	0	3(18)	2(126)	1(1)
Theridiidae	GWB	13	36	8(19)	17(152)	24(381)	21(336)	23(166)
Uloboridae	OWB	3	8	1(1)	3(18)	5(245)	6(122)	4(28)
Totals				274	1742	2411	2715	2665

The study adds four new families, Oonopidae and Orsolobidae, as well as the two recently erected families of Trachelidae and Euthichuridae (Ramirez 2014) to the check list record of farm Lajuma, and brings the total to 50 families (Appendix 3; Foord *et al*. 2002). In addition, 183 species were new records for the region. Specimens collected in this study lead to the description of a new species, *Australutica africana* Jocqué, 2008, and represents the first record of this zodariid genus outside Australia (Jocqué 2008). A series of corinnid specimens also collected in this study formed the basis of the description of a new genus *Vendaphaea,* endemic to the Soutpansberg (Haddad 2009).

Sample-based rarefaction curves did not approximate asymptotes, suggesting that not all species were sampled. Inventory completeness was more than 70% for all the habitats except the mosaic grassland (Table 3.2). The rarefaction curves did not converge with the estimators (Fig. 3.2). Taking into consideration the different vegetation patches that were sampled in this study, results of individual-based rarefaction curves showed that, when the curves of the different habitats are rarified to the lowest number of individuals collected in a habitat (1780 in the Woodland Mosaic), the highest species richness (175 species) was recorded in the Woodland (Table 3.3).

Table 3.2 Richness estimates for all five habitats collectively and independently. Sampling intensity is the ratio of individuals to species. Inventory completeness is the percentage of species that are not singletons. Adjusted estimate range is the range of all estimate values divided by the observed number of species. Mosaic Grassland (MG), Mosaic Woodland (MW), Short Forest (SF), Tall Forest (TF), and Woodland (W).

		MG	MW	SF	TF	W
Samples	74	14	15	15	15	15
Individuals (computed)	9752	271	1729	2396	2704	2652
Sobs (Mao Tau)	297	89	158	165	145	198
Species density (mean ± SE)		11.57± 1.99	37.06± 2.64	44.6± 3.58	41.6± 3.52	53.8± 2.47
Abundance (mean ± SE)		19.35± 3.03	115.26±8.01	159.73±12.09	180.26±18.57	176.8± 17.39
Singletons Mean	40	49	46	36	36	41
Doubletons Mean	27	13	17	19	18	35
Uniques Mean	52	57	53	47	45	55
Duplicates Mean	27	15	23	27	24	38
ACE Mean	300.17	173.52	198.85	193.13	180.82	230.53
ICE Mean	308.45	207.94	205.56	204.47	185.09	246.8

Chao1	300.63 ± 12.24	181.35 ± 38.01	220.24 ± 25.04	199.11 ± 14.99	181 ± 15.87	222.01 ± 9.84
Chao2	321.07 ± 18.33	197.3 ± 41.39	219.07 ± 22.46	205.91 ± 15.66	187.19 ± 16.57	237.8 ± 14.03
first order jackknife	322.32 ± 8.57	141.93 ± 12.67	207.47 ± 7.95	208.87 ± 10.75	187 ± 9.27	249.33 ± 10.28
second order jackknife	347.01	178.92	234.89	227.87	206.69	266.42
Bootstrap Mean	294.62	111.18	180.4	185.6	164.24	222.94
MMRuns Mean	285.06	302.3	203.68	209.99	168.66	236.93
MMMeans (1 run)	284.46	176.49	201.18	199.31	170.06	239.14
Sampling intensity	36.02	3.04	10.94	14.52	18.65	13.39
Inventory completeness	85.24	44.94	70.89	78.18	75.17	79.29
Adjusted estimate range	0.23	2.15	0.34	0.26	0.29	0.22

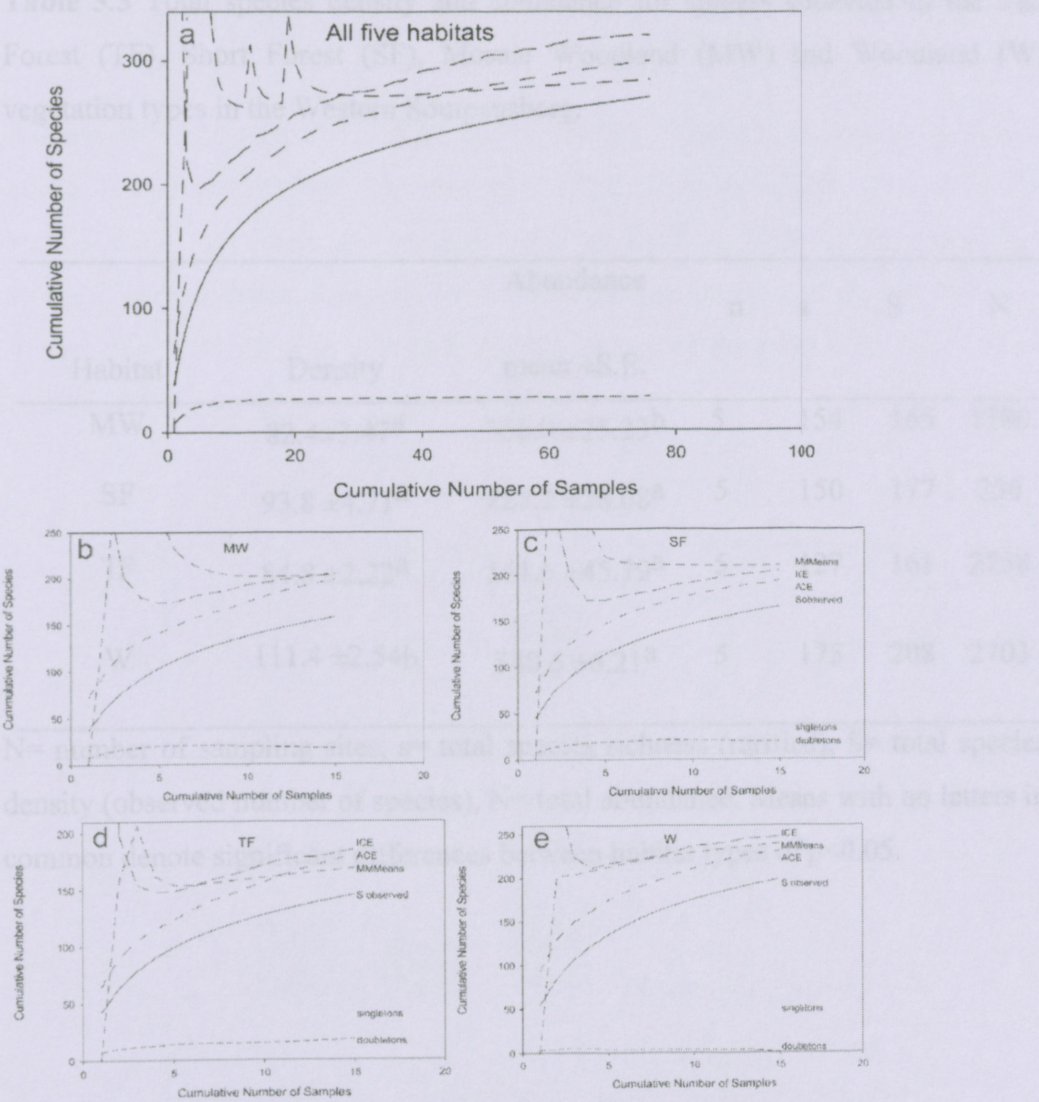

Figure 3.2 Plots comparing performance of three estimators with that of the observed number of species. For each curve a point is the mean of 100 values based on the 100 randomizations (Fig. 3.2a.) All 74 samples were collected at farm Lajuma for all habitats except the grassland. May 2004 Autumn, November 2004 Spring and March 2005 Summer samples combined for (3.2b.) Woodland Mosaic, (3. 2c.) Short Forest, (3.2d.) Tall Forest, (3.2e.) Woodland.

Table 3.3 Total species density and abundance for spiders collected in the Tall Forest (TF), Short Forest (SF), Mosaic Woodland (MW) and Woodland (W) vegetation types in the Western Soutpansberg.

Habitat	Density	Abundance mean ±S.E.	n	s	S	N
MW	82.4±3.47^{a}	356.0 ±25.23^{b}	5	154	165	1780
SF	93.8 ±4.71^{a}	487.2 ±28.68^{a}	5	150	177	236
TF	84.8 ±2.22^{a}	551.6 ±45.79^{a}	5	127	161	2758
W	111.4 ±2.54b	540.6 ±6.21^{a}	5	175	208	2703

N= number of sampling sites, s= total species richness (rarified), S= total species density (observed number of species), N= total abundance. Means with no letters in common denote significant differences between habitat types of $p<0.05$.

Almost 80% of specimens caught were juveniles, while the highest proportion of adults (34%) and highest numbers was collected in November (Fig. 3.3). The results show that plant and ground wanderer richness peaks in the Woodland, whereas web builders dominate spider assemblages in the Tall Forest and Short Forest (Fig. 3.4). Ground wanderer abundance varied the most with their activity peaking during spring (Fig. 3.5).

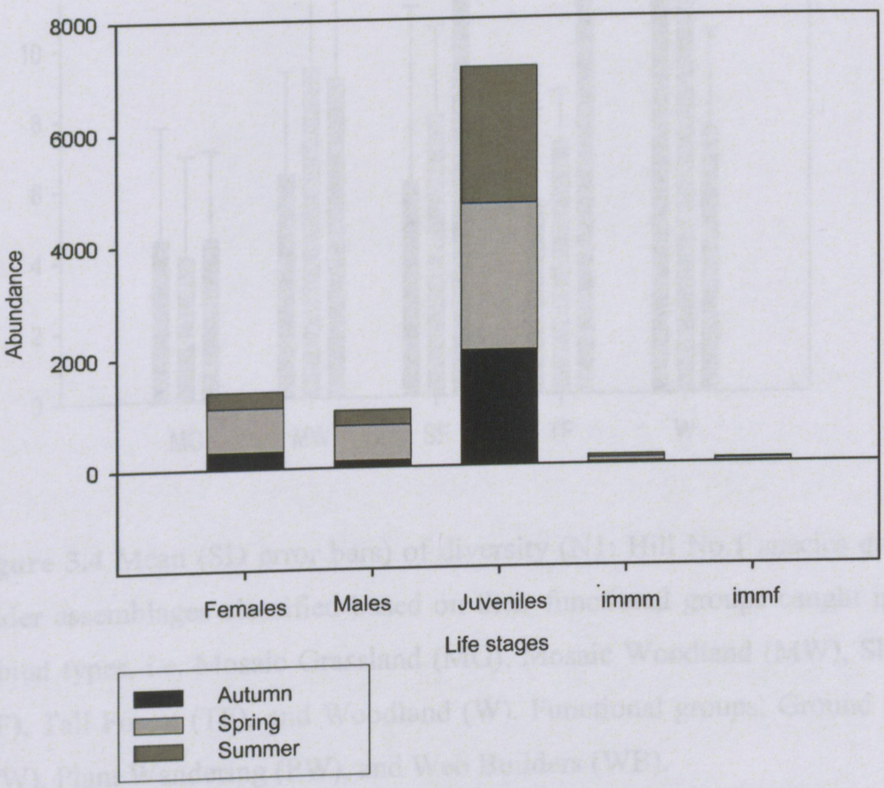

Figure 3.3 Abundances of life stages caught during each of the sampling periods, i.e. May 2004 = Autumn, November 2004 = Spring, March 2005 = Summer. immature male (immm), immature female.

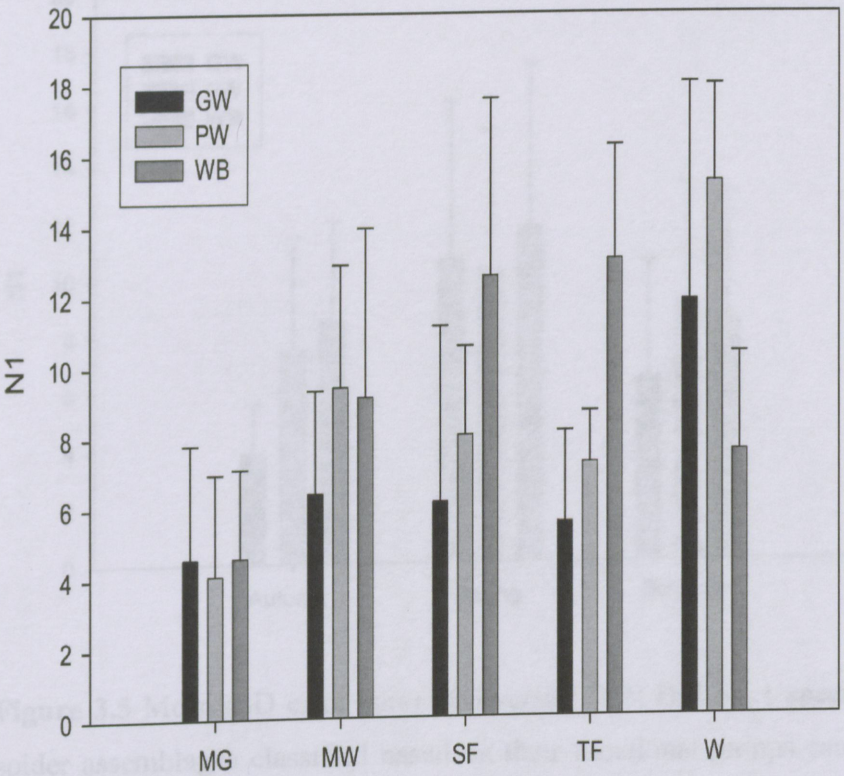

Figure 3.4 Mean (SD error bars) of diversity (N1: Hill No.1 species diversity) of spider assemblages classified based on their functional groups caught in different habitat types, i.e. Mosaic Grassland (MG), Mosaic Woodland (MW), Short Forest (SF), Tall Forest (TF), and Woodland (W). Functional groups: Ground wandering (GW), Plant Wandering (PW), and Web Builders (WB).

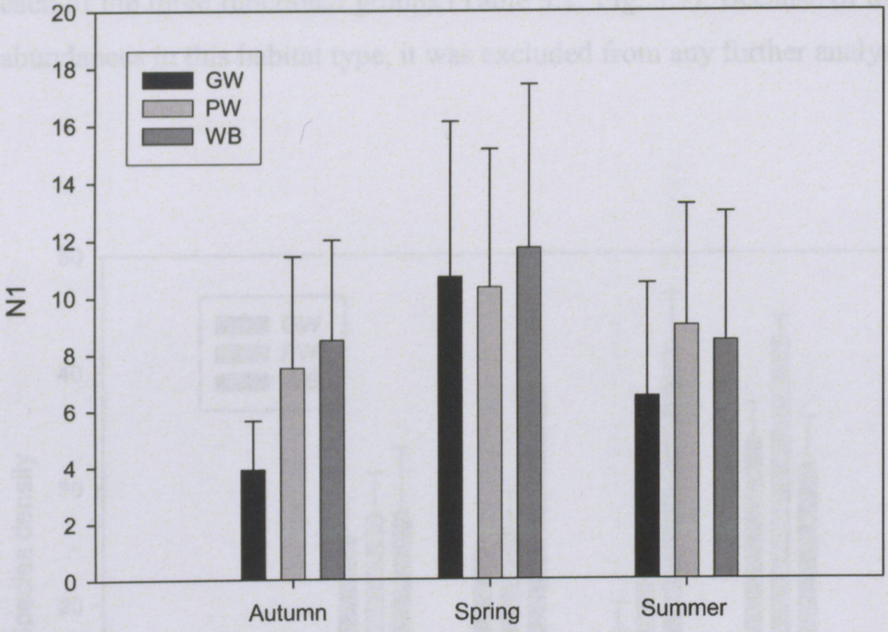

Figure 3.5 Mean (SD error bars) of diversity (N1: Hill No.1 species diversity) of spider assemblages classified based on their functional groups caught in different seasons. May 2004 = Autumn, November 2004 = Spring, March 2005 = Summer. Functional groups: Ground Wandering (GW), Plant Wandering (PW), and Web Builders (WB).

All functional groups were most diverse in Spring and decreased as the season progressed. Ground wanderers had the largest decrease in diversity while web builders and plant dwellers had a lower degree of seasonal variation. Woodlands had the highest species density while Mosaic Grasslands had the lowest (Fig. 3.6). Plant and ground wanderers had the highest species densities in the Woodlands, while web dwellers dominated Tall and Short Forests (Fig. 3.6). Plant wanderers were the most abundant in Woodlands; the Tall Forest had the most web builders, whereas ground wanderer abundance was not significantly affected by habitat type, except for Mosaic Grasslands, where considerably fewer spiders were caught in each of the three functional groups (Table 3.2; Fig. 3.7). Because of the low spider abundances in this habitat type, it was excluded from any further analyses.

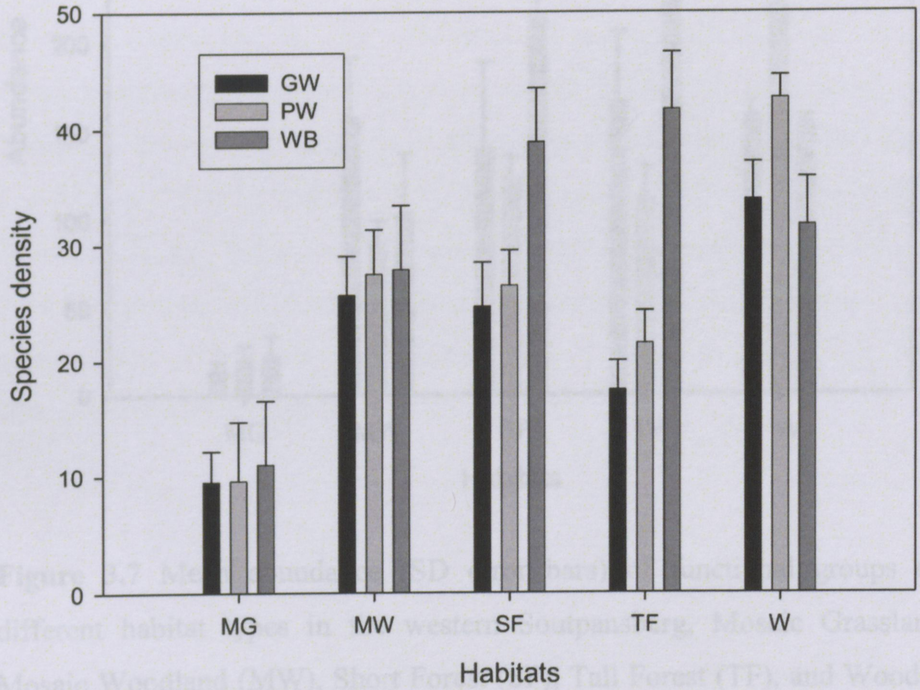

Figure 3.6 Mean (SD error bars) of species of density of functional groups caught in different habitat types. Mosaic Grassland (MG), Mosaic Woodland (MW), Short Forest (SF), Tall Forest (TF), and Woodland (W). Functional groups: Ground Wandering (GW), Plant Wandering (PW), and Web Builders (WB).

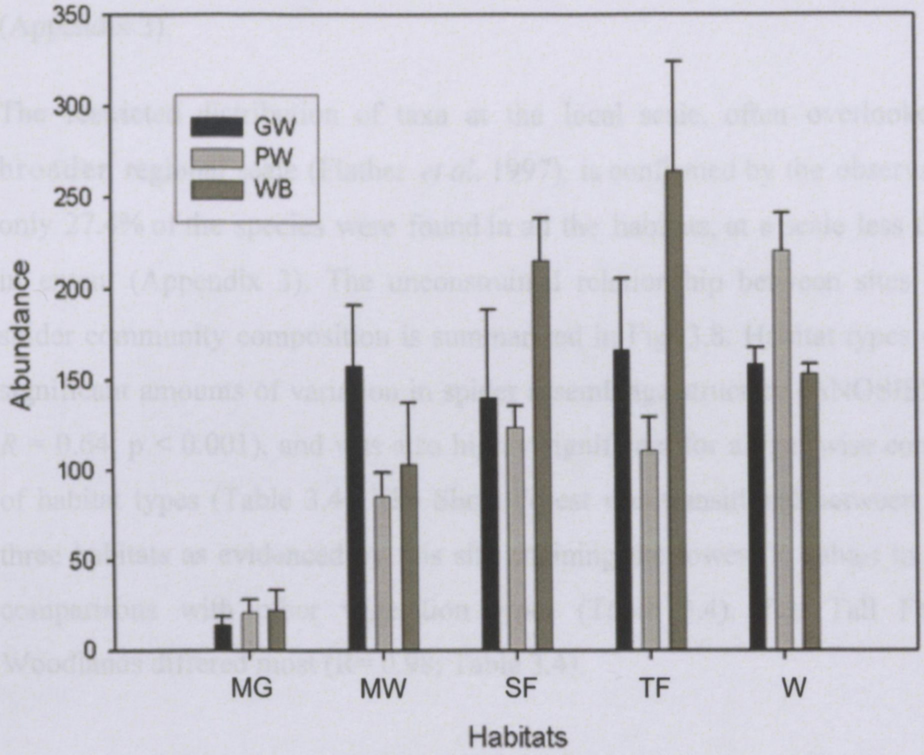

Figure 3.7 Mean abundance (SD error bars) of functional groups caught in different habitat types in the western Soutpansberg, Mosaic Grassland (MG), Mosaic Woodland (MW), Short Forest (SF), Tall Forest (TF), and Woodland (W). Functional groups: Ground Wandering (GW), Plant Wandering (PW), and Web Builders (WB).

Four families, Theraphosidae (MW), Migidae (TF), Phyxelididae (SF), and Sicariidae (TF), were restricted to single vegetation types. Eleven per cent of the species caught in the Mosaic Woodland were specific to this vegetation type, 10% for the Short Forest, 7.5% for the Tall Forest, and 16% in the Woodlands (Appendix 3).

The restricted distribution of taxa at the local scale, often overlooked at the broader regional scale (Flather *et al*. 1997), is confirmed by the observation that only 27.4% of the species were found in all the habitats, at a scale less than 2 km in extent (Appendix 3). The unconstrained relationship between sites based on spider community composition is summarized in Fig. 3.8. Habitat types explained significant amounts of variation in spider assemblage structure (ANOSISM Global $R = 0.64$, $p < 0.001$), and was also highly significant for all pairwise comparisons of habitat types (Table 3.4). The Short Forest was transitional between the other three habitats as evidenced by this site attaining the lowest R values in pair-wise comparisons with other vegetation types (Table 3.4). The Tall Forest and Woodlands differed most (R= 0.98; Table 3.4).

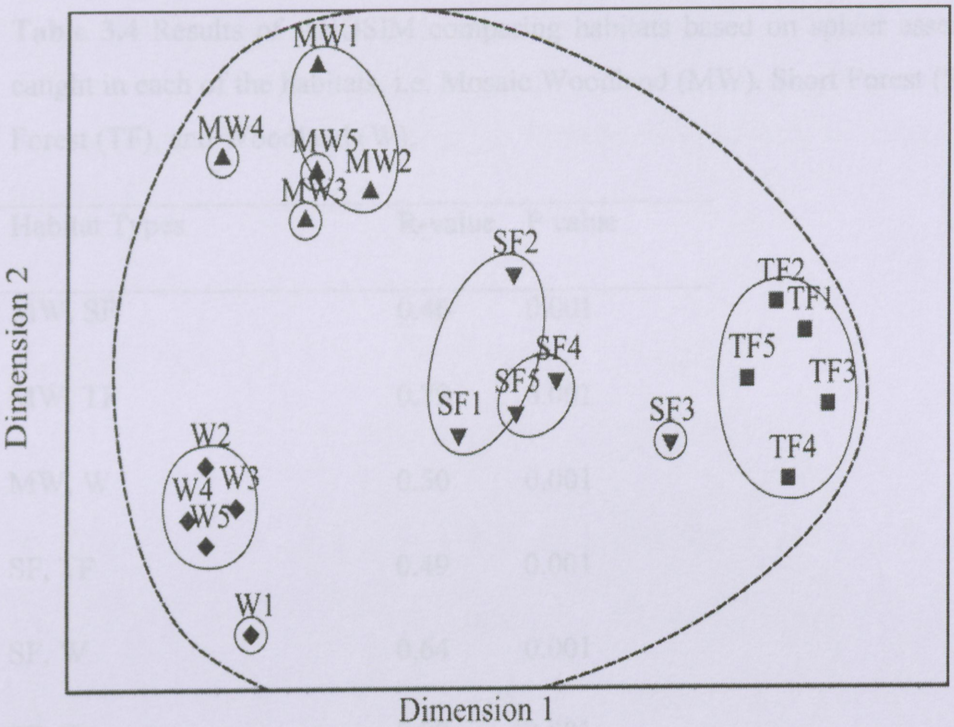

Figure 3.8 Multidimensional scaling (MDS) ordination plot of the five replicated sites in each of the four vegetation types based on multidimensional scaling of the spider assemblages. Each number indicates transects 1-5 in each habitat, i.e. Mosaic Woodland (MW), Woodland (W), Short Forest (SF), and Tall Forest (TF). Stress = 0.11.

Table 3.4 Results of ANOSIM comparing habitats based on spider assemblages caught in each of the habitats, i.e. Mosaic Woodland (MW), Short Forest (SF), Tall Forest (TF), and Woodland (W).

Habitat Types	R-value	P value
MW, SF	0.46	0.001
MW, TF	0.79	0.001
MW, W	0.50	0.001
SF, TF	0.49	0.001
SF, W	0.64	0.001
TF, W	0.98	0.001

An evaluation of the spider assemblages targeted by the different sampling methods suggests that tree beating and sweepnetting differ most in terms of their composition (R= 0.86), followed by leaf litter sifting and tree beating (R= 0.81; Table 3.5; Fig. 3.9). It is evident that active searching, leaf litter sifting and pitfall trapping overlap most in the assemblages they target, while sweepnetting and tree beating had the most distinct assemblages (Table 3.5; Fig. 3.9).

Table 3.5 Results of pairwise comparisons based on ANOSIM, testing for differences between assemblages targeted by the different sampling methods, i.e. Active search (A), Leaf litter sifting (L), Sweepnetting (S), Tree beating (T), Pitfall trapping (P).

Sampling Technique	R value	P value
A, L	0.23	0.001
A, P	0.21	0.001
A, T	0.66	0.001
A, S	0.48	0.001
L, P	0.29	0.001
L, T	0.81	0.001
L, S	0.68	0.001
P, T	0.64	0.001
P, S	0.43	0.001
T, S	0.86	0.001

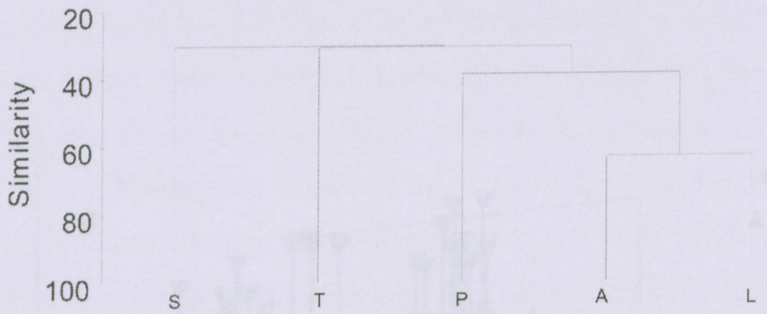

Figure 3.9 Agglomerative hierarchical cluster analysis of spider assemblages caught through the various sampling techniques, i.e. Sweepnetting (S), Tree beating (T), Pitfall trapping (P), Active search (A), and Leaf litter sifting (L).

The cluster analysis of the seasonal samples suggests hierarchical structuring of spider assemblages on a temporal scale (Fig. 3.10). The Tall Forest is least affected by seasonal changes in spider composition but does display seasonal structuring within. The Mosaic Woodlands, Woodlands and Short Forest are seasonally structured, with spider assemblages of one season being more similar between different habitats than to spider assemblages from the same habitat but from another season (Fig. 3.10). Within a season assemblages were more similar within a habitat, except for the Mosaic Woodland where turnover of spider species seems to be higher, resulting in significant changes in species composition throughout the year (Fig. 3.10).

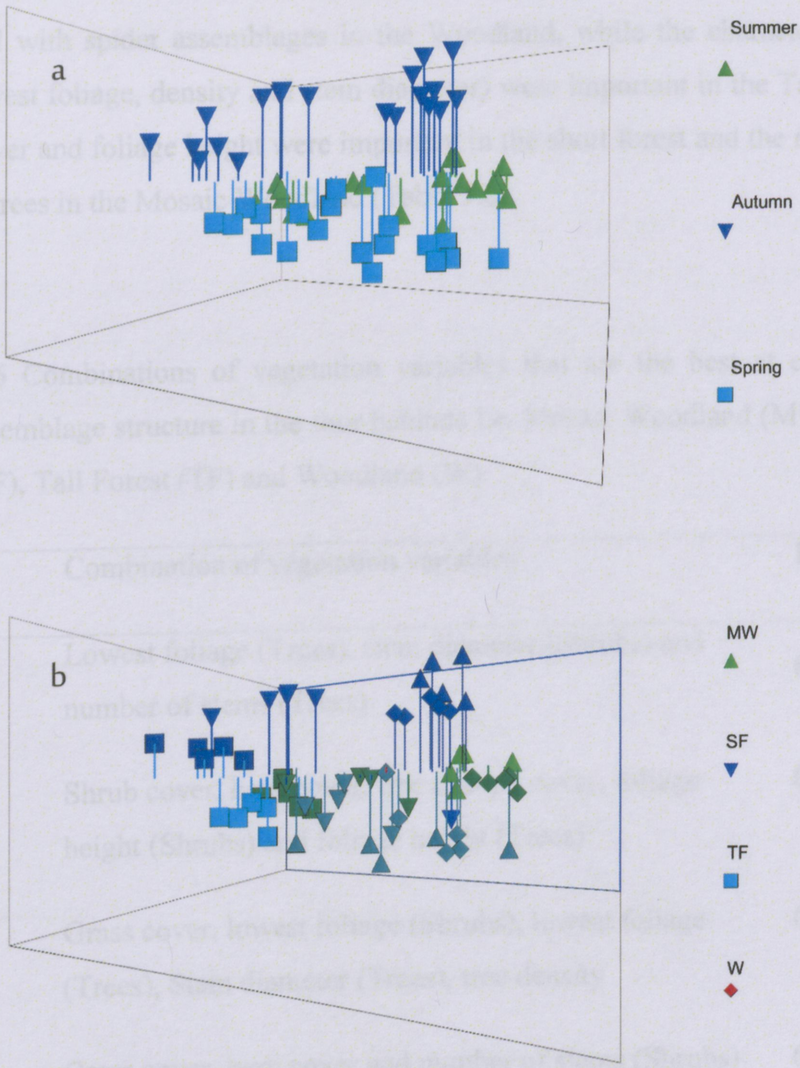

Figure 3.10 Non-metric multidimensional scaling (MDS) of four habitats (Mosaic Woodland (MW), Short Forest (SF), Tall Forest (TF), and Woodland) and five replicates within each habitat, displaying a) temporal and b) spatial structuring. Summer (March 2005), Autumn (November 2004), and Spring (March 2005). The first three axes of MDS ordination are displayed.

Spearman rank correlations between the ranked similarity matrices of environmental variables and that of the spider assemblages varied from 0.585 for the Mosaic Woodland to 0.827 for the Woodlands (Table 3.6). Grass cover, herb cover and number of stems of shrubs were the combination of variables best correlated with spider assemblages in the Woodland, while the characteristics of trees (lowest foliage, density and stem diameter) were important in the Tall Forest. Shrub cover and foliage height were important in the short forest and the number of stems of trees in the Mosaic Woodland (Table 3.6).

Table 3.6 Combinations of vegetation variables that are the best at explaining spider assemblage structure in the four habitats i.e. Mosaic Woodland (MW), Short Forest (SF), Tall Forest (TF) and Woodland (W).

Habitat	Combination of vegetation variables	P_w-value
MW	Lowest foliage (Trees), stem diameter (Shrubs) and number of stems (Trees)	0.585
SF	Shrub cover, herb cover, tree canopy cover, foliage height (Shrubs) and foliage height (Trees)	0.717
TF	Grass cover, lowest foliage (Shrubs), lowest foliage (Trees), Stem diameter (Trees), tree density	0.726
W	Grass cover, herb cover and number of stems (Shrubs)	0.827

Twenty- three species had significant IndVals > 70 (Table 3.7). The majority of indicators were associated with the Woodland (13), followed by Tall Forest (8), which was dominated by web builders such as *Nephila pilipes*, *Leucauge decorata* and *Glenognatha* sp. The Linyphiidae sp. 5 was restricted to Short Forest (Table 3.7) and *Neoscona blondeli* to the Mosaic Woodlands. Both of these are web dwellers.

Table 3.7 Percentage indicator values (IndVal>70) for spider species for each vegetation type throughout the year based on the spiders caught. Woodland, Tall Forest, Short Forest, and Mosaic Woodland. Functional guilds Web builders (WB), Ground wandering (GW), and Plant wandering (PW).

Species	**Guild**	**IndVal**	**Habitat**
Neoscona blondeli	WB	80.33	MW
Linyphiidae sp. 5	WB	71.44	SF
Anyphops sp. 1	GW	74.45	TF
Hermacha mazoena	WB	78.13	TF
Leucauge decorata	WB	78.37	TF
Nephila pilipes	WB	76.06	TF
Phlegra sp. 1	GW	71.43	TF
Quamtana bonamanzi	WB	77.37	TF

Scytodes clavata	GW	82.82	TF
Selenops brachycephalus	GW	91.77	TF
Camillina cordifera	GW	82.88	W
Diaea puncta	PW	75.52	W
Hersilia sericea	PW	80	W
Oxyopes hoggi	PW	81.55	W
Oxyopes jacksoni	PW	81.01	W
Oxyopes russoi	PW	81.43	W
Oxyopes schenkeli	PW	81.2	W
Runcinia aethiops	PW	74.07	W
Runcinia flavida	PW	87.8	W
Simorcus zuluanus	PW	85.21	W
Synema sp. 2	PW	84.96	W
Thomisus spiculosus	PW	100	W
Tibellus minor	PW	80	W

Uloborus lugubris and *Neoscona blondeli* contributed most to the difference between the Mosaic Woodland and the Short Forest, with *U. lugubris* being much more abundant in the Short Forest and *N. blondeli* very common throughout the Mosaic Woodland (Table 3.8). Although *U. lugubris* contributed more to average dissimilarity, *N. blondeli* contributed more consistently to these differences. Furthermore, *Theridion* sp. 1 and Linyphiidae sp. 5, both web dwellers, were very abundant in the Short Forest (Table 3.8).

It is evident that several species, such as *Selenops brachycephalus*, *Leucauge decorata*, *Quamtata bonamanzi*, *Glenognatha* sp. and *Scytodes clavata* were considerably more abundant in the Tall Forest than in other habitats. Two species commonly found on grasses, *Heriaeus crassispinus* and *Runcinia flavida*, were abundant in the Woodland, as well as a third thomisid, *Diaea puncta,* found on shrubs, grasses and trees. The lycosid, *Proevippa albiventris*, was commonly found in all four habitat types but varied in abundance, decreasing significantly from the Short Forest, Mosaic Woodland, Tall Forest to Woodlands (Table 3.8).

Table 3.8 SIMPER results listing species responsible for more than 10% of the cumulative dissimilarities between sites, sorted based on descending contribution to dissimilarities. Habitats: Mosaic Grassland (MG), Mosaic Woodland (MW), Short Forest (SF), Tall Forest (TF), and Woodland (W).

	Av.Abund	**Av.Abund**	**Av.Diss**	**Diss/SD**	**Contrib%**
Average dissimilarity: 54.22	MW	SF			
Uloborus lugubris	1	31.2	1.37	1.81	2.53
Neoscona blondeli	22.6	1.2	1.23	4.69	2.26
Theridion sp. 1	0.6	17	1.09	1.79	2.01
Trabea heteroculata	17.8	18.6	0.88	1.98	1.62
Linyphiidae sp. 5	0	15	0.86	0.84	1.59
Average dissimilarity: 62.27	MW	TF			
Selenops brachycephalus	0	32.6	1.79	4.47	2.88
Leucauge decorata	1	40.8	1.77	7.06	2.84
Quamtana bonamanzi	1	31.6	1.5	2.57	2.41
Scytodes clavata	0	21	1.41	3.86	2.27
Neoscona blondeli	22.6	0.8	1.3	5.24	2.09
Glenognatha sp.	0	14.2	1.2	7.58	1.92
Average dissimilarity: 54.69	MW	W			
Linyphiidae sp. 1	5.8	73.4	1.9	3.7	3.48
Neoscona blondeli	22.6	2.2	1.2	3.7	1.84

Diaea puncta	2.4	21.8	0.92	2.1	1.63
Proevippa albiventris	57.2	21	0.86	2.4	1.5
Heriaeus crassispinus	1	14.2	0.85	1.7	1.54
Runcinia flavida	0	5.4	0.68	8.2	1.25
Average dissimilarity: 50.66	SF	TF			
Selenops brachycephalus	2	32.6	1.34	2.81	2.65
Quamtana bonamanzi	2	31.6	1.33	2.25	2.62
Leucauge decorata	8.2	40.8	1.12	2.61	2.21
Theridion sp. 1	17	0	1.07	1.77	2.12
Anyphops sp.1	0.8	16.6	1.01	2.11	1.99
Average dissimilarity: 55.7	SF	W			
Linyphiidae sp. 1	17.6	73.4	1.22	2.76	2.2
Uloborus lugubris	31.2	0.6	1.21	1.94	2.18
Proevippa albiventris	58.8	21	0.94	2.18	1.69
Theridion sp. 1	17	0.8	0.89	1.83	1.6
Clubiona godfreyi	45.4	12.2	0.89	4.36	1.59
Runcinia flavida	0	5.4	0.62	8.33	1.12
Average dissimilarity: 65.98	TF	W			
Linyphiidae sp. 1	7.8	73.4	1.6	3.78	2.42
Leucauge decorata	40.8	0.8	1.52	7.97	2.3
Selenops brachycephalus	32.6	0.2	1.44	4.16	2.18

Scytodes clavata	21	0.2	1.12	3.53	1.7
Diaea puncta	0.6	21.8	1.08	2.84	1.63

CHAPTER 4. DISCUSSION

The main constraints in arachnid conservation are the large number of species and the variable taxonomic resolution between orders, families and genera; lack of expertise and capacity; and challenges associated with comprehensive field sampling of arachnids (Cardoso *et al*. 2011). Within a South African context, the SANSA initiative started in 1997, aimed to create an inventory of the arachnid fauna of South Africa (Dippenaar-Schoeman & Creamer 2000). The current study has made a significant contribution towards the existing South African National Survey of Arachnida (SANSA) database.

Spider abundance and richness was highest in November (spring) as compared to March (summer) in five habitat types, with 34% of the spiders caught in spring being adults decreasing to 25% later in the season (Fig. 3.3). This corresponds with previous findings (Muelelwa *et al*. 2010, Whitmore *et al*. 2002) and probably results from subadults, because of juveniles that mature during the winter months and attain adulthood in spring when mating occurs. By autumn, most of the adults, males in particular, have died or numbers are exceeded by juveniles. The proportion of adults in the present study also conformed to the results of several other studies (Coddington *et al*. 1991, 1996; Scharff *et al*. 2003).

Almost all the endemic taxa in this study were associated with Tall Forest. Biogeographically, Tall Forest is classified as Northern Mistbelt Forests (Mucina *et al*. 2005) and the sites sampled represent one of a group of distinct isolated patches scattered throughout the Soutpansberg, varying in both size and isolation. Two of the five families recorded outside Lajuma, in the Soutpansberg, namely Archaeidae and Cyatholipidae, are restricted to the Northern Mistbelt Forests. With the exception of one species (*Afrarchaea bergae* Lotz 1996; 2006), all the

Soutpansberg records for these two families are endemic species (Griswold 1987; Lotz 2003, 2006).

In contrast to studies in other regions, where high levels of plant endemism are mirrored by those of invertebrates (Botes *et al.* 2006; Haddad *et al*. 2006), no such relationship exists for Tall Forest. Only one plant taxon, *Streptocarpus parviflorus* subsp. *soutpansbergensis*, is endemic to Mistbelt Forests of the Soutpansberg (Hahn 2006). Most of the plant species endemic to the Soutpansberg are found in mountain sourveld, grasslands and xeric habitats, and 47% are succulents (Hahn 2006). An important process that affects all the vegetation types with high endemism is mist precipitation. This is especially true in times of drought, and many of the plant endemics (33.3%) are restricted to the mistbelt. Little is known about mist interaction with the environment that is influenced by moisture-laden air from the Indian Ocean, orography, and aerodynamics (Hahn 2006). All the new endemic or near endemic species described from specimens collected in this study were associated with the Woodlands, namely *Australutica africana* and *Vendaphaea lajuma*. These woodlands are associated with mistbelt vegetation.

The differences in vegetation structure between the vegetation types examined resulted in the formation of different spider communities in the four vegetation types. The turnover of species between Tall Forest and Woodlands were the greatest (Fig. 3.8). Similar to other studies, for example Rypstra (1986) and Jiménez-Valverde & Lobo (2007), our results suggest that vegetation structure, when compared among the habitats examined, is likely to contribute towards the differences between their associated spider assemblages. The large amount of variation explained by local variables, i.e. vegetation structure, is consistent with the suggestion that local processes play a large role in determining spider diversity, as competition is less likely to occur between these predators, and vegetation structure variables should therefore explain local diversity (Borges & Brown 2004).

This also confirms that spiders have predictable assemblages based on habitat structure (Uetz 1991; Borges & Brown 2004). In addition, results explained that spider assemblages varied considerably, whereby the turnover in the Woodlands was greatest (Table 3.6). Grass cover, herb cover and number of stems of shrubs were the combination of variables best correlated with spider assemblages in the Woodland, while the characteristics of trees (lowest foliage, density and stem diameter) were important in the Tall Forest. Shrub cover and foliage height were important in the Short Forest and the number of stems of trees in the Mosaic Woodland.

Although broad generalizations regarding the spider diversity related to the vegetation types examined cannot be made as a result of low replication in the sampling design, a few potential trends with wider significance for spider conservation can be highlighted based on those patches that were sampled, and should be investigated further to examine the generality of these trends. First, the restricted distribution of taxa at the local scale is often overlooked at the broader regional scales (Flather *et al.* 1997). Second, significant differences in the spider assemblage structure, largely explained by differences in vegetation structure, were found among the four vegetation types examined, emphasising the importance of local scale heterogeneity. Although the Tall Forests were comparatively species poor, they still had a considerable number of endemic taxa, emphasizing the role of providing stable conditions for the evolution and survival of endemic spider species in spite of their limited geographic range (Borges & Brown 2004).

The Soutpansberg region has been experiencing a dramatic increase in bush encroachment over the last 100 years because of human influence, namely elimination of keystone herbivores, introduction of goats, and suppression of fires as a result of afforestation (Hahn 2006). Grasslands, specifically, have been under increasing pressure in the Soutpansberg and very few remnant patches remain (Hahn 2006). Increased bush encroachment results in higher tree densities, and based on findings from this study, spider assemblages are likely to change into those represented by Mosaic Woodland and Short Forest, should such encroachment continue. These two vegetation types, generally, had lower species diversity, less indicator taxa and no endemic taxa. In addition, much of the variation of the spider assemblages in this study was explained by the two vegetation types most threatened in the Soutpansberg region, namely Woodlands and Tall Forests. Further bush encroachment would most certainly result in the loss of heterogeneity at this small scale and homogenization of spider assemblages, as well as the loss of endemic taxa. Woodlands and tall mist forest are the vegetation types which should form the focus of core areas in the proposed biosphere reserve, with associated management initiatives to prevent bush encroachment in these areas.

It has been shown by several other studies (Cardoso *et al*. 2008; Lovell *et al*. 2009) that the suite of methods used is the single most important factor in determining the abundance, species richness and composition of spiders sampled, hence this lead to real possibility to develop a protocol that has broad applicability. Findings in this study correspond with that of Muelelwa *et al*. (2010), which showed that beating targeted the most species, followed by ground collecting and leaf litter sifting. However, our findings contrast with the work in the Iberian Peninsula, where ground searching was the least effective method (Cardoso 2009), and this corresponds with the findings of the current study, where active search yielded comparatively low number of species. Pitfall trapping was the method that contributed the smallest number of unique species, but this pattern is consistent with Whitmore *et al*.'s (2002) results in the Savanna Biome. The species composition of leaf litter sifting and ground collecting overlapped significantly with that of pitfall catches. The wider significance of this result should, however, consider that the efficiency of any particular method may vary between habitats, vegetation types or biogeographic regions as a result of the relative dominance of different spider taxa and differences in vegetation structure. Furthermore, the use of litter sifting and canopy fogging as sampling techniques in future studies should be encouraged, as they have both been poorly utilised in past surveys and show considerable potential for the discovery of new taxa (e.g. Butler & Haddad 2011; Haddad & Wesolowska 2011; Wesolowska & Haddad 2013). Active searching at the base of grass tussocks and selection of pitfall sites in areas with different soil characteristics are also likely to yield new and unusual grassland species (Foord *et al.* 2011a).

The study by Halaj *et al.* 2000 shows that habitat structure dictates the distribution and community of arboreal arthropods, the result of the study lend strong support to the importance of habitat structural diversity in explaining general patterns of arthropod abundance and diversity on plants. In addition to the findings of the study, a strongly parallel patterns of spider species on a different habitat scale was observed, in western Oregon, more spider species are found on structurally more diverse tree species, and almost 60% of this variation is explained by the tree's structural heterogeneity (Halaj *et al.* 1998). Rypstra (1986) has documented strong correlations between the abundance of web-building spiders found on undergrowth vegetation and the biomass of Norway spruce. The pattern was consistent across three distinct communities, ranging from tropical Gabon through subtropical Peru to temperate sites in the northeastern United States. This clearly shows that spider abundance in tree canopies closely follows the availability (amount) of habitat substrate provided by host-tree species (Halaj *et al.* 1998). For example, although western hemlock appears structurally more complex than red alder, the disparity in the number of spiders that live on their branches may simply mirror differences in the branch biomass that both tree species can produce. Similarly, a greater spider abundance on higher-elevation Douglas-fir may be attributed to a greater biomass availability on this species at higher than lower sites (Halaj *et al*. 1998).

However, after accounting for the effect of branch biomass, various habitat variables such as prey availability, number of individual leaves, branching angles, or branch spread entered the prediction models. These may reflect fine-grained qualities of the habitat (microclimate, web-constructing sites or refugia), allowing a greater niche diversification and coexistence of more spider species (Halaj *et al*. 2000). For example, Greenstone (1984) documented a strong positive relationship between the diversity of web-building spiders and vegetation structural diversity across several habitat types ranging from tropical meadow in Costa Rica to scrub sites in California. Halaj *et al*. (2000) found that there were more spiders collected from Douglas-fir than noble fir at higher elevations; yet, noble fir branches of comparable length contained more biomass than Douglas-fir. Similarly, red cedar branches contained significantly more foliage biomass than western hemlock or Douglas-fir, but supported fewer spider species than either host-tree species (Halaj *et al*. 2000). Prey availability, or subtle differences in the branching pattern, resulting in a more favorable microclimate, may be responsible for this discrepancy (Halaj *et al*. 2000). Indeed, Douglas-fir branches at all higher elevation sites contained twice the number of total non-Araneae arthropods, and more than three times the densities of aphids than noble-fir branches; red cedar was the most prey-poor of all species (Halaj *et al*. 2000).

In conclusion, a few potential trends with wider significance for spider conservation should be investigated further to examine the generality of these trends based on those patches that were sampled. The loss of heterogeneity at this small scale and homogenization of spider assemblages as a result of bush encroachment is noted. Beating contributed the largest amount of unique species and pitfall trapping method contributed the least, pointing to the importance of optimising sampling protocols in order to maximize return on effort and cost.

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Appendix 2 Plant species and their abundance recorded in each of the replicates of the five habitat types.

Species	Sites																			
	mw1	mw2	mw3	mw4	mw5	sf1	sf2	sf3	sf4	sf5	tf1	tf2	tf3	tf4	tf5	w1	w2	w3	w4	w5
Acacia ataxacantha	6	3	6	3	2	8	25	11	21	16			1	2						
Acacia karroo						1			6	6			1			1				
Acokanthera oppositifolia		1		1	2															
Allophylus africana							7	1					4	1	2					
Ancylobotris capensis																			2	
Apodytes dimidiata					1															
Brachylaena discolor			1		1	2		4		2					1					
Canthium inerme			5			1				2					3					
Canthium mundianum	54			18	7	78	67						7	4			1			5
Carissa bispinosa											1			3						
Carissa tetramera						1														
Catha endulis								85	5	4										
Celtis africana		1							1	2			3		1					
Cessonia spicata					2															
Chionanthus															1					
Claredendrum glabrum						2		2	1											
Combretum moggii		3	1																	
Combretum molle																3	3	1		3
Combretum vendae		4															1			1
Croton sylvaticus										1	15	9	4	13	3					
Cussonia spicata							1				1	2			1	1				
Dichrostachys cinerea									3	2						5	3	2		2
Diospyros dichrophylla																	1			7
Diospyros lycioides		1														1	5			
Diospyros whyteana			1				3	6					2	4	3					
Dombeya rotundifolia													1			1	1			
Dovyalis zeyheri		1	5			8	2	2									6	1	2	9
Englerophytum magalismontanum																2	10		1	6
Erythrina lysistemon																1				
Euclea natalensis										2				1						
Eugenia capensis					1															
Euginia natalitia	4	6	3	1	4		2	1				11	9	6	6					
Faurea saligna																1		1		
Grewia flavescens																	1		1	1
Grewia occidentalis							6	1						3						
Gymnosporia buxifolia				8	13	2	1													
Gymnosporia havianus	1	5	13			19	45	60	2	2	5	10	4	21	32					
Helichrysum					1															
Heteropyxis natalensis																			1	
Hexalobus monopetalus					1															
Hyperacanthus amoenus	4	10	3	4	2	1	2									2	4	2	4	7
Keetia queinzii												2			1					
Maerua caffra	2	3	1		1												1			1
Maytenus undata	1		7	3	5	1	1			1									3	
Micrococca capensis															1					
Mimusops zeyheri			5							1		3		3	3	6			4	3
Mundulea sericea		1	25														6	10	1	3
Nuxia floribunda												2								
Olea capensis				2	4															
Olea europaea	1					2		2	2				1	1	4			1		
Olinia rochetiana		1	1																	
Opuntia ficus-indica																	2			
Pachystigma bowkeri						3					5	4	4	1			1			1
Pavetta grandifolia																1				
Plectrantus										5						10	4	9		
Podocarpus latifolius							2	3			2				3					
Rapanea melanophloeos	1										2				3					
Rhoicissus tomentosa							1													
Rhus chirindensis		1	1	2		2	4	1	11	4	2	5	3	4	4					1
Rhus coddii																1				
Rhus lucida					1					13										
Rhus magalismontanum																1	2			
Rhus queinzi		2	1	3		3	8	1	22				3			5	2			1
Rothmannia capensis		1	1		4												1			
Rothmannia globosa		10	2	5	3															
Scolopia zeyheri				1																
Strychnos	3				2															
Strychnos henningsii																1				
Tarenna zimbabwensis	3	3		1	6	2											1		3	1
Tricalysia capensis	1			5		1				2				1						
Trichilia dregeana											1	1								
Trimeria grandifolia							1	4							1					
Vangueria infausta																5			1	1
Vepris lanceolata							2						2	3			1			
Vitex rehmannii																2	4	1		2
Ximenia caffra																				2
Zanthoxylum capense																				2
Ziziphus mucronata						1	2	1					1							

Appendix 3 A check list of the spiders (Araneae) of farm Lajuma. The density of genera is given for the whole assemblage, whereas species densities are given for the whole assemblage and for the habitats (abundances are given in brackets). Functional group: AOWB= adapted orb-web; FGW=free-living ground wanderer; BGW= burrow -living ground wanderer; BPW=burrow-living plant wanderer; FPW=free living plant wanderer; burrow living plant wanderer; FWB=funnel-web; RWB=retreat-web; OWB=orb-web; GWB=gum foot-web; SHWB=sheet-web; SPWB=space-web. Habitat abbreviations: MG=Mosaic Grassland; MW=Mosaic Woodland; SF=Short Forest; TF=Tall Forest; W=Woodland. Symbols: •indicates a new genus and species, indicates a new species, a possible new species; all endemic taxa are indicated by grey arrow heads: endemic, near-endemic to the Soutpansberg. Foord et al. 2002 (p) lists the species recorded in that study, new records, post-2005, are also listed (p).

Family	Species /Functional group	Foord *et al.* 2002	New records: post-2005	Genera	Spp.	Vegetation type				
						MG	MW	SF	TF	W
Agelenidae	*Agelena australis* (Simon, 1896)/ FWB	p				5	2	0	0	7
	Benoitia ocellata (Pocock, 1900)/ FWB	p				9	16	0	0	1
	Maimuna deserticola (Simon, 1910)/ FWB		p							
	Total			2	2	2(14)	2(18)	0	0	2(8)
Amaurobiidae	*Pseudauximus annulatus* Purcell, 1908/ RWB	p				0	0	0	7	1
	Total			1	1	0	0	0	1(7)	1(1)
Anapidae	*Metanapis bimaculata* (Simon, 1895)/ OWB	p				0	0	0	0	0
	Total			0	0	0	0	0	0	0
Araneidae	•'*Acanthepeira*'-like/ OWB	0				0	1	9	1	0
	•Araneidae genus undetermined/ OWB	0				0	1	15	3	0
	Araneilla sp. 1/ OWB	p				35	4	3	1	14
	Araneus apricus (Karsch, 1884)/ OWB		p							
	Araneus legonensis Grasshoff & Edmund, 1979/ OWB	0				0	0	5	4	0
	Araneus nigroquadratus Lawrence, 1937/ OWB	0				0	1	2	0	2
	Araneus sp. 1/ OWB	0				1	17	22	25	0
	‡ *Araneus* sp. 2/ OWB	0				0	0	0	1	0

Family	Species/Functional group	Foord *et al.* 2002	New records: post-2005	Genera	Spp.	Vegetation type				
						MG	MW	SF	TF	W
	Araneus sp. 3/ OWB	0				1	1	1	0	1
	Araneus sp. 4/ OWB	0				2	1	0	1	0
	Araneus sp. 5/ OWB	0				0	0	1	0	0
	Araneus strupifera (Simon, 1885)/ OWB	p				0	0	0	0	0
	Argiope lobata (Pallas, 1772)/ OWB	0				1	0	0	0	0
	Caerostrissex cuspidate (Fabricius, 1793)/ OWB	p				0	3	7	19	3
	Chorizopes sp. 1/ OWB	0				0	0	0	0	2
	Cyclosa insulana (Costa, 1834)/ OWB	p				8	21	25	18	4
	Cyphalonotus larvatus (Simon, 1881)/ OWB	p				0	3	3	0	8
	Cyrtophora citricola (Forskål, 1775)/ OWB	p				0	1	0	0	0
	Hypsosinga lithyphantoides Caporiacco, 1947/	0				2	1	5	0	20
	Ideocaira sp.1/ OWB	0				0	1	0	2	0
	Lipocrea longissima (Simon, 1881)/ OWB	0				1	0	0	0	0
	Neoscona blondeli (Simon, 1885)/ OWB	p				7	113	7	4	11
	Neoscona penicillipes (Karsch, 1879)/ OWB	0				0	1	0	0	0
	Neoscona quincasea Roberts, 1983/ OWB	p				1	14	6	0	9
	Neoscona subfusca (C.L.Koch, 1837)/ OWB	p				1	27	11	6	20
	Neoscona triangula (Keyserling, 1864)/ OWB		p							
	Pararaneus spectator (Karsch, 1886)/ OWB	0				0	0	1	3	4
	Singa lawrencei (Lessert, 1930)/ OWB	0				2	0	0	0	2
	Total			18	28	13(65)	18(212)	17(127)	15(108)	14(101)
Caponiidae	*Caponia* sp./ FGW	p				0	2	2	1	2
	Total			1	1	0	1(2)	1(2)	1(1)	1(2)
Clubionidae	*Clubiona bevisi* Lessert, 1923/ FPW	0				0	18	2	12	5
	Clubiona godfreyi Lessert, 1921/ FPW	0				1	121	227	179	61
	Clubiona lawrencei Roewer, 1951/ FPW	p				0	0	0	0	0
	Clubiona pupillaris Lawrence, 1938/ FPW	0				0	7	6	9	5
	Clubiona sp. 1/ FPW	p				0	4	0	0	0
	Total			1	4	1(1)	4(150)	3(235)	3(200)	3(71)
Corinnidae	*Cambalida coriacea* Simon, 1909/ FGW	0				0	13	12	2	57
	Castianeira fulvipes Simon, 1896/ FGW	p				0	0	0	0	0
	Copaflavo plumose Simon, 1885/ FGW	0				0	23	16	13	23
	Copa sp. 2/ FGW	0				0	0	0	2	2
	Corinna natalis Pocock, 1898		p							
➢	•##Corinninae sp.1/ FGW	0				8	4	1	0	0
	Corinnomma lawrencei Haddad, 2006/ FGW	0				0	0	2	0	0
	Corinnomma radiata Haddad, 2006/ FGW	0				0	0	2	0	0
	Corinnomma semiglabrum (Simon, 1896)		p							
	Graptartia tropicalis Haddad, 2004/ FGW	0				0	6	3	0	1
➢	##*Hortipes contubernalis* Bosselaers & Jocqué, 2000/ FGW	p				0	0	0	3	0
➢	†##*Hortipes* sp. 4/ FGW	0				0	0	0	1	0
	Merenius simony Lessert, 1921/ FGW	0				0	1	0	0	0
	Pronophaea sp. 2/ FGW	0				0	21	0	0	1
	Pronophaea natalica Simon, 1897/ FGW	0				0	5	0	0	0
	Total			13	18	0	20(119)	13(52)	13(56)	17(103)
Ctenidae	*Ctenus transvaalensis* Benoit, 1981/ FGW	p				0	15	13	16	37
	Total			2	3	0	2(21)	2(25)	2(22)	3(51)
Cyrtaucheniidae	*Ancylotrypha nuda* (Hewitt, 1916)/ BGW	0				4	0	0	2	1
	Homostola pardalina (Hewitt, 1913)/ BGW	p				0	0	0	2	0
	Total			2	2	1(4)	0	0	2(4)	1(1)

Family	Species/Functional group	Foord *et al.* 2002	New records: post-2005	Genera	Spp.	Vegetation type MG	MW	SF	TF	W
Deinopidae	*Deino piscornigera* Gerstäecker, 1873/ AOWB	p				1	0	1	1	1
	Menneus camels Pocock, 1902/ AOWB	0				0	11	16	0	7
	Total			2	2	1(1)	1(11)	1(17)	1 (1)	1(8)
Dictynidae	*Devade* sp. 1/ RWB	p				0	0	6		1
	Mashimole leupi Lehtinen, 1967/ RWB	0				0	1	0		0
	Total			1	1	0	1(1)	1(6)		1(1)
Eresidae	*Dresse ruscolsoni* Tucker, 1920/ RWB	0				0	3	3		5
	Penestomus sp.(immature)/ FPW	p				0	0	0		0
	Total			2	1	0	1(3)	1(3)		1(5)
Eutichuridae	*Cheiracanthium africanum* Lessert, 1921/ FPW	p				3	10	2	0	24
	Cheiracanthium furculatum Karsch, 1879/ FPW	0				0	2	0	0	0
	Cheiracanthium vansoni Lawrence, 1936/ FPW	0				0	0	0	0	2
	Cheiramiona clavigera (Simon, 1897)/ FPW	0				0	10	9	7	39
	Cheiramiona lajuma Lotz, 2002/ FPW	p				1	24	36	19	124
	Cheiramiona krugerensis Lotz, 2002/ FPW	0				0	0	0	0	24
	Total			2	7	2(4)	4(36)	3(49)	2(26)	5(191)
Gnaphosidae	*Aneplasa* sp. 1									
	Aphanta ulaxinornata Tucker, 1923/ FGW	p				0	0	0		0
	Asemesthes numisma Tucker, 1923/ FGW	p				2	7	0		9
	Asemesthes payntei Tucker, 1923/ FGW	0				4	0	0		18
	Camillina cordifera (Tullgren, 1910)/ FGW	0				0	7	2		46
	Drassodes sp. 12/ FGW	p				3	0	1		0
	Drassodinae sp. 4/ FGW	0				0	28	4		39
	Echeminae sp. 10/ FGW	0				0	5	0		13
	Echeminae sp. 8/ FGW	0				1	3	2		8
	Echeminae sp. 9/ FGW	0				0	1	0		6
	Echemus erutus Tucker, 1923/ FGW	p				0	0	0		0
	Megamyrmekion transvaalense Tucker, 1923/ FGW	p				0	0	0		0
	Setaphis arcus Tucker, 1923/ FGW	p				0	0	1		3
	Setaphis subtilis (Simon, 1897)/ FGW	0				0	6	0		5
	Trachyzelotes jaxartensis (Kronenberg,1875/ FGW	0				0	1	0		0
	Xerophaeus sp./ FGW	p				0	0	0		0
	Zelotes Hewitti Tucker, 1923/ FGW	0				0	17	1		43
	Zelotes tuckeri Roewer, 1951/ FGW	p				0	0	0		0
	Zelotes ungulus Tucker, 1923/ FGW	0				1	2	1		8
	Zelotes sp. 7/ FGW	0				2	6	0		4
	Total			10	14	6(13)	11(86)	7(15)		12(207)
Hahniidae	*Hahnia tabulicola* Simon, 1898/ SPWB	p				0	2	0		1
	Total			1	1	0	1(2)	0		1(1)
Idiopidae	*Idiops castaneus* Hewitt, 1913/ BGW	p				0	0	0		0
	Total			0	0	0	0	0		0
Linyphiidae	Linyphiidae sp. 1/ SHWB	0				3	29	40		368
	Linyphiidae sp. 2/ SHWB	0				1	25	13		18
	‡ Linyphiidae sp. 4/ SHWB	0				0	1	0		0
	Linyphiidae sp. 5/ SHWB	0				0	0	2		3
	Linyphiidae sp. 7/ SHWB	0				0	0	0		1
	Linyphiidae genus_A sp.	p				0	0	0		0
	Linyphiidae genus_B sp.	p				0	0	0		0
	Mecynidis sp. 3/ SHWB	0				0	10	15		12
	Microlinyphia sp. 6/ SHWB	0				0	1	0		0
	Microlinyphia sterilis (Pavesi, 1883)/ SHWB	p				0	0	0		0
	Neriene natalensis Van Helsdingen, 1970/ SHWB	p				0	0	0		0
	Total			2	7	2(4)	5(68)	4(197)	4(70)	5(409)

Lycosidae	Lycosidae sp. 4/ FGW	0	1	20	9	4	8
	Lycosidae sp. 7/ FGW	0	1	0	0	0	0
	‡ *Minicosa neptuna* Alderweireldt & Jocqué, 2007/ FGW	0	0	0	0	0	1
	Pardosa crassipalpis Purcell, 1903/ FGW	p	0	0	2	0	0
	Pardosa leipoldti Purcell, 1903/ FGW	p	0	35	23	37	7
	Pardosa umtalica Purcell, 1903/ FGW	0	0	0	1	0	1
	Proevippa albiventris (Simon, 1898)/ FGW	0	1	287	296	199	106
	Proevippa fascicularis (Purcell, 1903)/ FGW	p	0	0	3	0	2

Family	Species/Functional group	Foord *et al.* 2002	New records: post-2005	Genera	Spp.	Vegetation type MG	MW	SF	TF	W
	Proevippa wanlessi (Russell-Smith, 1981)/ FPW	p				4	20	22	9	8
	Trabea heteroculata Strand, 1913/ FGW	0				2	89	93	88	16
	Trabea purcelli Roewer, 1951/ FGW	0				0	1	1	0	0
	Total			5	11	5(9)	6(452)	9(450)	5(337)	8(149)
Migidae ➤	† ##*Poecilomigas* sp. 1/ BPW	p				0	0	0	1	0
	Total			1	1	0	0	0	1(1)	0
Mimetidae	*Ero* sp. 2/ FPW	0				0	0	0	0	2
	‡ *Mimetus* sp. /FPW	p				0	0	0	0	0
	Mimetus cornutus Lawrence, 1947/ FPW	0				0	0	29	0	1
	Total			2	2	0	0	1(29)	0	2(3)
Mysmenidae	Undetermined sp.	p				0	0	0	0	0
	Total			0	0	0	0	0	0	0
Nemesiidae ➤		p				0	0	2	13	4
	Hermachama zoena Hewitt, 1915/ BPW	0				0	2	0	18	0
	Total			2	2	0	1(2)	1(2)	2(31)	1(4)
Nephiliidae	*Nephila fenestrate* Thorell, 1859/ OWB	p				0	0	2	69	3
	Nephila senegalensis (Walckenaer, 1842)/ OWB	0				0	1	0	13	0
	Total			2	2	0	1(1)	1(2)	2(82)	1(3)
Oecobiidae	*Uroecobius ecribellatus* 1976/RWB	p				0	0	0	0	0
	Total			0	0	0	0	0	0	0
Oonopidae	*Gamasomorpha* sp. 1	0				0	5	26	10	31
	Opopaea sp.		p							
	Total			1	1	0	1(5)	1(26)	1(10)	1(31)
Orsolobidae	*Afrilobus* sp.1	0				0	1	0	0	0
	Total			1	1	0	1(1)	0	0	0
Oxyopidae	*Hamataliwa fronticornis* (Lessert, 1927)/ FPW	0				0	3	4	0	5
	Hamataliwa kulczynskii (Lessert, 1915)/ FPW	p				0	0	1	0	0
	Oxyopes bedoti Lessert, 1915/ FPW	0				1	17	7	0	41
	*Oxyopes bothai*Lessert, 1915/ FPW		p							
	Oxyopes hoggi Lessert, 1915/ FPW	0				2	1	1	0	21
	Oxyopes jacksoni Lessert, 1915/ FPW	p				3	0	0	0	16
	Oxyopes longispinosus Lawrence, 1938/ FPW	p				1	5	3	9	21
	Oxyopes pallidecoloratus Strand, 1906/ FPW	p				0	7	0	0	8
	Oxyopes russoi Caporiacco, 1940/ FPW	0				6	7	1	1	38
	Oxyopess chenkeli Lessert, 1927/ FPW	0				1	0	0	1	9
	Oxyopes sp. 3/ FPW	0				0	1	0	0	1
	Oxyopes sp. 5/ FPW	0				1	0	0	0	0
	Oxyopes sp. 11/ FPW	0				0	1	1	0	0
	Oxyopes sp. 12/ FPW	0				1	0	0	0	0
	Peucetia viridis (Blackwall, 1858)/ FPW	p				0	4	1	0	6
	Total			3	14	8(16)	9(46)	8(19)	3(11)	10(166)
Palpimanidae	*Palpimanus transvaalicus* Simon, 1893/ FGW	p				1	6	18	8	10
	Total			1	1	1(1)	1(6)	1(18)	1(8)	1 (10)
Philodromidae	*Gephyrota* sp. 3/ FGW	0				0	0	0	0	1
	Philodromus browning Lawrence, 1952/ FGW	p				1	33	3	1	61

Family	Species/Functional group	Foord *Et al.* 2002	New records: post-2005	Genera	Spp.	Vegetation type				
						MG	MW	SF	TF	W
	Philodromus guineensis Millot, 1941/ FGW	0				2	44	14	2	51
	Suemus punctatus Lawrence, 1938/ FGW	p				0	3	0	0	2
	Tibellus minor Lessert, 1919/ FGW	p				0	0	0	0	14
	Tibellus sunetae Van den Berg & Dippenaar-Schoeman 1994/ FGW	0				0	2	0	0	11
	Total			4	6	2(3)	4(88)	2(35)	2(11)	6(150)
Pholcidae	‡ *Micropholcus* sp. 1/ SPWB	p				0	0	0	0	0
	Pholcus ciliates Lawrence, 1938/ SPWB	p				0	0	0	0	0
	Quamtana bonamanzi Huber, 2003/ SPWB	0				0	5	10	158	24
➢	#*Quamtana entabeni* Huber, 2003/ SPWB	0				0	0	4	6	0
➢	##*Quamtana lajuma* Huber, 2003/ SPWB	0				0	0	0	2	2
	Smeringopus natalensis Lawrence, 1947/ SPWB	p				0	0	0	0	0
	Smeringopus sp. 1/ SPWB	0				2	2	1	4	1
	Spermophora peninsulae Lawrence, 1964/ SPWB	p				0	0	0	0	0
	Total			2	4	1(2)	2(7)	3(15)	4(170)	3(27)
Phyxelididae	*Vidole sothoana* Griswold, 1990/ RWB	p				0	0	5	0	0
	Total			1	1	0	0	1(5)	0	0
Pisauridae	*Afropisaura rothiformis* (Strand, 1908)/ FWB	0				3	1	3	1	6
	Cispius problematicus Blandin, 1978/ FPW	p				0	0	0	0	0
	Euprosthenopsis pulchella (Pocock, 1902)/ FWB	p				0	0	0	0	0
	Total			1	1	1(3)	1(1)	1(8)	1(1)	1(6)
Prodidomidae	*Austrodomus* sp. 1/ FGW	p				0	0	0	0	0
	Theuma purcelli Tucker, 1923/ FGW	p				0	0	0	1	2
	Total			1	1	0	0	0	1(1)	1(2)
Salticidae	*Aelurillus* sp. 1/ GW	0				2	0	2	0	2
	Asemonea serrate Wesolowska, 2001/	0				0	0	1	0	0
	Asemonea sp. 1	0				0	0	7	18	0
	Baryphasah	0				9	7	12	2	27
	Brancus bevisi Lessert, 1925/ FPW	p				0	1	0	0	1
	Cosmophasis australis Simon, 1902/ GW	p				0	0	0	0	2
	Cosmophasis sp. 2/ GW	0				0	0	0	0	2
	Dendryphantes sp. 1/ F PW	0				1	2	6	7	2
	Festucula sp.(immature)	p				0	0	0	0	0
	Heliophanus debilis Simon, 1901/ F PW	0				15	2	4	8	18
	Heliophanus lesserti Wesolowska, 1986/ FPW		p							
	Heliophanus orchestra Simon, 1885/ FPW	p				0	1	3	4	1
	Hyllus argyrotoxus Simon, 1902/ FPW		p							
	Hyllus treleaveni Peckham & Peckham,1902PW	0				1	4	0	1	27
	Langona sp. 1		p							
	Marpissa sp. 1	0				0	1	0	0	3
	Myrmarachne ichneumon Simon, 1886	0				0	0	3	2	0
	Myrmarachne marshalli Peckham & Peckham, 1903	0				0	0	0	0	3
	Myrmarachne sp. 1	0				2	8	0	3	12
	Natta horizontalis Karsch, 1879/ GW	p				0	0	0	0	0
	Pachyballus sp. 1		p							
	Pachyballus transversus Simon, 1900	p				0	0	0	0	0
	Phlegra sp. 1/ GW	0				0	1	5	29	4
	Portia sp. 1	0				0	6	25	1	20
	Rhene machadoi Berland & Millot, 1941/ FPW	0				1	0	0	0	2
	Salticidae (undetermined genus) sp. 3	0				0	0	1	0	0
	Stenaelurillus sp. 1/ GW	p				3	0	0	0	2
	Thyene coccineovittata (Simon, 1885) / FPW	0				8	13	5	4	12
	Thyene dakarensis Lawrence, 1927/ FPW	0				2	0	0	0	1
	Thyene inflate (Gerstaecker, 1875) / FPW	p				5	2	1	21	5
	Thyene natalii (Peckham & Peckham, 1903)	0				0	3	8	0	7

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						MG	MW	SF	TF	W
	Thyenula aurantiaca (Simon, 1902) / FPW	p				28	31	49	15	110
	Thyenula sempiterna Wesolowska, 2000FPW		p							
	Total			19	27	13(81)	16(99)	17(157)	15(171)	23(290)
Scytodidae	*Scytodes clavata* Benoit, 1965 /FGW	0				0	0	16	105	1
	Scytodes fusca Walckenaer, 1837/ FGW	p				0	0	0	0	0
	Scytodes maritime Lawrence, 1938 / FGW	0				0	0	1	9	0
	Scytodes sp. 1/ FGW	p				0	0	0	0	0
	Scytodes sp. 2/ FGW	0				0	0	4	0	4
	Total			1	4	0	0	4(21)	3(114)	2(5)
Segestriidae	*Ariadna* sp .1/ RWB	0				1	1	1	1	2
	Total			1	1	1(1)	1(1)	1(1)	1(1)	1(2)
Selenopidae	*Anyphops* sp. 1 /FGW	0				1	0	4	83	2
	Selenops brachycephalus Lawrence, 1940/ FGW	p				0	0	10	163	1
	Selenops tenebrosus Lawrence, 1940/	p				0	0	0	0	0
	Selenops zuluanus Lawrence, 1940/ FGW	p				0	0	0	0	0
	Total			2	3	1(2)	1(1)	2(15)	3(247)	2 (5)
Sicariidae	*Loxosceles spiniceps* Lawrence, 1952 /FGW	p				0	0	0	2	0
	Total			1	1	0	0	0	1(2)	0
Sparassidae	*Olios*sp. 1/ FPW	0				0	0	2	0	5
	*Olios*sp. 2/ FPW	0				0	1	0	0	4
	*Olios*sp. 3/ FPW	0				0	3	0	0	3
	Palystes Johnstoni Pocock, 1896/ FPW	p				0	0	0	1	0
	Total			2	4	0	2(4)	1(2)	1(3)	3(12)
Tetragnathidae										
	Leucauge decorata (Blackwall, 1864)/ OWB	p								
	Leucauge festiva (Blackwall, 1866)/ OWB	p				1	0	1	0	0
	Leucauge levanderi (Kulczynski, 1901)/ OWB	p								
	Leucauge thomeensis Kraus, 1960/ OWB	0				0	0	9	77	1
	Pachygnatha sp. 1	p								
	Tetragnatha subsquamata Okuma, 1985/ OWB	0				0	0	9	49	0
	Total			6	7	2(2)	1(5)	5(69)	4(401)	2(5)
Theraphosidae	*Ceratogyrus bechuanicus* Purcell, 1902/ BGW	p				0	0	0	0	0
	Harpactirella flavipilosa Lawrence, 1936/ BGW	p				0	1	0	0	0
	Harpactira sp.1		p							
	Total			1	1	0	1(1)	0	0	0
	Argyrodes sp. 1/ GWB	0				0	1	13	13	0
	Argyrodes sp. 2/ GWB=*Rhomphaeanasica* (Simon, 1873)	0				2	4	11	8	4
	Chrysso sp. 10/ GWB	0				0	0	0	1	0
	Coleosoma sp. 1/ GWB	0				0	1	3	17	4
	Coscinida??sp. 1/ GWB	0				0	0	0	2	2
	Coscinida tibialis Simon, 1895/ GWB	p				1	6	8	15	0
	Crustulina sp. 1/ GWB	p				0	0	0	0	0
	Dipoena sp. 4/ GWB	0				0	0	0	3	1
	Dipoena sp. 7/ GWB	0				5	6	32	32	4
	Dipoenura sp. 1/ GWB	0				0	0	0	0	5
	Euryopis sp. 4/ G WB	0				0	0	0	0	9
	Euryopis sp. 5/ GWB	0				0	0	8	2	8
	Latrodectus geometricus C.L.Koch, 1841/ GWB	p				0	0	0	0	0
	Latrodectus renovulvatus Dahl, 1902		p							
	Phoroncidia sp. 2/ GWB	0				0	0	0	1	2
	Steatoda sp. 20/ GWB	p				0	0	2	0	0
	Steatoda sp. 6/ GWB	0				0	0	1	0	0
	Theridiidae sp. 22(genus undetermined)/ GWB	0				0	20	3	20	1
	Theridiidae sp. 23(genus undetermined)/ GWB	0				0	0	4	0	1

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	Theridiidae sp. 28(genus undetermined)/ GWB	0				0	0	3	2	2
	Theridion purcelli O.P.-Cambridge, 1904/ GWB	0				0	0	1	0	0
	Theridion sp. 1/ GWB	0				0	3	87	0	4
	Theridion sp. 3/ GWB	0				1	11	36	122	17
	Theridion sp. 11/ GWB	0				0	7	4	9	15
	Theridion sp. 12/ GWB	0				0	0	6	2	2
	Theridion sp. 13/ GWB	0				5	48	72	29	27
	Theridion sp. 14/ GWB	0				0	11	28	10	11
	Theridion sp. 15/ GWB	0				3	7	1	6	1
	Theridion sp. 16/ GWB	0				1	8	25	23	31
	Theridion sp. 18/ GWB	0				0	9	22	14	9
	Theridion sp. 21/ GWB	0				0	0	1	0	0
	Theridion sp. 30/ GWB	0				0	1	9	0	2
	Theridion sp. 31/ GWB	0				0	1	0	0	0
	Tidarren cuneolatum (Tullgren, 1910)/ GWB	p				1	8	1	5	4
	Total			13	36	14(19)	24(171)	23(393)	16(346)	28(170)
Trachelidae	*Cetonana simoni*, Lawrence, 1942/ FGW	p				0	16	14	30	9
	Thysanina transversa, Lyle & Haddad 2006/ FGW	0				0	1	3	5	1
	Trachelia sp. 1/ FGW	0				0	0	0	0	2
	Trachelia sp. 2/ FGW	0				0	1	0	0	3
	Trachelia sp. 3/ FGW	0				0	1	0	0	3
	Trachelinae sp. 1/ FGW	p				0	0	0	0	0
Total				3	5	0	4(19)	2(17)	2(35)	5(18)
Thomisidae	*Ansiea tuckeri* (Lessert, 1919)/ FPW	0				1	2	1	1	1
	Diaea puncta Karsch, 1884/ FPW	p				0	12	18	3	109
	‡ *Heriaes ynaema* sp. (new record SA)/ FPW	0				0	1	0	0	0
	Heriaeus crassispinus Lawrence, 1942/ FPW	0				0	1	2	1	2
	Heriaeus fimbriatus Lawrence, 1942/ FPW	p				1	5	18	6	71
	Heriaeus transvaalicus Simon, 1895	p								
	Heterogriffus sp .1/ FPW	0				0	1	0	0	0
	Misumenops rubrodecoratus Millot, 1941/ FPW	p				1	5	1	0	14
	Monaeses austrinus Simon, 1910/ FPW	p				0	0	0	0	4
	Oxytate argenteooculata (Simon, 1886)/ FPW	p				0	10	9	10	4
	Oxytate concolor (Caporiacco, 1947)/ FPW	p				0	16	1	1	0
	Oxytate phaenopomatiformis / FPW	0				0	1	0	0	0
	Oxytate ribes (Jézéquel, 1964)/ FPW	0				1	4	4	1	10
	Pactates compactes Lawrence, 1947/ FPW	0				0	0	0	0	1
	Pherecydes nicolaasi Dippenaar-Schoeman, 1980/ FPW	0				0	0	5	1	3
	Runcinia aethiops Simon, 1901/ FPW	p				1	4	0	0	15
	Runcinia flavida Simon, 1881/ FPW	p				3	0	0	0	27
	Simorcus zuluanus Lawrence, 1942/ FPW	0				1	0	1	0	12
	Smodicinus coroniger Simon, 1895	p								
	Synema decens (Karsch, 1878)	p								
	Synema diana (Audouin, 1826)	0				0	0	1	0	0
	Synema imitator (Pavesi, 1883)/ FPW	p				1	2	21	19	11
	Synema langheldi Dahl, 1907 (first male)/ FPW	0				0	2	16	1	2
	Synema nigrotibiale Lessert, 1919/ FPW	0				1	1	2	0	2
	Synema sp. 2/ FPW	0				0	2	1	0	16
	Synema vallotoni Lessert, 1923(first male)/FPW	0				0	0	4	0	2
	Thomisops bullatus Simon, 1895/ FPW	0				0	3	0	0	1
	Thomisops pupa Karsch, 1879/ FPW	p				0	0	2	0	0
	Thomisus dalmasi Lessert, 1919/ FPW	p								
	Thomisus daradiodes Simon, 1890/ FPW	p				1	0	0	0	1

	Thomisus granulatus Karsch, 1880/ FPW	p				1	0	1	1	3
	Thomisus kalaharinus Lawrence, 1936/ FPW	p				1	2	0	1	0
	Thomisus scrupeus (Simon, 1886)/ FPW	0				0	5	0	0	12
	Thomisus spiculosus Pocock, 1901/ FPW	0				0	0	0	0	16
	Tmarus africanus Lessert, 1919/ FPW	0				0	6	0	0	1
	Tmarus cameliformis Millot, 1942/ FPW	p				2	7	37	54	54
	Tmarus natalensis Lessert, 1925/ FPW	0				0	0	1	0	0
	Tmarus planetariums Simon, 1903/ FPW	0				0	0	0	1	0
	*Tmarus riccii*Caporiacco, 1941/ FPW	0				0	0	1	0	0
	†*Tmarus* sp. 1/ FPW	0				0	2	23	17	29
	Xysticus natalensis Lawrence, 1938/ FPW	p				0	0	0	0	2
	Total			17	39	14(17)	24(105)	23(175)	16(122)	27(439)
Trochanteriidae	*Platyoides walteri* (Karsch, 1886)/ FPW	p				0	0	3	0	1
	Total			1	1	0	0	1(3)	0	1(1)

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Uloboridae	*Hyptiotes akermani* Wiehle, 1964/ OWB	0				0	0	14	10	1
	Miagrammopes longicaudatus O.P-.Cambridge, 1882/ OWB	P				0	11	37	10	17
	Miagrammopes sp. 2/ OWB	0				0	0	0	1	0
	Uloborus lugubris Berland, 1939/ OWB	P				0	5	156	85	3
	*Uloborus plumipes*Lucas, 1845/ OWB	P				1	2	27	11	7
	Uloborus sp. 4/ OWB	0				0	0	11	5	0
	Total			3	8	1(1)	3(18)	4(248)	5(122)	3(29)
Zodariidae➤†##	*Australutica africana*/ FGW	0				0	0	0	0	5
	Caesetius sp. 1/ FGW	0				9	0	5	0	9
	Caesetius sp. 2/ FGW	0				2	0	0	0	3
	†*Cyrioctea* sp. 1/ FGW	0				0	0	1	0	0
	Diores auricular Tucker, 1920/ FGW	p				0	0	0	0	0
	‡ *Diores* sp. 1/ FGW	0				0	1	0	0	1
	‡*Psammoduon*(undetermined sp.)/ FGW	p				0	0	0	0	0
	Ranops sp. 1/ FGW	P								
	Thaumastochilus sp. 1/ FGW	0				0	0	10	2	4
	Total			5	8	2(11)	1(1)	2(16)	1(2)	4(22)
	Grandtotal			**156**	**277**	**95** (274)	**173** (1742)	**167** (2411)	**142** (2715)	**198** (2665)