Spider ecology in southwestern Zimbabwe, with emphasis on the impact of holistic planned grazing practices

Sicelo Sebata

Thesis submitted in satisfaction of the requirements for the degree Philosophiae Doctor in the Department of Zoology and Entomology, Faculty of Natural and Agricultural Sciences, University of the Free State

January 2020

Supervisors

Prof. Charles R. Haddad (PhD): Associate Professor: Department of Zoology and Entomology, University of the Free State, P.O. Box 339, Bloemfontein 9300, South Africa.

Prof. Stefan H. Foord (PhD): Professor: Department of Zoology, School of Mathematics and Natural Sciences, University of Venda, Private Bag X5050, Thohoyandou 0950, South Africa.

Dr. Moira J FitzPatrick (PhD): Regional Director: Natural History Museums of Zimbabwe, cnr Park Road and Leopold Takawira Avenue, Centenary Park Suburbs, Bulawayo, Zimbabwe.

STUDENT DECLARATION

I, the undersigned, hereby assert that the work included in this thesis is my own original work and that I have not beforehand in its totality or in part submitted it at any university for a degree. I also relinquish copyright of the thesis in favour of the University of the Free State.

S. Sebata   **31 January 2020**

SUPERVISOR DECLARATION

UNIVERSITY OF THE
FREE STATE
UNIVERSITEIT VAN DIE
VRYSTAAT
YUNIVESITHI YA
FREISTATA

UFS·UV

NATURAL AND
AGRICULTURAL SCIENCES
NATUUR- EN
LANDBOUWETENSKAPPE

Department of Zoology & Entomology

P.O. Box 339
Internal Box 49
Bloemfontein
9300

Republic of South Africa
E-mail: haddadcr@ufs.ac.za

☎: +27-(0)51-401-2568
🖷: +27-(0)51-401-9950

2020-01-31

To Whom It May Concern:

I, Prof Charles Richard Haddad, and on behalf of the co-supervisors, hereby approve the submission of the PhD thesis of Mrs Sicelo Sebata for assessment, and I confirm that the submitted work has not previously, either in part or in its entirety, been submitted to the examiners or moderators.

Yours truly,

Prof. Charles Haddad
Associate Professor: Entomology

DEDICATION

I would like to dedicate this thesis to all the spiders that lost their lives in the name of Science.

ABSTRACT

The current information on Zimbabwean spiders is fairly poor and is mostly restricted to taxonomic descriptions, while their ecology remains largely unknown. While taxonomic studies are very important, as many species are becoming extinct before they are described, a focus on the ecology of spiders is also essential, as it helps with addressing vital questions such as the effect of anthropogenic activities on spider fauna. Therefore, in order to address this research gap, assessment of the response of spiders to holistic management practises within Debshan Ranch, Shangani, Zimbabwe was done. Additionally, in order to establish baseline data on spider fauna, the standardised South African National Survey of Arachnida sampling protocol was utilised to assess its efficacy within the Khami World Heritage Site.

Spider sampling was done in three sub-projects: the first included sampling in several geographic distances around previously kraaled inclusions and control sites within the ranch, using sweep nets and pitfall traps, in six sampling periods from July 2017 until April 2018; the second included sampling that was done inside the previously kraaled inclusions and their surrounding areas dating back to at least ten months since cattle occupation in two sampling intervals early summer (November 2017) and late summer (March 2018), using pitfall traps; the third entailed sampling within Khami in three sampling periods (summer, winter and spring 2018) using six sampling methods, namely pitfall traps, beating sheets, litter sifting, sweeping, day hand collecting and night hand collecting.

The model that best explained changes in mean grass height (cm), as well as percentage grass cover around previously kraaled inclusions and the control sites, was that which included time since kraal removal, whereas inside the inclusions and their surroundings was that which included season and short duration kraaling. At the functional group level, only the web builder's genera richness responded negatively to short duration kraaling around the previously kraaled inclusions and their control sites. On the other hand, inside the previously kraaled inclusions and their surroundings only ground dwelling abundance responded negatively and significantly to short duration kraaling. The most important predictor amongst the vegetation structure variables around the previously kraaled inclusions and control sites was mean grass height (cm), which impacted genera richness and abundance of both ground dwellers and web builders. In contrast, genera richness and abundance of plant wanderers were positively associated with mean grass

height (cm). However, inside the previously kraaled inclusions and their surroundings the most important predictor was time since kraal removal and the vegetation structure variable percentage coarse woody debris cover, which responded positively to the ground dwellers. In addition, the late sampling season had significantly lower ground dwelling abundance compared to the early sampling season. Within the Khami World Heritage Site the riparian woodland had the highest species richness compared to the other biotopes. Similarly, the summer period also produced the highest diversity, with winter recording the lowest species richness. Night and day hand collecting had the highest observed species richness with adult individuals. In order to sample 50% of the spider assemblages, 15 samples were required to be collected in the mixed woodland, which represented the biotope requiring the fewest samples.

Seasonality effects explained a significant amount of variation in changes of mean grass height (cm) and percentage cover around previously kraaled sites and their control sites. However, when inter-seasonal variation was excluded by sampling previously kraaled sites within one season, short duration kraaling explained a significant amount of variation. Standardised sampling protocols aid in establishing databases of spider fauna which will in the long run ensure inclusion of spiders in biodiversity reports in Zimbabwe, which has historically not been the case, due to limited information.

Keywords: checklists, short duration kraaling, standardised sampling protocol, optimization, and Debshan Ranch

ACKNOWLEDGEMENTS

Many daughters have done exceedingly well but you exceed them all.

Proverbs 31 v 29

CONTENTS

CHAPTER 5: GENERAL DISCUSSION, CONCLUDING REMARKS AND RECOMMENDATIONS

Chapter 1

1.0 Introduction and Literature review

1.1 Araneae research in Zimbabwe

Spider research in Zimbabwe is a developing field, with the opening of the Department of Arachnology at the Natural History Museum of Zimbabwe (NHMZ) only occurring during the mid-1980s (National Museums and Monuments Zimbabwe 2015a). The establishment of the department was mainly due to the focus on spiders by the first curator of Invertebrates, Mrs Cathy Car in 1977, whose material built up the initial Arachnological department collections. In addition, during this period random and sporadic collecting of spiders within Zimbabwe was usually done by private collectors such as Reay Smithers, who donated most of his collections before leaving the country in 1978 (National Museums and Monuments Zimbabwe 2015a).

The first curator of Arachnids in Zimbabwe was Mrs Jacqueline Minshull, who was appointed curator in 1982. She built up the majority of the collection mainly through field trips around Zimbabwe, with a few specimens from neighboring countries. Subsequently, in 1992 Dr. Moira FitzPatrick took over, and has since been involved in research on the biogeography and natural history of spiders and scorpions, and taxonomic descriptions of mostly ground spiders (Gnaphosidae). The collection of Arachnids is the youngest in the Museum and houses over 150 000 specimen lots, with over 250 Holotypes and 100 Paratypes (National Museums and Monuments Zimbabwe 2015a).

The current information of the Zimbabwean spider fauna is fairly poor and is mostly restricted to taxonomic descriptions, while its ecology remains fairly unknown. Published records show that research on the spider fauna in Zimbabwe has focused on checklists (FitzPatrick 2001, Wesołowska & Cumming 2011, FitzPatrick & Dube 2018), urban diversity (Wesołowska & Cumming 2008), diversity within protected areas (Wesołowska & Cumming 2011, Sebata 2015, Sebata *et al.* 2015), natural history (Jocque & Dippenaar-Schoeman 1992, Wesołowska & Cumming 1999,2002, 2008), and taxonomic descriptions (FitzPatrick 1994, Wesolowska 1999a, FitzPatrick 2007, 2009, FitzPatrick & Sebata 2018). Taxonomic research has not been done within Zimbabwe only, as specimens contained within the national collection have been included in regional taxonomic revisions and faunistic papers, such as Jocque (1990), Lotz(1994, 2007a,

2007b) Wesolowska (1999b), Haddad & Wesołowska (2006), Haddad & Lyle (2008), Fourie *et al*. (2011), and Haddad & Mbo (2015) amongst others.

Furthermore, specimens contained in regional and international museums such as the KwaZulu-Natal Museum, Pietermaritzburg, South Africa (e.g. Hewitt 1915), the Royal Museum for Central Africa (MRAC) in Tervuren, Belgium (e.g. Lawrence 1940), Iziko South African Museum, Cape Town, South Africa (e.g. Tucker 1923), and the British Museum of Natural History, London, U.K. (e.g. Pocock 1901, Hyatt 1954) have been included in taxonomic descriptions and revisions, which have also augmented data on the spider fauna of Zimbabwe.

Apart from work done by Chari (2011) on the influence of large herbivores and vegetative termitaria on spider diversity in miombo woodlands, and that of Cumming & Wesolowska (2004) on habitat separation by jumping spiders in a suburban area in Harare, research focusing on ecology in Zimbabwe has largely remained unexplored. While taxonomic studies are important, as many species are becoming extinct before they are described (Costello 2015), there is also an essential need to focus on the ecology of spiders that will help in addressing some of the most essential questions on their role in terrestrial ecosystems and the impact that human activities have on their survival and functional significance.

In order to enable the inclusion of spiders into conservation programmes, there is need for correct and regularly updated checklists. For example, the South African National Survey of Arachnida inventories have enabled the production of a number of checklists (e.g. Foord *et al*. 2002, 2016, Wassenaar 2006, Haddad & Dippenaar-Schoeman 2009), which have expedited the inclusion of South African spiders for the first time ever in the National Spatial Biodiversity Assessment (NSBA) (Dippenaar-Schoeman *et al.* 2015) in 2010. However, in Zimbabwe, partial surveys mainly undertaken by private researchers and museum taxonomists have contributed limited checklists (Wesołowska & Cumming 2011, FitzPatrick & Dube 2018), which regrettably have not yet been included into any Government of Zimbabwe National Reports on biodiversity or conservation programmes.

There is therefore a critical need to increase the interest on these commonly ignored organisms in order to escalate research on spiders, as they are also worthy of protection. Spider research also enables the identification of species that are already receiving protection within protected areas and those that require conservation (Balmford & Gaston 1999). This assists in the development

of Red Data Lists for arachnids (Dippenaar-Schoeman *et al.* 2015). In order to understand the distributions and the diversity of the spider fauna, inventories should be conducted in all floral biomes using a variety of sampling techniques, which will enable researchers to determine endemic and threatened species (Dippenaar-Schoeman *et al.* 2015).

1.2 Spider species richness in Zimbabwe

The species richness of spiders known so far in Zimbabwe translates to 0.72% of the current global species richness (World Spider Catalog 2020). According to Jocqué *et al*. (2013) 349 species of spiders are documented in the country. This is far below the 2170 species recorded from South Africa (Dippenaar-Schoeman *et al.* 2015), 722 species recorded in Tanzania and the 533 from Kenya (Jocque *et al.* 2013). In contrast, it is higher than that of the 250 species of Botswana and 183 of Malawi (Jocque *et al.* 2013). Generally, the species richness of Zimbabwean spiders may be regarded as low in relation to the rest of the Afrotropical region, regardless of the fact that it is amongst the top ten countries in its species richness (Jocqué *et al.* 2013). This may be attributed to the underutilization of the Natural History Museum Zimbabwe collections by taxonomists, which is reflected by the bias towards few families that have been previously described. In addition vast areas of the country are still poorly sampled thus limiting the knowledge on distribution records and also limits identification of endemic species.

According to Jocqué *et al*. (2013) the documented species richness of each country is a reflection of the effort that is placed on the study of spiders rather than the size of the country, mainly because countries with approximately similar sizes have recorded very contrasting levels of species richness. Information on species richness of organisms is quite significant in conservation planning; efforts to consolidate knowledge on species have been initiated at a global level, by various projects, e.g. the Global Biodiversity Information Facility (GBIF). In Zimbabwe, checklists of spiders of the Matopos National park have been included into such databases (FitzPatrick & Dube 2018). Despite the effort of placing all known published records into such databases, the knowledge of all species is still relatively poor (Cardoso 2009).

The apparently low species richness of Zimbabwe was also explained by FitzPatrick (2001) who cited the presence of above 13 000 specimen lots of Zimbabwean spiders contained within the Natural History Museums collections that had been curated since the 1960s but still awaits identification. According to the National Museums and Monuments website (2015b), spider

species richness of Zimbabwe may reach at least 6000 species. So as to enhance the information of biodiversity, besides the collection of published records, several activities should be initiated, including surveying and monitoring invertebrates, examination of unidentified material in museums, and engagement of the public in order to increase awareness of the spider fauna, thus encouraging research on spiders (Dippenaar-Schoeman *et al*. 2015), which will most likely increase the species richness of spiders known from Zimbabwe.

Warui *et al.* (2004) also argues that the limitation of spider identifications to species level due to taxonomic impediments limits the possibility of bio geographical comparisons among studies. This can be seen for example, in a study done by Muvengwi *et al.* (2018) that focused on the abundance and diversity of macro-invertebrates on previously kraals sites in a semi-arid savanna in Zimbabwe, and that of Mashavakure *et al.* (2019) on the response of spiders under different tillage systems in Zimbabwe. Araneae were among the invertebrates reported on, but identification was done only to family resolution, therefore limiting their contribution to spider knowledge within the country. Another point to consider when comparing species richness among studies is the issue of utilising standardised sampling protocols in various surveys and studies across the globe, as comparison of studies that have utilised different sampling methods becomes more difficult (Dippenaar-Schoeman *et al.* 2015).

1.3 Spider ecology

Spider ecology is a wide-ranging issue that consists of feeding and reproductive ecology, dispersal, growth, survival, as well as the effects of spiders on the environment that they live in (Ramel 2020). Feeding ecology focuses on how spiders consume their prey. In general, the majority of spiders feed the same manner, with the narrow gut of spiders only able to accept liquid food, and solid food being kept out by two sets of filters (Ramel 2020). External digestion occurs in one of two forms, either by the spider pumping digestive juices from the gut into the prey, with the liquefied tissues of the prey sucked into the gut with the empty husk left behind. The second form of external digestion involves holding pulp, which is masticated and finely ground prey material, held in a pre-oral cavity formed by the chelicerae and the bases of the pedipalps (Turnbull 1973).

Hunting strategies are also a component of feeding ecology, which has received attention from most ecologists (Turnbull 1960). In general, spider families were initially grouped into two broad

groups based on their hunting strategies: the sedentary type that employ suspended silk in a permanent station (a web), and spiders that forego utilisation of a snare but range over the substrate in search of prey (Turnbull 1973). Webs are quite essential, especially for spider families that utilise them as their hunting ground, as they (i) provide early warning signs of dangers, thus forming a protective barrier from dangerous invaders; (ii) provide filters that intercept prey beyond the range of spiders' perception; and (iii) they place the prey in a place of disadvantage, thus enhancing the spider's attack efficiency. Prey-capturing strategies vary with type of web structure and spider behaviour. Generally, for web-builders prey attack usually occurs when a potential prey enters a web and alerts the spider of its presence by vibrations and stresses set up upon the web as it tries to escape.

Depending on the type of web, the spider usually approaches the prey in a leisurely and cautious fashion if the web contains adhesive qualities (Szlep 1961, Friedrich & Langer 1969, Eberhard 1971). However, when the web has no such qualities the spider usually has to act promptly in order to avoid prey escape (Turnbull 1973). For the majority of spiders except the Uloboridae and Heptathelidae (Kaston 1948), the prey is subdued by the emission of venom that usually paralyses the prey, sometimes with the spider casting silk over the prey thereafter (depending on the spider, silk sometimes is cast over the prey before emission of the venom) until its struggle subsides. The prey is either consumed on the spot or carried to a special station in the web and consumed there. These attack procedures by spiders are usually efficient, but not all prey are attacked with the same vigour and some may escape (Bristowe 1939-1941, 1958, Eberhard 1967).

According to Kajak (1965), substantial differences occur between the preys potentially available versus the prey actually captured. Potential prey differs with the type of web, but generally they should be organisms of appropriate size, they must possess surface characteristics that make them vulnerable to ensnarement by the web, and they must move in an appropriate fashion through the spaces occupied by the webs (Turnbull 1973). Furthermore, position of webs is determined by suitable microclimatic conditions to meet the physiological needs of the spider, provide a framework for web construction, and yield appropriate numbers of prey (Turnbull 1973, Alderweireldt 1994).

Despite the usefulness of webs to web-builders, insects have also developed several defence mechanisms against spider webs, such as (i) the Syrphidae that can that can perceive and avoid aerial webs; (ii) the Buprestidae and the Coccinellidae that have body forms and surfaces that minimise chances of ensnarement by webs; (iii) the Vespidae and the Bombidae that have weaponry or colouration that intimidates some spiders; (iv) the Pentatomidae and Corscidae that produce exudates that repel some spiders; as well as (v) the Scarabaeidae that are powerful enough to break through the webs (Turnbull 1973). Prey selection by spiders is also a component of feeding ecology. Spiders have been reported to be able to feed on almost all kinds of flies, earwigs, butterflies, moths, wasps, bees, woodlice, harvestmen, ants, beetles, as well as other spiders (Savory 1928). However, rejection of certain invertebrates by spiders has also been reported to occur (Bristowe 1939-1941). The latter has been attributed to the physiological state of the spider when it encounters the prey, and not necessarily as a result of the kind and quality of the prey (Bristowe 1939-1941).

Hunting spiders do not have to wait for the prey to be caught in a web, but rather venture to search for desirable prey. Several methods can be utilised, including ambushing, which is usually observed in the genus *Misumena* (Thomisidae) that lies awaiting in flowers for insects that seek pollen and nectar. The prey is drawn towards the venomous fangs when it moves within the vicinity of the long drawn out raptorial forelegs of the spider. Some ambushers have been reported to have limited abilities to change their colour in order to conform to colour of a chosen blossom (Chew 1961, Gabritschersley 1927). Other families such as the Gnaphosidae are active runners that pursue and overpower small prey (Haynes and Sisojenic 1966).

Jumping spiders (Salticidae) wander over surfaces and foliage, searching with powerful eyes for appropriate prey (Gardiner 1965, Phanael 1967). Despite the prey being perceived within several centimetres, the gap between the prey and the spider is reduced to a few millimetres by the creeping spider that leaps forward rapidly, seizing the prey with its jaws and injecting the venom, hanging on until the prey struggle ceases. Rapid runners include families such as the Lycosidae and Pisauridae, which usually have good eyesight and orient their prey to be within the range of vision of the two large frontal eyes, and charge forward to subdue their prey in a similar manner to that of jumping spiders. Some species of *Dolomedes* that live near water are able to remain under water over 30 minutes, and prey on aquatic insect larvae and even small fish (Vogel 1965).

Other less studied strategies are those of the short-sighted night hunters of the families Gnaphosidae and Clubionidae, for example.

Reproductive ecology is the science that deals with how spiders procreate. In general, spiders reproduce sexually. Females emit a sex pheromone attractive to males. In order to avoid being consumed as prey, most males have developed courtship rituals. In most cases, a single mating facilitates the fertilization of several batches of eggs produced within several weeks or months (Ramel 2020). The development of spiders focuses on the ontogeny of these invertebrates. They develop by going through a set of stages usually followed by a moult of the integument (Turnbull 1973). The juvenile is similar in form to the parent, but varies in spination, proportion of parts and colouration. The completion of the sexual organs marks the final moult, which brings the most significant change in spiders (Turnbull 1973). Moulting is usually a susceptible period for the spiders, as they are prone to predation because they are usually incapable of escaping or helping themselves.

Spider dispersal and movement deals with how spiders move from one place to another. Dispersal normally occurs by the process known as ballooning, with journeys spanning a distance of a few yards or many miles (Turnbull 1973). It is most common in juveniles, as it aids in division of family groups, thus avoiding overcrowding and cannibalism. Silk bridges are mostly utilised by web-building spiders, whereas hunting spiders' major mode of movement is through walking, with adult wolf spiders such as *Pardosa monticola* covering at most straight distances of almost 100 m over a lifetime (Bonte *et al.* 2003), with female natal dispersal of between 30-40 m of straight distances per day (Bonte *et al.* 2007).

Spider survival and mortality focuses on how spiders evade death through predation or other factors. Mortality rates of spiders are not known, but mortality factors include ballooning mishaps, death during moults, starvation, and predation by birds, rodents, insect parasites and predatory wasps. According to Gunnarson (1983), larger spiders (> 2.5 mm) seem to be more prone to predation by birds than smaller spiders. However, at higher spider densities larger spiders seem to survive better than smaller ones. Adverse weather such as cold temperatures has been reported to be also responsible for spider deaths, despite the presence of cold resistant spiders that survive the Arctic winters (Turnbull 1973). Spider survival is enhanced by factors such as mimicry seen in ant-like spiders (Wesolowska & Szeremeta 2001) and cryptic

colouration that tends to avoid predation, as well as change in colour by web-spinners that usually mimic the colour of the soil detritus upon disturbance from its web when evading potential predators.

Population and community ecology is another aspect that ecologists have focused on, with many authors attempting to determine the densities of spiders in natural and modified biotopes. Attempts to describe the structure of spider communities with explanations of perceived patterns have been commonly approached through the utilisations of species lists (FitzPatrick 2001, Dippenaar-Schoeman & Wassenaar 2002, Warui *et al*. 2004, Foord *et al.* 2016), which at times are accompanied by notes on the taxonomy, morphology, biotope and behaviour. These lists are essential, as they improve the knowledge on species distribution and morphological variation (Turnbull 1973). Usually such lists are mostly useful to taxonomists, although ecologists may attempt to understand the changes in numbers over space and time using several sampling methods (Turnbull 1973). In addition, several environmental parameters may also be measured with the intention to relate the kinds and numbers of spiders to these parameters (Turnbull 1973).

1.4 Ecological and economic importance of spiders

Spiders rank seventh in global diversity (Coddington & Levi 1991), with roughly 48 438 described species (World Spider Catalog 2020). They are amongst the most abundant organisms that are easy to collect, have short life cycles (Coddington *et al.* 1991), and are suitable indicators of disturbance (Marc *et al*. 1999, Ford *et al*. 2013). They are found in almost all types of biotopes (Turnbull 1973). Spiders are essential predators in all terrestrial ecosystems (Dippenaar-Schoeman 1998, 2001), feeding on diverse organisms that include bats (Nyffeler & Knornschild 2013), fish (Nyffeler & Pusey 2014) lizards and frogs (Nyffeler *et al*. 2017) amongst others. They also control natural populations of insects (Nyffeler & Birkhofer 2017), pests (Hoefler *et al.* 2006, Michalko *et al*. 2018), feeding on insects that generally infest homes such as cockroaches and mosquitoes (Nelson & Jackson 2006, Ndava *et al.* 2018). They are also a source of food to various predators that include birds (Peterson *et al.* 1989), snakes (Marques *et al.* 2006), as well as arachnids (Elgar & Fahey 1996, Wilder & Rypstra 2008).

Spider silk might be an integral part of the economy as it has chemical and biomedical properties (Eisoldt *et al.* 2011). It is also the strongest natural material, which has enabled habitation of a unique niche by the riverine spider *Caerostris darwini* (Agnarsson *et al.* 2010). Some families

contain species that have the potential to cause serious injury to humans, with some species being lethal (Hauke & Herzig 2017). Due to inadequate taxonomic and geographic distributional data (Muelelwa *et al.* 2010) and the generally poor interest in spiders, spiders have conservatively obtained interest from conservation professionals and the general public. Considering that spider diversity is remarkable in its own right, spiders are worthy of research and protection.

1.5 Standardized and optimised sampling

Inventories are mostly conducted in order to determine the distribution and composition of the flora and fauna in areas where such information is not known, whereas monitoring seeks to enable the appreciation of the trends or effects of management practises on such populations and habitations (Morrison *et al.* 2008). In biodiversity monitoring, standardised and regular repeated measurements of each biome and biota is usually recommended, which Teder *et al.* (2007) argues is lacking in most countries. A standardised sampling protocol is one which enables comparability of data when it is applied to sites of the same biotope, whereas an optimized protocol seeks to distribute the number of samples between methods in order to estimate the maximum possible species and species assemblages with minimum effort (Malumbres-Olarte *et al.* 2016). Optimised and standardised sampling has been shown to be more reliable than *ad hoc* sampling (Cardoso *et al.* 2009a) and can be utilised as an alternative to species richness estimators (Cardoso 2009).

One of the initial studies that endeavoured to estimate species richness of spiders was the first design of a sampling protocol that was tested in tropical forests Coddington *et al.* (1991). They proposed this sampling protocol as a result of the discrepancies that were found amongst collecting efforts between systematics and ecologists. The collecting efforts of museum taxonomic staff were efficient in representing local species richness, but were difficult to determine statistically, whereas that of ecologists were usually not representative of the total fauna. In their study, they were able to suggest a sampling protocol that was expected to be able to allow for comparability between studies in different areas of the world. The concept of the methodology involved the production of replicate samples (Sørensen *et al.* 2002), with the sampling effort standardised by one hour of collecting, employing an array of methods that were selected to sample various microhabitats. Since sampling protocols are required to obtain species

richness from all possible microhabitats without any bias, Coddington *et al.* (1996) recommended inclusion of pitfall traps and litter sifting, as the sampling protocol of Coddington *et al.* (1991) was reported to under-sample litter fauna.

Since Coddington *et al.*'s (1991) sampling protocol design, various studies have utilised this format in order to estimate species richness in several parts of the world, such as in the temperate regions (Coddington *et al.* 1996, Dobyns 1997, Toti *et al.* 2000, Scharff *et al.* 2003), tropical areas (Silva & Coddington 1996, Sørensen *et al.* 2002, Sørensen 2004, Coddington *et al.* 2009, Malumbres-Olarte *et al.* 2016), subtropical (Muelelwa *et al.* 2010) and Mediterranean biotopes (Jiménez-Valverde & Lobo 2006, Cardoso *et al.* 2007, 2008a). Several aspects that affect taxonomic composition of samples, number of adults and species of adults have been assessed. For example, sampling methods have been reported to be an essential element of the study design (e. g. Coddington *et al.* 1996, Sørensen *et al.* 2002, Cardoso *et al.* 2008a, 2008b, Muelelwa *et al.* 2010), and it has been suggested that if resources permit then all methods should be incorporated in the study design (Muelelwa *et al.* 2010).

However, combinations of the chosen methods should always be kept to a minimum to avoid complexity and should also be able to collect different species, hence minimizing species overlap (Coddington *et al.* 1991). For example, aerial searching, beating and sweeping have been reported to strongly overlap (Cardoso *et al.* 2008a, 2008b, 2009b), thus wasting resources that can be utilised to capture different species (Cardoso 2009). Various sampling methods have also been reported to be inefficient, such as aerial searching in two savanna vegetation types (Muelelwa *et al.* 2010), Winkler traps in the Ophathe Game Reserve (Haddad & Dippenaar-Schoeman 2015) and the bark trap method in the Mediterranean (Cardoso *et al.* 2008b). Thus, the methods included in a design should be carefully selected, considering that the efficacy of a particular sampling method may differ with biotope (Muelelwa *et al.* 2010).

Night sampling has been reported to yield higher spider species and samples than day sampling (e.g. Cardoso *et al.* 2008a, 2008b), thus combinations of time of day and method can be regarded as dissimilar sampling methods altogether (Cardoso *et al.* 2008b, Cardoso 2009). Seasonality is also an important aspect that needs to be accounted for in study designs, as sampling during the peak season has been reported to capture almost 50% of the yearly spider diversity (Jiménez-Valverde & Lobo 2006). The best period for sampling spiders in the Mediterranean is between

May and June (Cardoso *et al.* 2007), while in savanna ecosystems the best period for sampling is during early summer in November (Muelelwa *et al.* 2010). Collector experience has also been reported not to have any significant effect on the species richness and relative abundance caught (e.g. Coddington *et al.* 1991, Cardoso *et al.* 2008b). However, experienced collectors are usually the most productive (Cardoso *et al.* 2008b), especially as they usually know which microhabitats to target (Muelelwa *et al.* 2010). Thus, for any study design, the inclusion of at least one experienced collector who will maximise consistency between teams is recommended (Cardoso 2009).

Dobyns' (1997) study on sampling intensity revealed that repetitive collection is a more efficient strategy, while Sorenson *et al*. (2002) reported that plotless (unrestricted) and plot-based approaches caught a similar species composition and number of species per sample. However, Cardoso (2008b) reported a higher species richness obtained in plot less sampling, mainly because of the different sampling effort placed outside plots. According to Cardoso (2009), an adequate plot size for standardised sampling is one hectare. The study of these aspects has assisted in the improvement of standardised sampling protocols for spiders, which is closely approaching scientific maturity (Cardoso *et al.* 2008a), especially within the Mediterranean region. However, this is not the case within the savanna biome.

Recently, two field protocols nicknamed Conservation Oriented Biodiversity Rapid Assessment (COBRA) and South African National Survey Arachnida (SANSA) have been developed for the Mediterranean (Cardoso 2009) and South Africa (Dippenaar-Schoeman & Haddad 2008, Haddad & Dippenaar-Schoeman 2015) spiders, respectively. The COBRA sampling protocol recommends utilisation of five methods (i.e. beating trees, ground searching, aerial searching, sweeping and pitfall trapping) with at least 24 or 96 samples. A sample is determined by one-person hour of effective fieldwork (Cardoso 2009). According to Cardoso (2009), aerial collecting and ground collection are more productive at night, while beating and sweeping were variable, depending on the biotope. As a result, within the Mediterranean ideal protocols should comprise a larger percentage of aerial searching done at night, pitfall trapping, and both day time and night sweeping and beating.

In Africa, a sampling protocol based on that of Coddington *et al.* (1991) was initially established for use within the SANSA surveys (Dippenaar-Schoeman & Haddad 2008, Haddad &

Dippenaar-Schoeman 2015) which was tested in two savanna vegetation types (Muelelwa *et al.* 2010) and later led to the development of the SANSA standardised sampling protocol (Dippenaar-Schoeman & Haddad 2008, Haddad & Dippenaar-Schoeman 2015). The SANSA protocol entails that four biotopes characteristic of the chosen area be identified and six sampling methods (i.e. beating, sweeping, leaf litter sifting, day hand collecting, night hand collecting, and pitfall traps) are recommended for utilization in each biotope (Dippenaar-Schoeman & Haddad 2008, Haddad & Dippenaar-Schoeman 2015).

According to Haddad & Dippenaar-Schoeman (2015), active searching, pitfall traps and beating yield the highest species richness, and are therefore very important methods that should be incorporated into a design. However, Winkler traps were very inefficient, and therefore efforts for using such methods may as well be directed to other methods. Ideally canopy fogging also gives excellent results and can also be added, however it is usually an expensive method and less environmentally friendly (Kuria *et al.* 2010). Since the initial design of sampling protocols by Coddington *et al.* (1991), the utilisation of standardised sampling protocol has been a widely recommended concept. The SANSA sampling protocol has yielded impressive diversity of arachnids within South Africa, whereas its efficiency within the biomes of Zimbabwe is yet to be tested.

1.6 Spider sampling techniques

Spiders are the most widespread and ubiquitous arthropod predators that are found almost everywhere occupying all possible terrestrial microhabitats (Turnbull 1973). When monitoring invertebrates is the main goal of a study, several sampling methods should be used (e.g. pitfall traps, sweep nets, beating, active searching, dvac samples, leaf litter samples etc.), as no single technique is able to capture invertebrates from all microhabitats (Standen 2000). For example, pitfall traps are effective at sampling ground-active spiders but under sample foliage-dwellers (Green 1999). Beating sheets have also been reported to underrepresent web-building spiders (Costello & Dane 1995).The sampling methods included in the design of this study were chosen mainly because they are required by the SANSA standardised sampling protocol (Dippenaar-Schoeman & Haddad 2008, Haddad & Dippenaar-Schoeman 2015) and are described in the following paragraphs.

1.6.1 Pitfall traps

Pitfall traps are a common method used to sample ground-active invertebrates (Dippenaar-Schoeman & Wassenaar 2002, Russell-Smith 2002, Haddad & Butler 2018). They are usually containers that contain a killing agent or preservative such as ethylene glycol, and placed with their upper rim equal with the ground surface. The quantity of pitfall traps is dependent on the sampling protocol, i.e. 50 pitfall traps (Dippenaar-Schoeman & Haddad 2008, Haddad & Dippenaar-Schoeman 2015) used per biotope in the SANSA sampling protocol, or 48 pitfalls used in COBRA sampling protocol with 24 samples (Cardoso 2009). Pitfall trap catches are affected by various factors that include trap diameter (Brennan *et al.* 1999, 2005, Brown & Matthews 2016), layout (Perner & Schueler 2004), trap construction (Knapp & Ruzicka 2012), construction material (Luff 1975), or baits (Raffa & Hunt 1988).

Effective trap nights ranges from two to seven nights (Engelbrecht 2013), however these normally catch active species and not necessarily many rare taxa. Abundance and richness of spiders also increases with trap sizes of greater than ≥ 7.0 cm (Brennan *et al*. 1999, Work *et al.* 2002) Pitfall trap shape has also been shown to affect catches, with round uncovered pitfalls usually yielding higher catches of large bodied organisms than rectangular or covered traps (Spence & Niemela 1994). Distance between pitfall trap (interspacing) also affects catches, with distances of between 5 and 10 m catching higher numbers of invertebrates than those of 1 m interspacing (Ward *et al.* 2001).

According to Brown and Matthews *et al.* (2016), plastic pitfalls are preferable to glass and metal, as they are easily available, cheaper, less fragile, lighter and have been the most commonly used in previous years. Knapp and Kuzikka (2012) reported on trap construction (funnel or cup), where higher catches were obtained in cups than in funnels. In addition, Patrick and Hansen (2013) reported on higher catches by modified pitfalls known as ramp traps and highlighted their importance in areas where digging is difficult or impossible as they are simply placed on their substrate. According to Penner and Schueler (2004), the nested cross array is a favorable layout for sampling ground-dwelling spiders.

Nevertheless, in order to ensure comparability between studies a standardized pitfall trap design should be utilised (Brown & Matthews 2016). However, in certain cases standardization might result in catching non-target organisms. For instance in a study by Lehmitz *et al*. (2012) on the

distribution of mites. Be that as it may, optimal pitfall traps have also been shown to efficiently collect invertebrates with minimal bias, reducing by-catches of non-target animals (Lange *et al.* 2011, Csaszar *et al.* 2018) As a result, pitfall traps are a satisfactory method that can be utilized to capture invertebrates, as they usually capture more species than most of the other methods (Cardoso *et al.* 2008b), despite the fact that it may not capture all of the ground-dwelling arthropods (Driscoll 2010). Pitfall traps are also inexpensive, harvest great quantities of specimens that contain a wide variety of taxa, especially large specimens (Gibson *et al.* 1992), they are also the most productive method for collecting unique species (Cardoso *et al.* 2008b), and require little labor to operate (Ward *et al.* 2001).

1.6.2 Sweeping

Sweeping is a common passive method used to sample invertebrates linked with low-lying flora found in the understory (Haddad 2005). Sweep nets dislodge the specimens from the vegetation with a sweeping action (Dippenaar-Schoeman & Haddad 2014). In order to capture active invertebrates, a definite number of sweeps of a certain stroke are made (Delong 1932). The SANSA sampling protocol recommends a total of 500 sweeps of herbs, lows shrubs and grasses to be done in each biotope (Dippenaar-Schoeman & Haddad 2008). Sweeping is suitable for utilisation both during the day and night (Cardoso 2009, Guevara & Aviles 2009). Regardless of the fact that it is labor intensive (Yi *et al.* 2012) and requires experience (Spafford & Lortie 2013), it is however a robust method that captures a broad range of taxa (Orlofske *et al.* 2010) in a consistent, reliable and precise manner that provides an estimate of diversity (Spafford & Lortie 2013).

Sweep nets sample arthropods in a fast and inexpensive manner and have been used to actively sample hunting spiders and small web-building species (Basset *et al.* 1997). Sweep net catches have been reported to be affected by vegetation type, sweep speed, height, time of day and weather (Guevara & Aviles 2009). For example, spiders tend to orient themselves differently on a plant depending on the time of day (Delong 1932). Thus, sweeping should be conducted at different heights. Wet periods also tend to cause invertebrates to stick together, reducing the efficiency of sweep nets during periods of rain (Warui *et al.* 2005).

1.6.3 Beating sheets

Beating sheets are useful for capturing foliage-dwellers that are usually resting in the understory and mainly target taxa foraging on vegetation (Guevara & Aviles 2009). The specimens are usually knocked from the vegetation by beating it with a stick while the sheet is held under the vegetation. The number of beats are also dependent on the sampling protocol, i.e. 500 beats of tree and shrub foliage are recommended by the SANSA sampling protocol (Dippenaar-Schoeman & Haddad 2008) or 2 samples with a sample being determined by one-person hour of effective fieldwork is normally used in COBRA sampling protocol (Cardoso 2009). The contents of the beating sheet can then be collected either using small vials, or by a pooter or small paintbrush dipped in alcohol. Advantages of the method include higher productivity and repeatability (Coddington *et al.* 1991) and provision of a comparable number of species (Hatley & MacMahon 1980). Beating sheets are a simple and fast method used to sample invertebrates (Guevara & Aviles 2009). However, the sheets are biased towards small or active taxa and usually exclude ground arthropods (Guevara & Aviles 2009), and web builders are usually underestimated (Costello & Dane 1995).

1.6.4 Hand collecting

Hand collecting is an active method that involves visual searching on plants, under logs, rocks, bark and leaf litter or grass tussocks. Spiders are then collected by hand using vials (Dippenaar-Schoeman & Haddad 2014). According to Dippenaar-Schoeman & Haddad (2008) each team member is supposed to do two hours of hand collecting in every site. Just like pitfall traps, active searching catches large visible arthropods (Guevara & Aviles 2009), and is highly efficient both during the day and the night. Visual searches are advantageous in the sense that they are non-destructive, straight-forward and fast (Guevara & Aviles 2009). However, their limitations include exclusion of small non-obvious taxa, and that collector experience highly affects catches (Guevara & Aviles 2009).

1.6.5 Leaf litter

Ground-dwelling spiders can also be collected by sampling leaf litter whose sample size/volume can be determined. The SANSA sampling protocol recommends ten samples of litter that are taken randomly from underneath shrubs and trees and sieved over a wide cloth. In order to standardize samples, a sieve that is 45cm in diameter and 10 cm in depth with mesh gaps of 8

mm can be filled for one sample (e.g. Butler & Haddad 2011, Haddad & Dippenaar-Schoeman 2015, Haddad *et al*. 2019). Spider assemblages linked with litter arthropods have been determined worldwide (e.g. Stevenson & Dindal 1982, Burgess & Goddard 1999, Castro & Wise 2009, Butler & Haddad 2011, Cole *et al.* 2016). Leaf litter spider assemblages have been reported to vary along elevation gradients (Olson 1994, Vargas 2000), with biotope complexity (Stevenson & Dindal 1982, Haddad *et al.* 2019), and are affected by litter depth and complexity (Uetz 1979, Wagner *et al.* 2003, Butler & Haddad 2011), as well as fluctuations in environmental conditions (Frith & Frith 1990). Leaf litter samples are predominated by small-bodied invertebrates (Spence & Niemela 1994).

1.7 Bio-indicators

According to McGeoch (1998), bio-indicators are a group of species that are used to show levels of taxonomic variety within a site, monitor a specific ecosystem stress, as well as monitor changes within a local environment. The primary purpose of an indicator is to show a relationship with another abiotic or biotic variable (Jones & Eggleton 2000). In a review on how ecologists select indicators (Siddig *et al.* 2016), more than 70% of the selected indicators were invertebrates. It has been argued by Taylor and Doran (2001) that the credibility of any biodiversity monitoring programme is entirely dependent on the inclusion of invertebrates. Research indicates that more than a few assemblages of invertebrates have successfully been utilised as ecological indicators in recent decades, such as earthworms (Suthar 2009), ants (Andersen *et al.* 2004, Ribas *et al.* 2011), beetles (McGeoch *et al.* 2002, Cameron & Leather 2012, Shahabuddin *et al.* 2014), soil invertebrates (Paoletti *et al.* 1996, 2010), spiders (Marc *et al.* 1999, Haddad *et al*. 2009) and butterflies (Kyerematen *et al.* 2018), amongst others. The usefulness of each invertebrate group varies. For example, butterflies have been reported to show greater potential as bio-indicators than beetles and bats (Syaripuddin *et al.* 2015).

1.8 Spiders as bio-indicators

Spiders are good bio-indicators and they possess several qualities that were reviewed by Churchill (1997). Not only have spiders been used as bio-indicators to environmental disturbances such as fire (Pryke & Samways 2012, Podgaiski *et al.* 2013, Haddad *et al.* 2015), biotope changes (Haddad *et al.* 2009) and grazing (Ford *et al.* 2013, Fuller *et al.* 2014, Dennis *et al.* 2015, Schwerdt *et al.* 2018) but they have also been used to determine other widespread

environmental changes such as effects of leaf litter on spider fauna (Castro & Wise 2009, Butler & Haddad 2011, Podgaiski & Rodrigues 2016), effect of seasons (Janzen & May 1973, Niemela *et al*. 1994, Weeks & Holtzer 2000), rainfall gradients (Churchill 1998), quality of biotopes (Halaj *et al.* 1998). Spiders are therefore useful indicators of environmental factors and can be used in all studies. In this study, they will be utilised to indicate the effects of holistic planned grazing within a mixed cattle and wildlife ranch in Shangani, Zimbabwe.

1.9 Influence of grazing on spiders

Generally, the impact of grazers on invertebrates can be positive, negative and neutral (Gibson *et al.* 1992). *Negative:* this occurs when the total abundance is illustrated to decrease with grazing and there is a reduction in faunal composition (Gibson *et al.* 1992, Churchill & Ludwig 2004, Szineter & Samu 2012, Foord *et al.* 2013, Fuller *et al.* 2014). *Positive;* this scenario occurs when there is a significant increase in arthropod abundance with increasing disturbance (Seymour & Dean 1999). *Neutral*: this scenario occurs when there is no significant difference between grazed and ungrazed sites (Harris *et al.* 2003, Jansen *et al*. 2013). Grazing by livestock affects spider ecology and distribution, for example larger web spinning species have been shown to be more sensitive to grazing pressures (Gibson *et al.* 1992), mainly as a result of reduction of vegetation structure by the physical presence of cattle that destroys webs as cattle walk through them, resulting in the loss of locations to anchor webs (Rypstra 1983, Takada *et al.* 2008). Impact of deer on vegetation, spiders and prey availability have been shown to be sequential, according to Roberson *et al*. (2016), higher prey densities of prey were reported in grazed areas most probably due to the additional space utilisation by prey in the absence of structural impediment created by vegetation thereby allowing more prey to be able to fly freely unimpeded through the grazed plots.

The influence of grazing in most areas of Africa is still poorly known. Sparse information on the influence of cattle and wildlife grazing on spiders have been studied in South Africa (Seymour & Dean 1999, Jonsson *et al*. 2010, Jansen *et al*. 2013). In east Africa, a study was piloted on the impacts of large mammals on spider communities (Warui *et al.* 2005), results indicated reduced species richness and spider abundance that occurred as an effect of the reduction in vegetation complexity. Other similar studies conducted elsewhere, include that of Ford *et al.* (2013) on how management of grazing in saltmarshes drives functional group structure and invertebrate

diversity, Szinetar and Samu (2012) on the potential of intensive grazing enabling the invasion by disturbance-tolerant species in Hungary, and by Fuller *et al*. (2014) on the effects of grazing on ground-dwelling spiders in oak and yew woodland in Ireland. Greater numbers of active Linyphiidae have been reported in grazed marshes in England (Ford *et al.* 2013) and wolf spiders in the grasslands of the Mpumalanga province in South Africa (Jansen *et al.* 2013). This was mainly as a result of their capacity to disperse into disturbed or open biotopes (Ford *et al.* 2013). Cattle have also been reported to have a superior impact on the spider fauna compared to large mammalian herbivores (Warui *et al.* 2005), mostly due to their high densities compared to that of wildlife in most rangelands.

1.10 The Savory grazing method or Holistic resource management (HRM)

The Savory grazing method (SGM) or Holistic resource management (HRM) has been known through the use of many different variable terms. For example, in a meta-analysis of the global assessment of Holistic planned grazing (HPG), Hawkins (2017) utilises the terminology HPG and defines it as "time controlled, rotational grazing that utilises an adaptive versus prescriptive management". This is a very interesting and inclusive definition that attempts to incorporate the majority of the key principles of the SGM (Savory & Parsons 1980). However, the method has also been commonly been identified as short-duration grazing (Goodloe 1969, Holechek *et al.* 2000, Dormaar *et al.* 2018), rapid rotation grazing (Briske *et al.* 2008, 2011, Brown & Kothman 2009), time-controlled grazing (Willms *et al.* 1990), cell grazing (Earl & Jones 1996, McCosker 2000, Richards & Lawrence 2009), Savory grazing (Savory & Parsons 1980, Savory 1983) and HRM (Savory 1983, 1999, Baxter *et al.* 2015). In order to decrease the opposition from government in its implementation, Savory & Parsons (1980) stated that in the early stages of development HPG was initially known as short duration grazing. However, later on in his publication on the method, Savory (1983) refers to the method as the Savory Method or HRM (Savory 1983) and also endeavours to elaborate misconceptions and myths that surround the framework for holistic decision making and management.

According to Savory (1983) the Savory grazing method or holistic resource management has been surrounded by misinformation and myths which have led to it being referred to as a "wagon wheel system", "a cell grazing system", "short duration grazing", as well as a "rapid rotation grazing" etc. Savory further elaborates points that one has to take note of when referring

to the Savory grazing method or HRM which include the following: (i) that it is a wildlife management technique that can be practised even when there are no livestock in the land; (ii) it is a watershed management technique even in the absence of livestock land; (iii) it is a method of managing livestock on land whereby the latter is to reverse the desertification process economically with or without using fences; (iv) it is a method of managing livestock whether on ranges or on planted pastures in order to obtain greater production from the land and the animals at a greater profitability than conventionally; (v) it is a method of making conventional range management techniques economically sound where they are uneconomically unsound; (vi) it is not just another grazing system of which they have been so many.

The Savory grazing method, which is better understood by its alternate name HRM, has been suggested to be the answer to the desertification problem by Savory (1983). Despite the unit of land utilised which may either be a ranch, tribal area, dairy farm or national park, as well as the goal of management which may either be preservation of a rare semi-desert animal or plant species or to produce stable grassland with high livestock carrying capacity (numbers) in a tribal area amongst others. The same framework for holistic decision making and management can be used and is usually referred to as the Holistic management model (Savory & Butterfield 1999). The goal rests on four fundamental foundation blocks (Fig 1.1). In order to produce the desired goal the key to the management of all four foundational blocks lies in the manipulation of the soil surface in correlation with available resources. Whatever available resources are there they should be directed to the foundation blocks through the action of range influences (Fig 1.1), short of weather and natural catastrophe (which the latter two are usually beyond the control of man). With the application of Holistic resource management there are usually management guidelines which may be applied either daily or usually brought into play periodically as when a particular situation or problem arises or when undertaking long term plans and annual budgeting (Savory 1983).

SAVORY GRAZING METHOD

OR

HOLISTIC RESOURCE MANAGEMENT

GOAL

SUCCESSION   WATER CYCLE   MINERAL CYCLE   ENERGY FLOW

ECOLOGICAL FOUNDATION BLOCKS

FIRE   RANGE REST   GRAZING   ANIMAL IMPACT   WILDLIFE   MAN

RANGE INFLUENCES

TIME   STOCK DENSITY   HERD EFFECT   WEAK LINK   CAUSE & EFFECT   WHOLE ECOSYSTEM   FLEXIBILITY   BURNING   ECONOMICS

MANAGEMENT GUIDELINES
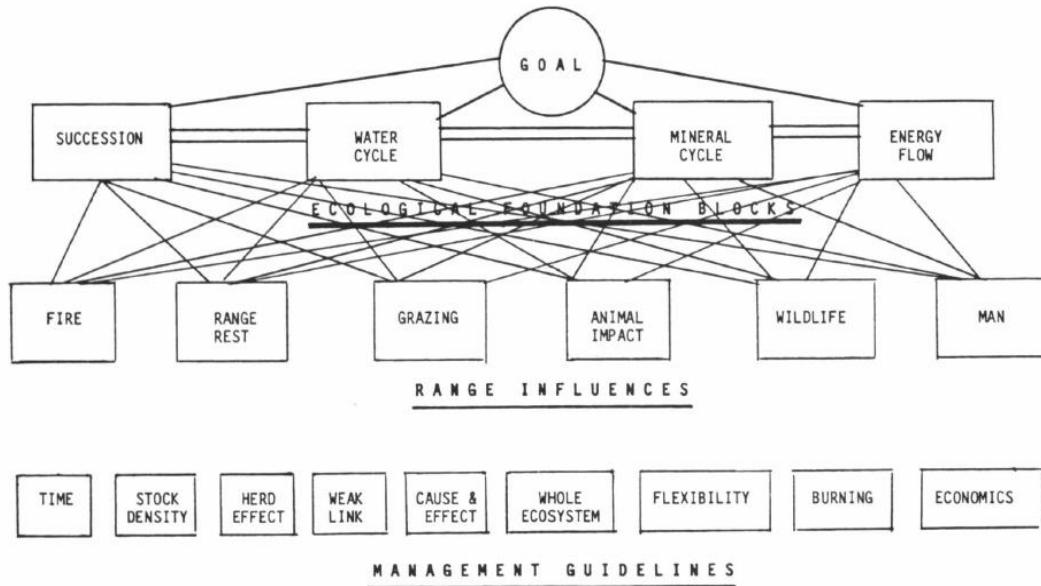
Fig. 1.1 The Holistic Management model   (Source: Savory 1983).

Holistic planned grazing is a planning process whose goal is to ensure productivity in the management of stock to improve animal vigour and land restoration (Baxter *et al*. 2015). Common inception involves the determination of paddocks that can either be fenced or herded, where boundaries can be determined by natural features. Early schemes in Zimbabwe involved paddocking that utilised grazing cells that were developed to overcome stock stress (Savory & Parsons 1980). However, paddocks can be applied without the wagon wheel design (whereby the shapes of the paddocks resemble a wagon wheel). In each paddock, water is an essential resource, together with livestock handling facilities (Holechek *et al.* 2000). A key feature is to group animals into a few large herds, preferably one large herd (Holechek *et al*. 2000, Baxter *et al.* 2015), which in the latter case ensures an increase in the recovery periods of the plants, as well as in the intensity of hoof impact, which is an essential feature for breaking the hard crust of the soil, allowing water penetration and plant decay (Savory 2013).

Typically, the grazing period within each paddock should be as short as possible to ensure the reduction of overgrazing, which has since been determined as an issue of the time that plants are exposed to grazing and the time it is next grazed (Savory 2013) rather than the number of

animals. It is usually recommended that livestock be moved more rapidly during periods of active plant growth than plant dormancy (Holechek *et al*. 2000). According to Savory & Parsons (1980), HRM is versatile as opposed to being a rigid approach, as it is adaptive and flexible. For example, in its application there is no set number of paddocks, animal numbers, and interval of grazing periods and preparation of pastures that is prescribed.

Holistic resource management is a tool that has generated intense and conflict-ridden discussion among practitioners of holistic planned grazing, rangeland scientists and other professionals. This debate was initiated due to the claim of Allan Savory, which attributes holistic planned grazing as the only tool that is left as an option to reverse two of the utmost perplexing and correlated global change processes, climate change and desertification (Briske *et al.* 2013). A large portion of the critics attribute their points to the lack of scientific evidence that is published to back up most of Savoury's claims (Nordborg 2016), which Savory admits to (Savory 2013), mainly because of the inability of most pastoralist and ranchers to publish their results in journals, mostly due to the lack of replication. However, in response to an appeal for evidence on the methodology and science behind Holistic management and HPG, a portfolio was published by the Savory Institute (Savory 2013).

Several reviews have attempted to assess Savory claims over the period of years, by comparing continuous grazing to diverse forms of rotational grazing systems (Nordborg 2016). One paper by Holechek *et al.* (2000) reviewed 13 North American studies undertaken between 1982 and 1999, with a purpose to summarise knowledge on short duration grazing with regards to soil, vegetation, livestock, and financial response, concluded that short duration was not grander than continuous grazing. In another review by Briske *et al*. (2008) that reviewed above 40 studies from USA and Africa, continuous grazing was also found not to be superior to rotational grazing with regards to plant production and livestock production. Other critics include scholars such as Briske *et al.* (2013), who argued on the lack of scientific evidence of the tool to reverse degradation. However, evidence suggesting acceleration of these processes. Hawkins (2017) also argues that the tool does not increase production, and hence does not permit all the set-up and effort it requires. Many studies that were reviewed yielded valid results in their comparison of continuous grazing to short duration grazing with regards to variable parameters, but nonetheless there is little research that has reported on the response of biodiversity, in particular spiders, to holistic planned grazing.

In contrast to these views, research also indicates that there are a number of studies that have yielded positive results with regards to holistic planned grazing. For example, Teague & Barnes (2017) concluded that utilisation of multiple paddocks has been shown to provide a tangible advantage over continuous grazing systems, especially when they are managed adaptively and there is good planning involved. In addition, higher soil organic matter and soil chemical properties (Sanjari *et al.* 2008, Teague *et al.* 2011, Ferguson *et al.* 2013), denser vegetation, higher percentage litter cover, higher basal diameters and higher relative frequency of most palatable species (Earl & Jones 1996, Weber & Gokhale 2011, Ferguson *et al.* 2013), higher soil moisture (Weber & Gokhale 2011), and increased plant biodiversity (Stinner *et al.* 1997) have been reported in rangelands that practise holistic management. Research on holistic planned grazing has utilised both field measurements and interviews. Nordborg (2016) argues that results of better quality are usually produced by field measurements, especially for effects that measure soil and vegetation parameters. In addition, utilisation of longer measurement series and more test sites compared to a single, short, once-off site would produce results of better quality. However, these arguments have not adequately addressed what the potential response of the spider fauna to holistic planned grazing would be.

1.11 Specific research problem

In Zimbabwe significant efforts have been made to come up with elaborate and comprehensive biodiversity inventories and monitoring programmes for most organism's such as mammals (Dunham *et al.* 2009, Jammes *et al.* 2009, Dunham & du Toit 2013), reptiles (Zisadza-Gandiwa *et al.* 2013, Sai *et al.* 2016), birds (Bird Life Zimbabwe 2010), plants (Mapaura & Timberlake 2004). However, the least documented are insects and arachnids (FitzPatrick 2001, Wesołowska & Cumming 2011, FitzPatrick & Dube 2018) with a limited distribution coverage.

The lack of knowledge on species and their distributions is a great hindrance to conservation (Oliveira *et al.* 2016). Therefore, there is need to complete this initial stage in order to develop management plans. The latter can only be established and applied once inventories or partial inventories are accomplished (Whitmore *et al.* 2002). Currently in Zimbabwe, conservation strategies only take into account plants and vertebrates, with the belief that protection efforts afforded to plant and vertebrates will benefit invertebrates as well. However, this method usually does not cater for the protection of rare and threatened invertebrate species (Lovell *et al.* 2009).

Currently, South Africa, is the country with the greatest identified species from Africa (Jocque *et al.* 2013), which has led to the tremendously improvement of its knowledge on spiders, and is likely to contribute to the knowledge on threatened spider species. However, such a scenario is still somehow an impossible reality to most other African countries, regardless of them being signatories of the Convention of Biological Diversity. Nevertheless, Dippenaar-Schoeman *et al.* (2015) highlights the direction of future research in South Africa that requires global collaborative studies that focus on focused research questions and standardized sampling protocols. Therefore, the need to fill such a gap motivated this study on spider ecology within south-western Zimbabwe, with an emphasis on holistic planned grazing practices.

1.12 Research objectives

1.12.1. General objective

To establish the response of spider fauna to holistic management practises within a mixed cattle and wildlife ranch situated at Shangani, Zimbabwe, test the efficacy of the SANSA protocol in estimating species richness within the Khami World Heritage Site, and propose an "optimized protocol" that can be used to determine species richness of a typical region.

1.12.2 Specific objectives

The specific objectives of this study are:

1. To describe the changes in mean grass height (cm) and percentage grass cover over the sampling periods, both inside and around previously kraaled sites and control sites at Debshan Ranch, Shangani, Zimbabwe.
2. To assess the effect of short duration kraaling as implemented through holistic management plan for both inside and around previously kraaled sites on spider assemblages at Debshan Ranch, Shangani, Zimbabwe.
3. To determine the importance of geographic distances from cattle kraals around previously kraaled sites on spider assemblages at Debshan Ranch, Shangani, Zimbabwe.
4. To assess the influence of time since kraal removal for both inside and around previously kraaled sites on spider assemblages at Debshan Ranch, Shangani, Zimbabwe.
5. To evaluate the significance of vegetation structure variables on spider community structure.
6. To estimate species richness in four selected biotopes within the Khami World Heritage site

using coverage based rarefaction.

7. To propose an "optimised protocol" that can be used to determine species richness in other regions.

1.12.3 Research scope

The research focused on three aspects. The first aspect involved monitoring of three cattle herds that had approximately 350 livestock stationed in three selected sites that were all located within the miombo woodlands dominated by *Brachystegia* spp. and *Julbernadia* spp., following a time sequence of recovery after cattle have intensely grazed a particular area. The second aspect included evaluating the impacts of the cattle kraal itself at a local scale by monitoring of previously kraaled sites dating back to a period of 10 months since kraal removal, with sampling done in two sampling seasons (early summer and late summer). The third aspect included testing the efficacy of the SANSA sampling protocol within a world heritage site in Khami by attempting to compare the species richness between selected biotopes, seasons and selected sampling methods used to sample spider fauna, with an attempt to optimize the sampling protocol.

1.12.4 Relevance of the research

A study on spider ecology is justified on several grounds. First, it has been established in the literature review that very little information is available on the spider fauna of Zimbabwe. This study is likely to produce new species that when described will aid in updating of the national species list and distribution patterns of spiders within Zimbabwe, thus improving the status of Zimbabwean spider knowledge.

Secondly, there is insufficient information on how African spider assemblages respond to holistic management practices. The latter which has stimulated heated, divisive debate in conference sessions and journals among conservationists, with many scientists questioning the scientific basis of holistic management practices (Briske *et al.* 2013).

Thirdly, spider communities (species richness, abundance and structure) are good indicators for whole-biocenosis biodiversity (Schwerdt *et al.* 2018). According to Oxbrough *et al.* (2007), spider communities are affected by the structure of the flora and disturbances. At the same time, their abundance and species richness shows a relationship to other invertebrates and birds, and

thus can be applied in assessments of environmental changes (Sauberer *et al.* 2004). The findings of the research will be quite useful, especially to ranch managers with regards to the impact of holistic planned grazing as a tool that is believed to reduce biodiversity loss and reverse climate change.

Fourthly, in order to improve taxonomic and distributional data there is need to design an optimised sampling protocol (Coddington *et al.* 1991) that will allow for comparability amongst ecological surveys. Therefore, this study will also try to optimize a sampling protocol for standardised inventories, with particular interest to answering an unrequited question raised in Muelelwa *et al.*'s (2010) study on the role that biotope structure plays in optimization, as well as two related concepts of sampling intensity and sampling combinations.

Lastly, but definitely not least, spider diversity is exciting in its own right, and therefore worthy of research and protection.

1.13 References

Agnarsson, I., Kuntner, M. & Blackledge, T.A. (2010) Bioprospecting finds the toughest biological material: extraordinary silk from a giant riverine orb-spider. *PLoS One*, **5**, 1–8.

Alderweireldt, M. (1994) Prey selection and prey capture strategies of Linyphiid spiders in high-input agricultural fields. *Bulletin of the British Arachnological Society*, **9**, 300-308.

Andersen, A.N., Fisher, A., Hoffmann, B., Read, J.L. & Richards, R. (2004) Use of terrestrial invertebrates for biodiversity monitoring in Australian rangelands, with particular reference to ants. *Australian Ecology*, **29**, 87–92.

Balmford, A. & Gaston, K.J. (1999) Why biodiversity surveys are good value. *Nature*, **398**, 204–205.

Basset, Y., Springate, N.D., Aberlenc, H.P. & Delvare, G. (1997) *A review of methods for sampling methods in tree canopies: Canopy Arthropods*. Edited by Stork, N.E., Adis, J & Didham, R.K. Chapman and Hall, London.

Baxter, R., Hastings, N., Law, A. & Glass, E.J. (2015) *What is holistic planned grazing?* Savory Institute, United States of America.

Bird Life Zimbabwe. (2010) *Important bird areas in Zimbabwe: national status and trends 2009*. Genesis Advertising and Design, Harare, Zimbabwe.

Bonte, D., Lens, L., Maelfait, J.P., Hoffmann, M. & Kuijken, E. (2003) Patch quality and connectivity influence spatial dynamics in a dune wolfspider. *Oecologia*, **135**, 227–233.

Bonte, D., Travis, J.M.J., De Clera, N.D., Zwertvaegher, I. & Lens, L. (2008) Thermal conditions during juvennille development affect adult dispersal in a spider. *Proceedings of the National Academy of Sciences of the United Statesof America*, **105**, 17000-17005.

Bonte, D., Van Belle, S. & Maelfait, J.P. (2007) Maternal care and repro-ductive state-dependent mobility determine natal dispersal in a wolf spider. *Animal Behaviour*, **74**, 63–69.

Brennan, K.E.C., Majer, J.D. & Moir, M.L. (2005) Refining sampling protocols for inventorying invertebrate biodiversity:influence of drift-fence length and pitfall trap diameter on spiders. *Journal of Arachnology*, **33**, 681–702.

Brennan, K.E.C., Majer, J.D. & Reygaert, N. (1999) Determination of an optimal pitfall trap size for sampling spiders in a western Australian Jarrah forest. *Journal of Insect Conservation*, **3**, 297–307.

Briske, D.D., Bestelmeyer, B.T., Brown, J.R., Fuhlendorf, S.D. & Polley, H.W. (2013) The Savory method can not green deserts or reverse climate change. *Rangelands*, **35**, 72–74.

Briske, D.D., Derner, J.D., Brown, J.R., Fuhlendorf, S.D., Teague, W.R., Havstad, K.M., Gillen, R.L., Ash, A.J. & Willms, W.D. (2008) Rotational grazing on rangelands: reconciliation of perception and experimental evidence. *Rangeland Ecology & Management*, **61**, 3–17.

Briske, D.D., Sayre, N.F., Huntsinger, L., Fernandez-Gimenez, M., Budd, B. & Derner, J.D. (2011) Origin, persistence, and resolution of the rotational grazing debate: integrating human dimensions into rangeland research. *Rangeland Ecology and Management*, **64**, 325–334.

Bristowe, W.S. (1939-1941) *The Comity of spiders.* Vol 1: 1-228, 1939; Vol 11: 229-560, 1941. Ray Society, London, United Kingdom.

Bristowe, W.S. (1958) *The world of spiders*. 1st Edition. Collins, London, United Kingdom.

Brown, G.R. & Matthews, I.M. (2016) A review of extensive variation in the design of pitfall traps and a proposal for a standard pitfall trap design for monitoring ground-active arthropod biodiversity. *Ecology and Evoluton*, **6**, 3953–3964.

Brown, J. & Kothman, M. (2009) Rotational grazing and the profession and practice of rangeland management. *Rangelands*, **31**, 3–4.

Burgess, N.D. & Goddard, J. (1999) Surface and leaf-litter arthropods in the coastal forests of Tanzania. *African Journal of Ecology*, **37**, 355–365.

Butler, V.P & Haddad, C.R. (2011) Spider assemblages associated with leaf litter of three tree species in central South Africa (Arachnida : Araneae). *African Journal of Ecology*, **49**, 301–310.

Cameron, K.H. & Leather, R. (2012) How good are carabid beetles (Coleoptera, Carabidae) as indicators of invertebrate abundance and order richness? *Biodiversity and Conservation*, **21**, 763–779.

Cardoso, P. (2009) Standardization and optimization of arthropod inventories-the case of Iberian spiders. *Biodiversity and Conservation*, **18**, 3949–3962.

Cardoso, P., Crespo, L.C., Carvalho, R., Rufino, A.C. & Henriques, S.S. (2009a) Ad-hoc vs. standardized and optimized arthropod diversity sampling. *Diversity*, **1,** 36–51.

Cardoso, P., Gaspar, C., Pereira, L.C., Silva, I., Henriques, S., Ricardo, R. & Sousa, P. (2008a) Assessing spider species richness and composition in Mediterranean cork oak forests. *Acta Oecologica,* **33**, 114–127.

Cardoso, P., Henriques, S.S., Gaspar, C., Crespo, L.C., Carvalho, R., Schmidt, J.B., Sousa, P. & Szuts, T. (2009b) Species richness and composition assessment of spiders in a Mediterranean scrubland. *Journal Insect Conservation,* **13**, 45–55.

Cardoso, P., Scharff, N., Gaspar, C., Henriques, S.S., Carvalho, R., Castro, P.H., Schmidt, J.B., Silva, I., Szuts, T., Castro, A. & Crespo, L.C. (2008b) Rapid biodiversity assessment of spiders (Araneae) using semi-quantitative sampling : a case study in a Mediterranean forest. *Insect Conservation and Diversity*, **1**, 71–84.

Cardoso, P., Silva, I., De Oliveira, N.G. & Serrano, A.R. (2007) Seasonality of spiders (Araneae) in Mediterranean ecosystems and its implications in the optimum. *Ecological Entomology,* **32**, 516–526.

Castro, A. & Wise, D.H. (2009) Influence of fine woody debris on spider diversity and community structure in forest leaf litter. *Biodiversity Conservation*, **18**, 3705–3731.

Chari, L.D. (2011) *The influence of large vegetated termitaria and large herbivores on spider (Araneae) diversity in miombo woodlands.* Master of Science dissertation, University of Zimbabwe, Harare, Zimbabwe.

Chew, R.M. (1961) Ecology of the spiders of a desert community. *Journal of the NewYork Entomological Society*, **69**, 5-41.

Churchill, T.B. (1997) Spiders as ecological indicators : an overview for Australia. *Memoirs of the Museum of Victoria,* **56**, 331–337.

Churchill, T.B. (1998) *Spiders as ecological indicators in the Australian tropics: family distribution patterns along rainfall and grazing gradients.* In Selden, P.A (ed). Proceedings of the 17th European Colloquium of Arachnology, Edinburgh 1997. Burnham Beeches, Bucks, British Arachnological Society.

Churchill, T.B. & Ludwig, J.A (2004) Changes in spider assemblages along grassland and savanna grasslands gradients in Northern Australia. *Rangeland Journal,* **26**, 3-16.

Coddington, J.A., Agnarsson, I., Miller, J.A., Kuntner, M. & Hormiga, G. (2009) Undersampling bias: the null hypothesis for singleton species in tropical arthropod surveys. *Journal of Animal Ecology,* **78**, 573–584.

Coddington, J.A., Griswold, C.E., Davila, D.S., Peftaranda, E. & Larcher, S.F. (1991) *Designing and testing sampling protocols to estimate biodiversity in tropical ecosystems.* In Dudley, E.C (ed) the Unity of Evolutionary Biology: Proceedings of the Fourth International Congress of Systematic and Evolutionary Biology, Dioscorides Press, Portland, Oregon.

Coddington, J.A. & Levi, H.W. (1991) Systematics and evolution of spiders (Araneae). *Annual Review of Ecology and Systematics*, **22**, 565–592.

Coddington, J.A., Young, L.H. & Coyle, F.A. (1996) Estimating spider species richness in a southern Appalachian cove hardwood forest. *Journal of Arachnology*, **24**, 111–128.

Cole, R.J., Holl, K.D., Zahawi, R.A., Wickey, P. & Townsend, A.R. (2016) Leaf litter arthropod responses to tropical forest restoration. *Ecology and Evolution*, **6**, 5158–5168.

Costello, M.J. (2015) Biodiversity: the known, unknown, and rates of extinction. *Current Biology*, **25**, 368–371.

Costello, M.J. & Dane, K.M. (1995) Spider species composition and seasonal abundance in San Joaquin valley grape vineyards. *Enviromental Entomology*, **24**, 823–831.

Csaszar, P., Torma, N., Galle-Szasjak, N., Tolgyesi, C. & Gulle, R. (2018) Efficiency of pitfall traps with funnels and/or roofs in capturing ground-dwelling arthropods. *European Journal of Entomology*, **115**, 15-24.

Cumming, M.S. & Wesolowska, W. (2004) Habitat separation in a species-rich assemblage of jumping spiders (Araneae: Salticidae) in a suburban study site in Zimbabwe. *Journal of Zoology*, **262**, 1–10.

Delong, D.M. (1932) Some problems encountered in the estimation of insect populations by the sweeping method. *Annals Entomological Society of America*, **25**, 13–17.

Dennis, P., Skartveit, J., Kunaver, A. & McCracken, D.I. (2015) The response of spider (Araneae) assemblages to structural heterogeneity and prey abundance in sub-montane vegetation modified by conservation grazing. *Global Ecology and Conservation*, **3**, 715–728.

Dippenaar-Schoeman, A.S. (1998) *Spiders as predators of citrus pests*. In Bedford, E.C.G., Van den Berg, M.A. & De Villiers, E.A. (Eds). Citrus pests in the Republic of South Africa. Pretoria, Agricultural Research Council. South Africa.

Dippenaar-Schoeman, A.S. (2001) *Spiders as predators of pests of tropical and non-citrus*

subtropical crops. In Van den Berg, M.A., De Villiers, E. A. & Joubert, P.H. (ed). Pests of Tropical and non-citrus subtropical crops in the Republic of South Africa. Nelspruit, ARC-Institute for Tropical and Subtropical Crops, South Africa.

Dippenaar-Schoeman, A.S. & Haddad, C.R. (2008) South African National Survey of Arachnida surveys: procedure, gap analysis, survey teams and the public. *South African National Survey of Arachnida Newsletter*, **5**, 4–8.

Dippenaar-Schoeman, A.S. & Haddad, C.R. (2014) *Spiders of the grassland biome.* Plant Protection Research Institute Handbook No 19. ARC Plant Protection Research Institute, Pretoria, South Africa.

Dippennaar-Schoeman, A.S., Haddad, C.R., Foord, S.H., Lyle, R., Lotz, L.N. & Marais, P. (2015) South African National Survey of Arachnida (SANSA): review of current knowledge, constraints and future needs for documenting spider diversity (Arachnida : Araneae). *Transactions of the Royal Society of South Africa*, **70**, 245–275.

Dippenaar-Schoeman, A.S. & Wassenaar, T.D. (2002) A checklist of the ground-dwelling spiders of coastal dune forests at Richards Bay, South Africa (Arachnida : Araneae). *Bulletin of British Arachnological Society*, **12**, 275–279.

Dobyns, J.R. (1997) Effects of sampling intensity on the collection of spider (Araneae) species and the estimation of species richness. *Environmental Entomology*, **26**, 150–162.

Dormaar, J.F., Smoliak, S., Willms, W.D., Journal, S. & May, N. (2018) Vegetation and soil responses to short-duration grazing on fescue grasslands. *Society for Range Management*, **42**, 252–256.

Driscoll, D.A. (2010) Few beetle species can be detected with 95% confidence using pitfall traps. *Australian Ecology*, **35**, 13–23.

Dunham, K. & Du Toit, A. (2013) Using citizen based survey data to determine densities of large mammals : a case study from Mana Pools National Park, Zimbabwe. *African Journal of Ecology*, **51**, 431–440.

Dunham, K., Van Der Westhuizen, E., Van Der Westhuizen, H.F. & Gandiwa, E. (2009) *Aerial survey of elephants and other large herbivores in Gonarezhou National Park (Zimbabwe) Zinave National Park (Mozambique) and surroundings.* Gonarezhou Conservation project : Parks and Wild Management Authrority and Frankfurt Zoological Society, Harare, Zimbabwe.

Earl, J.M & Jones, C.E. (1996) The need for a new approach to grazing management-is cell grazing the answer? *Rangeland Journal*, **18**, 327–50.

Eberhard, W. (1967) Attack behaviour of diguetid spiders and the origin of prey wrapping in spiders. *Psyche*, **74**, 173-81.

Eberhard, W.G. (1971) The ecology of the web of *Uloborus diversus* (Araneae: Uloboridae). *Oecologica*, **6**, 328-42.

Eisoldt, L., Smith, A. & Scheibel, T. (2011) Decoding the secrets of spider silk. *Materials Today*, **14**, 80–86.

Elgar, M.A. & Fahey, B.F. (1996) Sexual cannibalism, competition, and size dimorphism in the orb-weaving spider *Nephila plumipes* Latreille (Araneae:Araneoidea). *Behavioral Ecology*, **7**, 195–198.

Engelbrecht, I. (2013) Pitfall trapping for surveying trapdoor spiders: the importance of timing, conditions and effort. *Journal of Arachnology*, **41**, 133–142.

Ferguson, B.G., Diemont, S.A.W., Alfaro-Arguello, R., Martin, J.F., Nahed-Toral, J., Álvarez-Solis, D. & Pinto-Ruiz, R. (2013) Sustainability of holistic and conventional cattle ranching in the seasonally dry tropics of Chiapas, Mexico. *Agricultural Systems*, **120**, 38–48.

FitzPatrick, M.J. (1994) A new species of *Eilica* Keyserling, 1891 from Zimbabwe (Araneae: Gnaphosidae). *Arnoldia Zimbabwe*, **10**, 19–21.

FitzPatrick, M.J. (2001) The spider fauna of Zimbabwe (Arachnida: Araneae). *Arnoldia Zimbabwe*, **10**, 177–188.

FitzPatrick, M.J. (2007) A taxonomic revision of the Afrotropical species of *Zelotes* (Arachnida : Araneae : Gnaphosidae). *Bulletin of the British Arachnological Society*, **14**, 97–172.

FitzPatrick, M.J. (2009) A revision of the afrotropical species of *Setaphis* and the description of a new genus (Arachnida: Araneae: Gnaphosidae). *Navorsinge Van Die Nasionale Museum Bloemfontein*, **25**, 73–106.

FitzPatrick, M.J. & Dube, S. (2018) Scorpiones, Solifugae and Araneae of the Matobo World Heritage Site, Zimbabwe. v1.4. Dataset/Checklist. Natural History Museum of Zimbabwe.

FitzPatrick, M.J. & Sebata, S. (2018) The description of the female of *Ibala minshulae* FitzPatrick, 2009 (Araneae: Gnaphosidae). *Arnoldia Zimbabwe*, **10**, 368–371.

Foord, S.H., Dippenaar-Schoeman, A.S., Jocque, R., Haddad, C.R., Lyle, R. & Webb, P. (2016) South African National Survey of Arachnida : a checklist of the spiders (Arachnida, Araneae) of the Lekgalameetse Nature Reserve, Limpopo province, South Africa. *Koedoe*, **58**, 1–8.

Foord, S.H., Dippenaar-Schoeman, A.S. & Van Der Merwe, M. (2002) A check list of the spider fauna of the Western Soutpansberg, South Africa (Arachnida : Araneae). *Koedoe*, **45**, 35–43.

Ford, H., Garbutt, A., Jones, L. & Jones, D.L. (2013) Grazing management in saltmarsh ecosystems drives invertebrate diversity, abundance and functional group structure. *Insect Conservation and Diversity*, **6**, 189–200.

Fourie, R., Haddad, C.R. & Jocque, R. (2011) A revision of the purse-web spider genus *Calommata* Lucas, 1837 (Araneae, Atypidae) in the Afrotropical Region. *ZooKeys*, **95**, 1–28.

Friedrich, B.V.L. & Langer, R.M. (1969) Fine structure of cribellate spider silk. *American Zoology*, **9**, 91-96.

Frith, D. & Frith, C. (1990) Seasonality of litter invertebrate populations in an Australian upland tropical rain forest. *Biotropica*, **22**, 181–190.

Fuller, L., Newman, M., Irwin, S., Kelly, T. & Halloran, J.O. (2014) Ground-dwelling spider diversity in rare European oak and yew woodlands and the impact of grazing. *Biodersity Conservation*, **23**, 1911–1929.

Gabritschersky, E. (1927) Experiments on color changes and regeneration in the crab-spider. *Misumena vatia Journal of Experimantal Biology*, **47**, 251-67

Gardner, B.T. (1965) Observations on three species of *Phidippus* jumping spiders (Araneae: Salticidae). *Psyche*, **72**, 133-47.

Gibson, C.W.D., Hambler, C. & Brown, V.K. (1992) Changes in spider (Araneae) assemblages in relation to succession and grazing management. *Journal of Applied Ecology*, **29**, 132–142.

Goodloe, S.I.D. (1969) Short duration grazing in Rhodesia. *Journal of Range Management*, **29**, 369–373.

Green, J. (1999) Sampling method and time determines composition of spider collections. *Journal of Arachnology*, **27**, 76–182.

Guevara, J. & Aviles, L. (2009) Elevational changes in the composition of insects and other terrestrial arthropods at tropical latitudes: a comparison of multiple sampling methods and social spider diets. *Insect Conservation and Diversity*, **2**, 142–152.

Gunnarsson, B. (1983) Winter mortality of spruce-living spiders: effect of spider interactions and bird predation. *Oikos*, **40**, 226-233.

Haddad, C.R. (2005) Ecology of spiders (Arachnida: Araneae) inhabiting *Themeda triandra* forsskal grassland in semi-arid South Africa. *Navorsinge Van Die Nasionale Museum Bloemfontein*, **21**, 25–36.

Haddad, C.R. & Butler, V.P. (2018) Ground-dwelling spider assemblages in contrasting habitats in the central South African grassland biome. *Koedoe*, **60**, 1–12.

Haddad, C.R & Dippenaar-Schoeman, A.S. (2009) A checklist of the non-acarine arachnids (Chelicerata:Arachnida) of the De Hoop Nature Reserve, Western Cape. *Koedoe*, **51**, 1–9.

Haddad, C.R. & Dippenaar-Schoeman, A.S. (2015) Diversity of non-acarine arachnids of the Ophathe game reserve, South Africa: testing a rapid sampling protocol. *Koedoe*, **57**, 1–15.

Haddad, C.R., Foord, S.H., Fourie, R. & Dippenaar-Schoeman, A.S. (2015) Effects of a fast-burning spring fire on the ground-dwelling spider assemblages (Arachnida: Araneae) in a central South African grassland habitat. *African Zoology*, **50**, 281–292.

Haddad, C.R., Honiball, A.S., Dippenaar-Schoeman, A.S., Slotow, R. & Rensburg, B.J.V. (2010) Spiders as potential indicators of elephant-induced habitat changes in endemic sand forest, Maputaland, South Africa. *African Journal of Ecology*, **48**, 446–460.

Haddad, C.R., De Jager, L.J.C. & Foord, S.H. (2019) Habitats and cardinal directions are key variables structuring spider leaf litter assemblages under *Searsia lancea*. *Pedobiologia*, **73**, 1-32.

Haddad, C.R. & Lyle, R. (2008) Three new genera of tracheline sac spiders from southern Africa (Araneae: Corinnidae). *African Invertebrates*, **49**, 37–76.

Haddad, C.R. & Mbo, Z. (2015) Five new species of the Afrotropical dark sac spider genus *Messapus* Simon, 1898 (Araneae: Corinnidae). *Zootaxa*, **4057**, 385–398.

Haddad, C. & Wesolowska, W. (2006) Notes on taxonomy and biology of two *Stenaelurillus* species from Southern Africa (Araneae:Salticidae). *Annales Zoologici*, **56**, 575–586.

Halaj, J., Ross, D.W. & Moldenke, R. (1998) Habitat structure and prey availability as predictors of the abundance and community organization of spiders in western Oregon

forest canopies. *Journal of Arachnology*, **26**, 203–220.

Harris, R.Y. & Beattie, A.J. (2003) Impacts of grazing and burning on spider assemblages in dry eucalyptus forest of north eastern New South Wales. Australia. *Australian Ecology*, **28**, 526-538.

Hatley, C.L. & Macmahon, J.A. (1980) Spider community organization: seasonal variation and the role of vegetation architecture. *Enviromental Entomology*, **9**, 632–639.

Hauke, T.J. & Herzig, V. (2017) Dangerous arachnids-fake news or reality? *Toxicon*, **138**, 173–183.

Hawkins, H.J. (2017) A global assessment of holistic planned grazing^TM compared with season-long, continuous grazing: meta-analysis findings. *African Journal of Range and Forage Science*, **34**, 65–75.

Haynes, D.L., Sisojevic, P. (1966) Predatory behaviour of *Philodromous rufus* Walchenaer (Araneae: Thomisidae). *Canadian Entomology*, **98**, 113-33.

Hewitt, J. (1915) New South African Arachnida. *Annals of the Natal Museum*, **3**, 289–327.

Hoefler, C.D., Chen, A. & Jakob, E.M. (2006) The potential of a jumping spider, *Phidippus clarus*, as a biocontrol agent. *Horticultural Entomology*, **99**, 432–436.

Holechek, J.L., Gomes, H., Molinar, F., Galt, D., Valdez, R., Holechek, J.L., Gomes, H., Molinar, F., Galt, D. & Valdez, R. (2000) Short-duration grazing: the facts in 1999. *Society for Range Management*, **22**, 18–22.

Hyatt, K.H. (1954) The African spiders of the family Ctenidae in the collections of the British Museum (Natural History). *Annals and Magazine of Natural History*, **12**, 877–894.

Jammes, C., Valeix, S., Bourgarel, M., Murindagomo, F. & Fritz, H. (2009) Seasonal density estimates of common large herbivores in Hwange National Park. *African Journal of Ecology*, **47**, 804–808.

Jansen, R., Makaka, L., Little, I.T. & Dippenaar-Schoeman, A.S. (2013) Response of ground-dwelling spider assemblages (Arachnida, Araneae) to Montane grassland management practices in South Africa. *Insect conservation and diversity*, **6**, 572–589.

Janzen, D.H. (1973) Sweep samples of tropical foliage insects: effects of seasons, vegetation types, elevation, time of day, and insularity. *Ecology*, **54**, 687–708.

Jimenez-Valverde, A. & Lobo, J.M. (2006) Establishing reliable spider (Araneae, Araneidae and Thomisidae) assemblage sampling protocols: estimation of species richness, seasonal

coverage and contribution of juvenile data to species richness and composition. *Acta Oecologica*, **30**, 21–32.

Jocque, R. (1990) A revision of the Afrotropical genus *Diores* (Araneae, Zodariidae). *Annales du Musee royal de l' Afrique Centrale*, **260**, 1–81.

Jocque, R., Alderweireldt, M. & Dippenaar-Schoeman, A.S. (2013) *Biodiversity: an African perspective.* In Penney, D. (ed). Spider research in the 21st century : trends and perspectives. Siri Scientific Press, Manchester, England.

Jocque, R. & Dippenaar-Schoeman, A.S. (1992) Two new termite-eating *Diores* species (Araneae, Zodariidae) and some observations on unique prey immobilization. *Journal of Natural History*, **26**, 1405–1412.

Jones, D.T. & Eggleton, P. (2000) Sampling termite assemblages in tropical forests: testing a rapid biodiversity assessment protocol. *Journal of Applied Ecology*, **37**, 191–203.

Jonsson, M., Bell, D., Hjalten, J., Rooke, T. & Scogings, P.F. (2010) Do mammalian herbivores influence invertebrate communities via changes in the vegetation ? Results from a preliminary survey in Kruger National Park, South Africa. *African Journal of Range and Forage Science*, **27**, 39-44.

Kajak, A. (1965) An analysis of food reloations between spiders (*Araneus cornutus* Clerk and *Araneus quadratus* Clerk) and their prey in a meadow. *Ekologia Polska Seria A*, **13**, 717-64.

Kajak, A. (1965) Quantitative analysis of relations between spiders (*Araneus cornutus* Clerk, and *Araneus quadratus* Clerk) and their prey. *Bulletin of the Polish Academy of Sciences*, **13**, 515-22.

Kaston, B.J. (1948) Spiders of Conneticut. *Bulletin of the Connecticut State Geological and Natural History Survey,* **70**, 1-874

Knapp, M. & Ruzicka, J. (2012) The effect of pitfall trap construction and preservative on catch size, species richness and species composition of ground beetles (Coleoptera: Carabidae). *European Journal Entomology*, **109**, 419–426.

Kuria, S.K., Villet, M.H., Palmer, T.M. & Stanton, M.L. (2010). A comparison of two sampling methods for surveying mammalian herbivore impacts on beetle communities in the canopy of *Acacia drepanolobium* in Kenya. *African Entomology*, **18**, :87–98.

Kyerematen, R., Adu-Acheampong, S., Acquah-Lamptey, D., Anderson, R.S., Owusu, E.H. & Mantey, J. (2018) Butterfly diversity: an indicator for environmental health within

Tarkwa gold mine, Ghana. *Environment and Natural Resources Research*, **8**, 69–83.

Lange, M., Gossner, M.M. & Weissener, W.W. (2011) Effect of pitfall trap type and diameter on vertebrate by-catches and ground beetle (Coleoptera: Carabidae) and spider (Araneae) sampling. *Methods in Ecology and Evolution*, **2**, 185-190.

Lawrence, R.F. (1940) The genus *Selenops* (Araneae) in South Africa. *Annals of the South African Museum*, **32,** 555–608.

Lehmitz, R.D., Russell, K., Hohberg, A., Christian, W.E. (2012) Active dispersal of oribatid mites into young soils. *Applied Soil Ecology*, **55**, 10–19.

Lotz, L.N. (1994) Revision of the genus *Latrodectus* (Araneae: Theridiidae) in Africa. *Navorsinge Van Die Nasionale Museum Bloemfontein*, **10**, 27–56.

Lotz, L.N. (2007a) The genus *Cheiracanthium* (Araneae: Miturgidae) in the Afrotropical region 1. revision of known species. *Navorsinge Van Die Nasionale Museum Bloemfontein*, **23**, 1–76.

Lotz, L.N. (2007b) The genus *Cheiracanthium* (Araneae: Miturgidae) in the Afrotropical region 2. description of new species. *Navorsinge Van Die Nasionale Museum Bloemfontein*, **23**, 145–184.

Lovell, S., Hamer, M., Slotow, R. & Herbert, D. (2009) An assessment of the use of volunteers for terrestrial invertebrate biodiversity surveys. *Biodiversity and Conservation*, **18**, 3295–3307.

Luff, M.L. (1975) Some features influencing the efficiency of pitfall traps. *Oecologia*, **19**, 345–357.

Malumbres-Olarte, J., Scharff, N., Pape, T., Coddington, J.A., Nikolaj, S., Thomas, P., Coddington, J.A. & Cardoso, P. (2016) Gauging megadiversity with optimized and standardized sampling protocols: a case for tropical forest spiders. *Ecology and Evolution*, **7**, 494-506.

Mapaura, A. & Timberlake, J. (2004) *A checklists of Zimbabwean vascular plants.* Southern African Botanical diversity Network Report No.33. Sabonet, Pretoria and Harare.

Marc, P., Canard, A. & Ysnel, F. (1999) Spiders (Araneae) useful for pest limitation and bioindication. *Agriculture, Ecosytsystems and Environment*, **74**, 229–273.

Marques, O.A.V., Sawaya, R.J., Stender-Oliveira, F. & Franca, F.G.R. (2006) Ecology of the colubrid snake *Pseudablabes agassizii* in south-eastern South America. *Herpertological*

Journal, **16**, 34–49.

Mashavakure, N., Mashingaidze, A.B., Musundire, R., Nhamo, N., Gandiwa, E., Thierfelder, C. & Muposhi, V.K. (2019) Spider community shift in response to farming practices in a sub-humid agroecosystem of Southern Africa. *Agriculture, Ecosystems and Environment*, **272**, 237–245.

McCosker, T. (2000) Cell grazing-the first 10 years in Australia. *Tropical Grasslands*, **34**, 207–218.

McGeoch, M.A. (1998) The selection, testing and application of terrestrial insects as bioindicators. *Biological Reviews of the Cambridge Philosophical Society*, **73**, 181–201.

McGeoch, M.A., Van Rensburg, B.J. & Botes, A. (2002) The verification and application of bioindicators: a case study of dung beetles in a savanna ecosystem. *Journal of Applied Ecology*, **39**, 661–672.

Michalko, R., Pekar, S. & Entling, M.H. (2018) An updated perspective on spiders as generalist predators in biological control. *Oecologia*, **182**, 21–36.

Morrison, M.L., Block, W.M., Strickland, M.D., Collier, B.A. & Peterson, M.J. (2008) *Inventory and Monitoring studies. In wildlife study design*. Springer Series on Environmental managemnt. Springer New York, United States of America.

Muelelwa, M.I., Foord, S.H., Dippenaar-Schoeman, A.S. & Stam, E.M. 2010. Towards a standardized and optimized protocol for rapid biodiversity assessments: spider species richness and assemblage composition in two savanna vegetation types. *African Zoology*, **45**, 273–290.

Muvengwi, J., Chikorowondo, G., Mbiba, M. & Gandiwa, E. (2018) Diversity and abundance of macro-invertebrates on abandoned cattle kraals in a semi-arid savanna. *Ecology and Evolution*, **8**, 8477-8483.

National Museums and Monuments Zimbabwe. (2015a) *Natural History Museum of Zimbabwe: A guide.* Friends of the Museum. Bulawayo, Zimbabwe.

National Museums and Monuments Zimbabwe. (2015b) Arachnids, online at http://www.nmmz.co.zw/research/arachnids, accessed on the 16th of June 2018.

Ndava, J., Llera, S.D. & Manyanga, P. (2018) The future of mosquito control: the role of spiders as biological control agents: a review. *International Journal of Mosquito Research*, **5**, 6–11.

Nelson, X.J. & Jackson, R.R. (2006) A predator from East Africa that chooses malaria vectors as preferred prey. *PLoS One*, **1**, 1–15.

Niemela, J., Pajunen, T., Haila, Y., Punttila, P. & Halme, E. (1994) Seasonal activity of Boreal forest-floor spiders (Araneae). *Journal of Arachnology*, **22**, 23–31.

Nordborg, M. (2016) *Holistic management-a critical review of Allan Savory's grazing method.* Technical report. SLU/EPO- Centre for Organic Food and Farming and Chalmer. Swedish University of Agricultural Sciences and Chalmers.

Nyffeler, M. & Birkhofer, K. (2017) An estimated 400– 800 million tons of prey are annually killed by the global spider community. *Science in Nature*, **104**, 1–12.

Nyffeler, M., Edwards, G.B. & Krysko, K.L. (2017) A vertebrate-eating jumping spider (Araneae: Salticidae) from Florida, United States of America. *Journal of Arachnology*, **45**, 238–241.

Nyffeler, M. & Knornschild, M. (2013) Bat predation by spiders. *PLoS One*, **8**, 1–15.

Nyffeler, M. & Pussey, B.J. (2014) Fish predation by semi-aquatic spiders: a global pattern. *PLoS One*, **9**, 1–13.

Oliveira, U., Paglia, A.P., Brescovit, A.D., De Carvalho, C.J.B., Silva, D.P., Rezende, D.T., Leite, F.S., Batista, A.N., Barbosa, J.P.P., Stehmann, R., Ascher, J.S., Vasconcelos, F., Marco, P., Lowenberg-Neto, P., Dias, P.G., Ferro, V.G. & Santos, A. (2016) The strong influence of collection bias on biodiversity knowledge shortfalls of Brazilian terrestrial biodiversity. *Diversity and Distributions*, **22**, 1232-1244.

Oliver, I. & Beattie, J. (1993) A possible method for the rapid assessment of biodiversity. *Conservation Biology*, **7**, 562–568.

Olson, D. (1994) The distribution of leaf litter invertebrates along a Neotropical altitudinal gradient. *Journal of Tropical Ecology*, **10**, 129–150.

Orlofske, J.M., Ohnesorg, W.J. & Debinski, D.M. (2010) Potential terrestrial arthropod indicators for tallgrass prairie restoration in iowa. *Ecological Restoration*, **28**, 250–253.

Oxbrough, A.G., Gittings, T., O'Halloran, J., Giller, P.S. & Kelly, T.C. (2007) Biodiversity of the ground-dwelling spider fauna of afforestation habitats. *Agriculture, Ecosystems and Environment*, **120**, 424–431.

Paoletti, M.G., Bressan, M., Professor, C.A. & Edwards, C.A. (1996) Soil Invertebrates as bioindicators of human disturbance. *Critical reviews in Plant Sciences*, **15**, 21-62.

Paoletti, M.G., Inca, A.D., Tonon, S., Migliorini, C., Petruzzelli, G., Pezzarossa, B., Gomiero, T. & Sommaggio, D. (2010) Soil invertebrates as bio-indicators in a natural area converted from agricultural use: the case study of Vallevecchia-Lugugnana in North-Eastern Italy. *Journal of Sustainable Agricultre*, **34**, 38–56.

Patrick, L.B. & Hansen, A. (2013) Comparing ramp and pitfall traps for capturing wandering spiders. *Journal of Arachnology*, **41**, 404–406.

Perner, J. & Schueler, S. (2004) Estimating the density of ground-dwelling arthropods with pitfall traps using a nested-cross array. *Journal of Animal Ecology*, **73**, 469–477.

Peterson, T.A., Osborne, D.R. & Taylor, D.H. (1989) Tree trunk arthropod faunas as food resources for birds. *Ohio Journal of Science*, **89**, 23–25.

Phanael, G.J. (1967) *Orientation and biorhythms in jumping spiders (Araneidae: Salticidae).* PhD thesis, University of Florida, United States of America.

Pocock, R.I. (1901) XL111. Diagnoses of some new species of spiders from Mashonaland. *Annals and Magazine of Natural History*, **7**, 337–340.

Podgaiski, L.R., Joner, F., Lavorel, S., Moretti, M., Ibanez, S., Mendonc, M.D. & Pillar, V.D. (2013) Spider trait assembly patterns and resilience under fire-induced vegetation change in South Brazilian Grasslands. *PLoS One*, **8**, 1–11.

Podgaiski, L.R. & Rodrigues, G. (2016) Spider community responds to litter complexity: insights from a small-scale experiment in an exotic pine stand. *Iheringia Serie Zoologia*, **107**, 1–8.

Pryke, J.S. & Samways, M.J. (2012) Importance of using many taxa and having adequate controls for monitoring impacts of fire for arthropod conservation. *Journal of Insect Conservation*, **16**, 177–185.

Raffa, K.F. & Hunt, D.A. (1988) Use of baited pitfall traps for monitoring pales weevil, *Hylobius pales* (Coleoptera: Curculionidae). *Great Lakes Entomologist*, **21**, 123-125.

Ramel, G. (2020) Reproduction in spiders: the earthlife web, online at http://www.earthlife.net/chelicerata/s-reproduce.html, accessed on 22nd of February 2020.

Ramel, G. (2020) Spider ecology: the earthlife web, online at http://www.earthlife.net/chelicerata/s-ecology.html, accessed on 22nd of February 2020.

Ramel, G. (2020) The feeding ecology of spiders: the earthlife web, online at http://www.earthlife.net/chelicerata/s-feeding.html, accessed on 22nd of February 2020.

Ribas, C.R., Solar, R.R.C., Campos, R.B.F., Schmidt, F.A., Valentim, C.L. & Schoereder, J.H. (2011) Can ants be used as indicators of environmental impacts caused by arsenic? *Journal of Insect Conservation*, **16**, 413–421.

Richards, C. & Lawrence, G. (2009) Adaptation and change in Queenslands rangelands: cell grazing as an emerging ideology of pastoral ecology. *Land use policy*, **26**, 630–639.

Roberson, E.J., Chips, M.J., Carson, W.P. & Rooney, T.P. (2016) Deer herbivory reduces web-building spider abundance by simplifying forest vegetation structure. *PeerJ*, **4**, 1–15.

Rypstra, A.L. (1983) The importance of food and space in limiting web-spider densities: a test using field enclosures. *Oecologia*, **59**, 312-316.

Russell-Smith, A. (2002) A comparison of the diversity and composition of ground active spiders in Mkomazi game reserve, Tanzania and Etosha National park, Namibia. *Journal of Arachnology*, **30**, 383–388.

Sai, M., Utete, B., Chinoitezvi, E., Moyo, G.H. & Gandiwa, E. (2016) A survey of the abundance, population structure and distribution of Nile Crocodiles *Crocodylus niloticus* using day ground surveys in Sengwa wildlife research. *Herpetological Conservation and Biology*, **11**, 426–433.

Sanjari, G., Ghadiri, H., Yu, B. & Ciesiolka, C.A.A. (2008) Comparing the effects of continuous and time-controlled grazing systems on soil characteristics in Southeast Queensland. *Australian Journal of Soil Research*, **46**, 443–441.

Sauberer, N., Peter, K., Abensperg-Traun, M., Berg, H., Bieringer, G., Milasowszky, N., Moser, D., Plutzar, C., Pollheimer, M., Storch, C., Tro, R. & Zechmeister, H. (2004) Surrogate taxa for biodiversity in agricultural landscapes of eastern Austria. *Biological Conservation*, **117**, 181–190.

Savory, T.H. (1928) *The Biology of spiders.* London: Sidgwick and Jackson, England.

Savory, A. (1983) The Savory Grazing Method or Holistic Resource Management. *Rangelands*, **5**, 155–159.

Savory, A. & Butterfield, J. (1999) *Holistic Management: a new framework for decision making* (2nd Edition). Island Press, Washington DC, Covelo, California, United States of America.

Savory, A. (2013) Response to request for information on the science and methodology underpinning holistic management and holistic planned grazing, online at

http://www.savoryinstitute.com, accessed on the 12th January 2015.

Savory, A. & Parsons, S.D. (1980) The Savory grazing method. *Rangelands*, **2**, 234–237.

Scharff, N., Coddington, J.A., Griswold, C.E., Gustavo, H. & Bjorn, P.D.P. (2003) When to quit? estimating spider species richness in northern European deciduous forest. *Journal of Arachnology*, **31**, 246–273.

Schwerdt, L., De Villalobos, A.E. & Miles, F.P. (2018) Spiders as potential bioindicators of mountain grasslands health: the Argentine tarantula *Grammostola vachoni* (Araneae, Theraphosidae). *Wildlife Research*, **45**, 64–71.

Sebata, S. (2015) Species composition and abundance of the spider fauna of the Hillside dams conservancy, Bulawayo, Zimbabwe. *Researcher*, **7**, 44–49.

Sebata, S., Muteveri, T. & Ndlovu, E. (2015) Spider diversity and abundance in an array of protected and distributed areas of Mguza Nature Reserve, Bulawayo, Zimbabwe. *Report of the proceedings of the Environmental Youth Symposiums*, **1**, 7-9.

Seymour, C.L. & Dean, W.R.J. (1999) Effects of heavy grazing on invertebrate assemblages in the Succulent Karoo, South Africa. *Journal of Arid Environments*, **43**, 267–286.

Shahabuddin, Hasanah, U. & Nahdi, E. (2014) Effectiveness of dung beetles as bioindicators of environmental changes in land use gradient in Sulawesi, Indonesia. *Biotropia*, **21**, 33-63.

Sidding, A.A.H., Ellison, A.M., Ochs, A., Villar-Leeman, C. & Lau, M.K. (2016) How do ecologists select and use indicator species to monitor ecological change？ insights from 14 years of publication in Ecological Indicators. *Ecological Indicators*, **60**, 223–230.

Silva, D.A. & Coddington, J.A. (1996) *Spiders of Pakitza (Madre de Dios, Peru): species richness and notes on community structure.* Edited by Wilson, D.E & Sandoval, A. The Biodiversity of Southern Peru Smithsonian Institution, Washington, DC, United States of America.

Sørensen, L.L. (2004) Composition and diversity of the spider fauna in the canopy of a montane forest in Tanzania. *Biodiversity and Conservation*, **13**, 437–452.

Sørensen, L.L., Coddington, J.A. & Scharff, N. (2002) Inventorying and estimating subcanopy spider diversity using semiquantitative sampling methods in an Afromontane forest. *Environmental Entomology*, **31**, 319–330.

Spafford, R.D. & Lortie, C.J. (2013) Sweeping beauty: is grassland arthropod community composition effectively estimated by sweep netting？ *Ecology and evoluton*, **3**, 3347–3358.

Spence, J.R. & Niemela, J.K. (1994) Sampling carabid assemblages with pitfall traps: the madness and the method. *Canadian Entomologist*, **126**, 881–894.

Standen, V. (2000) The adequacy of collecting techniques for estimating species richness of grassland invertebrates. *Journal of Applied Ecology*, **37**, 884–893.

Stevenson, B.G. & Dindal, D.L. (1982) Effect of leaf shape on forest litter spiders: community organization and microhabitat selection of immature *Enoplognatha ovata* (Clerck)(Theridiidae). *Journal of Arachnology*, **10**, 165–178.

Stinner, D.H., Stinner, B.R. & Martsolf, E. (1997) Biodiversity as an organizing principle in agroecosystem management: case studies of holistic resource management practitioners in the United States of America. *Agricultre, Ecosystems and Environment*, **62**, 199–213.

Suthar, S. (2009) Earthworm communities a bioindicator of arable land management practices: a case study in semiarid region of India. *Ecological Indicators*, **9**, 588–594.

Syaripuddin, K., Sing, K. & Wilson, J.J. (2015) Comparison of butterflies, bats and beetles as bioindicators based on four key criteria and DNA barcodes. *Tropical Conservation science*, **8**, 138–149.

Szinetar, C. & Samu, F. (2012) Intensive grazing opens spider assemblage to invasion by disturbance-tolerant species. *Journal of Arachnology*, **40**, 59–70.

Szlep, R. (1961) Developmental changes in the web spinning instict of Uloboridae: construction of the primary-type web. *Behaviour*, **17**, 60-70.

Takada, M., Baba, Y.G., Yanagi, Y., Terada, S. & Miyashita, T. (2008) Contrasting responses of web-building spiders to deer browsing among habitats and feeding guilds. *Environmental Entomology*, **37**, 938-946

Taylor, R.J. & Doran, N. (2001) Use of terrestrial invertebrates as indicators of the ecological sustainability of forest management under the Montreal Process. *Journal of Insect Conservation*, **5**, 221–231.

Teague, R. & Barnes, M. (2017) Grazing management that regenerates ecosystem function and grazingland livelihoods. *African Journal of Range and Forage* Science, **34**, 77-86.

Teague, W.R., Dowhower, S.L., Baker, S.A., Haile, N., Delaune, P.B. & Conover, D.M. (2011) Grazing management impacts on vegetation, soil biota and soil chemical, physical and hydrological properties in tall grass prairie. *Agriculture, Ecosystems and Environment*, **141**, 310–322.

Teder, T., Moora, M., Roosaluste, E., Zobel, K. & Meelis, P.A. (2007) Monitoring of biological diversity: a common-ground approach. *Conservation Biology*, **21**, 313–317.

Toti, D.S., Coyle, F.A. & Miller, J.A. (2000) A structured inventory of appalachian grass bald and heath bald spider assemblages and a test of species richness estimator perfomance. *Journal of Arachnology*, **28**, 329–345.

Tucker, R.W.E. (1923) The Drassidae of South Africa (Arachnida). *Annals of the South African Museum*, **19**, 251–437.

Turnbull, A.L. (1973) Ecology of the true spiders (Araneomorphae). *Annual Reviews Entomology*, **18**, 305–348.

Turnbull, A.L. (1960) The prey of the spider *Linyphia triangularis* (Clerk) (Araneae, Linyphiidae). *Canadian Journal of Zoology*, **38**, 859-873.

Uetz, G.W. (1979) The influence of variation in litter habitats on spider communities. *Oecologia*, **40**, 29–42.

Vargas, A.J. (2000) Effects of fertilizer addition and debris removal on leaf litter spider communities at two elevations. *Journal of Arachnology*, **28**, 79–89.

Vogel, B.R. (1965) *Fishing spiders* (*Dolomedes*) *at Powdermill nature reserve: Educational. Release. No. 59.* Powdermill Nature reserve, Carnegie Museum, Pittsburgh, Unites States of America.

Wagner, J.D., Toft, S. & Wise, D.H. (2003) Spatial stratification in litter depth by forest floor spiders. *Journal of Arachnology*, **31**, 28–39.

Ward, D.F., New, T.R. & Yen, A.L. (2001) Effects of pitfall trap spacing on the abundance, richness and composition of invertebrate catches. *Journal of Insect Conservation*, **5**, 47–53.

Warui, C.M., Villet, M.H. & Young, T.P. (2004) Spiders (Arachnida: Araneae) from black cotton soil habitats of a highland savanna biome in Laikipia, central Kenya. *Journal of Afrotropical Zoology*, **1**, 13–24.

Warui, C.M., Villet, M.H., Young, T.P. & Jocque, R. (2005) Influence of grazing by large mammals on the spider community of a Kenyan savanna biome. *Journal of Arachnology*, **33**, 269-279.

Wassenaar, T.D. & Dippenaar-Schoeman, A.S. (2006) A checklist of spiders from the herbaceous layer of a coastal dune forest ecosystem at Richards Bay, KwaZulu-Natal, South Africa (Arachnida: Araneae). *African Invertebrates*, **47**, 63–70.

Weber, K.T. & Gokhale, B.S. (2011) Effect of grazing on soil-water content in semiarid rangelands of southeast Idaho. *Journal of Arid Environments*, **75**, 464–470..

Weeks, R.D. & Holtzer, T.O. (2000) Habitat and season in structuring ground-dwelling spider (Araneae) communities in a shortgrass steppe ecosystem. *Enviromental Entomology*, **29**, 1164–1172.

Wesolowska, W. (1999a) New and little known species of jumping spiders from Zimbabwe. *Arnoldia Zimbabwe*, **10**, 145–176.

Wesolowska, W. (1999b) A revision of the spider genus *Menemerus* in Africa (Araneae : Salticidae). *Genus*, **10**, 251–353.

Wesolowska, W. & Cumming, M.S. (1999) The first termitivorous jumping spider (Araneae: Salticidae). *Bulletin of the British Arachnological Society*, **11**, 204–208.

Wesolowska, W. & Cumming, M.S. (2002) *Mashonarus guttatus,* gen. and sp.n., the second termitivorous jumping spider from Africa (Araneae: Salticidae). *Bulletin of British Arachnological Society*, **12**, 165–170.

Wesolowska, W. & Cumming, M.S. (2008). Taxonomy and natural history of a species rich assemblage of jumping spiders (Araneae: Salticidae): a long-term study of a suburban site in Zimbabwe. *Annales Zoologici*, **58**, 167–230.

Wesolowska, W. & Cumming, M.S. (2011) New species and records of jumping spiders (Araneae, Salticidae) from Sengwa Wildlife Research Area in Zimbabwe. *Journal of Afrotropical Zoology*, **7**, 75–104.

Wesolowska, W. & Szeremeta, M. (2001) A revision of the antlike Salticid genera *Enoplomischus* Giltay, 1931, Kima Peckham & Peckham, 1902 and *Leptorchestes* Thorell, 1870 (Araneae: Salticidae). *Insect Systematics and Evolution*, **32**, 217-240.

Whitmore, C., Slotow, R., Crouch, T.E. & Dippenaar-Schoeman, A.S. (2002) Diversity of spiders(Araneae) in a savanna reserve, Northern province, South Africa. *Journal of Arachnology*, **30**, 344–356.

Wilder, S.M. & Rypstra, A.L. (2008) Sexual size dimorphism predicts the frequency of sexual nannibalism within and among species of spiders. *American Naturalist*, **172**, 431-440.

Willms, W.D., Smoliak, S. & Dormaar, J.F. (1990) Vegetation response to time-controlled grazing on mixed and fescue prairie. *Journal of Range Management*, **43**, 513–517.

Work, T.T., Buddle, C.M. & Spence, J.R. (2002) Pitfall trap size and capture of three taxa of

litter-dwelling arthropods: implications for biodiversity studies. *Enviromental Entomology*, **31**, 438–448.

World Spider Catalog. (2020) World Spider Catalog. Version 20.0. Natural History Museum Bern, online at https://wsc.nmbe.ch, accessed on 22 January 2020.

Yi, Z., Jinchao, F., Dayuan, X., Weiguo, S. & Axmacher, J.C. (2012) A comparison of terrestrial arthropod sampling methods. *Journal of Resources and Ecology*, **3**, 174–182.

Zisadza-Gandiwa, P., Gandiwa, E., Jakarasi, J., Van Der Westhuizen, H. & Muvengwi, J. (2013). Abundance, distribution and population trends of Nile crocodile *Crocodylus niloticus* in Gonarezhou National Park, Zimbabwe. *Water South Africa*, **39**, 165–170.

Chapter 2

Temporal and vegetation effects on spider assemblages around previously kraaled inclusions and control sites at Debshan Ranch, Zimbabwe

2.1 Abstract

The multiple benefits of holistic planned grazing are attracting considerable attention. These, among others, include increased weed control, soil organic matter, and grass health. However, its impact on the arthropod fauna has not been studied yet. A matched pair design (short duration kraaled inclusions vs. unkraaled sites) in three selected sites within the miombo woodlands at Debshan Ranch in western Zimbabwe was used to assess the response of spider diversity to holistic planned grazing at six time intervals: before, during, 1, 3, 6 and 10 months after cattle introduction. At each of the six sites, sampling points were positioned 50, 100, 200 and 400 m along four perpendicular transects leading away from the cattle kraal, for a total of 16 sampling points. At every sampling point spiders were collected by means of sweep nets and pitfall traps. A total of 4192 individuals representing 173 species in 145 genera and 33 families were caught. Results indicated contrasting response to previously kraaled sites and to vegetation structure variables by each functional group. Mean grass height (cm) and time since kraal removal were the most important predictors of spider diversity. Ground dwellers genera richness contributed 36.98 % of the variation explained by the models which was above the 13.17 % and 23 .47 % contributed by the web builders and plant dwelling respectively. Similarly, the ground dwelling abundance model fixed factors also contributed the greatest variance of 66.62 % while the web-builders and plant-dwelling contributing to only 14.92 % and 28.98 % respectively. In addition 14 of the 25 individual genera of the ground dwelling responded contrastingly to vegetation structure variables, which were suggestive of the level of sensitivity of this functional group to disturbance, making them a useful functional group that can be utilised to focus on the impact of holistic planned grazing even at a local scale.

2.2 Introduction

Rangelands are ecologically important as a source of food for most wildlife and domestic animals (Walker & Janssen 2002), clean water, recreational space, natural medicines and religious sites (Havstad *et al*. 2007), amongst other services. Yet, in most developing countries

they are prone to degradation, mainly as a result of overgrazing (Neely & Butterfield 2004), poor range management and poor government policies (Savory & Parsons 1980). Globally, it is estimated that 3.4 billion ha of lands are foraged by livestock, and 73 % of these rangelands are projected to be suffering from soil dilapidation (Gabathuler *et al*. 2009). Regardless of the tremendous research efforts on the problem by universities, research stations, ranchers and conservation professionals (Savory & Parsons 1980), range deterioration remains one of the major problems affecting agriculture globally. As a result, range management has not only been a concern to ranchers within industrialized nations such as the United States of America, but also to most developing countries on the African continent (Savory & Parsons 1980).

According to Abel & Blaikie (1989), range management in Zimbabwe is a critical issue that dates as far back as the pre-colonial period, though attempts to manage rangelands were initiated only during the colonial through to the post-colonial period. The majority of these rangeland management approaches were often met with suspicion and hostility by most local communities (Cousins 1992), with only a few selected areas where they were deemed popular and met with no opposition (Weinrich 1975). Initial attempts included the concept of centralization, whereby communal lands were divided into blocks of arable and grazing lands (Weinrich 1975). This was later on followed by destocking practices that involved issuing permits that regulated the average number of cattle held per family, which at times led to a reduction of almost 60 % of the herd (Passmore 1972). The Native Land Husbandry Act (NLHA) of 1951 was received with great opposition by the local communities, as it was also another attempt utilised to control stock through the issuing of grazing licenses' (Duggan 1980). An idea that was received with less resistance was the introduction of communal grazing schemes, which included identification of land that was usually fenced within each community and managed through short duration grazing (Froude 1974).

Currently, the country has three main grazing systems that are in practice, namely communal grazing systems (CG), small-scale farming systems (SSC) and large scale farming systems (LSC). According to Gusha *et al*. (2017), the CG is by far the most deleterious system in Zimbabwe when compared to SSC and LSC. This is mainly due to the lack of a rest period, due to the absence of fences that allow for plant recovery, leading to repeated consumption of plants until they are stressed or die. As a result, overgrazing occurs with a decrease of palatable species, an upsurge of unpalatable species and soil erosion, leading to rangeland deterioration (Neely &

Butterfield 2004). Despite its impact on biodiversity, grazing still remains as an essential conservation tool utilised to maintain plant communities by most rangeland management practices (Lyons 2017).

Rangelands are the main inexpensive source of livestock feed (Gambiza & Nyama 2006) for most communities. However, in the absence of proper management they continue to be prone to deterioration. Stoddart (1967) defines range management as "the care of natural grazing lands that involves planning and administering the use of rangelands to obtain maximum livestock or game production, consistent with the conservation of the range resources". His view is that a technical range manager should have a methodology in place that will assist in determining the actual forage producing performance rather than its capacity, which when overestimated may lead to deterioration. Thus, improvements for sustainable utilisation of range are of utmost importance to ranch managers and local communities. One rangeland management practice that has been advocated among Zimbabweans in recent decades to manage rangelands sustainably is the utilisation of livestock with increased stocking rates (Savory 1983).

The early Savory's method management schemes were mainly between private consultants and ranchers in the early 1980s that utilised grazing cells to overcome stress resulting from the frequent moves (Savory & Parsons 1980). In 1992, a partnership between the Africa Centre for Holistic Management in Zimbabwe and the Wange Community, on an 8000 ha area of state and privately owned land known as Dimbangombe Ranch, was initiated (Neely & Butterfield 2004). In order to improve soil aeration, seed germination, water penetration, productivity and species diversity within the ranch, the Savory method was implemented (Neely & Butterfield 2004). After two years the project recorded improved forage production and ground cover. In 2010, the center earned the Buckminster Fuller Challenge and, in 2014, the Humanitarian Water and Food Award as a "leading edge innovator in food security" (The Senior Reporting Farmer 2015). As a result of the success of the Savory method within Matabeland North, other regions such as the Chivu communal area within the Manicaland province, which is also a dry area with little forage and poor rainfall, sought to put in place plans with the aid of a local NGO to also introduce holistic planned grazing within this community, which is situated 529 km south-east of Dimbangombe (The Zimbabwean 2015).

The Holistic management approach in Zimbabwe has not only been introduced within communal lands, but it has also been introduced within large-scale commercial schemes such as the Debshan cattle ranch, located within the Matabeland North province, Insiza district, Shangani, Zimbabwe. This ranch initially utilized a conventional paddocking system, but it changed its grazing system to Holistic planned grazing in 2012 (Nyoni 2015). This was largely as a consequence of the upsurge in the elephant (*Loxodonta africana*) population that destroyed paddock fencing, thus rendering fence management costly, as well as the abundant number of cattle that were being exterminated by predators such as leopards (*Panthera pandus*) and hyena (*Crocuta crocuta*) (Mberi 2013). When utilised as a management tool, holistic planned grazing seeks to improve animal vigor and land restoration (Baxter *et al.* 2015).

Grazing is amongst several other factors that influence rangelands (Savory 1983), causing changes in abundance, population structure and diversity of plants (Bullock *et al*. 2001, Landsberg *et al*. 2003). The treading by animals has an effect on soil texture while the mosaic pattern of biotopes is usually as a result from selective grazing by animals (Szmatona-Turi *et al*. 2018). Furthermore, manure from livestock contributes to increasing biodiversity as the former provides food for several arthropod species (Jankielsohn *et al.* 2001). The selection of grazing species is important due to differences of grazing strategies of mammals. Sheep prefers short vegetation, while cattle consume tall vegetation, as well (Kelemen 1997). Similarly, cattle have been reported to have a superior impact on the spider fauna compared to large mammalian herbivores, mostly as a result of their high densities compared to that of wildlife in most rangelands (Warui *et al*. 2005). Presence or absence of spider species has been attributed to changes in vegetation structure as a result of grazing (Dennis 2003, Churchill &Ludwig 2004). High structural heterogeneity occurs within low to moderate intensities, which is most likely to promote spider community conservation (Foord *et al.* 2008). A major cause in decrease of abundance and richness of ground dwelling arthropods is overgrazing (Gibson *et al.* 1992, McCraken and Tallowin 2004), and intensively managed pastures have been shown to harbour depleted arthropod fauna, whereas extensive management is usually associated with high levels of biodiversity (Wettstein and Schmid 1999).

A key element of the holistic management approach is the utilisation of short duration overnight corrals (Savory & Butterfield 1999, Sibanda *et al.* 2016, Huruba *et al.* 2018), with period of occupation varying from 7 days to at times more than 4 week periods (Huruba *et al.* 2018,

Veblen & Porensky 2019). Research focusing on the effects of short duration kraaling have grown rapidly in the recent decades, with reports of improved quality of grasses due to the increased quantities of edible species, diversity and grass nutrient composition (Huruba *et al.* 2018) in abandoned kraal sites. It also enhances kraal visitation after abandonment by other wildlife. For example, the increased attraction of previously kraaled sites to warthogs (*Phacochoerus aethiopicus*) has been attributed to the decrease in biomass and basal cover in these sites. In addition, the high quality nutrient resprouts that occur in most woody plant stems as a result of being feed on by livestock have seen the highest utilisation of the kraals after abandonment by the mesoherbivore, impala (*Aepyceros melampus*), as early as within a period of 4 weeks after kraal removal.

A study done by Sibanda *et al.* (2016) within the Dimbangombe Ranch also supports the concept of short duration kraaling, as they reported improved grass quality, higher biomass and greater species diversity within kraaled sites. Other benefits of the holistic management approach, as identified by Savory & Parsons (1980) include the improvement of ranch economics, livestock improvement, range improvements and increased carrying capacity of the land. Despite the growing interest in holistic management approach, limited research is available on the response of invertebrates in general to holistic planned grazing. Therefore, in order to address the information gap, evaluation of the effects of short duration kraaling and vegetation structure on the diversity of spider assemblage of the miombo woodlands at Debshan Ranch, Zimbabwe was done.

The aims of this study were to: i) describe changes in mean grass height (cm) and percentage grass cover over the six sampling periods around previously kraaled inclusions and their control sites; ii) to determine the effect of previously kraaled inclusions on spider assemblages; iii) to determine the significance of geographic distances on spider assemblages around previously kraaled inclusions and their control sites; (iv) to determine the influence of time since kraal removal on spider assemblages around previously kraaled inclusions and their control sites; (v) to evaluate the effect of vegetation structure on spider assemblages around previously kraaled inclusion and their control sites.

2.3 Materials and Methods

2.3.1 Study area and period

The study was piloted at Debshan Ranch (29º 15′ E, 19º 35′ S), a privately owned commercial cattle ranch, from July 2016 to April 2017. The property has a total coverage of 800 km^2 (Huruba *et al*. 2017). It is located 100 km North-East of Bulawayo along the Bulawayo-Harare road within Insiza district in Matabeleland North Province. Lying adjacent to it, to the south-west is the Shangani Mine that has nickel deposits, while the eastern boundary is formed by the Shangani River (Fig. 2.1). The ranch lies at an elevation of between 1230 and 1414 m above sea level (Dunham *et al.* 2003).
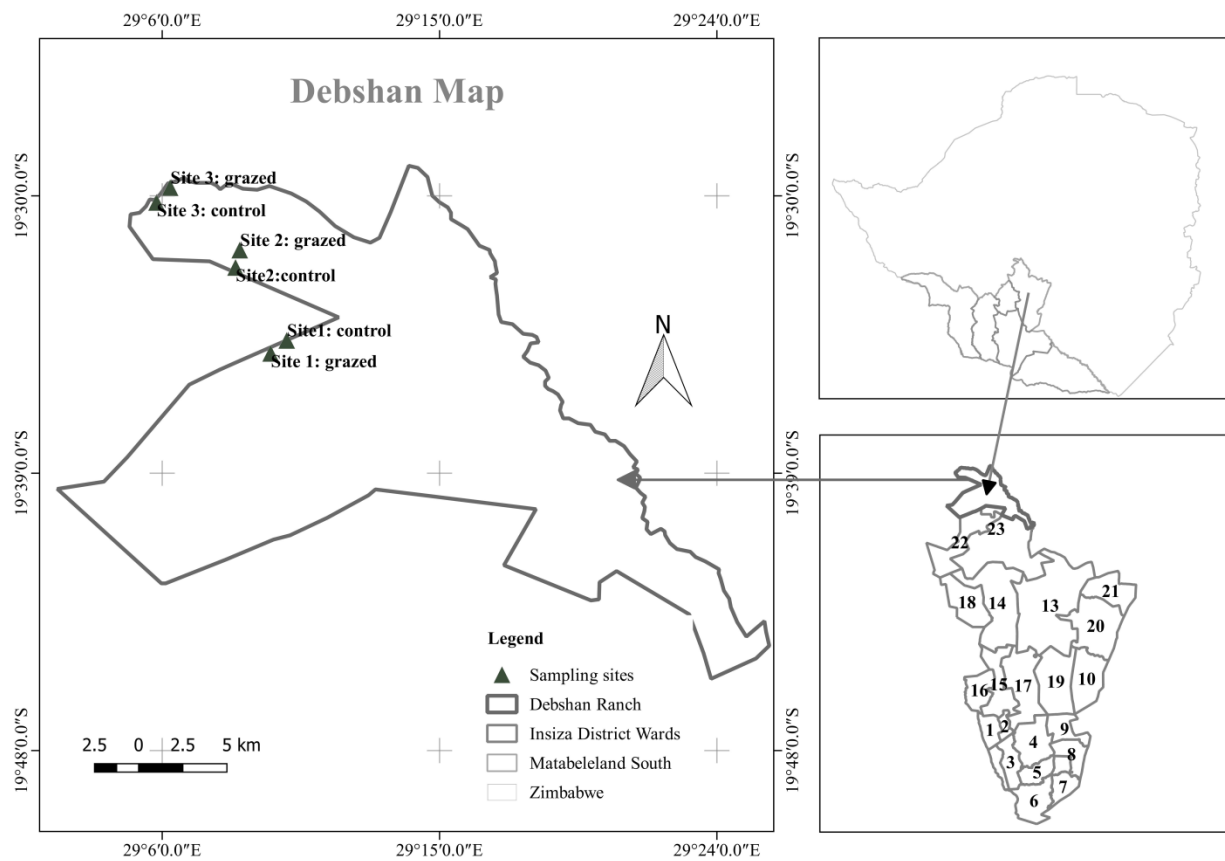
Fig. 2.1 Location of Debshan Ranch in Zimbabwe and the distribution of spider sampling sites within the miombo woodlands. Sites followed a matched pair design (short duration kraaled inclusions vs. unkraaled sites) (1 km apart).

The ranch practiced conventional paddocking for over a hundred years, with a stock rate of approximately 8 000 cattle per year preceding the National Land Reform programme, until 2012 when it switched to a holistic management approach (Mberi 2013). According to Nyoni (2015),

in the paddocking scheme, the cattle (approx. 120 head) were held in paddocks (approx. 324 ha) for approximately 30 days and then moved to another paddock after sufficient grazing. However, upon introduction of holistic management approach, livestock were split into bigger herds (approx. 350 head). Each herd was allocated to four herders that monitor it during the day, leading it to either grazing areas or water points. The cattle are kept within mobile kraals at night (Fig. 2.2). These kraals are usually small (50 x 60 m) but can be as huge as 100 x 100m dependent on the herd size, season of the year, age class of the cattle and topography (that is whether sparsely or densely vegetated) (Mberi 2013). The kraals are relocated to a new site every two weeks, and usually have a water supply within them for the duration of the occupation.

Figs 2.2 (A) Cattle inside kraal enclosures and (B) facilities such as herder's housing and lighting adjacent to the kraal at Debshan Ranch in Zimbabwe.

Zimbabwe has a subtropical climate that experiences three broad seasons: dry, hot season (spring; August to mid-November); wet, hot season (summer; mid-November to March); a dry, cold season (winter; April to July) (Climate-data.org 2019). Shangani gets an average annual rainfall of 639 mm. Most of the precipitation falls in December, averaging 144 mm, while no rainfall occurs in July. During the rainy season the rainfall pattern is subject to recurrent seasonal droughts and severe dry spells. It also characterized by an annual average temperature of 18°C, the hottest month being October, with an average temperature of 21°C, and the coldest month being July, with an average temperature of 12.4 °C. Average daily humidity is 55 % (Climate-data.org 2019)

The ranch falls under agro-ecological natural region IV (Cousins 1992). This region covers an area of 155 707 km^2, which translates to 39.9% of the total land area of Zimbabwe. It is a widespread livestock production area that also cultivates certain crops that are drought-tolerant such as sorghum, millet and rapoko. It is also a semi-extensive area suitable for forestry, wildlife and tourism (Mugandani *et al.* 2012). The landscape of the ranch is moderately undulant, and comprises coarse-grained loamy sands that are also yellowish-brown resulting from granite, forming soils that are usually infertile and poorly drained. In the south-eastern section of the ranch, the Mahwe section, the rocks are ultramafic or mafic and give rise to productive red soils, whereas along the Shangani River and other main rivers there are also productive dark brown clay soils (Robertson 2013).

The soils support flora types that are normally dispersed in a catenae pattern (Dunham *et al.* 2003) that comprises of bushlands, grasslands, wetlands and woodlands. This includes the following vegetation types: 1) *Julbernadia-Stereochlaena* woodland, with a sparse herbaceous layer dominated by unpalatable unproductive grasses such as *Stereochlaena cameroni*; 2) *Combretum hereroense-Hyparrhenia* mixed bushlands, dominated by *Combretum hereroense* and other species of *Combretum* mixed with *Vachellia* species, with a herbaceous layer of *Hyparrhenia filipendula*; 3) *Colophospermum* bushlands, with small patches of heavy textured fertile sodic soils, dominated by *Colophospermum mopane* with sparse grasses; 4) *Terminalia-Schizachyrium* bushlands and wooded grassland that include crests with deeper soils, dominated by *Terminalia sericea*, with perennial grasses such as *H. filipendula* and *Schizachyrium jeffreysii*; 5) Riverine woodland, dominated by *Vachellia karroo* on fertile dark brown clay, mostly found along the Shangani River; 6) *Hyperthelia* vlei grassland, characterised by small depressions with heavy dark clay soils mostly invaded by shrubs, with dense perennials such as *Hyperthelia dissoluta* (Robertson 2013).

2.3.2 Sampling design and sampling methods

Spiders were sampled using sweep nets and pitfall traps at six sites (Fig. 2.1), using a matched-pair design (short duration kraaled inclusions vs. unkraaled sites) with three herds (replicates), that had approximately 350 cattle per herd. To reduce heterogeneity, all six sites selected were within the miombo woodlands that were dominated by *Brachystegia* and *Julbernadia* species. Sampling was done at six time intervals (surveys), i.e. before, during, one, three, six and ten

months after cattle introduction, from July 2016 to April 2017. At each site, sampling points were positioned 50, 100, 200 and 400 m from the cattle kraal (shelter) in each cardinal direction (North, East, South and West), for a total of 16 sampling points at each site (Fig. 2.3A).
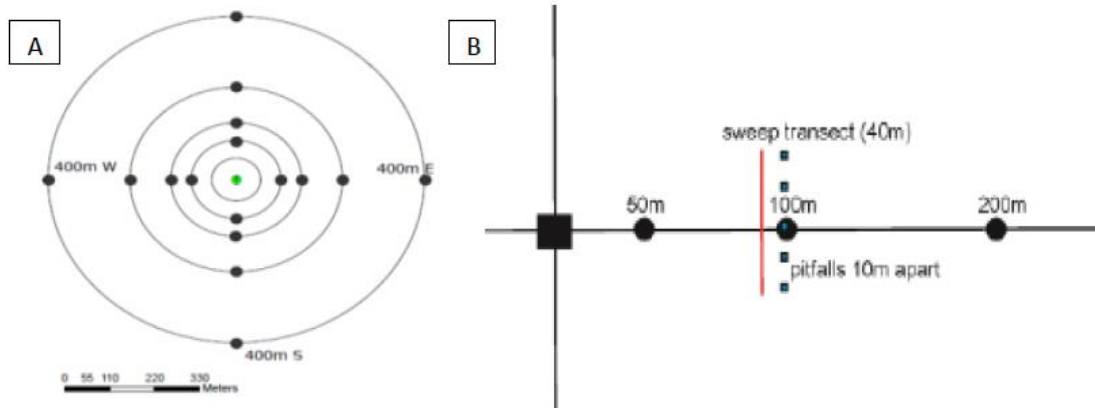
Figs 2.3 (A) Spider sampling setup at each site in Debshan Ranch, Zimbabwe and (B) placement of five pitfall traps and the sweeping transect at each sampling point.

2.3.2.1 Pitfall traps: Ground-active spiders were sampled using pitfall traps. Each trap was made of glass bottles with a diameter of 9 cm at the opening and 15 cm deep placed inside a plastic PVC pipe and buried to their rims in the ground (Fig. 2.4 A). At each sampling point, five pitfalls, 10m apart, were laid out transverse to the transect (Fig. 2.3B), leading to a total of 480 pitfall traps laid down in each of the six sampling surveys. The glass bottles were filled with 100 ml of propylene glycol. Traps were left open for 14 days and emptied in each sampling period. At the end of the collecting period the contents were collected and emptied into suitable bottles for sorting in the laboratory.

2.3.2.2 Sweep nets: Sweep-netting involved using a sweep net (50 cm in diameter with a 75 cm handle) that was swung back and forth across the grass for 50 sweeps per sampling point along each 40 m transect parallel to pitfalls (5 m from the pitfall trap line) (Fig. 2.3B). The sweep net handle was rotated so that the net stayed shut between sweeps (Fig. 2.4 B). Each sweep covered an arc of roughly 180° through the vegetation (Coddington *et al.* 1996). After 25 sweeps, samples were emptied on a plain sheet of cloth and all visible invertebrates were aspirated with a pooter. Each sample was kept separately in 70% ethyl alcohol. A total of 96 samples were collected in each of the six sampling periods.
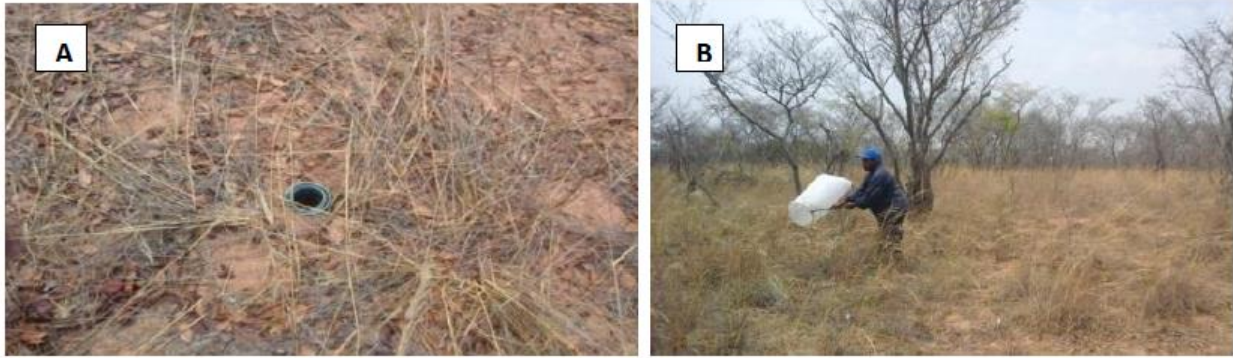
Figs 2.4 (A) Example of a pitfall trap used to sample ground-active spiders and (B) Sweeping in one of the sampling points at Debshan Ranch in Zimbabwe.

2.3.3 Identification of specimens

All adult specimens were sorted into morphospecies and identified to species where possible, with most juveniles identified to genus level. All preliminary sorting was done by the student following identification keys and taxonomic guides in Dippenaar-Schoeman & Jocqué (1997). Dr. M. J FitzPatrick and Professor Charles Haddad assisted with the identification of spiders. Spiders were broken into nine functional groups defined by Foord *et al.* (2011b): ground wanderers, plant wanderers, funnel web builders, gum-foot web builders, modified orb-web builders, retreat-web builders, sheet-web builders, space-web builders and tube-web builders. According to Polis & McCormick (1986) functional groups include spider species that potentially compete for jointly exploited limited resources. Due to the small data sets in the finer resolutions of web builders, they were all condensed into web builders. Therefore, the resultant functional groups were web builders, plant dwelling and ground dwelling. Voucher specimens for all the mature collected species were deposited at the Arachnid collection in the Department of Arachnology, Natural History Museum in Bulawayo, Zimbabwe.

2.3.4 Vegetation structural variables

Vegetation structure was quantified by placing a 1 m x 1 m quadrat over each pitfall trap and a photograph was taken (Fig. 2.5A). During each sampling period, a total of 240 images were taken in the three previously kraaled sites (Site 1: grazed; 2: grazed; 3: grazed) and 240 images were taken in the three unkraaled sites (Site 1: control; Site2: control and 3: control) (Fig 2.1) which gave a total of 2880 images taken for the six sampling periods at Debshan ranch. Images were analysed to calculate percentage cover of each vegetation structure variable that included

bare ground, leaf litter, grass cover, coarse woody debris, cow dung, and rock. Visual estimates of each vegetation structure variable were estimated utilising methodology described in Dethier *et al.* (1993). Within the 40 m transect used for sweeping, twenty grasses placed a meter apart at each sampling point were marked with plastic ear tags (Fig. 2.5B), and measured at the tallest vertical point. Average grass height for each sampling point was determined from the measurements.
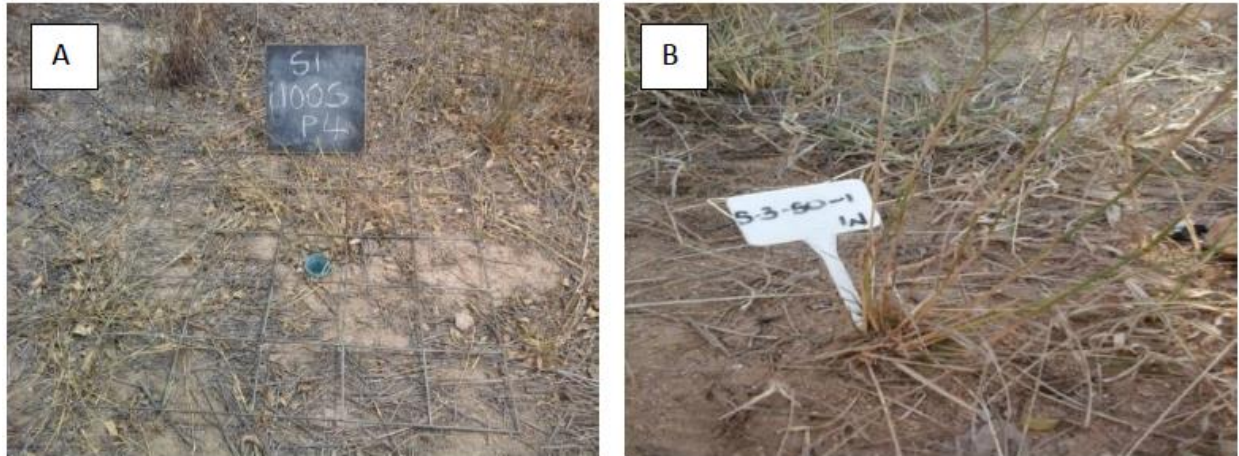
Fig. 2.5 (A) Picture illustrating placement of quadrat over a pitfall trap utilised to measure vegetation structure variables, with (B) picture illustrating a single sampling point used to measure grass height (cm) through marking by tags at Debshan Ranch in Zimbabwe.

2.3.5 Data analysis

All analysis were done using R statistical software version 3.6.0 (R Core Team 2019). In each replicate treatment five of the pitfall traps were pooled across for each survey. To determine if sampling of the spiders was representative of the overall community, coverage was calculated using the iNEXT package (Hsieh *et al.* 2016), as its inverse is essentially the likelihood that the subsequent sampled individual might be an unsampled species (Chao & Jost 2012).

To describe changes in mean grass height (cm) and percentage grass cover between the six sampling periods (before, during, 1, 3, 6 and 10 months after cattle introduction), mean grass height (cm) and percentage grass cover were included as the response variables, short duration kraaling, time since kraal removal, distance from kraal were included as the fixed variables with herd included as the random factor. Several competing models were selected that were based on prior hypothesis about variables which were suspected to potentially explain changes in the mean grass height (cm) and percentage grass cover. The model that best explained mean grass

56

height (cm) and percentage grass cover was selected using an Information theoretic approach based on Akaike Information Criterion (AIC) (Burnham & Anderson 2002) value by at least 2 units. According to Whittingham *et al.* (2006) the latter is superior to traditional stepwise procedures as in the presence of several competing models it accounts for uncertainties concerning model structure and parameter estimation on the observed data set. Percentage grass cover were arcsine transformed before analysis and linear mixed models with Gaussian distribution, using the lmer function of the lme4 package (Bates *et al.* 2013) were utilised. For mean grass height (cm) it was log transformed before the analysis (Harrison *et al.* 2018) in order to reduce heteroscedasticity.

As juvenile spiders are often difficult to identify up to species level and to include all the specimens in the analysis, genera richness was utilised in this study as it has been shown to be a good surrogate for species richness (Foord *et al.* 2013). To assess whether genera richness and abundance varied with short duration kraaling, distance from kraal, time since kraal removal and vegetation structure (Table 2.1) around previously kraaled inclusions and their control sites, Generalised Linear Mixed Models (GLMM) with the log-link function and Poisson error distribution, were used with the 'glmer' function of the lme4 package (Bates *et al.* 2013). Evidence of non-linearity in the response of spider diversity to mean grass height (cm) showed that the addition of a quadratic term improved the model (Zar 1998).

According to Whitmore *et al.* (2002), combined biodiversity has the possibility of masking spider diversity effects, as a result, assemblages were placed into three functional groups namely; web-builders, plant-dwellers and ground-dwellers that were analysed separately. Web-builders genera richness and abundance, ground-dwelling genera richness and abundance and plant-dwelling genera richness and abundance were set as the dependent variables, short duration kraaling, distance from kraal, time since kraal removal, % leaf litter cover, % cow dung, % coarse woody debris, % rocks and mean grass height (cm) as fixed variables, with herd as random factor.

Table 2.1 Site (UNK: unkraaled sites; KRA: kraaled sites) characteristics summarised by replicates for each of the vegetation structure variables measured at Debshan Ranch, Zimbabwe around previously kraaled inclusions and their control sites and used as explanatory variables in the analysis on spider fauna.

Site	Rep	n	Bare soil	Vegetation	Leaf litter	Rock	Coarse woody debris	Cow dung	Grass height (cm)
UNK	1	24	43.85±27.56	42.55±28.26	11.72±15.58	0.66±6.02	1.41±2.32	0.00	78.01±33.12
	2	24	38.39±26.83	39.58±28.37	21.06±20.06	0.00	0.88±1.82	0.01±0.18	85.94±28.01
	3	24	24.25±24.26	59.82±29.38	14.28±15.92	0.37±1.71	1.24±2.12	0.04±0.40	96.28±33.85
KRA	1	24	31.73±27.16	53.12±31.05	13.83±16.36	0.05±0.66	1.04±2.23	0.04±0.38	82.50±35.46
	2	24	40.23±28.56	42.90±30.60	15.72±20.49	0.00	0.90±1.79	0.07±0.60	76.58±32.70
	3	24	32.24±27.87	54.96±30.63	11.74±14.71	0.42±2.32	0.51±1.28	0.06±0.35	78.8±32.89

Grass height: mean± SD vegetation height (cm). Values for bare soil, vegetation, leaf litter, rock, coarse woody debris and cow dung are mean ± SD % cover.

In order to ensure that variables that differ in their units of measurement do not vary in their influence in the model mean grass height (cm) and percentage litter cover were scaled. Percentage vegetation cover had a moderately strong positive correlation with mean grass height (cm) (r = 0.5, p < 0.001)(Fig. 2.6) and was therefore not included in the analyses regardless of its significance in predicting spider diversity (Warui *et al.* 2005). Model validation was done by inspection of model residuals (Zuur *et al.* 2010, Zuur & Ieno 2016). In order to assess how much of the variance was explained by the models, conditional and marginal R^2 values were calculated (Nakagawa & Schielzeth 2013).
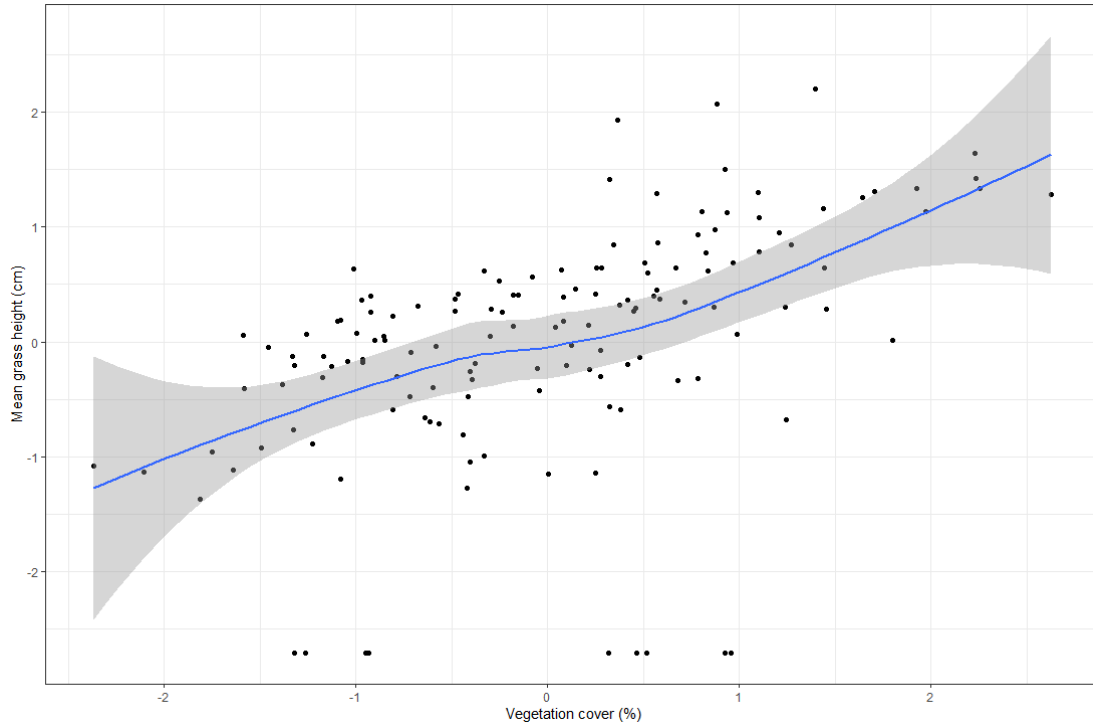
Fig. 2.6 Correlation (r= 0.5, p < 0.001) between percentage vegetation cover (veg) and mean grass height (cm) measured around previously kraaled sites at Debshan Ranch, Zimbabwe.

Variation in spider community assemblages was analysed through the utilisation of generalised linear latent models (Niku *et al.* 2019). The package Boral was used to perform model based unconstrained ordination (Hui 2016), which has been shown to outperform most distance-based multivariate analyses (Hui *et al.* 2015). The gllvm package was utilised to perform model selection and residual analysis (Niku *et al.* 2019). Spider communities in each functional group were modeled using two latent variables with negative binomial distribution to visualise the grouping of communities across kraaling and time since kraal removal. In order to determine the potential role of, and interaction between, short duration kraaling, distance from kraal and time since kraal removal on spider communities, multivariate generalised linear models were performed with the function "manyglm" in the package mvabund (Wang *et al*. 2012). This technique fits many GLM simultaneously to variables, and 'anova.manyglm' was utilised for hypothesis testing. Univariate test statistics and their corresponding p-values were calculated for each species in order to determine their respective contributions to each model. Model residuals, normality and independence, and constant-mean variance relationships were inspected in all cases.

2.4 RESULTS

2.4.1 Community composition

A total of 4192 spiders, representing 173 species, 145 genera and 33 families, were collected during the study period. All species are new records for Debshan Ranch, as there had been no prior sampling at this locality. Ten species and one genus are possibly new based on identifications of specialists (Appendix 2.1). Pitfall traps accounted for 2540 individuals, overall, the ground-dwelling fauna was dominated by *Stenaelurillus guttiger* (Simon, 1901) (18.3%). Sweep-nets accounted for 1652 individuals; overall grass-dwelling fauna was dominated by *Monaeses* spp.) (6.5%). Sample based rarefaction curves showed higher genera richness of spider assemblages in the unkraaled sites as compared to the kraaled sites although confidence intervals did overlap (Fig. 2.7). Furthermore, sample coverage for both the unkraaled and the kraaled sites was relatively high with unkraaled sites and the kraaled sites recording (98.59%) and (98.69%) coverage respectively, suggesting that sampling captured a significant portion of the spider assemblages at Debshan Ranch.
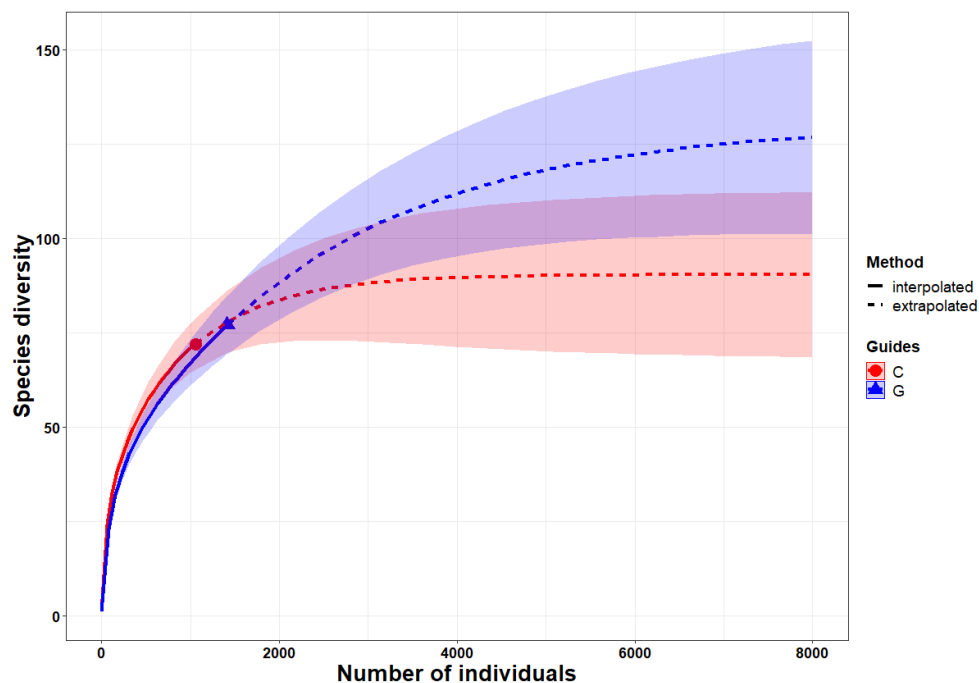
Fig. 2.7 Sample sized based rarefaction and extrapolation sampling curves with 95% confidence intervals based on a bootstrap method with 200 replications comparing spider genera richness between the unkraaled sites (C) and the kraaled sites (G). Solid lines = rarefaction; dotted lines = extrapolation; confidence intervals = shaded areas.

The most abundant functional group were the free-living ground dwellers (n = 2309, 55.08%), which were dominated by the genera *Stenaelurillus* (27.8%) (Salticidae) and *Asemesthes* (16.61%) (Gnaphosidae), then plant dwellers (n = 1043, 24.88%) which were also dominated by the genera *Thyene* (16.20%) (Salticidae), and *Monaeses* (11.31%) (Thomisidae) and the least abundant functional group were the orb-web builders (n= 499, 11.9%). The remaining functional group were mostly different web builders that constituted less than 8% (341) of the total spider fauna, individually contributing less than 3% (Fig. 2.8).
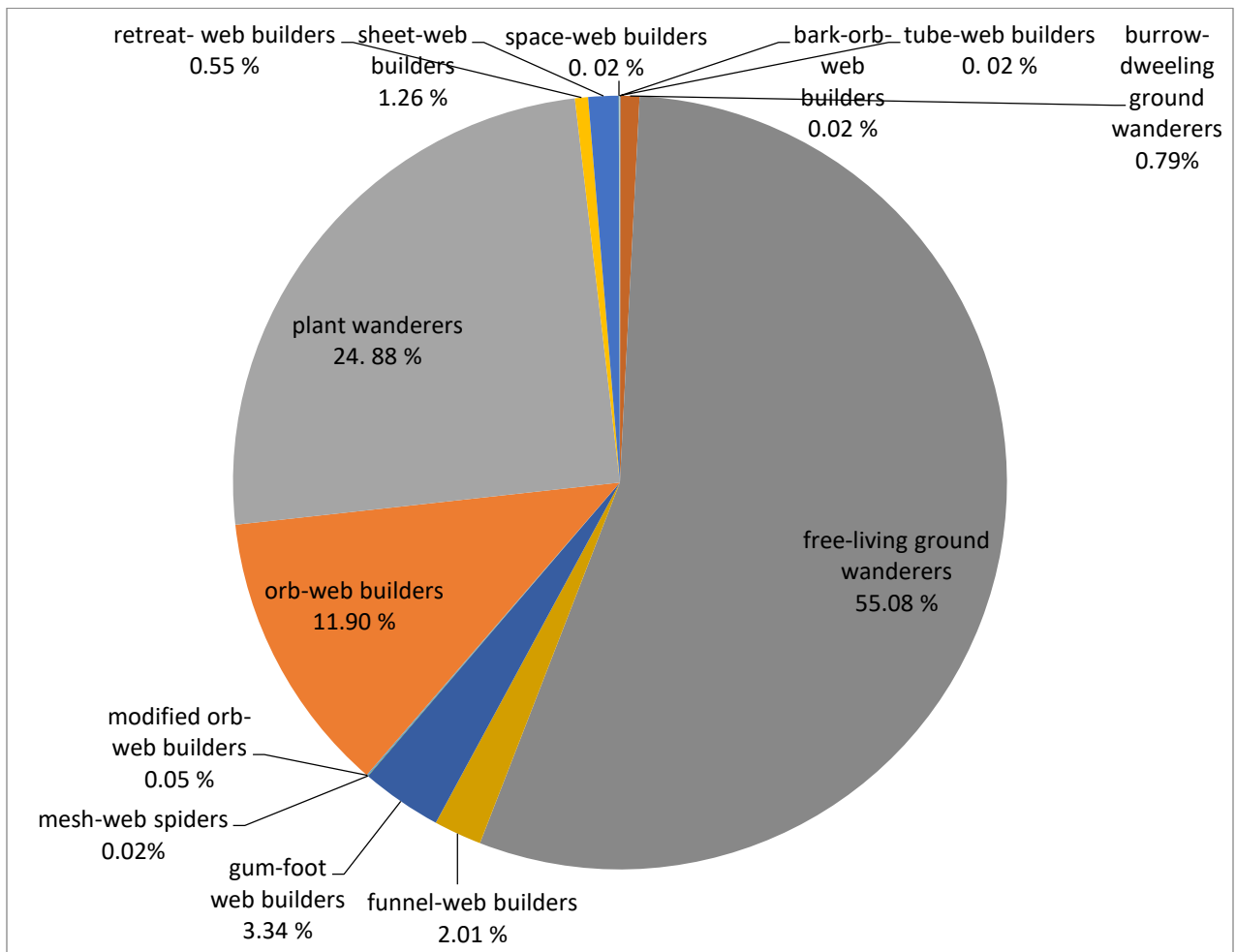
Fig. 2.8 Relative proportion of each functional group sampled by pitfall trapping and sweep-netting within the miombo woodlands at Debshan Ranch, Zimbabwe from July 2017 till April 2018. Classification of spiders into functional groups was adapted from (Foord *et al.* 2011a).

2.4.2 Changes in mean grass height (cm) and percentage grass cover during the sampling periods

Mean grass height (cm) decreased continuously from the first sampling period until 6 months after kraal removal, with 10 months since kraal removal recording the highest mean grass height (cm) in all the sampling periods (Fig. 2.9). This variation in mean grass height (cm) within the miombo woodlands was best elucidated by the model that included time since kraal removal compared to the other models (Table 2.2).
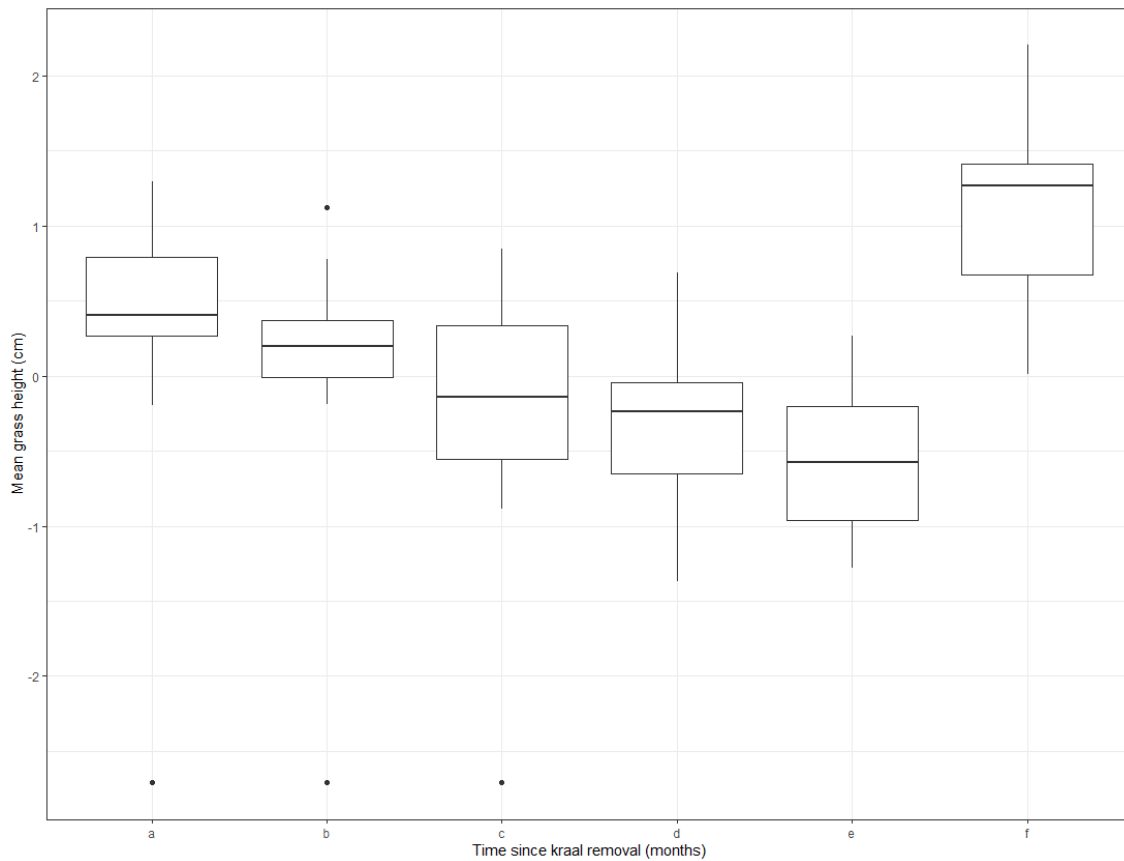
Fig. 2.9 Differences of mean grass height at Debshan Ranch, Zimbabwe over the six sampling period from July 2016 to April 2017. The box bounds the 25th and 75th percentiles, while the whiskers represent $\pm$ 1.5 IQR (interquartile range). Dots represent outliers. Sample size = 24 observations in each period since kraal removal. Abbreviations; a = before cattle introduction into kraal, b = during cattle occupation of the kraal, c = 1 month since kraal removal, d = 3 months since kraal removal, e = 6 months since kraal removal, f = 10 months after kraal removal.
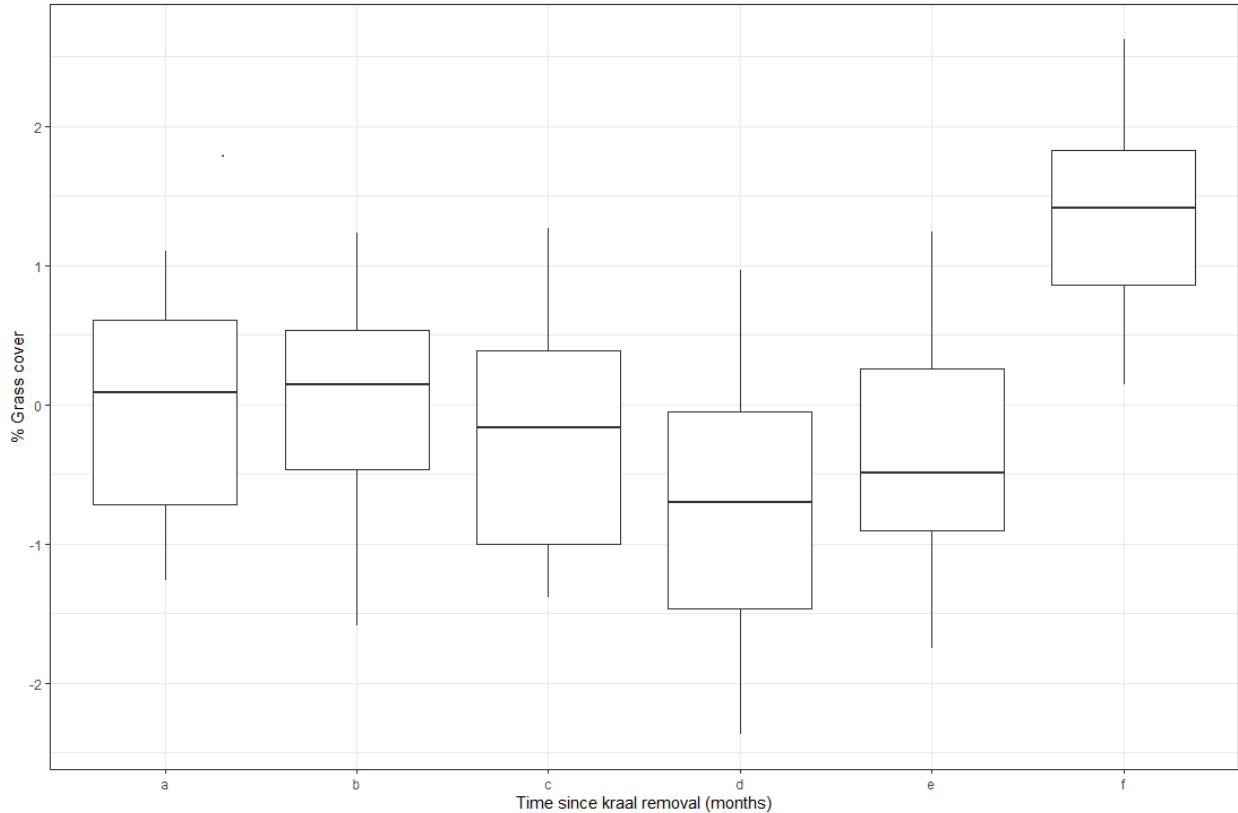
Fig. 2.10 Differences of grass cover at Debshan Ranch, Zimbabwe over the six sampling period from July 2016 to April 2017. The box bounds the 25th and 75th percentiles, while the whiskers represent ± 1.5 IQR (interquartile range). Dots represent outliers. Sample size = 24 observations in each period since kraal removal. Abbreviations; a = before cattle introduction into kraal, b = during cattle occupation of the kraal, c = 1 month since kraal removal, d = 3 months since kraal removal, e = 6 months since kraal removal, f = 10 months after kraal removal.

Percentage grass cover recorded during the sampling period of before cattle introduction and during cattle occupation were similar in their percentage grass cover, with a decrease in percentage grass cover occurring during the 1st and 3rd months since kraal removal period, however the 6 months since kraal period recorded a higher percentage cover than 3 months since kraal removal period but was less than the previous sampling periods. The period of 10 month since kraal removal recorded the highest grass percentage cover amongst all other sampling methods (Fig. 2.10). This variation in percentage grass cover within the miombo woodlands was best elucidated by the model that included time since kraal removal compared to the other models (Table 2.2).

Table 2.2 Generalised linear mixed models that explain the effects of holistic planned grazing, distance from kraal and time since kraal removal on mean grass height (cm) and percentage grass cover measured at Debshan Ranch, Zimbabwe. Holistic planned grazing had two levels (short duration kraaled inclusions vs. unkraaled sites), distance from kraal had four levels (50m, 100, 200m and 400m) away from the kraal and time since kraal removal had six level (before, during, 1, 3, 6, 10 months since cattle occupation). Abbreviations; AIC= Akaikes information criterion.

Models	K	Log likelihood	AICc	Delta AIC	weight
Mean grass height (cm)					
Time since kraal removal	8	-212.459	442.0	0.00	0.589
Constant	3	-218.650	443.5	1.49	0.280
Time since kraal removal+ Short duration kraaling treatment	9	-213.247	445.8	3.85	0.086
Short duration kraaling treatment	4	-219.399	447.1	5.10	0.046
Percentage grass cover					
Time since kraal removal	8	97.411	-177.8	0.00	0.943
Time since kraal removal + Short duration kraaling treatment	9	95.750	-172.2	5.6	0.057

2.4.3 Abundance and diversity patterns

The most important predictor was mean grass height (cm) and was negatively correlated with ground dwelling genera richness and abundance and web-builders genera richness and abundance. In contrast, plant dwelling genera richness and abundance were positively associated with mean grass height (cm). With the exclusion of plant dwelling genera richness and abundance, time since kraal removal had a positive relationship with ground dwelling genera richness and abundance and web-builders genera richness and abundance. Leaf litter cover was positively correlated with ground dwelling genera richness and abundance (Table 2.3).

2.4.3.1 Ground dwellers richness and abundance

Ground dwelling genera richness was negatively and significantly influenced by mean grass height (cm) (Fig. 2.11, Table 2.3). It was also positively influenced by time since kraal removal and percentage leaf litter cover (Table 2.3). Fixed variables of the ground dwelling genera richness data explained 36.98 % of the variation with the random factors only explaining an additional 1.18% of the variance. The abundance of ground-dwellers was also negatively and significantly influenced by mean grass height (cm) and positively influenced by percentage leaf

litter cover. The fixed variables of the ground dwellers abundance data explained 66.62% of the variance while the random factor explained only an additional 1.62 % of the variation.
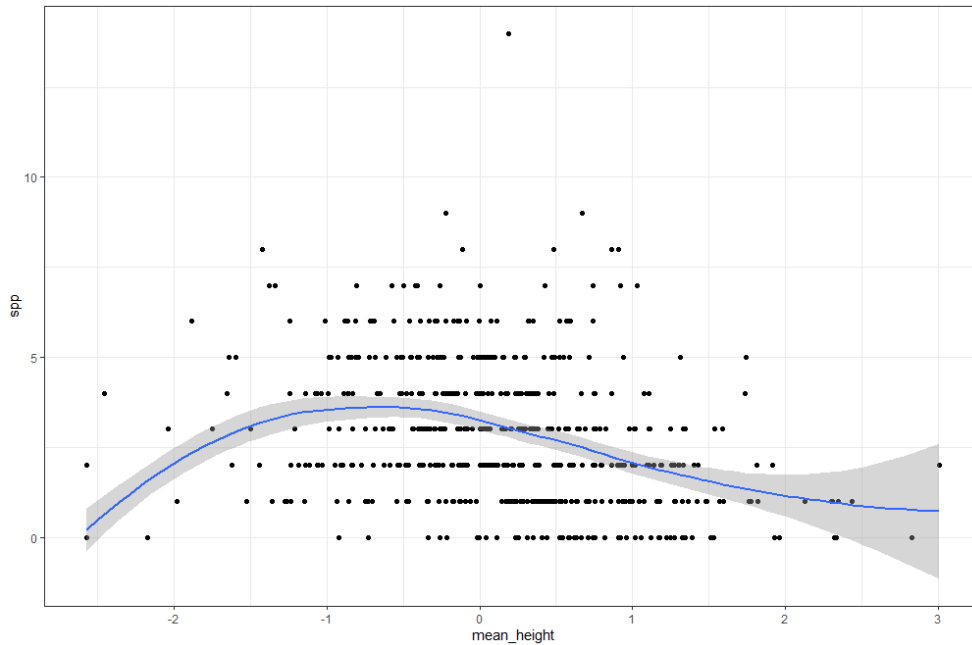
Fig. 2.11 Ground dwelling spiders genera richness sampled at Debshan Ranch, Zimbabwe as a function of mean grass height (cm) with geom smooth line and 95% confidence intervals.

2.4.3.2 Web-builders richness and abundance

Web-builders genera richness was negatively and significantly influenced by short duration kraaling with lower genera richness in the kraaled sites than the unkraaled sites. It was also positively influenced by time since kraal removal, however recovery of the genera richness over time within the kraaled sites was slower compared to that of the unkraaled sites. Furthermore, genera richness of the web-builders also responded with a hump-shaped response to mean grass height (cm) with diversity peaking at intermediate heights but they did not decrease with increasing grass height (cm) as observed for the ground dwellers. Fixed variables of the web-builders data explained 13.17% of the variation, while random effects did not contribute any significant variation. Web-builders abundance did not respond to short duration kraaling; however it was also negatively and significantly influenced by mean grass height (cm) and positively influenced by time since kraal removal. Fixed variables for the web-builders abundance data explained 14.92% of the variation with the random factor only explaining a further 0.15% of the variance (Table 2.3).

Table 2.3 Variables that were kept by model selection by Akaikes information criterion (AIC) on the best model on results of Generalised linear mixed models on spider functional groups: ground wanderers, web-builders and plant dwelling richness.

	Estimate [a]	Z value	p-value
Ground dwelling richness			
Intercept	1.213±0.061	20.005	<0.001
Time since kraal removal (months)	0.067±0.029	2.337	<0.05
Mean grass height (cm)	-0.310±0.028	-11.162	<0.001
Percentage leaf litter cover	0.094±0.026	3.699	<0.001
Ground dwelling abundance			
Intercept	1.916±0.063	30.410	<0.001
Mean grass height (cm)	-0.369±0.021	-17.386	<0.001
Percentage leaf litter cover	0.177±0.018	10.078	<0.001
Web-builders richness			
Intercept	0.498±0.054	9.226	<0.001
Short duration kraaling	-0.157±0.074	-2.117	<0.05
Time since kraal removal (months)	0.231±0.058	4.006	<0.001
Mean grass height (cm)	-0.232±0.035	-6.608	<0.001
Short duration kraaling x time since kraal removal (months)	-0.146±0.086	-1.692	0.091
Web-builders abundance			
Intercept	0.748±0.051	14.585	<0.001
Time since kraal removal (months)	0.183±0.051	3.613	<0.001
Mean grass height (cm)	-0.209±0.030	-7.166	<0.001
Short duration kraaling x time since kraal removal	-0.1334±0.075	-1.784	0.0744
Plant dwelling richness			
Intercept	0.535±0.040	13.526	<0.001
Mean grass height (cm)	-0.183±0.032	-5.632	<0.001
Plant dwelling abundance			
Intercept	0.837±0.034	24.726	<0.001
Mean grass height (cm)	-0.165±0.027	-6.178	<0.001

[a] Estimates on Table 2.3 include ± standard error

2.4.3.3 Plant dwelling richness and abundance

In contrast to other functional groups plant dwelling richness and abundance responded significantly and positively only to mean grass height (cm), also with a hump-shaped response. Fixed variables of the plant dwelling richness data explained 23.47% of the variation, whereas

random effects also did not contribute any significant variation as observed for the web-builders. The fixed variables of the plant dwelling abundance data explained 28.98% of the variation whereas the random effects did not explain any additional variance.

2.4.4 Spider assemblage composition

The multivariate generalised linear models of functional genera composition revealed significant effects of time since kraal removal in all functional groups (Table 2.4). This suggest that genera composition in each functional groups differed significantly between the different periods of time since kraal removal, however each functional group responded contrastingly with time since kraal removal as revealed by the ordination plots (Fig.2.12). The ordination of web builders showed clearly distinct spider assemblages with those sampled less than 2 months and 2 months after kraal removal intermediate between assemblages sampled during 6 and 8 months after kraal removal. However, such distinction was less evident for the plant dwelling genera composition. Only the ground dwelling spiders were the only functional group that was also significantly affected by distance from kraal in its genera composition assemblage. The genera composition of all functional groups also responded in a similar manner to three vegetation structure variables (Table 2.4). Only 25 out of the 145 genera showed significance in abundance according to changes in the predictor variables (Appendix 2.2). Distance from kraal, percentage cow dung cover and percentage rock cover had no significant impact on any of the considered genera. Univariate analysis revealed five genera that were significantly impacted by short duration kraaling, 11 by time since kraal, removal, while 12 were positively and negatively impacted by percentage leaf litter cover, five by mean grass height (cm) and a single species by percentage coarse woody debris cover (Appendix 2.2).

Table 2.4. Results of multivariate generalized linear model of the abundance of all spider genera of ground wanderers, web-builders and plant dwelling identified in this study around inclusions of previously kraaled sites factors (as well as their interaction) and vegetation structure variables. Abbreviations: Kraal–short duration kraaling; Distance–distance from kraal (metres); Time-time since kraal removal (months). Significant differences are represented by Asterisks with p value from 0.05-0.01 =*, 0.1-0.001 = ** and < 0.001 = ***

Term	WB		GW		PW	
	Wald	p-value	Wald	p-value	Wald	p-value
Kraal	5.275	0.228	6.818	0.059	3.632	0.827
Distance	6.448	0.083	**7.750**	**0.048***	4.875	0.452
Time	**12.917**	**0.019***	**15.685**	**0.001*****	**11.288**	**0.001*****
Kraal: Distance	4.315	0.562	4.332	0.857	3.962	0.609
Kraaling: Time	5.598	0.052	4.819	0.668	5.120	0.067
Distance :Time	5.239	0.155	4.218	0.979	3.868	0.675
Kraaling: Distance :Time	3.195	0.871	3.532	0.942	3.685	0.461
Leaf litter cover (%)	**9.495**	**0.001*****	**13.339**	**0.001*****	**8.678**	**0.001*****
Rock cover (%)	5.788	0.257	6.685	0.220	5.383	0.187
Coarse woody debris cover (%)	7.163	0.057	7.703	0.211	5.579	0.293
Cow dung cover (%)	4.808	0.594	7.044	0.122	4.516	0.437
Mean grass height (cm)	**7.362**	**0.014***	**11.100**	**0.001*****	**7.800**	**0.003****

Fig. 2.12 Ordination plots of posterior medians of the first two latent variables of (A) web builders spider communities and (B) plant dwelling spider communities of time since kraal removal caught at Debshan Ranch, Zimbabwe. ● a = less than 2 months after kraal removal, ▲ b = 2 months after kraal removal, ■ c = 4 months after kraal removal, + d= 6 months after kraal removal, e = 8 months after kraal removal, * f = 10 months after kraal removal.

2.5 DISCUSSION

This study, sought to detect the drivers of spider assemblages within the miombo woodlands at Debshan Ranch, Zimbabwe around previously kraaled sites and their control sites. Different functional groups responded contrastingly to temporal and vegetation effects mean grass height (cm) was the most important predictor of spider diversity while time since kraal removal was the most important predictor in genera composition. Mean grass height (cm) and time since kraal removals are important predictors of spider assemblages that can be manipulated in a manner that ensures conservation of most invertebrates.

2.5.1 Changes in mean grass height (cm) and percentage grass cover during the sampling periods.

Findings of this study suggest that seasonality played a major role in describing vegetation structure changes of mean grass height (cm) and percentage grass cover. The best model that explained the observed decrease in mean grass height (cm) and percentage grass cover within the miombo woodlands at Debshan contained time since kraal removal variable. Due to the fact that the sampling periods spanned over the three main seasons of Zimbabwe (Climate-data.org 2019), change in vegetation structure within the sampled sites may be attributable to seasonality more than the holistic management approach. According to D`Onofrio *et al.* (2019) percentage grass cover has a positive relationship with rain fall frequency which in turn varies with seasonality. In Zimbabwe, the highest rainfall occurs during the summer season (Moven 2016), with over 80% of the yearly rainfall expected in Zimbabwe occurring during the November to March period (Mberego & Gwenzi 2014), which in this study was also the same period that also recorded the highest mean grass height (cm) as well as percentage grass cover.

Despite the presence of the models that combined time since kraal removal and the short duration kraaling treatment as also being part of the models that had a delta of less than ten from the best model for both mean grass height (cm) and percentage grass cover, the latter models also had a very low Aikaike weight, therefore their contribution to the observed change in mean grass height (cm) and percentage grass cover is insignificantly small. It is therefore safe to say, that in order to obtain a better reflection of mean grass height (cm) and percentage grass cover without the seasonality effect, sampling periods should therefore be done within one season, preferably

the summer season which is the optimum period for catching spider assemblages within the sub-tropical regions (Muelelwa *et al. 2*010).

2.5.2 Abundance and diversity patterns.

In this study, unkraaled sites were expected to have higher spider assemblages than kraaled sites grounded on the judgments of De Keer & Maelfait (1988), that spiders mostly oviposit or overwinter in ungrazed areas as a result of avoiding unfavorable climatic conditions, thus implying higher abundances in such areas. Contrary to the projection, it was found that ground dwelling and plant dwelling diversity and web builder's abundance showed no response to the effect of short duration kraaling. However, only web builder's genera richness responded negatively to short duration kraaling. According to Warui *et al.* (2005), this result may be due to the biology of spiders caught in each functional group, For example, foliage-dwellers normally caught by sweep-nets depend mostly on vegetation cover, however, pitfall traps that capture mostly ground-active invertebrates, are normally not associated with vegetation cover. However, the lack of short duration kraaling effect on plant dwelling spiders in this study is somehow unexpected considering that vegetation dwelling spiders are also shaped by vegetation (Lafage *et al.* 2019).

Structure of vegetation is a key element that affects habitat choice by most spiders (Štokmane & Spuņģis 2016, Lafage *et al*. 2019), for example, Robertson *et al*. (2016) showed the importance of vegetation structure for most web builders whereby plots without deer recorded a sevenfold increase of the web scaffold index. The latter is basically a measure that quantifies the structural complexity of vegetation that is available as a support structure for web construction, whether it be a branch, twig leaf or log (Miyashita *et al.* 2004). However grazing leads to modification of the vegetation by the cattle through trampling (Fuller *et al.* 2014) reducing the suitability of the biotope to web builder's requiring anchor substrates (Roberson *et al.* 2016), thereby rendering the biotope less suitable to harbour spiders fauna especially web builders.

Spider genera richness and abundance of all functional groups responded with a hump-shaped pattern to mean grass height (cm). Spider genera richness decreased with increasing mean grass height (cm) for both ground dwelling and web-builders; however the opposite was true for plant wanderers. This is in conflict with previous studies whereby within a meta-analysis study increased habitat complexity lead to significant increases in most arthropods, especially web-

building and hunting spider abundance, as a result of vegetation structure importance to most arthropods (Langellotto & Denno 2004). In this study, only the plant dwelling responded positively with mean grass height (cm) suggesting the importance of vegetation structure to this functional group, which is also consistent to previous studies such as that of Whitmore *et al.* (2002). Spiders have been reported to benefit from greater vegetation structure and cover mainly through the provision of hunting grounds for most hunters, web attachments points, shelter from predators. For example, taller swards in ungrazed areas provide increased humidity for most arthropods (Eyre 2005) and space for most climbing spiders (Lenoir & Lennartsson 2010). Surprisingly, web builders in this study responded negatively with increase in vegetation height, suggesting that despite the importance of vegetation structure to most web builders maybe the architecture provided by the grasses did not provide adequate and suitable web anchor structures for the full range of all possible web builders thus reducing their richness (Gomez *et al.* 2016).

The response of ground dwelling spiders was as expected as the majority are mostly generalists that require open habitats and disturbed habitats (Pedley *et al.* 2014), which the latter normally exhibits short or little vegetation that is suitable for their foraging mode as running predatory invertebrates (Bonte *et al.* 2003). Recently however, ground dwellers have also been reported to be associated with vegetation, according to Lafage *et al.* (2019) ground dwellers are negatively influenced by vegetation structure within the wet oligotrophics of Western France. However, this negative relationship was attributed to changes in leaf litter characteristics rather than vegetation height itself, as grass height was highly and positively correlated to litter depth in their study. According to Uetz (1979) the activity-density of Lycosidae are negatively influenced by increasing litter depth and in support of their results the Lycosidae were also the most abundant spiders that was found within the oligotrophics of Western France. However, in this study, vegetation cover was correlated to mean grass height (cm) and not to percentage leaf litter cover. As a result, the effect of vegetation structure on all functional is most probable as a result of the mean grass height (cm) or the percentage vegetation cover and not that of percentage leaf litter cover.

Ground dwelling spiders have been reported to be affected by several factors such as disturbance, moisture, leaf litter and ph. (Seyfulina 2005, Butler & Haddad 2011, Engelbrecht 2013, Gallé *et al.* 2017) amongst others. Similar to previous studies that also reported an increase of ground dwellers with an increase in litter depth and complexity (Uetz 1976, 1979), ground

dwelling spiders in this study were the only functional group that were positively correlated with leaf litter cover, despite the former two variables having not been measured in this study. This result may be attributable to the ground dwelling spiders belonging to the Gnaphosidae family that construct silk retreat in leaf litter in which they normally rest in during the day, as well as the Lycosidae family which has also been found to be highly correlated with leaf litter (Lafage *et al.* 2019). According to Bultman & Uetz (1982), litter depth and not complexity is the key major predictor of ground dwelling species richness, the opposite being true for web builders. Therefore the lack of influence of the litter cover to web builder's is most probably as a result of the lack of complex spatial heterogeneous leaf litter which is an important determinant of most web builder's (Bultman & Uetz 1982). Other studies however, have reported that web builders are more sensitive to depth of leaf litter (Bultman *et al.* 1982), however in this study web builder's did not respond to leaf litter cover which possibly could be explained by the possible shallow depth of the leaf litter cover which was nevertheless not measured in this study but could be a possible predictor. Furthermore, leaf litter and coarse woody debris provide structural complexity that also provide sites for web attachments for web builder's (Bultman & Uetz 1982). For example, most spider belonging to the family Araneidae construct orb-webs generally above forest floors and anapids and symphytognathids constructs their webs often in the leaf litter (Coddington *et al.* 2009), similarly anapids and symphytognathids were not caught in this study hence their absence could also have contributed to the lack of web builder's diversity response to litter cover.

In this study percentage coarse woody debris cover and percentage cow dung cover did not influence spider genera richness and abundance despite the various roles they play within ecosystems. For example, spider assemblages have also been reported to favor microhabitats that include coarse woody debris (Goldin & Hutchinson 2013, 2014) more than open spaces, as a result of higher soil moisture and reduced temperatures. In addition potential prey for spiders, in the form of saproxylic arthropods, are normally also found amongst logs (Ulyshen & Hanula 2009), thus making such microhabitats suitable for hosting high abundances of spider assemblages. Cow dung has also been shown to benefit the emergence and increase of other invertebrates such as Diptera, which have been recorded to be more pronounced in high density sheet plots (Dennis *et al*. 2015). In other studies, however, the effect of dung deposition has been described to be of little importance to most predators when compared to vegetation height

(Lenoir & Lennartsson 2010). Therefore the lack of influence of the spider genera richness and abundance by both percentage coarse woody debris and cow dung deposits, despite their importance to most invertebrates is most probably as a result of the relatively low percentage composition of the variables (Table 2.1) that were not sufficient and adequate to harbor any spider assemblages.

With the exception of the plant wanderers, time since kraal removal was the second most significant predictor of spider genera abundance and richness. With increasing time since kraal removal there was an increase in both spider genera richness and abundance. The effect of abandoned kraals has been well studied for many other organisms (Sibanda *et al.* 2016, Huruba *et al.* 2017, 2018), including invertebrates (Chikorowondo *et al.* 2017, 2018), with the exception of spiders. Abandoned kraal sites have been reported to have higher soil nutrient content of potassium (K), nitrogen (N) and phosphorous (P) mainly as a result of the deposits of the dung from the livestock (Kizza *et al.* 2010, Chikorowondo *et al.* 2017), which in turn as a result of the increased nutrient content within the kraals leads to faster growth of grasses and vegetation structure thereby increasing grass cover and mean grass height (cm) which has been noted to also provide essential services to most invertebrates (Gibson *et al.* 1992), enhancing the development of different functional groups.

Contrary to the expected hypothesis that spider diversity would be more diverse with distance from kraal; spider genera richness and abundance in all the functional groups were not significantly affected by distance from the cattle night shelter (Table 2 & Table 3). In other studies, however, spiders have been reported to be shaped by distance. For example, in the Australian tropics Churchill (1998) reported an increase in Zodariidae abundance with decreasing grazing gradient. Bonte *et al*. (2000) also reported distance from rough vegetation as an significant factor that regulates the occurrence of species in short-grazed grasslands in mesophylic calcareous grasslands. Marin (2013) reported a negative association of spider diversity in the rainy season, while no effect of distance was reported during the dry season within coffee plantations. This was as a result of greater spider mobility observed during the dry season compared to the rainy season. However, this was not the case in this study, under a grazing gradient of less than 400m, for spider genera richness and abundance of functional groups. A possible reason for the lack of dissimilarity in spider genera richness and abundance in all the functional groups with regards to distance from kraal has been suggested in previous

74

research to be due to insufficient sampling, that may result in the missing of rare species (Jimenez-Valverde & Lobo 2006) which in their absence may reduce the dissimilarity of assemblages (Jimenez-Valverde *et al.* 2009) hence assemblage will be homogenous. However, in this study sample coverage in both the unkraaled and kraaled sites was above 98%, indicating that undersampling was less likely to be a contributing factor in these results.

2.5.3 Spider assemblage composition.

 Only the ground dwelling genera composition responded to distance from kraal. Apart from the environment, geographical distance has been reported to be amongst one of the most significant predictors of compositional similarity amongst various taxa (e.g angiosperms Qian *et al.* 2005; rock-restricted cichlid fishes Genner *et al.* 2004; leaf beetles Baselga *et al.* 2007). Similar to the results in this study, the impact of geographical distance was predicted to be greater in orgarnisms with lower dispersal capacities. According to a classification done by Jimenez-Valverde *et al.* (2009), based on the balloning ability in spiders, Araneidae (mostly orb-web builders) were grouped as high vagility spiders; Thomisidae and part of Salticidae (plant wanderers) were grouped as medium vagility spiders whilst the Gnaphosidae and partly Salticidae (ground wanderers) were grouped as low vagility spiders. Since the ground dwelling spiders were mostly dominated by Gnaphosidae and Lycosidae families which are low vagility spiders, their dispersal ability was thus limited in this study and most probably a contributory cause of the lack of compositional similarity amongst the ground dwelling spiders in the different distances from the kraal.

Around the inclusions of previously kraaled sites time since kraal removal had a significant influence on genera composition in all the functional groups, however when considering the most abundant genera that was recorded in all sampling periods (*Stenaelurillus*). The short duration kraaling effect initially reduced abundance of *Stenaelurillus*, however in the sampling period of 1 month after kraal removal there was an increase in abundance with subsequent reduction of abundance that lead to the lowest abundance recorded at 10 months after kraal removal (Appendix 2.3). This outcome may suggest two things, for example its either the effect of time since kraal removal on spider diversity is the true reflection of the time effect however, as a result of the emergence of *Capheris fitzsimonsi* (Lawrence, 1936) at 6 months after kraal removal due to its life cycle this result may be suggestive of simply being a reflection of

seasonality more than treatment effect, as spiders have been reported to be affected by seasons (Lubin 1978, Weeks & Holtzer 2000, Campuzano & Ibarra-n 2019).

Only percentage leaf litter cover, percentage coarse woody debris cover and mean grass height (cm) explained spider genera composition of all the measured vegetation structure variables. Leaf litter depth has been broadly considered to be a significant predictor of spider assemblages, due to its ability to affect prey abundance, provide suitable spider retreats for selected ground dwelling spiders as well as reduction of temperature and humidity (Bultman & Uetz 1982, Bultman *et al.* 1982, Uetz 1991). Foliage dwellers and grass dwelling assemblages have also been suggested to use leaf litter as an overwintering site (Butler & Haddad 2011). As a result, leaf litter cover has been shown to be an important vegetation structural element more than other measured vegetation structure elements in this study in terms of habitat selection by spider genera composition. Univariate analysis only revealed significant effect of mean grass height (cm) in the structure of only the web builders and plant dwelling spiders but not ground dwellers(Appendix 2.2), thus contributing to the evidence of the importance of vegetation structure to both web-builders and plant dwelling genera composition. Genera such as *Araneus* are closely associated to low vegetation, the latter which was amongst the most abundant caught genera in this study.

2.6 References

Abel, N.O.J. & Blaikie, P.M. (1989) Land degradation, stocking rates and conservation policies in the communal rangelands of Botswana and Zimbabwe. *Land Degradation & Development*, **1**, 101–123.

Baselga, A. & Jimenez-Valverde, A. (2007) Environmental and geographical determinants of beta diversity of leaf beetles (Coleoptera: Chrysomelidae) in the Iberian Peninsula. *Ecological Entomology*, **32**, 312–318.

Bates, D., Machler, M., Bolker, B.M. & Walker, S.C. (2015) Fitting linear mixed-effects models using lme4. *Journal of Statistical Software*, **67**, 1–48.

Baxter, R., Hastings, N., Law, A. & Glass, E.J. (2015.) *What is holistic planned grazing?* Savory Institute, United States of America.

Bolker, B.M., Brooks, M.E., Clark, C.J., Geange, S.W., Poulsen, J.R., Stevens, M.H.H. & White, J.S. (2008) Generalized linear mixed models: a practical guide for ecology and evolution. *Trends in Ecology & Evolution*, **30**, 1–9.

Bonte, A.D., Maelfait, J. & Hoffmann, M. (2000) The Impact of grazing on spider communities in a mesophytic calcareous dune grassland. *Journal of Coastal Conservation*, **6**, 135–144.

Bonte, D., Lens, L., Maelfait, J.P., Hoffmann, M. & Kuijeken, E. (2003) Patch quality and connectivity influence spatial dynamics in a dune wolfspider. *Oecologia*, **135**, 227–233.

Bullock, J.M., Franklin, J., Stevenson, M.J., Silvertown, J., Coulson, S.J., Gregory, S.J & Tofts, R. (2001) A plant trait analysis of responses to grazing in a long term experiment. *Journal of Applied Ecology*, **38**, 253-267.

Bultman, T., Uetz, G.W. & Brady, A. (1982) A comparison of cursorial spider communities along a successional gradient. *Journal of Arachnology*, **10**, 23–33.

Bultman, T. & Uetz, G. (1982) Abundance and community structure of forest floor spiders following litter manipulation. *Oecologia*, **55**, 34–41.

Burnham, K.P. & Anderson, D. (2002) *Model selection and inference: a practical information-theoretic approach.* 2nd Edition, Springer-Verlag, New York, United States of America.

Butler, V.P. & Haddad, C.R. (2011) Spider assemblages associated with leaf litter of three tree species in central South Africa (Arachnida: Araneae). *African Journal of Ecology*, **49**,

301–310.

Campuzano, E.F., Ibarra-Nunez, G., Machkour-M`Rabet, S., Moron-Rios, A. & Jimenez, M.L. (2019) Diversity and seasonal variation of ground and understory spiders from a tropical mountain cloud forest. *Insect Science*, **27**, 826-844.

Chao, A. & Jost, L. (2012) Coverage-based rarefaction and extrapolation: standardizing samples by completeness rather than size. *Ecology*, **93**, 2533–2547.

Chikorowondo, G., Muvengwi, J., Mbiba, M. & Gandiwa, E. (2017) Influence of abandoned cattle enclosures on plant assemblages and herbivory in a semi-arid savanna. *Ecological Research*, **36**, 1023–1033.

Chikorowondo, G., Muvengwi, J., Mbiba, M. & Gandiwa, E. (2018) Functional diversity of macroinvertebrates on abandoned cattle enclosures in a semi-arid African savannah. *African Journal of Ecology*, **56**, 629-640

Churchill, T.B. (1998) *Spiders as ecological indicators in the Australian tropics : family distribution patterns along rainfall and grazing gradients*. In PA Selden, (ed). Proceedings of the 17th European Colloquium of Arachnology, Edinburgh 1997. Burnham Beeches, Bucks, British Arachnological Society.

Climate-data.org. (2019) Climate: Shangani, online at https://en.climate-data.org/africa/ zimbabwe/matabeleland-south/shangani-214843/, accessed on 16th of October 2019.

Coddington, J.A., Agnarsson, I., Miller, J.A., Kuntner, M., Hormiga, G., Washington, D.C. & Coddington, J.A. (2009) Undersampling bias: the null hypothesis for singleton species in tropical arthropod surveys. *Journal of Animal Ecology*, **78**, 573–584.

Coddington, J.A., Young, L.H. & Coyle, F.A. (1996) Estimating spider species richness in a southern Appalachian cove hardwood forest. *Journal of Arachnology*, **24**, 111–128.

Cousins, B. (1992) Managing communal rangeland in Zimbabwe: experience and lessons. *Journal of Experimental Psychology: General*, **136**, 23-42

D`onofrio, D., Sweeney, L., Hardenberg, J.V. & Baudena, M. (2019) Grass and tree cover responses to intra-seasonal rainfall variability vary along a rainfall gradient in African tropical grassy biomes. *Scientific Reports*, **9**, 2334.

De Keer, R. & Maelfait, J. (1988) Laboratory observtions on the development and reproduction of Erigone atra, Blackwall, 1833 (Araneae: Linyphiidae). *Bulletin of the British Arachnological Society*. **7**, 237-242.

Dennis, P., Skartveit, J., Kunaver, A. & McCracken, D.I. (2015) The response of spider (Araneae) assemblages to structural heterogeneity and prey abundance in sub-montane vegetation modified by conservation grazing. *Global Ecology and Conservation*, **3**, 715–728.

Dethier, M.N., Graham, E.S., Cohen, S. & Tear, L.M. (1993) Visual versus random-point percent cover estimations: objective is not always better. *Marine Ecology Progress Series*, **96**, 93–100.

Dippenaar-Schoeman, A.S. & Jocque, R. (1997) *African spiders: an identification manual*. Plant Protection Research Institute Handbook No.9, Agricultural Research Council, Pretoria, South Africa.

Duggan, W.R. (1980) The Native Land Husbandry Act of 1951 and the Rural African Middle Class of Southern Rhodesia. *African Affairs*, **79**, 227–239.

Dunham, K.M., Robertson, E.F. & Swanepoel, C.M. (2003) Population decline of tsessebe antelope (*Damaliscus lunatus*) on a mixed cattle and wildlife ranch in Zimbabwe. *Biological Conservation*, **113**, 111–124.

Engelbrecht, I. (2013) Pitfall trapping for surveying trapdoor spiders : the importance of timing, conditions and effort. *Journal of Arachnology*, **41**, 133–142.

Eyre, M. (2005) Habitat diversity in the conservation of the grassland *Auchenorrhyncha* (Homoptera: Cercopidae, Cicadellidae, Cixidae, Delphacidae) of Northern Britain. *Journal of Insect Conservation*, **9**, 309–317.

Foord, S.H., Mafadza, M.M., Dippennaar-Schoeman, A.S. & Van Rensburg, B.J. (2008) Micro-scale heterogeneity of spiders (Arachnida: Araneae) in the Soutpansberg, South Africa: a comparative survey and inventory in representative habitats. *African Zoology*, **43,** 156–174.

Foord, S.H., Dippenaar-Schoeman, A., & Haddad, C.R. (2011a) The faunistic diversity of spiders (Arachnida: Araneae) of the Savanna Biome in South Africa. *Transactions of the Royal Society of South Africa*, **66**, 170–201.

Foord, S.H., Dippenaar-Schoeman, A.S. & Haddad, C.R. (2011b) South African spider diversity: African perspectives on the conservation of a mega-diverse group. In Grillo, O. & Venora, G. (ed). Changing diversity in changing environment. Rijeka, InTech Publishing.

Foord, S.H., Dippenaar-Schoeman, A.S. & Stam, E.M. (2013) Surrogates of spider diversity,

leveraging the conservation of a poorly known group in the Savanna Biome of South Africa. *Biological Conservation*, **161**, 203–212.

Froude, M. (1974) Veld management in the Victoria Tribal Areas. *Rhodesian Agricultural Journal*, **71**, 29-37.

Fuller, L., Newman, M., Irwin, S., Kelly, T. & Halloran, J.O. (2014) Ground-dwelling spider diversity in rare European oak and yew woodlands and the impact of grazing. *Biodiversity and Conservation*, **23**, 1911–1929.

Gabathuler, E., Liniger, H., Hauert, C. & Giger, M. (2009) *Benefits of Sustainable Land Management.* World Overview of Conservation Approaches and Technologies & Centre for Development and Environment. University of Berne, Bern, Switzerland.

Galle, R., Urak, I., Nikolett, G.S. & Hartel, T. (2017) Sparse trees and shrubs confers a high biodiversity to pastures : case study on spiders from Transylvania. *Plos One*, **12**, 1–12.

Gambiza, J. & Nyama, C. (2006) *Country pasture/forage resources profiles : Zimbabwe.* Food and Agriculture Orgarnisation of the United Nations, Rome.

Genner, M.J., Taylor, M.I., Cleary, D.F.R., Hawkins, S.J., Knight, M.E. & Turner, F. (2004) Beta diversity of rock-restricted cichlid fishes in Lake Malawi: importance of environmental and spatial factors. *Ecography*, **27**, 601–610.

Gibson, C.W., Hambler, C. & Brown, V. (1992) Changes in spider (Araneae) assemblages in relation to succession and grazing management. *Journal of Applied Ecology*, **29**, 132–142.

Goldin, S.R. & Hutchinson, M.F. (2013) Coarse woody debris modifies surface soils of degraded temperate eucalypt woodlands. *Plant Soil*, **370**, 461–469.

Goldin, S.R. & Hutchinson, M.F. (2014) Coarse woody debris reduces the rate of moisture loss from surface soils of cleared temperate Australian woodlands. *Soil Res*, **52**, 637–644.

Gómez, J.E., Lohmiller, J. & Joern, A. (2016) Importance of vegetation structure to the assembly of an aerial web-building spider community in North American open grassland. *Journal of Arachnology*, **44**, 28–35

Gusha, J., Masocha, M. & Mugabe, P.H. (2017) Impact of grazing system on rangeland condition and grazing capacity in Zimbabwe. *The Rangeland Journal*, **39**, 19–225.

Harrison, X.A., Donaldson, L., Correa-Cano, M.E., Evans, J., Fisher, D.N., Goodwin, C. E.D., Robinson, B.S., Hodgson, D.J. & Inger, R. (2018) A brief introduction to mixed effects modelling and multi-model inference in ecology. *PeerJ*, **6**, 1–32.

Havstad, K.M., Peters, D.P.C., Skaggs, R., Brown, J., Bestelmeyer, B., Fredrickson, E., Herrick, J. & Wright, J. (2007) Ecological services to and from rangelands of the United States. *Ecological Economics*, **64**, 261–268.

Hsieh, T.C., Ma, K.H. & Chao, A. (2016) iNEXT : an R package for rarefaction and extrapolation of species diversity (Hill numbers). *Methods and Evolution*, **7**, 1451–1456.

Hui, F.K.C., Taskinen, S., Pledger, S., Foster, S.D. & Warton, D.I. (2015) Model-based approaches to unconstrained ordination. *Methods in Ecology and Evolution*, **6**, 399–411.

Huruba, R., Mlambo, T., Mundy, P.J., Sebata, A. & MacFadyen, D.N. (2018) Short duration overnight cattle kraaling in natural rangelands : implications for grass composition, quality, above ground biomass, species diversity and basal cover. *Agriculture, Ecosystems and Environment*, **257**, 144–151.

Huruba, R., Mundy, P.J., Sebata, A., Purchase, G.K. & MacFadyen, D.N. (2017) Impala, *Aepyceros melampus:* does browse quality influence their use of sites originally utilised as short-duration kraals in a southern African savanna ? *Rangeland Journal*, **39**, 113–121.

Jankielsohn, A., Scholtz, C.H. & Louw, S.V (2001) Effect of habitat transformation on dung beetle assemblages: a comparison between a South African nature reserve and neighboring farms. *Environmental Entomology*, **30**, 474–483

Jansen, R., Makaka, L. & Little, I.A. (2013) Response of ground-dwelling spider assemblages (Arachnida: Araneae) to montane grassland management practices in South Africa. *Insect conservation and diversity*, **6**, 572–589.

Jimenez-Valverde, A., Baselga, A., Melic, A. & Txasko, N. (2009) Climate and regional beta-diversity gradients in spiders: dispersal capacity has nothing to say? *Insect Conservation and Diversity*, **3**, 51–60.

Jimenez-Valverde, A. & Lobo, J.M. (2006) Establishing reliable spider (Araneae, Araneidae and Thomisidae) assemblage sampling protocols: estimation of species richness, seasonal coverage and contribution of juvenile data to species richness and composition. *Acta Oecologica*, **30**, 21–32.

Kizza, S., Totolo, O., Pekkins, J. & Areola, O. (2010) Analysis of persistence soil nutrient status in abandoned cattle kraals in a semi arid area in Botswana. *Scientific Research and Essays*, **5**, 3613–3622.

Lafage, D., Djoudi., E.A., Perrin, G., Gallet, S. & Petillon, J. (2019) Responses of

ground-dwelling spider assemblages to changes in vegetation from wet oligotrophic habitats of Western France. *Arthropod-Plant Interactions*, **13**, 653–662.

Landsberg, J., James, C.D., Morton, S.R., Muller, W.J. & Stol, J. (2003) Abundance and composition of plant species along grazing gradients in Australian rangelands. *Journal of Applied Ecology*, **40**, 1008-1024.

Langellotto, G.A. & Denno, R.F. (2004) Responses of invertebrate natural enemies to complex-structured habitats: a meta-analytical synthesis. *Oecologia*, **139**, 1–10.

Lenoir, L. & Lennartsson, T. (2010) Effects of timing of grazing on arthropod communities in semi-natural grasslands. *Journal of Insect Science*, **10**, 1–24.

Lubin, Y.D. (1978) Seasonal abundance and diversity of web-building spiders in relation to habitat structure on Barro Colorado Island, Panama. *Journal of Arachnology*, **6**, 31–51.

Lyons, A. (2017) *The impacts of contrasting grazing management on biodiversity in upland calcareous grasslands*. Doctoral dissertation, Edge Hill University, United Kingdom.

Marin, L.E.R. (2013) *The diversity and ecology of spiders in coffee agroecosystems in the Soconusco Region, Chiapas, Mexico.* Doctoral dissertation, University of Michigan, Mexico.

Mberego, S. & Gwenzi, J. (2014) Temporal patterns of precipitation and vegetation variability over Zimbabwe during extreme dry and wet rainfall seasons. *Journal of Applied Meteorology and Climatology*, **53**, 2790–2804.

Mberi, M. (2013) *Debshan Ranch and the holistic alternative-an overview.* Presentation at the 4th Annual Route Research Conference 29 October 2013 Debeers Campus Johannesburg, South Africa.

McCraken, D.I, Tallowin, J.R. (2004) Swards and structure: the interactions between farming practices and bird food resources in lowland grasslands. *Ibis*, **146**, 108–114

Miyashita, T., Takada, M. & Shimazaki, A. (2004) Indirect effects ofherbivory by deer reduce abundance and species richness of web spiders. *Ecoscience*, **11**, 74–79.

Moven, M. (2016) *Inter annual summer rainfall variability over Zimbabwe and its possible mechanism.* MSc thesis College of Atmospheric Science, Nanjing University of Information Science and Technology, China.

Muelelwa, M.I., Foord, S.H., Dippenaar-Schoeman, A.S. & Stam, E.M. (2010) Towards a standardized and optimized protocol for rapid biodiversity assessments : spider species

richness and assemblage composition in two savanna vegetation types. *African Zoology*, **45**, 273–290.

Mugandani, R., Wuta, M., Makarau, A. & Chipindu, B. (2012) Re-classification of agro-ecological regions of Zimbabwe in conformity with climate variability and change. *African Crop Science Journal*, **20**, 361–369.

Mysterud, A., Aaserud, R., Hansen, L., Akra, K., Olberg, S. & Austrheim, G. (2010) Large herbivore grazing and invertebrates in an alpine ecosystem. *Basic and Applied Ecology*, **11**, 320–328.

Nakagawa, S. & Schielzeth, H. (2013) A general and simple method for obtaining R^2 from generalized linear mixed-effects models. *Methods in Ecology and Evolution*, **4**, 133–142.

Neely, C.L. & Butterfield, J. (2004) Holistic management of African rangelands. *Leisa Magazine*, **20**, 26–28.

Niku, J., Hui, F.K.C., Taskinen, S. & Warton, D.I. (2019) gllvm-fast analysis of multivariate abundance data with generalized linear latent variable models in R. *Methods in Ecology and Evolution*, **10**, 2173–2182.

Nyoni, P. (2015) *Leopard population density and community attitudes towards leopards in and around Debshan Ranch, Shangani, Zimbabwe.* MSc dissertation, Rhodes University, Pretoria, South Africa.

Passmore, G. (1972) *The National Policy of Community Development in Rhodesia.* Department of Political Science, University of Rhodesia, Salisbury.

Pedley, S.M. & Dolman, P.M. (2014) Multi-taxa trait and functional responses to physical disturbance. *Journal of Animal Ecology*, **83**, 1542–1552.

Qian, H., Ricklefts, R.E. & White, P.S. (2005) Beta diversity of angiosperms in temperate floras of eastern Asia and eastern North America. *Ecology Letters*, **8,** 15–22

R Core Team. (2019) R: A language and environment for statistical computing. Version 2.15.1. R Foundation for Statistical Computing, Vienna, Austria.

Roberson, E.J., Chips, M.J., Carson, W.P. & Rooney, T.P. (2016) Deer herbivory reduces web-building spider abundance by simplifying forest vegetation structure. *PeerJ*, **4**, 1–15.

Robertson, E.F. (2013) *Vegetation monitoring sites on Shangani Ranch: methods and descriptions of grass species composition and ecological condition for sites 1-27, March-April 2013*. Report prepared for E. Oppenheimer and Son (Pty) Limited and Debshan Pvt

Ltd, Shangani, Zimbabwe.

Savory, A. (1983) The Savory Grazing Method or Holistic Resource Management. *Rangelands*, **5**, 155–159.

Savory, A. & Butterfield, J. (1999) *Holistic Management: a new framework for decision making.* (2nd Edition). Island Press, Washington, D.C, Covelo, California, United States of America.

Savory, A. & Parsons, S.D. (1980) The Savory Grazing Method. *Rangelands*, **2**, 234–237.

Seyfulina, R.R. (2005) Microhabitat effect on spider distribution in winter wheat agroecosystem (Araneae). *European Arachnology*, **1**, 161–172.

Sibanda, P., Sebata, A., Mufandaedza, E. & Mawanza, M. (2016) Effect of short-duration overnight cattle kraaling on grass production in a southern African savanna. *African Journal of Range and Forage Science*, **33**, 217-223.

Stoddart, L.A. (1967) What Is Range Management? *Journal of Range management*, **20**, 304–307.

Štokmane, M. & Spungis, V. (2016) The influence of vegetation structure on spider species richness, diversity and community organization in the Apšuciems calcareous fen, Latvia. *Animal Biodiversity and Conservation*, **39**, 221–236.

Szmatona-Turi, T., Vona-Turi, D., Urban, L & Magos, G. (2018) Effect of grazing intensity on diversity of ground-dwelling spiders of grassy and shrubby habitats. *Acta Zoologica Bulgarica*, **70**, 195-202.

The Senior Reporting Farmer. (2015) *Land programme betters food security.* The Sunday News, Zimbabwe.

The Zimbabwean. (2015) *Holistic management restores lost biodiversity.* The Zimbabwe Situation, Zimbabwe.

Uetz, G.W. (1976) Gradient analysis of spider communities in a streamsidc forest. *Oecotogia*, **22**, 373–385.

Uetz, G.W. (1979) The influence of variation in litter habitats on spider communities. *Oecologia*, **40**, 29–42.

Uetz, G.W. (1991) *Habitat structure and spider foraging. Habitat structure: the physical arrangement of objects in space.* In Bell, S.S., McCoy, E.D and Mushinsky, H.R.(ed). Chapman and Hall, London, United Kingdom.

Ulyshen, M.D. & Hanula, J.L. (2009) Responses of arthropods to large-scale manipulations of dead wood in lobolly pine stands of the southeastern United States. *Environmental Entomology*, **38**, 1005–1012.

Veblen, K. & Porensky, L.M. (2019) Thresholds are in the eye of the beholder : plants and wildlife respond differently to short-term cattle corrals. *Ecological Applications*. **29**, 1–15.

Walker, B.H. & Janssen, M.A. (2002) Rangelands, pastoralists and governments : interlinked systems of people and nature. *Philosophicsal Transactions : Biological Science*, **357**, 719–725.

Wang, Y., Naumann, U., Wright, S.T. & Warton, D.I. (2012) mvabund –an R package for model-based analysis of multivariate abundance data. *Methods in Ecology and Evolutio,*. **3**, 471–474.

Warui, C.M., Villet, M.H., Young, T.P. & Jocque, R. (2005) Influence of grazing by large mammals on the spider community of a Kenyan savanna biome. *Journal of Arachnology*, **33**, 269–279.

Weeks, R.D. & Holtzer, T.O. (2000) Habitat and season in structuring ground-dwelling spider (Araneae) communities in a shortgrass steppe ecosystem. *Enviromental Entomology*, **29**, 1164–1172.

Wettstein, W. & Schmid, B. (1999) Conservation of arthropod diversity in montane wetlands: effect of altitude, habitat quality and habitat fragmentation on butterflies and grasshoppers. *Journal of Applied Ecology*, **36**, 363–373

Weinrich, A.K. (1975) *African farmers in Rhodesia.* Oxford Univerity Press for International African Institute, London .

Weyman, G.S., Sunderland, K.D. & Jepson, P.C. (2002) A review of the evolution and mechanisms of ballooning by spiders inhabiting arable farmland. *Ethology Ecology and Evolution*, **14**, 307–326.

Whitmore, C., Slotow, R., Crouch, T.E. & Dippenaar-Schoeman, A.S. (2002) Diversity of Spiders (Araneae) in a savanna reserve, Northern province, South Africa. *Journal of Arachnology*, **30**, 344-356

Whittingham, M.J., Stephens, P.A., Richard, B. & Freckleton, R.P. (2006) Why do we still use stepwise modelling in ecology and behaviour ? *Journal of Animal Ecology,* **75**, 1182–1189.

Wise, D.H. (1993) *Spiders in Ecological Webs* (Cambridge Studies in Ecology). Cambridge, Cambridge University Press, Cambridge, United Kingdom.

Zar, J.H. (1998) *Bio statistical analysis,* 4th Edition. Upper Saddle River, N.J: Prentice-Hall, United States of America.

Zuur, A.F. & Ieno, E.N. (2016) A protocol for conducting and presenting results of regression-type analyses. *Methods in Ecology and Evolution*, **7**, 636–645.

Zuur, A.F., Ieno, E.N. & Elphick, C.S. (2010) A protocol for data exploration to avoid common statistical problems. *Methods in Ecology and Evolution*, **1**, 3–14.

Appendix 2.1: Checklist of spiders caught at Debshan Ranch from July 2017 to April 2018 with grazing treatment and functional group indicated. C= unkraaled sites, G= kraaled sites, GW= ground wanderers, PW= plant wanderers, W= web builders.

Scientific name	C GW	C PW	C W	C Total	G GW	G PW	G W	G Total	Grand Total
Agelenidae			2	2			1	1	3
Mistaria lawrencei (Roewer, 1955)			1	1			1	1	2
Benoitia ocellata Pocock, 1900			1	1					1
Ammoxenidae	13			13	41			41	54
Ammoxenus daedalus Dippenaar & Meyer,1980	5			5	15			15	20
Ammoxenus sp.	8			8	27			27	35
Araneidae			251	251			208	208	459
Araneidae sp.			7	7			11	11	18
Araneus sp.			18	18			33	33	51
Argiope australis Walckenaer, 1805			29	29			22	22	51
Argiope sp.			30	30			30	30	60
Argiope trifasciata Forsskål, 1775			2	2			2	2	4
Caerostris sp.			1	1					1
Crytophora citricola Forsskål, 1775			1	1					1
Crytophora sp.			1	1					1
Cyclosa sp.			3	3					3
Cyphalonotus sp.							1	1	1
Eriovixia sp.							1	1	1
Eriovixia sp.1							1	1	1
Gasteracantha sp.			5	5			2	2	7
Hyposinga sp.			2	2			2	2	4
Hyposinga sp.1			2	2			1	1	3

Taxon					
Ideocaira sp.			1	1	1
Isoxya sp.	2	2			2
Kilima sp.	1	1			1
Larinia chloris Audouin, 1826	37	35	23	22	60
Larinia sp.	3	3	5	5	8
Lipocrea sp.	2	2	4	4	6
Nemoscolus affinis Lessert, 1933	1	1			1
Nemoscolus cotti Lessert, 1933	1	1	1	1	2
Nemoscolus sp.	27	27	27	27	54
Nemoscolus sp.1	1	1			1
Neoscona sp.	22	22	20	20	42
Neoscona subfusca Koch, 1837	3	3			3
Nephila inaurata Walckenaer, 1841	10	10	4	4	14
Nephila senengalensis Walckenaer, 1842	11	11	10	10	21
Nephila sp.	10	10			10
Poltys sp.	1	1			1
Singa albordosata Kauri,1950	18	18	8	8	26
Singa sp.	2	2	1	1	3
Caponiidae			**1**	**1**	**1**
Caponia sp.			1	1	1
Cheiracanthiidae	**19**	**19**	**25**	**25**	**44**
Cheiracanthium furculatum Karsch, 1879	3	3			3
Cheiracanthium n.sp.			1	1	1
Cheiracanthium sp.	16	16	24	24	40
Clubionidae	**2**	**2**	**3**	**3**	**5**
Clubiona sp.	2	2	3	3	5
Corinnidae	**1**	**1**	**6**	**6**	**7**

Taxon								
Copa flavoplumosa Simon, 1885					4		4	4
Copa sp.	1			1	1		1	2
Graptartia granulosa Simon, 1886					1		1	1
Ctenidae	**10**			**10**	**5**		**5**	**15**
Afroneutria n.sp.	1			1				1
Ctenidae n.gn sp.1	2			2				2
Ctenus sp.	7			7	5		5	12
Cyrtaucheniidae	**2**			**2**	**4**		**4**	**6**
Ancylotrypa nuda Hewitt,1966					1		1	1
Ancylotrypa sp. 2	2			2				2
Ancylotrypa sp.3					2		2	2
Homostola sp.					1		1	1
Dictynidae			**10**	**10**		**8**	**8**	**18**
Archaeodictyna sp.			9	9		8	8	17
Dictyna sp.			1	1				1
Dipluridae						**1**	**1**	**1**
Allothele sp.1						1	1	1
Eresidae			**3**	**3**		**3**	**3**	**6**
Gandanameno purcelli Tucker, 1920						1	1	1
Paradonea sp.1			1	1		1	1	2
Stegodyphus africanus Blackwall, 1866			1	1				1
Stegodyphus sp. 2			1	1		1	1	2
Gnaphosidae	**344**	**1**		**345**	**276**		**276**	**621**
Asemesthes fodina Tucker, 1923					2		2	2
Asemesthes lineatus Purcell, 1908	55			55	36		36	91
Asemesthes paynteri Tucker, 1923	101			101	83		83	184

Species	C1	C2	C3	C4	C5	C6
Asemesthes windhukensis Tucker, 1923	2		2	7	7	9
Asemesthes sp.	49		49	54	54	103
Camillina maun Platnick & Murphy, 1987	1		1			1
Drassodes solitarius Purcell, 1907	30		30	14	14	44
Drassodes splendens Tucker, 1923	8		8	16	16	24
Drassodes sp. 3	14		14	10	10	24
Ibala bulawayensis Tucker, 1923				4	4	4
Ibala declani FitzPatrick, 2009	1		1			1
Ibala minshullae FitzPatrick, 2009	8		8	8	8	16
Ibala n.sp. 4	2		2			2
Ibala sp. 5	15		15	3	3	18
Leptodrassex n.sp. 1				1	1	1
Leptodrassus sp. 2	1		1			1
Megamyrmaekion transvaalense Tucker, 1923	2		2			2
Nomisia sp.	1		1	1	1	2
Nomisia varia Tucker, 1923				1	1	1
Odontodrassus sp.				1	1	1
Pterotricha sp.		1	1			1
Scotophaeus relegatus Purcell, 1907	1		1			1
Setaphis makalali FitzPatrick, 2005	1		1			1
Theuma fusca Purcell,1907	32		32	19	19	51
Theuma sp.	3		3	1	1	4
Trephopoda sp.1				1	1	1
Upognampa sp.				1	1	1
Urozelotes rusticus L.Koch, 1872	1		1			1
Xerophaeus aurariarum Purcell, 1907	1		1			1
Xerophaeus druryi Tucker, 1923				1	1	1

Xerophaeus sp.	3		3	4	4	7
Zelotes bastardi Simon, 1896				2	2	2
Zelotes brennanorum FitzPatrick, 2007	3		3			3
Zelotes corrugatus Purcell, 1907	1		1	1	1	2
Zelotes mosioatunya FitzPatrick, 2007				1	1	1
Zelotes n.sp.	7		7	2	2	9
Zelotes scrutatus O.P. Cambridge, 1872	1		1			1
Zelotes sp.	1		1	1	1	2
Idiopidae	**7**		**7**	**3**	**3**	**10**
Ctenolophus sp. 1	3		3			3
Ctenolophus sp. 2	2		2	1	1	3
Segregara sp.1	2		2	2	2	4
Lycosidae	**215**	**1**	**216**	**198**	**198**	**414**
Allocosa lawrencei Roewer,1951	4		4			4
Allocosa sp.	73		73	65	65	138
Allocosa sp.1	1		1	2	2	3
Allocosa umtalica Purcell, 1903	5		5	5	5	10
Evippa sp.				1	1	1
Evippomma sp.	10		10	6	6	16
Evippomma squamulatum (Simon, 1898)				1	1	1
Foveosa foveolata Purcell, 1903				1	1	1
Hippasa sp.		1	1			1
Hogna sp.	4		4	2	2	6
Langona sp.				1	1	1
Lycosa gigantea Roewer, 1960	6		6			6
Lycosa palliata Roewer, 1960	1		1			1
Lycosa sp.				1	1	1

Lycosidae sp.	37	37		39	39	76
Lycosidae sp.1	1	1				1
Lycosidae sp.3	1	1		1	1	2
Lycosidae sp.5				1	1	1
Pardosa crassipalpis Purcell, 1903	22	22		22	22	44
Pardosa injucunda O.Pickard-Cambridge, 1876				1	1	1
Pardosa manubriata Simon, 1898	5	5		3	3	8
Pardosa sp.	28	28		31	31	59
Proevippa albiventris Simon, 1898	1	1		3	3	4
Proevippa fascicularis Purcell, 1903	2	2		1	1	3
Proevippa sp.	9	9		3	3	12
Schizocosa darlingi Pocock, 1898	3	3		2	2	5
Trabea sp.				3	3	3
Zenonina albocaudata Lawrence, 1952				1	1	1
Zenonina mystacina Simon, 1898	1	1				1
Zenonina sp. 3	1	1		2	2	3
Nemesidae				**1**	**1**	**1**
Hermacha sp. 1				1	1	1
Nemesidae sp.	1	1				1
Oxyopidae	**122**	**122**		**133**	**133**	**255**
Hamataliwa sp.	9	9		5	5	14
Oxyopes bothai Lessert, 1915	11	11	1	16	16	28
Oxyopes jacksoni Lessert, 1915	3	3		1	1	4
Oxyopes pallidecolaratus Strand, 1906				2	2	2
Oxyopes russoi Caporiacco, 1940	1	1		2	2	3
Oxyopes sp.	89	89		94	94	183
Oxyopes sp.1	1	1		1	1	2

Oxyopes sp.2						2		2	2
Oxyopes sp.3		1		1		4		4	5
Oxyopes sp.4		1		1					1
Oxyopes sp.5		1		1					1
Peucetia sp.		6		6		5		5	11
Peucetia striata Karsch, 1878						1		1	1
Philodromidae	**2**	**50**		**52**	**3**	**41**		**44**	**96**
Philodromidae sp.		1		1					1
Hirriusa arenacea Lawrence, 1927	2			2	3			3	5
Philodromus sp.		12		12		12		12	24
Tibellus minor Lessert, 1919						1		1	1
Tibellus sp.		37		37		28		28	65
Pholcidae							1	**1**	**1**
*Smeringopu*s sp.							1	1	1
Pisauridae	**1**		**75**	**76**	**1**		**57**	**58**	**134**
Chiasmopes sp.			1	1					1
Euprosthenops sp.			63	63			43	43	106
Euprothenopsis sp.			4	4			6	6	10
Maypacius sp.			3	3			5	5	8
Nilus margaritatus Pocock, 1898	1			1	1			1	2
Perenethis simoni Lessert, 1901			3	3			3	3	6
Perenethis symmetrica Lawrence, 1927			1	1					1
Salticidae	**451**	**144**	**2**	**597**	**396**	**163**	**1**	**560**	**1157**
Baryphas ahenus Simon, 1902						1		1	1
Belippo n.sp.	1			1					1
Bianor albobimaculatus Lucas, 1846	1			1	1			1	2
Brancus muticus Simon, 1902						1		1	1

Evarcha flagellaris Haddad & Wesolowska, 2011	8		8	4		4	12
Evarcha ignea Wesolowska & Cumming, 2008	1		1				1
Evarcha prosimilis Wesolowska & Cumming, 2008	1		1	1		1	2
Evarcha sp.	12		12	18		18	30
Evarcha striolata Wesolowska &Haddad, 2009	1		1	1		1	2
Evarcha zimbabwensis Wesolowska & Cumming, 2008	1		1	3		3	4
Habrocestum sp.				3		3	3
Heliophanus pistaciae Wesolowska, 2003					2	2	2
Heliophanus sp.		2	2	2		2	4
Heliophanus sp.1				1		1	1
Heliophanus transvaalicus Simon, 1901		2	2	4		4	6
Hermippus tenebrosus Jocque, 1986				1		1	1
Hyllus argyrotoxus Simon, 1902		4	4	9		9	13
Hyllus dotatus Peckham & Peckham, 1903		2	2	4		4	6
Hyllus sp.		41	41	54		54	95
Langelurillus minutus Wesolowska & Cumming, 2011	3		3	1		1	4
Langona sp.	15		15	13		13	28
Langona tortuosa Wesolowska, 2011	6		6	11		11	17
Langona zimbabwensis Wesolowska & Cumming, 2011	1		1	1		1	2
Mexcala angolensis Wesolowska, 2009	1		1				1
Nigorella hirsuta Wesolowska, 2009	1		1				1
Nigorella manica Peckham & Peckham, 1903	3		3				3
Parajotus obscurofemoratus Peckham & Peckham, 1903				1		1	1
Pellenes bulawayoensis Wesolowska, 2000	1		1	3		3	4
Pellenes tharinae Wesolowska, 2006	1		1	4		4	5
Phlegra langanoensis Wesolowska & Tomasiewicz, 2008				1		1	1
Phlegra procera Wesolowska & Cumming, 2008				1		1	1

Taxon							
Phlegra simplex Wesolowska & Russell-Smith, 2000	2		2	3		3	5
Rhene sp.					1	1	1
Salticidae sp.	29		29	28		28	57
Stenaelurillus guttiger Simon, 1901	256		256	216		216	472
Stenaelurillus termitophagus (Wesoloska & Cumming, 1999)	46		46	36		36	82
Stenaelurillus sp. 3				3		3	3
Stenaelurillus sp. 4	38		38	37		37	75
Stenaelurillus sp. 5	16		16	3		3	19
Thyene australis Peckham & Peckham, 1903		1	1				1
Thyene inflata Gerstacker, 1873		10	10		6	6	16
Thyene natali Peckham & Peckham, 1903		1	1		1	1	2
Thyene sp.		36	36		40	40	76
Thyene thyenioides Lessert, 1925		41	41		33	33	74
Thyenula sp.	4		4	1		1	5
Tusitala sp.		4	4		4	4	8
Scytodidae	**4**		**4**	**4**		**4**	**8**
Scytodes quarta Lawrence, 1927				1		1	1
Scytodes sp.	4		4	3		3	7
Segestridae					1	**1**	**1**
Ariadna sp. 1					1	1	1
Selenopidae				**4**		**4**	**4**
Selenops sp. 1				3		3	3
Selenops sp. 2				1		1	1
Sicariidae	**1**		**1**	**1**		**1**	**2**
Loxoscelessimillima Lawrence, 1927	1		1				1
Loxosceles sp.				1		1	1

Sparassidae	**12**	**12**	**13**		**13**	**25**
Olios brachycephalus Lawrence, 1938			3		3	3
Olios correvoni Lessert, 1921	1	1				1
Olios sp. 3	10	10	9		9	19
Pseudomicrommata vitigera Simon, 1897	1	1				1
Pseudomicrommata sp. 2			1		1	1
Tetragnathidae	**1**	**1**	**1**		**1**	**2**
Tetragnatha sp.	1	1	1		1	2
Theraphosidae			**3**		**3**	**3**
Ceratogyrus sp. 1			1		1	1
Ceratogyrus sp. 2			2		2	2
Theridiidae	**83**	**83**	**1**	**59**	**60**	**143**
Achaearaneae sp.			1		1	1
Argyrodes convivans Lawrence, 1937	6	6		3	3	9
Argyrodes sextuberculosis Strand, 1908				1	1	1
Argyrodes sp.	3	3		3	3	6
Argyrodes sp.1	4	4		1	1	5
Argyrodes zonatus Walckenaer, 1841	3	3				3
Enoplognatha sp. 1				1	1	1
Enoplognatha sp. 2				1	1	1
Euryopis episinoides Walckenaer, 1847	1	1				1
Euryopis sp. 2	3	3				3
Lactrodectus geometricus C.L. Koch, 1841	6	6		7	7	13
Lactrodectus renivulvatus Dahl, 1902	10	10		6	6	16
Lactrodectus rhodesiensis Mackay, 1972	2	2				2
Lactrodectus sp. 4	23	23		13	13	36
Phoroncidia eburnea Simon, 1895	4	4				4

	M	F	J	Total	M	F	J	Total	Total
Phoroncidia sp.			4	4			4	4	8
Steatoda sp.			2	2			2	2	4
Theridiidae sp.	1		10	11			10	10	21
Theridion sp.			2	2			6	6	8
Thymoites sp. 1						1		1	1
Thomisidae	**46**	**160**		**206**	**13**	**154**	**1**	**168**	**374**
Diaea sp.		4		4		4		4	8
Firmicus sp.		1		1					1
Heriaeus crassispinus Lawrence, 1942		1		1					1
Heriaeus n.sp.		1		1		1		1	2
Misumenops rubrodecoratus Millot, 1942		2		2		2		2	4
Misumenops sp.		2		2					2
Monaeses austrinus Simon, 1910		1		1					1
Monaeses gibbus Dippennar-Schoeman, 1984		2		2					2
Monaeses griseus Pavesi, 1897		2		2		1		1	3
Monaeses paradoxus Lucas, 1846		3		3		1		1	4
Monaeses sp.		48		48		60		60	108
Mystaria savannensis Lewis & Dippennar-Schoeman, 2014		1		1					1
Oxytate sp.		3		3		3		3	6
Pactactes sp. 1		1		1					1
Pherecydes sp.		1		1					1
Runcinia flavida Simon, 1881		26		26		23		23	49
Runcinia sp.						1		1	1
Simorcus cotti Lessert, 1936						1		1	1
Simorcus sp.		1		1					1
Synema sp.		6		6		5		5	11
Thomisops sp.		9		9		7		7	16

Thomisus congoensis Comellini, 1957	1	1			1
Thomisus daradioides Simon, 1890			1	1	1
Thomisus granulatus Karsch, 1880	2	2	2	2	4
Thomisus scrupeus Simon, 1886			1	1	1
Thomisus sp.	32	32	30	30	62
Thomisus stenningi Pocock, 1900	1	1			1
Tmarus africanus Lessert, 1919			1	1	1
Tmarus sp.	9	9	10	10	19
Xysticus havillandi Lawrence, 1942	39	39	10	10	49
Xysticus sp.	6	6	2	2	8
Trachelidae	1	1			1
Thysanina sp.	1	1			1
Uloboridae	18	18	19	19	37
Miagrammopes sp.	18	18	18	18	36
Uloborus sp.			1	1	1
Zodariidae	165	165	119	119	284
Capheris decorata Simon,1904	1	1			1
Capheris fitzsimonsi Lawrence, 1936	61	61	46	46	107
Diores magicus Jocque & Dippennar-Schoeman, 1992			2	2	2
Diores n.sp.	4	4	1	1	5
Diores rectus Jocque, 1990	3	3	2	2	5
Diores salisburyensis Tucker, 1920			1	1	1
Diores sp.	8	8	12	12	20
Diores sp.2	6	6	1	1	7
Hermippus sp.	18	18	12	12	30
Hermippus tenebrosus Jocque, 1986	31	31	17	17	48
*Mallinella n.*sp. 1	3	3	1	1	4

Mallinella sp. 2	12			12	10			10	22
Mallinella sp. 3	16			16	11			11	27
Palfuria n.sp. 1	1			1	1			1	2
Palfuria sp. 2					1			1	1
Ranops caprivi Jocque, 1991					2			2	2
Grand Total	**1262**	**511**	**446**	**2219**	**1080**	**532**	**361**	**1973**	**4192**

Appendix 2.2: Summary results of univariate multivariate generalized linear model of the abundance of all genera of ground wanderers, web-builders and plant dwellers identified in this study.

	Short duration kraaling		Time since kraal removal (months)		Mean grass height (cm)		(Mean grass height^2)		Percentage leaf litter cover		Percentage coarse woody debris cover	
	Wald	Pr (>wald)	Wald	Pr (> wald)	Wald	Pr (> wald)			Wald	Pr (> wald)	Wald	Pr (> wald)
Web builder's												
Araneus	4.441	0.019			4.163	0.031						
Euprosthenopsis	3.393	0.030										
Nephila	7.063	0.019							3.622	0.003		
Neoscona	3.745	0.021										
Stegodyphus											3.343	0.029
Ground wanderer's												
Allocosa			4.662	0.001								
Ammoxenus	3.571	0.016							3.166	0.024		
Asemesthes			4.951	0.001					4.467	0.001		
Capheris			6.978	0.001								
Drassodes			5.571	0.0001								
Evippomma									2.998	0.038		
Hermippus									3.664	0.003		

Ibala									3.845	0.003		
Langona									3.218	0.019		
Mallinella			4.64	0.001								
Pardosa									3.145	0.026		
Salticidae			3.203	0.024								
Stenaelurillus			3.931	0.002					6.416	0.001		
Theuma			3.458	0.016								
Plant wanderer's												
Cheiracanthium			4.191	0.001								
Hamataliwa									2.925	0.038		
Hyllus			4.298	0.001	4.046	0.012			4.444	0.001		
Monases			5.119	0.001			3.898	0.011				
Thomisus					3.516	0.018			2.999	0.031		
Thyene							3.497	0.026				

Appendix 2.3: Summary results of top ten most abundant species recorded in all six sampling intervals at Debshan Ranch from July 2017 to April 2018, a = less than two months after kraal removal, b= two months after kraal removal,, c = 4 months after kraal removal, d= 6 months after kraal removal, e = 8 months after kraal removal and f = 10 months after kraal removal.

d

Monases sp. 8%
C. fitzsimonsi 1%
Allocosa sp. 6%
Oxyopes sp. 10%
A.paynteri 20%
S. guttiger 55%

e

S. guttiger 21%
A.paynteri 9%
C. fitzsimonsi 39%
Oxyopes sp. 11%
Allocosa sp. 17%
Monases sp.

f

Monases sp. 3%
C. fitzsimonsi 1%
S. guttiger 11%
A.paynteri 8%
Allocosa sp. 42%
Oxyopes sp. 35%
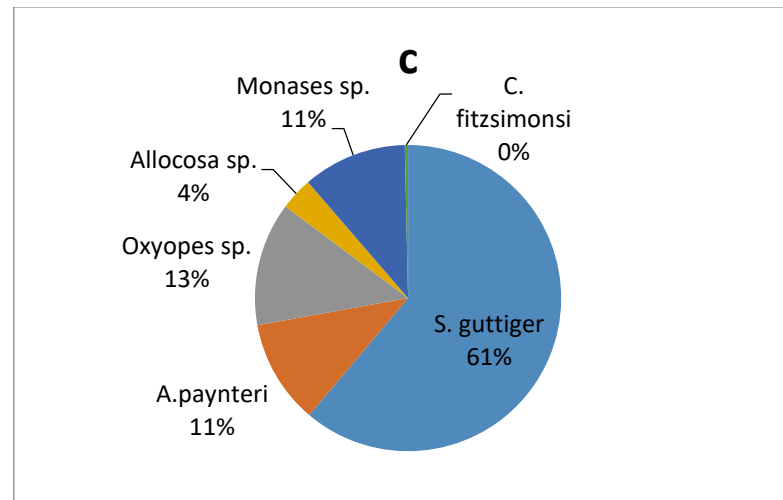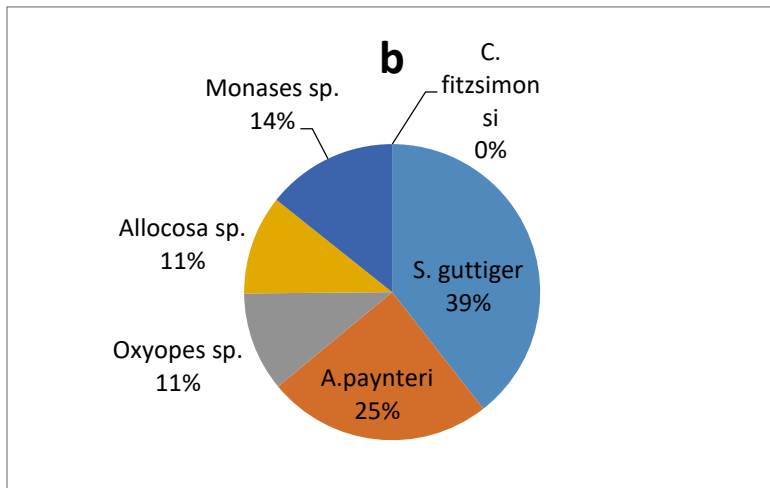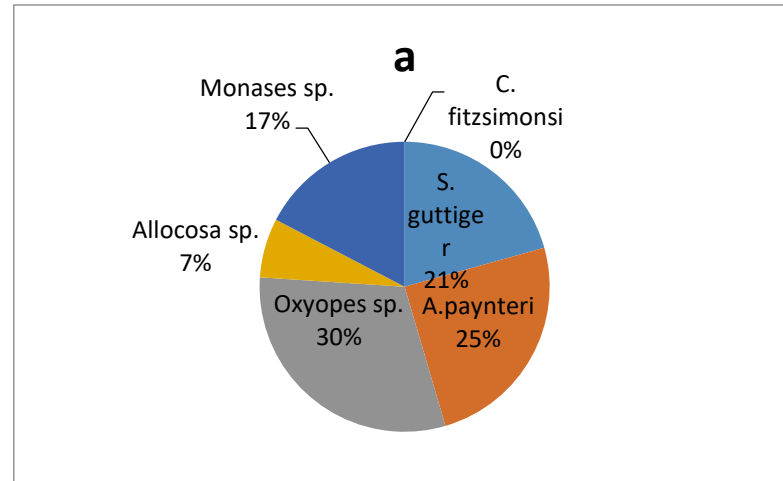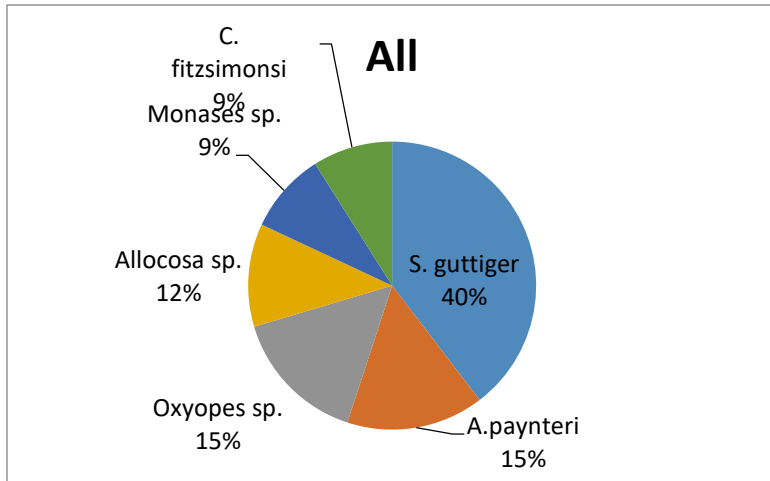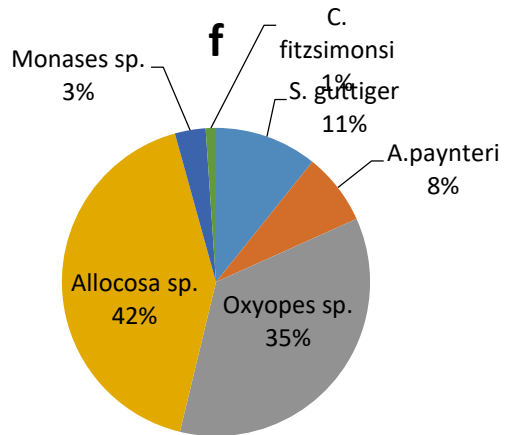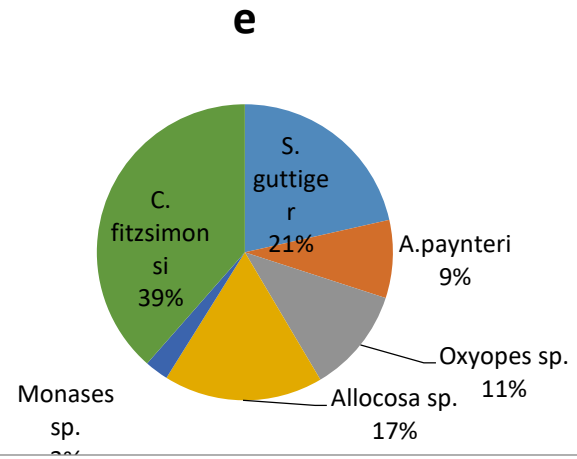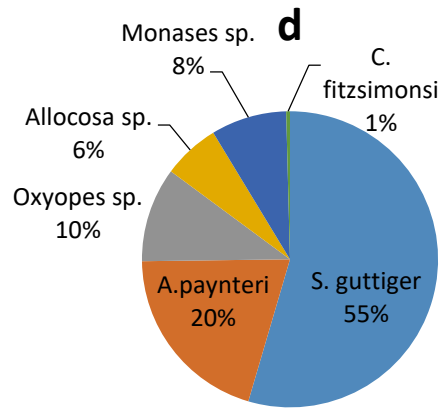
Chapter 3

Short-term spider community responses to cattle disturbance from previously kraaled inclusions and their surroundings in Zimbabwe

3.1 Abstract

The influence of short duration kraaling has been well documented for several organisms that include plants, wildlife and macro-invertebrates however, limited information is available on its impact on spider assemblages. A matched pair design (inside previously kraaled inclusions vs. outside previously kraaled inclusions) in twenty two sites, using four herds (H_1, H_6, H_7 and H_{Nguni}) for a total of 88 sampling points were utilized at Debshan Ranch, Zimbabwe to assess the effect of short duration kraaling, time since cattle removal and vegetation structure, on spider fauna at two time intervals (surveys): early rainy season and late rainy season. At each of the four replicates, spiders were sampled using pitfall traps. Five pitfall traps were installed both inside and outside the kraal disposed in a cross format placed 50 m apart from the center of each kraal for a total of 880 pitfall traps. A total of 634 spiders were captured, comprising 80 species, 60 genera belonging to 25 families. The most abundant family was Lycosidae (49.4%; 25 species), followed by Gnaphosidae (15.3%; 14 species), and Salticidae (13.88%; 9species). Generalised linear mixed models revealed that both time since kraal removal and vegetation structure percentage coarse woody debris cover were relatively important predictors of spider diversity, while only spider abundance responded to short duration kraaling and season. Multivariate generalized linear models revealed that community composition was significantly related to percentage coarse woody debris cover and that this relationship varied with time since kraal removal and with season. Coarse woody debris was the most significant predictor of spider genera richness, abundance and genera composition proving its importance to spider assemblages.

3.2 Introduction

Cattle enclosures, normally referred to as bomas (Stelfox 1986), livestock corrals (Augustine *et al.* 2009) or kraals (Huruba *et al.* 2018), have been part of daily management within livestock practices for decades (Augustine 2003). They are used as overnight protection enclosures from theft and livestock predators in most African rangelands. In addition, kraals have also been used as confinements that enable milk extraction, as well as the concentrated production of manure

that can be used in crop production (Abagale & Ayuegabe 2015). Kraals are enclosures that hold livestock (Borg 1996), which are usually made out of material that ranges from thorn scrub branches (Augustine *et al.* 2009), fences (Stelfox 1986) and sometimes boma sheeting (Nyoni 2015). They are commonly designed either as circular or rectangular, the former being preferred to the latter, as it avoids the bunching up of livestock in its corners (Borg 1996).

During the day, the cattle are normally permitted to graze in nearby areas, normally far away from the kraals and at night when they return inside the kraals their dung and urine redistributes the nutrients that were obtained during the day into soil nutrient rich patches of heterogeneity (Augustine 2003). These cattle enclosures can either be short-term in duration, usually lasting for a period of seven days or less (Huruba *et al.* 2018), or long-term, where kraal owners utilise the same location for decades within the vicinity of water resources such as boreholes or shallow pans (Kizza & Areola 2010). The former is a recently developed practice that has since been incorporated as part of the innovative management approach called holistic planned grazing (Savory & Parsons 1980, Savory 1983), while the latter comprises the traditional kraaling culture in most semi-arid regions in Southern Africa.

The influence of abandoned kraal sites has been well documented (Kizza & Areola 2010, Sibanda *et al.* 2016, Chikorowondo *et al.* 2017, Huruba *et al.* 2017, Chikorowondo *et al.* 2018, Huruba *et al.* 2018 and Muvengwi *et al.* 2018). Initial research focused on the importance of kraals as nutrient reserves that can be utilised in crop production (Kangalawe *et al.* 2008). Abandoned kraal sites enhance the soil nutrient status, as many studies have recorded higher nutrient levels, including nitrogen, phosphorous and potassium, in abandoned kraals when related to the surrounding control plots (Muchiru *et al.* 2009, Kizza *et al.* 2010, Abagale & Ayuegabe 2015, Chikorowondo *et al.* 2017, Huruba *et al.* 2018, Muvengwi *et al.* 2018). Cattle manure has also been shown to be a good biological resource that should be utilised economically in crop production, as it is environmentally friendly and reduces hazards of synthetic fertilizers (Abagale & Ayuegabe 2015). According to Augustine (2003), abandoned kraaled sites retain their nutrient enrichment for at least four decades, therefore, their placement and relocation rates should be adequately planned in order to attract wildlife to underutilised areas (Stelfox 1986). Chikorowondo *et al.* (2017) recommends the need for a conservation monitoring program of such nutrient-rich patches, which serve as good grazing hotspots for most herbivores.

The nutrient-rich patches enhanced by kraaling also have implications for vegetation diversity. Huruba *et al*. (2018) in a study conducted at Debshan Ranch in Shangani, found evidence to suggest that formerly kraaled sites had greater quantities of edible grass species. Similar findings were made at Dimbangombe Ranch in Hwange by Sibanda *et al.* (2016) who recorded higher abundances of the palatable grasses *Urochloa mosambicensis* and *Panicum maximum* in abandoned kraal sites. In contrast, Chikorowondo *et al.* (2018) did not find any significant difference in plant diversity between abandoned kraal sites and control plots, which was mainly attributed to their lack of utilisation of enclosure plots of kraals in their sampling design. It has also been reported that grasses on abandoned kraals are favored to those in controls plots by wild herbivores (Chikorowondo *et al.* 2017). However, the composition of wildlife that frequents these abandoned kraals is not quite certain (Huruba *et al.* 2018), but there have been indication of herbivores such as warthog (*Phacochoerus africanus*) and impala *Aepyceros melampus* (Huruba *et al.* 2017) that have been spotted in abandoned kraal sites. Their presence has been attributed to the strong resprouting response of grasses due to kraaling, as well as the increase of palatable grass species in these nutrient-rich patches. Short-duration overnight kraaling is an effective approach that can be utilised not only to augment soil nutrient status (Abagale & Ayuegabe 2015), which minimises environmental pollution, but also as a management tool within rangelands, as it provides heterogeneous grass patches that are dominated mostly by palatable grass species therefore improving grass production.

Apart from the work on the diversity and abundance of macro-invertebrates (Muvengwi *et al.* 2018), and that on the functional diversity of macro-invertebrates (Chikorowondo *et al.* 2018) limited knowledge is available on the influence of previously kraaled enclosures on invertebrates, in particular that of spiders. Regardless that spiders are amongst the most diverse organisms (Palem *et al.* 2017) with almost 48 438 described species (World Spider Catalog 2020). They occupy almost all possible microhabitats (Turnbull 1973, Foelix 2011). Ecologically, spiders are important as generalist predators (Nyffeler & Birkhofer 2017) that commonly feed on insects, the latter forming the majority of agricultural pests, therefore rendering spiders as beneficial pest control agents (Nentwig & Kobelt 2010). Spiders have also been reported to feed on small animals such as bats (Nyffeler & Knornschild 2013), aquatic organisms such as fish (Nyffeler & Pusey 2014) and even feed on other spiders (Wise 2006). Economically, spiders are also good produces of silk (Widhe *et al*. 2011) that might have uses in

the biomedical field. Spider diversity has also been shown to be dependent on several factors: (Foelix 2011) vegetation structure (Baldissera *et al*. 2004, Roberson *et al*. 2016), dispersal capacity and settlement (Rodrigues *et al*. 2009, Lin *et al.* 2016), prey availability and competitive exclusion (Dennis *et al*. 2015, Rodriguez-Artigas *et al.* 2016), being distant insignificant (Horvath *et al.* 2009).

Spiders possess several qualities that make them good bio indicators (Churchill 1997, Marc *et al.* 1999). As a result there have been used as bio-indicators of environmental disturbances such as fire (Pryke & Samways 2012, Haddad *et al.* 2015) habitat changes (Haddad *et al*. 2009), habitat quality (Halaj *et al.* 1998) and grazing (Ford *et al.* 2013, Fuller *et al.* 2014, Schwerdt *et al.* 2018). They have also been used to determine other widespread environmental changes that include spider fauna associated with leaf litter (Castro & Wise 2009, Butler & Haddad 2011, Podgaiski *et al.* 2013), seasonality (Niemela *et al.* 1994, Weeks & Holtzer 2000, Mineo *et al.* 2010) and rainfall gradients (Churchill 1998). Generally, the impact of grazers on invertebrates can be positive, negative and neutral (Gibson *et al.* 1992). A negative impact occurs when the total abundance is illustrated to decrease with grazing and there is a reduction in faunal composition (Gibson *et al.* 1992, Churchill & Ludwig 2004, Szineter & Samu 2012, Foord *et al.* 2013, Fuller *et al.* 2014), whereas a positive impact occurs when there is a significant increase in arthropod abundance with increasing disturbance (Seymour & Dean 1999) and a neutral impact occurs when there is no significant difference between grazed and ungrazed sites (Harris *et al* 2003, Jansen *et al*. 2013).

According to Podgaiski *et al.* (2013), spiders have the potential to reoccupy impacted habitats within a period of less than one month, and therefore they are a great taxon for testing the effect of short duration kraaling (7 days) on spider assemblages inside previously kraaled inclusions and their surroundings. The aims of the study were to: i) describe the changes in mean grass height (cm) over the two sampling periods early summer (November) and late summer (March) inside previously kraaled inclusions and their surroundings; (ii) to determine effect of short duration kraaling on spider assemblages inside previously kraaled inclusions and their surroundings; (iii) to determine the influence of time since kraal removal on spider assemblages inside previously kraaled inclusions and their surroundings; (iv) to evaluate the effect of vegetation structure variables (% vegetation cover, % coarse woody debris cover, % rock cover, % cow dung cover, % leaf litter and mean grass height (cm) on spider assemblages.

3.3 Materials and methods

3.3.1 Study area

For a full site description of the Debshan Ranch (location, climate, soils, natural vegetation and fauna) refer to Chapter 2.

3.3.2 Sampling design, methods and period

Ground-dwelling spiders were collected by pitfall traps (refer to Fig. 2.4A; Chapter 2) in 88 sites (Appendix 3.1) located within four replicates, which were represented by Herds 1, 6, 7 and Nguni (Fig. 3.1) that contained approximately 350-396 cattle per herd (Huruba *et al.* 2018). Each site represented a spatial node of boma footprints of the holistic program based on a chronological time series of kraal occupation, dating back to ten months (Appendix 3.1). The latter signified the full cycle of resting the land in the holistic planned grazing programme at Debshan Ranch. In order to avoid seasonality influences of spiders (Whitmore *et al.* 2002, Muelelwa *et al*. 2010), samples within each replicate, were taken at the same time during two sampling intervals: early-summer (November 2017) and late-summer (March 2018) as these are the periods where spider activity is considered to be relatively high within the savanna region (Muelelwa *et al.* 2010).
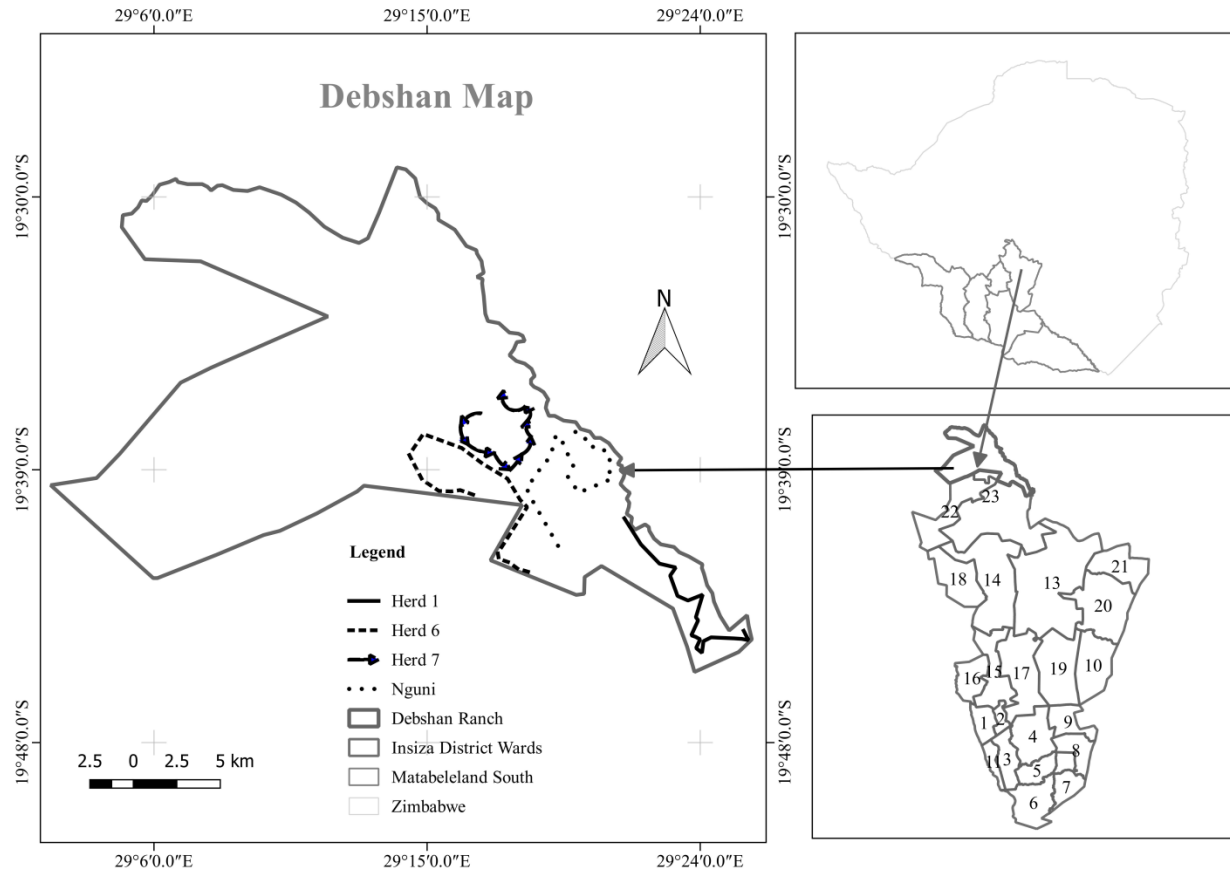
Fig. 3.1 Map of Debshan Ranch and its location within Zimbabwe, showing the distribution of the spider sampling points sampled within each replicate herd; Herd 1, Herd 6, Herd 7 and Herd Nguni during the early rainy season (November 2017) and the late rainy season (March 2017) .

At each site, a matched-pair design (inside previously kraaled inclusion vs. outside previously kraaled inclusion) containing 10 pitfall traps positioned in an cross format, with five pitfalls placed inside the kraal and five pitfall traps placed outside the kraal (Fig. 3.2), making a total of 880 (88 sites x 10 pitfalls) pitfall traps in total in all sites. Each pitfall trap was 10 m apart whereas the centre pitfall traps in each treatment were 50 m apart. Each trap consisted of glass bottles 14 cm deep and 9 cm wide at the mouth, placed inside a plastic PVC pipe and buried to their rims in the ground (refer to Fig. 2.4A; Chapter 2). The glass bottle was filled with 100 ml of 70% propylene glycol. Pitfall traps were left open for 14 days and emptied twice in each sampling period. At the end of the collecting period, the contents were collected and emptied into suitable vessels for sorting in the laboratory.
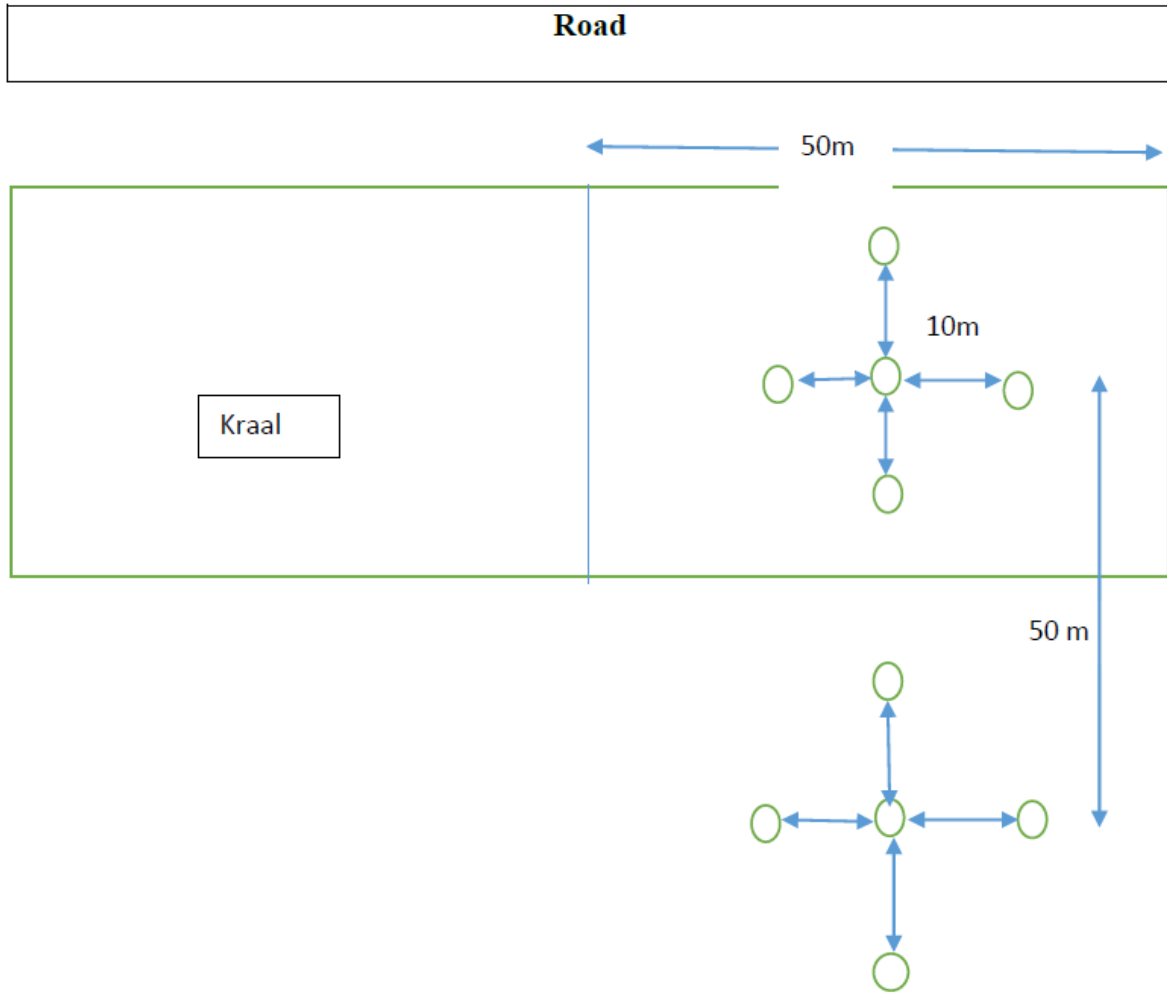
Fig. 3.2 Format used for the arrangement of pitfall traps used to sample spider diversity inside and outside the previously kraaled sites at Debshan Ranch, Zimbabwe. Traps were placed 10 m apart and at least 25 m from the kraal boundaries.

3.3.3 Identification of specimens

Adult specimens were sorted into morphospecies and identification was done to species level were possible, with all the juveniles identified to at least the genus level. Preliminary spider sorting was done by the student following identification keys by Dippenaar- Schoeman & Jocqué (1997). Professor Charles Haddad and Dr. Moira J FitzPatrick also assisted with the identification process. All adult spider specimens collected in the study are preserved in the Department of Arachnology, Natural History Museum in Bulawayo.

3.3.4 Vegetation structure variables

Vegetation structure was quantified by placing a 1m x 1m quadrat over each pitfall and a photograph was taken (refer to Fig. 2.5A; Chapter 2). During each sampling period, a total of 880 images were taken in the 88 previously kraaled sites (Appendix 3.1) during the early summer period (November 2017) and a total of 660 images were taken in the 66 previously kraaled sites during the late summer period (March 2018), which gave a total of 1540 images taken during the two sampling periods at Debshan ranch. Images were analysed to calculate percentage cover of each vegetation structure variable that included bare ground, leaf litter, vegetation, coarse woody debris, cow dung, and rock. Visual estimates of each vegetation structure variable were estimated utilising methodology described in Dethier *et al.* (1993). In each of the selected 88 previously kraaled sites a 40 m transect was demarcated inside and outside the kraal, in each transect 10 grasses placed a meter apart were selected and marked with plastic ear tags and measured at the tallest vertical point (refer to Fig. 2B; Chapter 2). In each sampling point the average grass height (cm) was determined from the measurements.

3.3.5 Data analysis

The R statistical software version 3.6.0 (R Core Team 2019) was utilised to carry out all statistical analyses. Coverage was calculated using iNEXT package (Hsieh *et al.* 2016) as it enables the estimation of the percentage of the total species that is obtained in a sample, and its counterpart is essentially the likehood that the next sampled individual might be a formerly unsampled species (Chao & Jost 2012).

In order to assess vegetation structure changes inside previously kraaled inclusions and their surroundings, comparison of the mean grass height (cm) between the two sampling periods early summer (November 2017) and late summer (March 2018) season was done. Mean grass height

(cm) was included as the response variable, kraaling treatment, time since kraal removal, season were included as the fixed variables with herd included as the random factor. Several competing models were selected that were based on prior hypothesis about variables which were suspected to potentially explain the changes in mean grass height (cm). The model that best explained the response of mean grass height (cm) was selected using an Information theoretic approach based on Akaike Information Criterion (AIC) (Burnham & Anderson 2002) value by at least 2 units. Mean grass height (cm) was log transformed before the analysis (Harrison *et al.* 2018) in order to reduce heteroscedasticity and Generalised linear mixed models (GLMM) with the identity link function and normally distributed errors, using the lmer function of the lme4 package (Bates *et al.* 2013) were utilised.

Factors affecting spider abundance and genera richness were examined using GLMM log-link function and Poisson error distribution with the glmer function of the lme4package (Bates *et al.* 2013). Model validation was done by inspection of residual analyses (Zuur *et al.* 2010, Zuur & Ieno 2016). Model selection was done using an Information theoretic approach based on Akaike Information Criterion (AIC) (Burnham & Anderson 2002) value by at least 2 units, as it has been considered to be superior to traditional stepwise procedures due to its ability to account for uncertainties regarding model structure and parameter estimation in the observed data set in the presence of several competing models (Whittingham *et al.* 2006). Short duration kraaling, time since kraal removal, sampling season, and vegetation structure variables (Table 3.1) were modeled as fixed factors with herd included as random factors. In order to determine the amount of discrepancy that was elucidated by the model, the Marginal R^2 (owing to fixed effects only) and the conditional R^2 (owing to both random and fixed effects) was also calculated (Nakagawa & Schielzeth 2013).

Table 3.1 Site (UNK: outside previously kraaled inclusions; KRA: inside previously kraaled inclusions) characteristics summarised by replicates for each of the vegetation structure variables measured at previously kraaled inclusions and their surroundings at Debshan Ranch, Zimbabwe and used as explanatory variables in the analysis on spider fauna.

Site	n	Rep	Bare soil	Vegetation	Leaf litter	Rock	Coarse woody debris	Cow dung	Grass height (cm)
UNK	6	H1	23.96±6.99	64.89±5.43	8.43±5.13	1.81±0.73	28.25±22.95	5.50±6.40	67.33±11.33
	6	H6	22.26±2.47	61.99±4.77	12.31±6.37	2.29±1.64	37.25±12.34	11.25±14.45	80.87±21.28
	6	H7	19.56±3.58	58.90±9.21	20.53±7.72	0.08±0.18	18.40±15.74	0.20±0.45	66.25±10.49
	6	HNG	17.22±1.28	70.56±2.91	10.14±2.30	0.54±0.51	41.80±16.10	21.60±20.94	92.92±31.30
KRA	6	H1	32.38±1.06	44.05±7.34	14.01±4.50	0.77±0.53	39.75±18.01	280±192.93	49.01±21.38
	6	H6	26.84±6.17	53.77±12.50	13.19±7.98	1.09±0.53	40.17±18.07	157.67±91.68	64.62±36.64
	6	H7	36.15±10.84	37.42±11.35	18.80±14.23	0.47±0.69	41.17±28.82	99.00±67.58	38.16±14.52
	6	HNG	26.26±6.83	49.03±10.20	17.07±7.40	1.48±0.76	56.83±20.60	176.33±51.91	67.7±39.74

Grass height: mean± SD vegetation height (cm). Values for bare soil, vegetation, leaf litter, rock, coarse woody debris, and cow dung are mean ± SD % cover.

Variation in spider community assemblages was analysed through the utilisation of Generalised linear latent models (Niku *et al.* 2019). The gllvm package was utilised to perform the latent variable modeling as it allows for the selection of a model that captures key data properties through model selection and residual analysis (Niku *et al.* 2019). In order to visualise clusters of spider communities across the treatments and also by time since kraal removal, ground dwelling communities were modeled with two latent variables with negative binomial distribution as it provided a better fit. In order to determine the potential role of short duration kraaling and time since kraal removal together with the interaction of short duration kraal and time since kraal removal on spider communities, multivariate generalised linear model were performed using the function "manyglm" in the package mvabund (Wang *et al.* 2012). The method fits many GLM simultaneously to variables and 'anova.manyglm' was utilised for hypothesis testing. Univariate test statistic and their corresponding p-values were calculated for each species in order to determine their respective contribution in each model. Model residuals, normality and independence and constant-mean variance relationships were inspected in all cases.

3.4 RESULTS

3.4.1 Community composition

A total of 634 individuals in 80 species represented in 60 genera and 24 families were collected during the study period. Of the 634, 451 were adults identified to species and 183 juveniles identified to genera level. Four hundred and thirty were collected during the early season in December 2017 while 204 were collected during the late season in March 2018 (Appendix 3.2). The most common family were the Lycosidae and the Gnaphosidae (Table 3.2). Nearly one quarter of the mature spiders was either *Allocosa umtalica* (Purcell, 1903) (101 specimens) or *Asemethes payntheri* (Tucker, 1923) (61 specimens). The four most common species consisted of 40% of the overall sum of spiders. One species is possibly new based on identifications of specialists (Appendix 3.2); forty five species were encountered only once. Sample coverage for both the inside of the previously kraaled inclusions and their surrounding areas was relatively high (> 94%).

3.4.2 Changes in mean grass height (cm) during the sampling periods

In both sampling seasons mean grass height (cm) was relatively lower inside the previously kraaled inclusions than their surrounding vegetation (Fig. 3.3). In the late summer season there was an initial drop in mean grass height (cm) inside the previously kraaled inclusions during the period of 2 months after kraal removal, however after 4 months there an increase in mean grass height (cm) until 8 months which recorded similar mean grass height (cm) in both the inside and outside the kraals with10 months after kraal removal recording greater mean grass height (cm) inside the previously kraaled inclusions (Fig. 3.3). The early summer season however indicated different results, despite the initial increase in mean grass height (cm) after 2 months of kraal removal the mean grass height (cm) inside the previously kraaled inclusions was relatively lower than the outside of the kraals even at 8 and 10 months after kraal removal (Fig. 3.3). These changes were best explained by a model that combined the short duration kraaling treatment and sampling season compared to other models (Table 3.3).

Table 3.2 Family compositions of ground dwelling spider fauna collected from Debshan Ranch, Zimbabwe.

Family	Total collected	% of total	Total species	% of total
Agelenidae	1	0.16	-	-
Ammoxenidae	17	2.68	1	0.13
Araneidae	6	0.95	3	3.75
Barychelidae	1	0.16	-	-
Caponiidae	1	0.16	-	-
Corinnidae	9	1.42	1	0.13
Ctenidae	5	0.79	1	0.13
Cyrtaucheniidae	1	0.16	-	-
Cheiracanthidae	4	0.63	2	2.5
Gnaphosidae	101	15.93	15	18.75
Hersiliidae	1	0.16	1	0.13
Liocranidae	5	0.79	2	2.5
Lycosidae	313	49.37	20	25
Migidae	1	0.16	1	1.25
Oxyopidae	29	4.57	5	6.25
Philodromidae	8	1.26	5	6.25
Pisauridae	13	2.05	3	3.75
Salticidae	88	13.88	9	11.25
Segestridae	1	0.16	1	1.25
Selenopidae	1	0.16	1	1.25
Sparassidae	8	1.26	3	3.75
Theridiidae	1	0.16	-	-
Thomisidae	1	0.16	-	-
Zodariidae	18	2.84	6	7.5
Total	634	100	80	100
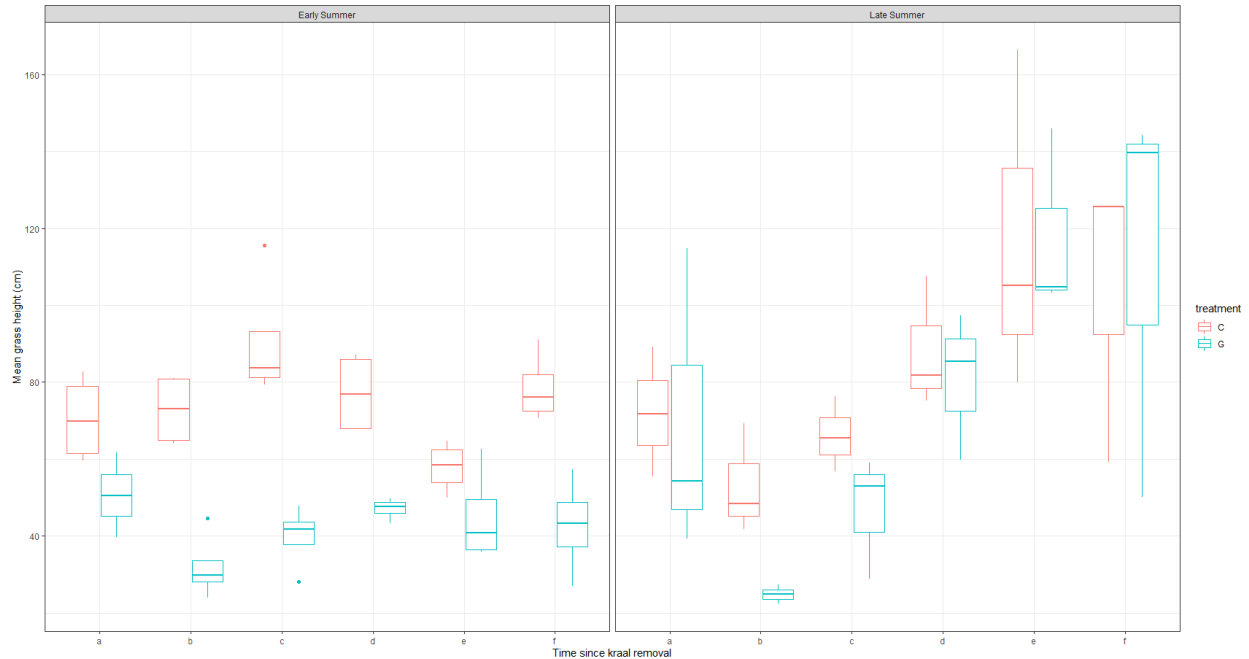
Fig. 3.3 Changes in mean grass height (cm) at Debshan Ranch, Zimbabwe over the two sampling periods from July 2016 to April 2017.The box bounds the 25th and 75th percentiles, while the whiskers represent ± 1.5 IQR (interquartile range). Dots represent outliers. Sample size = 6. Abbreviations; for sampling periods a = period less than two months after kraal removal, b = 2 months after kraal removal, c = 4 months after kraal removal, d = 6 months after kraal removal, e = 8 months after kraal removal, f = 10 months after kraal removal; for treatment G = inside previously kraaled site and C = outside previously kraaled sites.

Table 3.3 Generalised linear mixed models that explain the effects of holistic planned grazing, time since kraal removal and season on mean grass height (cm) and percentage grass cover sampled at Debshan Ranch, Zimbabwe. Holistic planned grazing had two levels (kraaled vs. unkraaled), time since kraal removal had six level (before, during, 1, 3, 6, 10 months since cattle occupation) and season had two levels (early and late summer).

Models	K	Log likelihood	AICc	Delta AIC	weight
Mean grass height					
Kraaling + Season	5	-36.906	84.6	0.00	0.745
Kraaling	4	-39.461	87.45	2.84	0.180
Kraaling+ Time since kraal removal+ Season	10	-33.251	89.68	4.975	0.062
Kraaling treatment + Time since kraal removal	9	-36.111	92.71	8.10	0.013

3.4.3 Spider abundance and diversity patterns

Ground dwelling genera richness was significantly and positively influenced by percentage coarse woody debris cover (GLMM, $z = 2.826$; $p < 0.01$). Time since kraal occupation had a significant impact on spider genera richness of the ground dwelling spiders ($z = 3.136$; $p < 0.01$) with diversity peaking four months after the kraal is removed (Fig. 3.4). Fixed variables explained 16.15 % of the variation with the random variables explaining only a further 0.81 % of the variation. Ground dwelling abundance was significantly lower inside the previously kraaled inclusions ($z = -3.665$; $p < 0.001$) than their surrounding vegetation. Time since kraal occupation had a significant impact on spider abundance ($z = 2.613$; $p < 0.01$) of the ground dwellers, with the highest abundance occurring 10 months after kraal removal (Fig. 3.5). The late sampling season had significantly lower ground dwelling abundance ($z = -3.021$; $p < 0.01$) than the early sampling season. The ground dwelling abundance was also significantly and positively influenced by percentage coarse woody debris cover (Fig. 3.6). Fixed variables explained 50.99 % of the variation with the random variables explaining only a further 5.41 % of the variation.
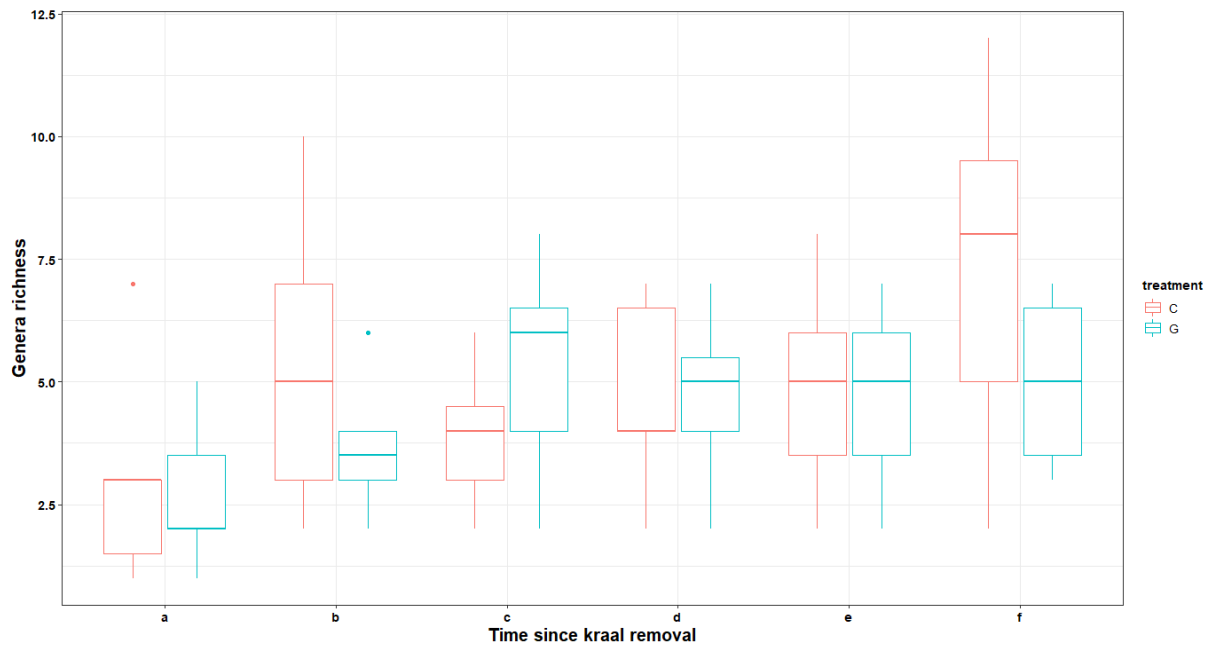
Fig. 3.4 Boxplot of spider genera richness of ground dwelling caught at Debshan Ranch as a function of kraaling and time since kraal removal. The box bounds the 25th and 75th percentiles, while the whiskers represent ± 1.5 IQR (interquartile range). Dots represent outliers. Sample size = 6. Abbreviations: for sampling periods a = period less than two months after kraal removal, b = 2 months after kraal removal, c = 4 months after kraal removal, d = 6 months after kraal removal, e = 8 months after kraal removal, f = 10 months after kraal removal; for treatment G = inside previously kraaled site and C = outside previously kraaled sites.
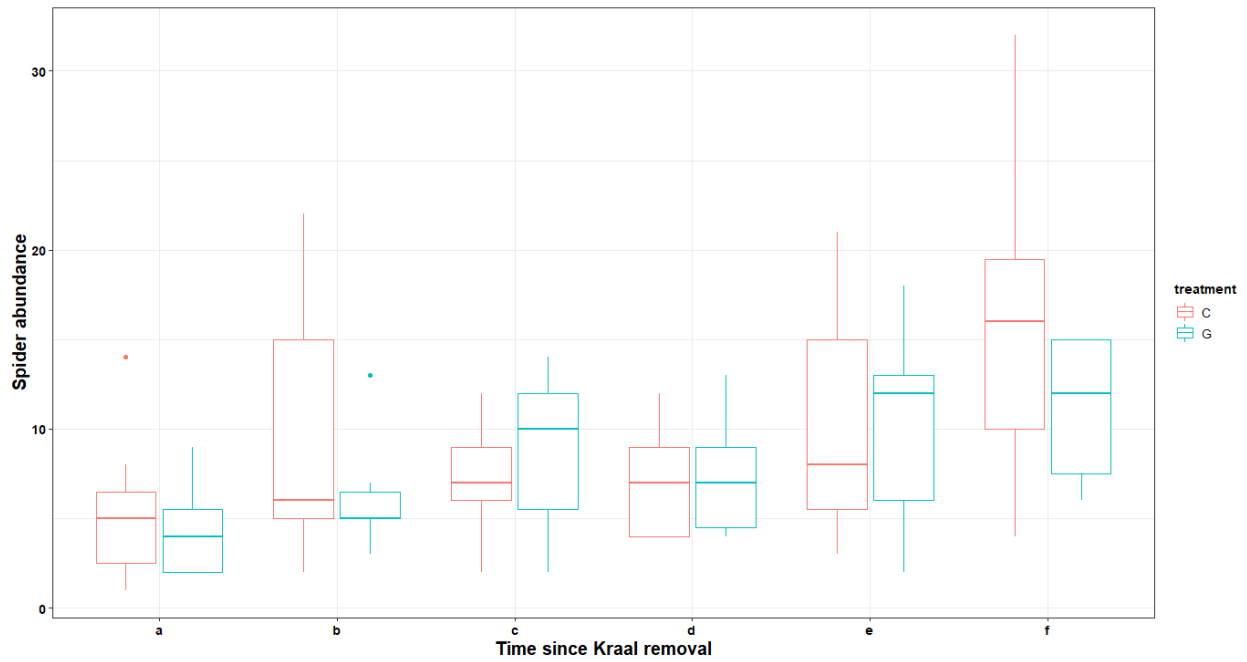
Fig. 3.5 Boxplot of spider genera richness of ground dwelling caught at Debshan Ranch as a function of kraaling and time since kraal removal. The box bounds the 25th and 75th percentiles, while the whiskers represent ± 1.5 IQR (interquartile range). Dots represent outliers. Sample size = 6 observations in each period since kraal removal. Abbreviations: for sampling periods, a = period less than two months after kraal removal, b = 2 months after kraal removal, c = 4 months after kraal removal, d = 6 months after kraal removal, e = 8 months after kraal removal, f = 10 months after kraal removal; for treatment G = inside previously kraaled site and C = outside previously kraaled sites.
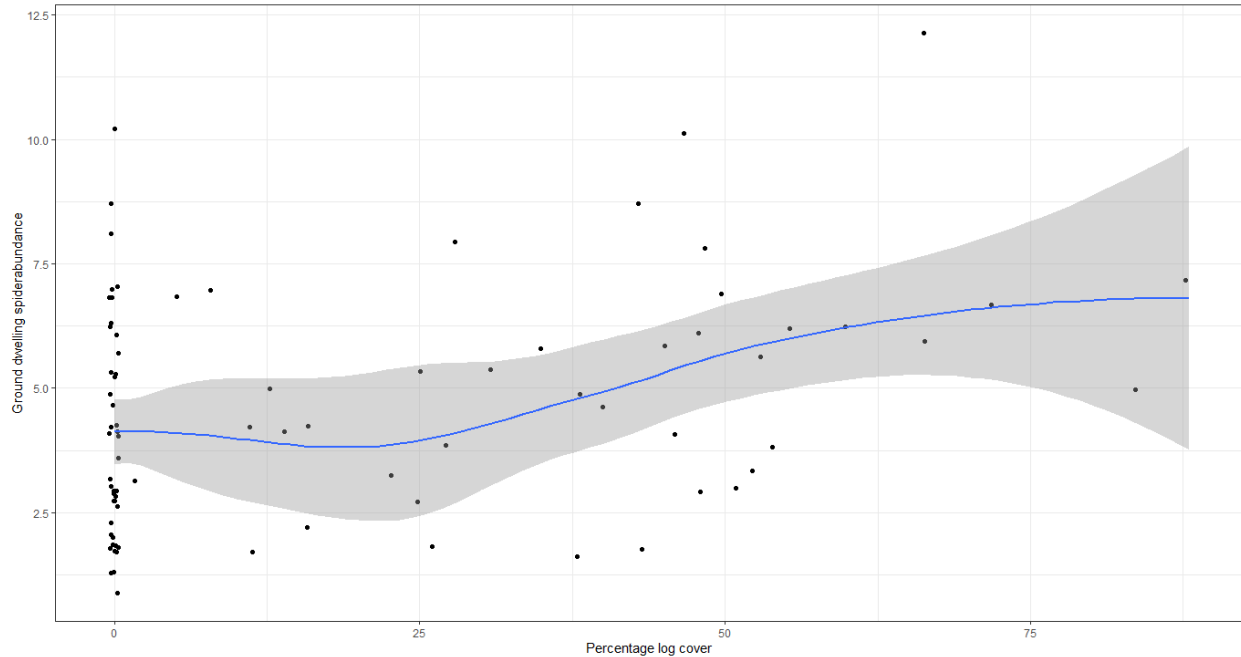
Fig. 3.6 Ground dwelling 'spider abundance as a function of percentage coarse woody debris cover sampled at Debshan Ranch, Zimbabwe with log regression line and 95% confidence intervals.

3.4.4 Spider assemblage composition

The multivariate generalised linear models revealed significant effects of season and time since kraal removal on genera composition; however no significant differences were detected amongst between spider communities of samples from inside the previously kraaled inclusions and their surrounding vegetation. In addition, only the vegetation structure variable percentage coarse woody debris cover revealed significant effects on the genera composition (Table 3.4) amongst the seven measured vegetation structure variables. Out of the 60 genera, mvabund univariate analysis identified significant differences in abundances of 5 genera according to the changes in the predictor variables. Two genera responded to the predictor season *Stenaelurillus* (Wald statistic= 3.855, p =0.005) and *Asemesthes* (Wald statistic = 4.997, p =0.001), with another two genera responding to the vegetation structure variable percentage coarse woody debris cover *Asemesthes* (Wald statistic = 5.167, p =0.002) and *Copa* (Wald statistic = 3.13, p = 0.031) and one genus responded to the predictor time since kraal removal *Pardosa* (Wald statistic = 5.076, p =0.020).

Table 3.4 Results of model based analysis of multivariate abundance data (mvabund) of ground dwelling spider communities' response to kraaling treatment (inside previously kraaled inclusions vs. outside previously kraaled inclusions), time since kraal removal (< 2, 2, 4, 6, 8 and 10 months), season [early sampling season (December) and late sampling season (March)] and vegetation structure (% vegetation cover) . Significant differences are represented by Asterisks with p value from 0.05-0.01 =*, 0.1-0.001 = ** and < 0.001 = ***

Term	Res.df	Df.diff	Wald	p-value
Short duration kraaling treatment	81	1	4.563	0.324
Time since kraal removal (months)	76	5	9.414	0.006**
Season	75	1	7.596	0.001***
Mean grass height (cm)	81	1	7.398	0.131
Coarse woody debris cover (%)	80	1	9.188	0.001***
Rock cover (%)	79	1	5.774	0.199
Cow dung cover (%)	78	1	4.936	0.428
Vegetation cover (%)	77	1	5.923	0.054
Leaf cover (%)	76	1	4.591	0.415

The model based unconstrained ordination showed clear separation of communities of spiders at both early and late sampling season, with some separation within each season varying in percentage coarse woody debris cover (Fig. 3.7). The distinct assemblage composition was mostly contributed by mostly low abundance genera (Fig. 3.7 & Appendix 3.1). Distinct spider assemblage composition was also quite clear with time since kraal removal, particularly with genera composition sampled at 6 and 10 months after removal in the early sampling season, however this distinction was less apparent in the late sampling season (Fig. 3.8).
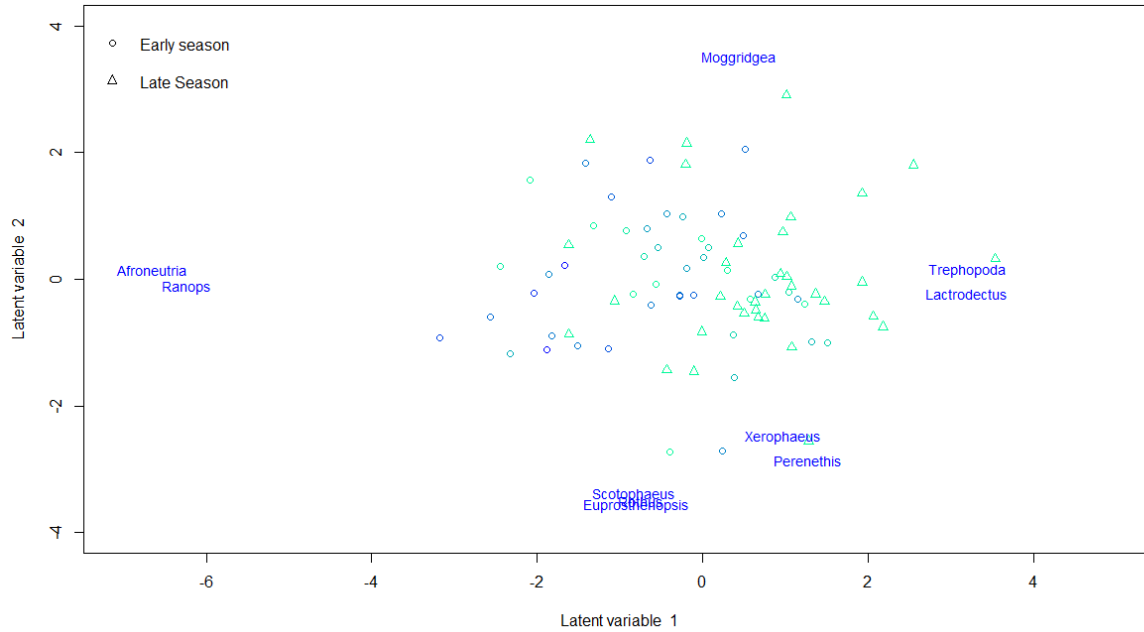
Fig. 3.7. Ordination plots of posterior medians of the first two latent variables of spider communities from ground dwelling spiders sampled during early summer season (November) and late summer season (March 2018) at Debshan Ranch, Zimbabwe and the ten most influential species.
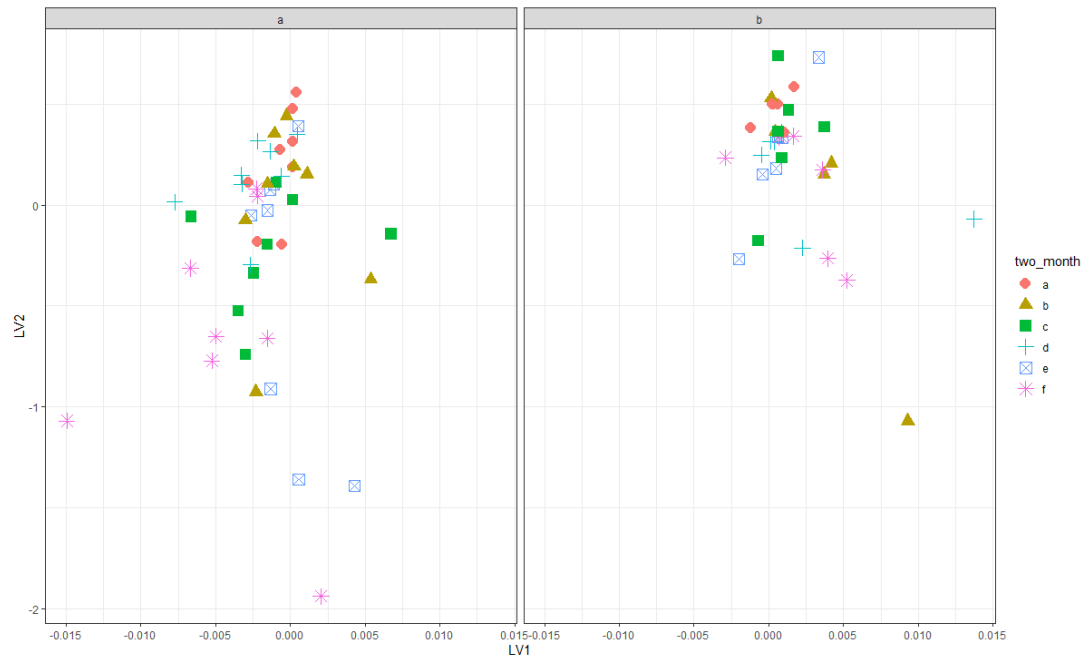
Fig. 3.8. Ordinations of ground dwelling spider communities sampled at Debshan Ranch, Zimbabwe of time since kraal removal, conditional on sampling season, a = < 2 months, b = 2 months, c = 4 months, d = 6 months, e = 8 months and f= 10months after kraal removal.

3.5 DISCUSSION

The study assessed the drivers of ground dwelling species richness and abundance within the miombo woodlands of Debshan Ranch in Zimbabwe. GLMM revealed that both time since kraal removal and percentage coarse woody debris cover were relatively important to the structure of spider diversity, while only spider abundance responded to short duration kraaling and season. Multivariate generalized linear models revealed that community composition was significantly related to percentage coarse woody debris cover and that this association varied with time since kraal removal and with season. Results indicate the importance of coarse woody debris to most ground dwelling spiders.

3.5.1 Community composition

In this study the species richness of ground dwelling spider species caught was well above than that recorded elsewhere. For, example, Silva & Ott (2017) recorded 77 species within the Neotropical region, in South America and Fuller *et al*. (2014) found 66 species within rare European oak and yew woodlands in Ireland. However, it was well below others such as Jansen *et al.* (2013) who recorded 86 species in the Montane grasslands in South Africa. In relation to the above studies the ground dwelling species richness can be considered as relatively high, bearing in mind that it was collected from a single stratum using pitfall-traps and collection was done only during the optimum collecting period that is suitable for most African grasslands (Muelelwa *et al.* 2010) and not throughout the year. However, higher species richness of 177 species were obtained in the first phase of this project using two sampling methods namely sweepnets and pitfall traps. Considering that spiders are found in almost all microhabitats (Turnbull 1973), if a combination of almost all sampling methods were to be utilised, greater species richness would most likely have been obtained.

When considering abundance, the number of spiders collected within the miombo woodlands at Debshan Ranch (n = 634) was relatively low when compared to previous studies such as that of Chari (2011) who collected (n = 3139) within the miombo woodlands of the Chizarira National Park and Chivero Bird Sanctuary in Zimbabwe and that of Jansen *et al.* (2013) who collected (n = 1145) in the Montane grasslands in South Africa. This outcome, may be attributed to the general lack of leaf litter which reduced double-fold with time since kraal removal (Appendix 3.3), in the previously kraaled sites within the miombo woodlands, as a result spider fauna that is

closely associated with leaf litter (Butler & Haddad 2011) was relatively low which inherently led to the observed low abundance. In addition, despite sampling during the optimum periods of spider activity (Muelelwa *et al.* 2010), whereby at least 50 % of the annual diversity is usually caught (Jimenez-Valverde & Lobo 2006), the number of sampling periods conducted in this study was only twice and relatively lower than the repetitive sampling periods done by Jansen *et al.* (2013), thus also contributing to the low abundance catch in this study.

The most dominant families reported within the miombo woodlands at Debshan Ranch (Table 3.2) are also fairly typical to those found in earlier studies in African grasslands (Russell-Smith 1981, Jansen *et al.* 2013, Haddad *et al.* 2015). Species belonging to the family Lycosidae strongly dominated the pitfall traps catches which are a common method used to sample ground dwelling spiders (Dippenaar-Schoeman & Wassenaar 2002, Russell-Smith 2002, Haddad & Butler 2018) rangelands. Lycosidae also well-known as wolf spiders are mostly ground dwelling spiders that are usually found resting under debris or stones on the ground (Dippenaar-Schoeman & Jocque 1997) and have been reported to have co-evolved with grasslands (Jocque & Alderweireldt 2005), in savanna habitats they are frequently found in agro ecosystems such as in strawberry beds (Dippenaar-Schoeman 1979), savanna grasslands and woodlands (Russell-Smith 1981) but they are less commonly sampled in bushvelds (Russell-Smith 1987).

The second most dominant group in this study were the Gnaphosidae also known as flat bellied ground spiders (Dippenaar-Schoeman *et al.* 2013) and are the fourth most diverse family globally with 2539 described species in 159 genera (World Spider Catalog 2020). There are mostly nocturnal generalists predators ground dwelling spiders (Dippenaar-Schoeman *et al.* 2013) with some genera such as *Aphantaulax* (Simon 1878) collected in trees and shrubs (Haddad *et al.* 2013). When at rest they hide in silk retreats that are constructed in leaf litter under rocks and logs (Haddad *et al.* 2013). The flat bellied ground spiders have been reported to dominate the majority of grasslands (Lotz *et al*. 1991, Haddad & Dippenaar-Schoeman 2002, 2006, Butler & Haddad 2011), however they are not prevalent in moist grasslands of Mpumalanga (Jansen *et al.* 2013).

The most globally diverse family with approximately 6173 described species in 649 genera is the Salticidae (World Spider Catalog 2020) and in this study it was the third dominant family caught by pitfall traps. The Salticidae also known as jumping spiders are diurnally active hunters that

conceal themselves in densely woven silk, usually attached to vegetation during periods of inactivity (Dippenaar-Schoeman & Jocque 1997). In order to enhance their chances of survival within their ideal habitats, grass dwelling species normally have elongate, pale bodies while ground dwelling species are cryptically colored (Haddad & Wesolowska 2011). African species have evolved specialised diets which have seen them able to feed on termites, ants, mosquitoes including others spiders (Li *et al.* 1997, Haddad & Wesołowska 2006, Nelson & Jackson 2006, Pekar & Haddad 2011).

3.5.2 Changes in mean grass height (cm) during the sampling periods.

The observed initial decrease in mean grass height (cm) inside the previously kraaled inclusions during both sampling seasons was mainly as a result of short duration of kraaling that has been reported to lead to lower grass biomass and basal cover in plant assemblages (Huruba *et al.* 2018) due to the trampling and grazing by the cattle typically leading to greater exposed soils. Similar to other studies such as Reid & Ellis (1995) grass establishment within the ranch was also rapid following kraal removal (Appendix 3.3). This is most probably as a result of the rich patches of nutrient reserves that develop in each previously kraaled inclusion due to the dung and urine deposition by the livestock occupying kraals (Augustine 2003). According to Veblen and Porensky (2019) grass cover increased faster during shorter periods of cattle occupation (4 and 7 day treatments) than on cattle corals where cattle occupation was longer (14 and 28 day treatments), due to hospitable plant establishment that occurs as a result of moderate cow dung deposits in shorter periods of cattle occupation bomas compared to the excessive deposits recorded within bomas of longer periods of cattle occupation. In addition, within short duration kraals there is usually less trampling of the original grass allowing for rapid regrowth after rains compared to excessive trampling over longer periods.

3.5.3 Spider abundance and diversity patterns.

In this study, short duration kraaling was expected to reduce spider abundance and genera richness within the previously kraaled inclusions as compared to their surrounding areas. Contrary to the prediction, short duration kraaling did not have any significant impact on spider genera richness however, there was significantly lower spider abundance of the ground dwelling spiders inside the previously kraaled inclusions than their surrounding vegetation (Fig. 3.4). The lack of differences between the inside and the outside of the previously kraaled inclusions may

be as a result of several possibilities, one such reasoning may be due to the fact that spiders have tremendous powers of dispersal, with many species of the Lycosidae and Gnaphosidae family having the ability to balloon especially as juveniles (Mrzljak & Wiegleb 2000). According to Pedley & Dolman (2014) dispersal abilities by spiders allows for dispersion, however, dispersal movements rarely exceed big distances, For example, the *Pardosa monticolla* (Keyserling, 1892) species has been estimated to be able to do not more than 280 m over its life time (Bonte *et al.* 2003), with the female able to do at least 30-40 m per day during natal dispersal (Bonte *et al.* 2007).

Thus the lack of differences between the inside of the kraals and the surrounding vegetation could possibly have been as a result of the high dispersal power that is achieved by either walking or ballooning (Weyman *et al.* 2002), which enables spatial exchange between populations of spiders assemblages as the sites between the treatments were separated by at least less than a 50m distance. Furthermore, active hunting spiders, especially cursorial spiders (running than ambush or stalkers) have been reported to increase in abundance with increase of disturbance (Pedley & Dolman 2014). Similar results were also obtained by Chikorowondo *et al.* (2018) where related functional response groups of above ground macro invertebrates between abandoned kraals and control plots were obtained within a semi-arid savannah in south-eastern Zimbabwe. The lack of treatment effect was attributed to the possibility of dispersal between the sites by the macro invertebrates which the majority consisted of legs and wings considering the short distance of approx. 150m between their sites.

Secondly, the most abundant species collected in this study were generalist, which are usually found in high numbers in open and disturbed habitats (Mallis & Hurd 2005), as they are the first species to normally inhabit disturbed lands (Pedley & Dolman 2014). For example, *Pardosa* species have been reported to achieve dense population in open barren lands (Buddle & Rypstra 2003, Mallis & Hurd 2005). In this study two species *Pardosa manubriata* (Simon, 1898) (15%) and *Pardosa crassipalpisi* (Purcell 1903) (5%) were the second and third most abundant species that were caught within the miombo woodlands after *Allocosa umtalica* (Purcell, 1903) (32%). Similarly, *Asemesthes paynteri* (Tucker, 1923) and *Nomisia varia* (Tucker, 1923) which were also abundant are usually also found in high numbers in disturbed habitats. Therefore the presence of the *Pardosa* and *Asemethes* species in high numbers within the ranch may be suggestive that the land is heavily disturbed and degraded to start with due to the many years of

being a cattle ranch. However, since sampling was only concentrated within the miombo woodlands, further research in other types of woodlands that are found within the ranch is recommended, in order to establish whether such generalists spiders are also found throughout the ranch or whether they were concentrated only within the miombo woodlands.

Spider genera richness and abundance tended to increase with time since kraal removal with genera richness peaking at 4 months after the kraal is removed and the greatest spider abundance recorded 10 months since kraal removal (Fig. 3.4). In this study, short duration kraaling enabled the deposition of moderate cow dung within previously kraaled sites which indirectly led to the increase in grass cover with time since kraal removal. Grass cover is essential for the development of most beneficial arthropods in orchards (Silva *et al.* 2010), and most invertebrates, including spiders have been reported to benefit from greater grass cover which provides essential services to most invertebrates (Gibson *et al.* 1992a, 1992b). The lowest genera richness was recorded during the period when cattle occupied the kraals (Fig. 3.4), most probably as a result of the trampling effect of the livestock (Gibson *et al.* 1992a, 1992b) which resulted in the development of bare soils with low percentage vegetation cover (Appendix 3.3).

As a result of the trampling action of the hooves of livestock lower grass cover was recorded inside previously kraaled sites. The presence of bare soil translates to reduced raw materials for minerals and nutrients for most orgarnisms (Rampai 2017). Bare ground cover has been reported to be negatively associated to spider species richness under trees (Barton *et al.* 2017). In this study the simplified vegetation structure found within the short duration kraals had major implications for the ground dwelling spiders at Debshan Ranch. At the time of abondonment kraals are usually bare patches that are mostly covered by dung above the soil surface layer (Sibanda *et al.* 2016). Contrary to this study O`DaSilva & Ott (2017) recorded similar results of spider abundance and genera richness even after 15 months after the removal of the cattle. They attributed the lack of differences mainly due to their sampling design that contributed to negative effects such as edge effects (Murcia 1995, Rodrigues *et al.* 2014) owing to the size of the fenced ungrazed areas and the low sampling effort obtained from lower number of pitfall traps used leading to low power outcome of statistical tests (Brennan *et al.* 1999, Work *et al.* 2002).

3.5.4 Spider assemblage composition

Time since kraal removal had a significant impact on spider genera composition. Spider assemblages sampled at 6 month after kraal removal were quite distinct from those sampled at 10 months after kraal removal in the early summer period, however this distinction was not so evident in the late summer period (Fig.3.8). This outcome can be attributed to the change in grass cover over time since kraal removal which was more prominent during the summer sampling period, as the early summer period recorded greater mean as compared to the late summer season that had less of the percentage grass cover. As a result of the increase of grass cover with time since removal heterogenous microhabitatses were created that are beneficial to most arthropods (Gibson *et al.* 1992a, 1992b) in each sampling period, contributing to also the different species composition in each time interval. In addition, the rains received during the late summer season might also have contributed to the low diversity of spider assemblages caught during that period that also led to the lack of distinct assemblages in different sampling periods as observed during the early sampling period.

Disturbances such as grazing and trampling can lead to changes in vegetation structure (Naeth *et al.* 1991), in this study only percentage coarse woody debris cover responded significantly and positively to spider assemblages amongst the six measured vegetation structure variables. Coarse woody debris is an essential component in conservation biology and ecosystem functioning (Harmon *et al.* 1986) as it supports high diversity of orgarnisms and also contributes to accumulation of orgarnic matter (Ulyshen & Hanula 2007). Various species of parasitoids and predators (Ehnstrom 2001, Grove 2002) are also supported by coarse woody debris. For example, arthropods species such as saproxylic arthropods (Barton *et al.* 2017) depend on dead wood as a resource and arthropod species normally found in litter are more abundant closer to dead wood than further away from it (Ulyshen & Hanula 2007), making coarse woody debris to be an important microhabitat for spider assemblages. Due to their ability to house invertebrates that are prey for most spiders (Olive 1982), logs are also a suitable surface for attachment of webs (Riechert & Gillespie 1986, Roberson *et al.* 2016) and also the microclimatic condition in areas adjacent to logs normally has lower temperatures and humidity (Savely 1939) and also due to the accumulation of leaf litter around logs such microhabitats acts as protective areas for some invertebrates (Harmon *et al.* 1986). The average annual temperatures of the Debshan Ranch is

approximmately 18 ºC and the microhabitats that are created by the coarse woody debris make them suitable for spider fauna.

In this study percentage litter cover and cow dung cover did not influence spider assemblages despite the various roles they play within ecosystems (Butler & Haddad 2011, Paschetta *et al.* 2013). For example leaf litter is not only essentiall to micro-orgarnisms (Rampai 2017) but it also provides habitat and food to most invertebrates including spiders. For example, leaf litter can be used as overwintering sites by some grass and foliage dwelling species from the Araneidae and Thomisidae as these were reported to be present in leaf litter fauna (Butler & Haddad 2011). Short duration kraaling also enhances nutrient recycling through fertilization by the cow dung, as a result complex communities are created. Denis *et al* (2015) reported an increase of dung beetles under the highest sheep stocking density within sub- montane vegetation in Scotland (Dennis *et al.* 2015) which were most probably associated to the presense of the dung. Despite, that rocks cover is also an essential hiding place for most ground dwelling spiders such as the flat bellied spiders (Gnaphosidae) and the wolf spider (Lycosidae) as they have been reported to be associated to the presence of hiding places such as rocks and litter (Paschetta *et al*. 2013), in this study rocks cover was also not a significant predictor of spider assemblages despite its ecological importance to most invertebrates.

Despite the sampling being done within the summer season, seasonality was an important predictor of spider assemblages with lower spider abundance caught in the late rainy season (March) than the early rainy season (December). The findings of this study are also similar to the work of Muelelwa *et al.*(2010) who also found higher spider abundance and species richness in the early summer (November) as compared to autumn (March), they attributed their results to the maturing of the juveniles and sub adults over winter which are captured as adults which are normally ready for mating during the early summer raining season with the late summer rainy season having the majority of the adults having died and being flooded by juveniles (Foord *et al.* 2008, Muelelwa *et al.* 2010). However, in this study the difference in the catch of the spider abundance was mainly possibly due to the heavy rains that were received during the late summer season, as rainfall has been reported to have a negative effect on spider activity (Queiroz & Gasnier 2017) which therefore led to the reduction of the abundance of spider that were caught during rainfall periods, in this case the late summer season.

3.6 References

Abagale, F.K. & Ayuegabe, E. (2015) Enhancing soil nutrient status using dynamic kraaling strategy in Northern Ghana. *Elixir International Journal*, **88**, 36279–36283.

Augustine, D. J. (2003) Long-term, livestock-mediated redistribution of nitrogen and phosphorus in an East African savanna. *Journal of Applied Ecology*, **40**, 137–149.

Augustine, D. J., Veblen, K.E., Goheen, J.R., Riginos, C. & Young, T.P. (2009) Pathways for positive cattle–wildlife interactions in semiarid rangelands. *Smithsonian Contributions to Zoology*, **632**, 55–71.

Baldissera, R., Ganade, G. & Fontoura, S.B. (2004) Web spider response along an edge between pasture and Araucaria forest. *Biological Conservation*, **118**, 403-409.

Barton, P.S., Evans, M.J., Foster, C.N., Cunningham, S.A. & Manning, A.D. (2017) Environmental and spatial drivers of spider diversity at contrasting microhabitats. *Austral Ecology*, **42**, 700-710.

Bates, D., Machler, M., Bolker, B.M. & Walker, S.C. (2015) Fitting linear mixed-effects models using lme4. *Journal of Statistical Software*, **67**, 1–48.

Baxter, R., Hastings, N., Law, A. & Glass, E.J. (2015) *What is holistic planned grazing?* Savory Institute, United States of America.

Bolker, B.M., Brooks, M.E., Clark, C.J., Geange, S.W., Poulsen, J.R., Stevens, M.H.H. & White, J.S. (2008) Generalized linear mixed models: a practical guide for ecology and evolution. *Trends in Ecology & Evolution*, **30**, 1–9.

Bonte, D., Van Belle, S. & Maelfait, J.P. (2007) Maternal care and reproductive state-dependent mobility determine natal dispersal in a wolf spider. *Animal Behaviour*, **74**, 63–69.

Bonte, D., Lens, L., Maelfait, J.P., Hoffmann, M. & Kuijeken, E. (2003) Patch quality and connectivity influence spatial dynamics in a dune wolfspider. *Oecologia*, **135**, 27–233.

Borg, R. (1996) *Corrals for handling beef cattle.* Alberta, Canada: Alberta , Food and Rural Development, Edmonton, Alberta, Canada.

Brennan, K.E.C., Majer, J.D. & Reygaert, N. (1999) Determination of an optimal pitfall trap size for sampling spiders in a Western Australian Jarrah forest. *Journal of Insect Conservation*, **3,** 1-11.

Buddle, C.M. & Rypstra, A.L. (2003) Factors initiating emigration of two wolf spider species

(Aranae: Lycosidae) in an agroecosystem. *Environmental Entomology*, **32**, 88–95.

Burnham, K.P. & Anderson, D. (2002) *Model selection and inference: a practical information-theoretic approach.* 2nd Edition, Springer-Verlag, New York, United States of America.

Butler, V.P. & Haddad, C.R. (2011) Spider assemblages associated with leaf litter of three tree species in central South Africa (Arachnida: Araneae). *African Journal of Ecology*, **49**, 301–310.

Castro, A. & Wise, D.H. (2009) Influence of fine woody debris on spider diversity and community structure in forest leaf litter. *Biodiversity Conservation*, **18**, 3705–3731.

Chao, A. & Jost, L. (2012) Coverage-based rarefaction and extrapolation : standardizing samples by completeness rather than size. *Ecology*, **93**, 2533–2547.

Chari, L.D. (2011) *The influence of large, vegetated termitaria and large herbivores on spider (Araneae) diversity in miombo woodlands.* Master of Science dissertation. University of Zimbabwe, Harare, Zimbabwe.

Chikorowondo, G., Muvengwi, J., Mbiba, M. & Gandiwa, E. (2017) Influence of abandoned cattle enclosures on plant assemblages and herbivory in a semi-arid savanna. *Ecological Research*, **32**, 1023–1033.

Chikorowondo, G., Muvengwi, J., Mbiba, M. & Gandiwa, E. (2018) Functional diversity of macroinvertebrates on abandoned cattle enclosures in a semi-arid African savannah. *African Journal of Ecology*, **56,** 629–640.

Churchill, T.B. (1997) Spiders as ecological indicators: an overview for Australia. *Memoirs of the Museum of Victoria*, **56**, 331–337.

Churchill, T.B. (1998) *Spiders as ecological indicators in the Australian tropics : family distribution patterns along rainfall and grazing gradients.* In PA Selden, (ed). Proceedings of the 17th European Colloquium of Arachnology, Edinburgh 1997. Burnham Beeches, Bucks, British Arachnological Society.

Dennis, P., Skartveit, J., Kunaver, A. & McCracken, D.I. (2015) The response of spider (Araneae) assemblages to structural heterogeneity and prey abundance in sub-montane vegetation modified by conservation grazing. *Global Ecology and Conservation*, **3,** 715–728.

Dethier, M.N., Graham, E.S., Cohen, S. & Tear, L.M. (1993) Visual versus random-point

percent cover estimations: objective is not always better. *Marine Ecology Progress Series*, **96**, 93–100.

Dippenaar-Schoeman, A.S. (1979) Spider communities in strawberry beds: seasonal changes in numbers and species composition. *Phytophylactica*, **11**, 1-4.

Dippenaar-Schoeman, A.S., Foord, S.H. & Haddad, C. (2013) *Spiders of the Savanna Biome of South Africa.* University of Venda, Thohoyandou and Agricultural Research Council, Pretoria, South Africa.

Dippenaar-Schoeman, A.S. & Jocque, R. (1997) *African spiders: an identification manual.* Plant Protection Research Institute Handbook No. 9, Agricultural Research Council, Pretoria.

Dippenaar-Schoeman, A.S. & Wassenaar, T.D. (2002) A checklist of the ground-dwelling spiders of coastal dune forests at Richards Bay, South Africa (Arachnida: Araneae). *Bulletin of British Arachnological Society*, **12**, 275–279.

Ehnstrom, B. (2001) Leaving dead wood for insects in boreal forests –suggestions for the future. *Scandinavian Journal Forest Research*, **3**, 91–98.

Foelix, R.F. (2011) *Biology of Spiders.* Oxford University Press, New York, United States of America.

Foord, S.H., Mafadza, M., Dippennar-Schoeman, A.S. & Van Rensburg, B.J. (2008) Micro-scale heterogeneity of spiders (Arachnida: Araneae) in the Soutpansberg, South Africa: a comparative survey and inventory in representative habitats. *African Zoology*, **43**, 156–174.

Ford, H., Garbutt, A., Jones, L. & Jones, D.L. (2013) Grazing management in saltmarsh ecosystems drives invertebrate diversity, abundance and functional group structure. *Insect Conservation and Diversity*, **6**, 189–200.

Fuller, L., Newman, M., Irwin, S., Kelly, T. & Halloran, J.O. (2014) Ground-dwelling spider diversity in rare European oak and yew woodlands and the impact of grazing. *Biodersity Conservation*, **23**, 1911–1929.

Gibson, C.W.D., Brown, V.K., Losito, L. & McGavin, G.C. (1992a) The response of invertebrate assemblies to grazing. *Ecography*, **15**, 166–176.

Gibson, C.W., Hambler, C. & Brown, V. (1992b) Changes in spider (Araneae) assemblages in relation to succession and grazing management. *Journal of Applied Ecology*, **29**, 132–

142.

Grove, S.J. (2002) Saproxylic insect ecology and the sustainable management of forests. *Annual Review of Ecology and Systematics*, **33**, 1–23.

Haddad, C.R. & Butler, V.P. (2018) Ground-dwelling spider assemblages in contrasting habitats in the central South African grassland biome. *Koedoe*, **60**, 1–12.

Haddad, C.R. & Dippenaar-Schoeman, A.S. (2002) The influence of mound structure on the diversity of spiders (Araneae) inhabiting the abandoned mounds of the snouted harvester termite *Trinervitermes trinervoides. Journal of Arachnology*, **30**, 403–408.

Haddad, C.R. & Dippenaar-Schoeman, A.S. (2006) Spiders (Arachnida: Araneae) inhabiting abandoned mounds of the snouted harvester termite *Trinervitermes trinervoides* (Sjo¨stedt) (Isoptera: Termitidae: Nasutitermitinae) in the Free State, with notes on their biology. *Navorsinge van die Nasionale Museum, Bloemfontein*, **22**, 10–15.

Haddad, C.R., Foord, S.H., Fourie, R. & Dippenaar-Schoeman, A.S. (2015) Effects of a fast-burning spring fire on the ground-dwelling spider assemblages (Arachnida: Araneae) in a central South African grassland habitat. *African Zoology*, **50**, 281-292.

Haddad, C.R., Foord, S.H., Lotz, L.N. & Lyle, R. (2013) The faunistic diversity of spiders (Arachnida: Araneae) of the South African grassland biome. *Transactions of the Royal Society of South Africa*, **68**, 97–122.

Haddad, C.R., Honiball, A.S., Dippenaar-Schoeman, A.S., Slotow, R. & Rensburg, B.J.V. (2009) Spiders as potential indicators of elephant-induced habitat changes in endemic sand forest, Maputaland, South Africa. *African Journal of Ecology*, **48**, 446–460.

Haddad, C.R. & Wesolowska, W. (2011) New species and new records of jumping spider (Araneae: Salticidae) from central South Africa. *African Invertebrates*, **52**, 51–134.

Haddad, C.R. & Wesolowska, W. (2006) Notes on taxonomy and biology of two *Stenaelurillus* species from Southern Africa(Araneae:Salticidae). *Annales Zoologici*, **56,** 575–586.

Halaj, J, Ross, D.W. & Moldenke, R. (1998) Habitat structure and prey availability as predictors of the abundance and community organization of spiders in western Oregon forest canopies. *Journal of Arachnology*, **26**, 203–220.

Harmon, M.E., Franklin, J.F., Swanson, F.J., Sollins, P., Gregory, S.V, Lattin, J.D., Anderson, N.H., Cline, S.P., Aumen, N.G., Sedell, J.R., Lienkaemper, G.W.,

Cromack, K.J. & Cumminis, K.W. (1986) Ecology of coarse woody debris in temperate ecosystems. *Advances in Ecological Research*, **15**, 133–302.

Harrison, X.A., Donaldson, L., Correa-Cano, M.E., Evans, J., Fisher, D.N., Goodwin, C.E.D., Robinson, B.S., Hodgson, D.J. & Inger, R. (2018) A brief introduction to mixed effects modelling and multi-model inference in ecology. *PeerJ*, **6**, 1–32.

Hsieh, T.C., Ma, K.H. & Chao, A. (2016). iNEXT : an R package for rarefaction and extrapolation of species diversity (Hill numbers). *Methods and Evolution*, **7**, 1451–1456.

Huruba, R., Mundy, P.J., Sebata, A., Purchase, G.K. & MacFadyen, D.N. (2017) Impala, *Aepyceros melampus :* does browse quality influence their use of sites originally utilised as short-duration kraals in a Southern African savanna ? *Rangeland Journal*, **39**, 113–121.

Huruba, R., Mlambo, T., Mundy, P.J., Sebata, A. & MacFadyen, D.N. (2018) Short duration overnight cattle kraaling in natural rangelands : Implications for grass composition, quality, above ground biomass, species diversity and basal cover. *Agriculture, Ecosystems and Environment*, **257**, 144–151. .

Horvath, R., Magura, T., Szinetar, C. & Tothemeresz, B. (2009) Spiders are not less diverse in small and isolated grasslands, but less diverse in overgrazed grasslands: a field study (East Hungary, Nyérség). *Agriculture, Ecosystems and Environment*, **130**, 16-22.

Jansen, R., Makaka, L. & Little, I. (2013) Response of ground-dwelling spider assemblages (Arachnida: Araneae) to montane grassland management practices in South Africa. *Insect conservation and diversity*, **6**, 572–589.

Jocque, R. & Alderweireldt, M. (2005) Lycosidae: the grassland spiders. In: Deltshev, C, Stoev, P, (eds). European Arachnology, *Acta Zoological Bugarica*, **1**, 125–130.

Kangalawe, R.Y.M., Christiansson, C. & Östberg, W. (2008) Changing land-use patterns and farming strategies in the degraded environment of the Irangi Hills, Central Tanzania. *Agriculture Ecosystem Environment*, **125**, 33–47.

Kizza, S. & Areola, O. (2010) Analysis of the effects of kraal manure accumulation on soil nutrient status through time. *Journal of Soil Science and Environmental Management*, **1**, 217–226.

Kizza, S., Totolo, O., Perkins, J. & Areola, O. (2010) Analysis of persistence soil nutrient status in abandoned cattle kraals in a semi arid area in Botswana. *Scientific Research and Essays*, **5**, 3613–3622.

Li, D., Jackson, R.R. & Barrion, A. (1997) Prey preferences of *Portia africana*, *P. labiata* and *P. schulzi*, araneophagic web-building jumping spiders (Araneae: Salticidae) from Kenya, the Philippines and Sri Lanka. *New Zealand Journal of Zoology*, **24**, 333–349.

Lin, S., Vasseur, L.M. & You, S. (2016) Seasonal variability in spider assemblages in traditional and transgenic rice fields. *Environmental Entomology*, **45**, 537-546.

Lotz, L., Seaman, M.T. & Kok, D.J. (1991) Surface-active spiders (Araneae) of a site in semi-arid central South Africa. *Navorsinge van die Nasionale Museum, Bloemfontein*, **7**, 530–540.

Mallis, R.E. & Hurd, L.E. (2005) Diversity among ground dwelling spider assemblages: habitat generalist and specialists. *Journal of Arachnology*, **33**, 101–109.

Marc, P., Canard, A. & Ysnel, F. (1999) Spiders (Araneae) useful for pest limitation and bioindication. *Agriculture, Ecosystems & Environment*, **74**, 229–273.

Mineo, M.F., Del-Claro, K. & Brescovit, A.D. (2010) Seasonal variation of ground spiders in a Brazilian Savanna. *Zoolofia*, **27**, 353–362.

Mrzljak, J. & Wiegleb, G. (2000) Spider colonization of former brown coal mining areas - time or structure dependent? *Landscape & Urban Planning*, **51**, 131–146.

Muchiru, A.N., Western, D. & Reid, R.S. (2009) The impact of abandoned pastoral settlements on plant and nutrient succession in an African savanna ecosystem. *Journal of Arid Environments*, **73**, 322–331.

Muelelwa, M.I., Foord, S.H., Dippenaar-Schoeman, A.S. & Stam, E.M. (2010) Towards a standardized and optimized protocol for rapid biodiversity assessments : spider species richness and assemblage composition in two savanna vegetation types. *African Zoology*, **45**, 273–290.

Murcia, C. (1995) Edge effect in fragmented forests: implications for conservation. *Trends in Ecology and Evolution*, **10**, 58–62.

Muvengwi, J., Chikorowondo, G., Mbiba, M. & Gandiwa, E. (2018) Diversity and abundance of macro-invertebrates on abandoned cattle kraals in a semi-arid savanna. *Ecology and Evolution*, **8**, 8477-8483.

Naeth, M.A., Bailey, A.W., Pluth, D.J., Chanasyk, D.S. & Hardin, R.T. (1991) Grazing impacts on litter and soil organic matter in mixed prairie and fescue grassland ecosystems of Alberta. *Journal of Range Managemet*, **44**, 7–12.

Nakagawa, S. & Schielzeth, H. (2013) A general and simple method for obtaining R^2 from generalized linear mixed-effects models. *Methods in Ecology and Evolution*, **4**, 133–142.

Neely, C.L. & Butterfield, J. (2004) Holistic management of African rangelands. *Leisa Magazine*, **20**, 26–28.

Nelson, X.J. & Jackson, R.R. (2006) A predator from East Africa that chooses malaria vectors as preferred prey. *PLoS One*, **1**, 1-15.

Nentwig, W. & Kobelt, M. (2010) Spiders (Araneae). Chapther 7.3. In : Roques, A., Kenis, M., Lees, D., Lopez-Vaamonde, C, Rabitsch, W., Rasplus, J-Y. & Roy, D. (eds). Alien terrestrial arthropods of Europe. *BioRisk*, **4**, 131–147.

Niemela, J., Pajunen, T., Haila, Y., Punttila, P. & Halme, E. (1994) Seasonal activity of boreal forest-floor spiders (Araneae). *Journal of Arachnology*, **22**, 23–31.

Niku, J., Hui, F.K.C., Taskinen, S. & Warton, D.I. (2019) gllvm-Fast analysis of multivariate abundance data with generalized linear latent variable models in R. *Methods in Ecology and Evolution*, **10**, 2173-2182.

Nyffeler, M. & Birkhofer, K. (2017) An estimated 400–800 million tons of prey are annually killed by the global spider community. *The Science of Nature*, **104**, 1-12.

Nyffeler, M. & Knornschild, M. (2013) Bat predation by spiders. *PLoS One*, **8**, 1–15.

Nyffeler, M. & Pusey, B. J. (2014) Fish predation by semi-aquatic spiders : a global pattern. *PLoS One*, **9**, 1-13.

Nyoni, P. (2015) *Leopard population density and community attitudes towards leopards in and around Debshan Ranch, Shangani, Zimbabwe.* MSc dissertation, Rhodes University, South Africa.

Olive, C.W. (1982) Behavioral responses of a sit-and-wait predator to spatial variation in foraging gain. *Ecology*, **63**, 912–920.

Palem, H., Kanike, S. & Purushottam, V.R.S. (2017) Diversity of spider fauna (Arachnida: Araneae) in different ecosystems, Eastern Ghats, Southern Andhra Pradesh, India. *Journal of Life Science*, **4**, 51–60.

Paschetta, M., Morgia, V.L., Masante, D., Negro, M., Rolando, A. & Isaia, M. (2013) Grazing history influences biodiversity: a case study on ground-dwelling arachnids (Arachnida: Araneae, Opiliones) in the Natural Park of Alpi Marittime (NW Italy). *Journal of Insect Conservation*, **17**, 339–356.

Pedley, S.M. & Dolman, P.M. (2014) Multi-taxa trait and functional responses to physical disturbance. *Journal of Animal Ecology*, **83**, 1542–1552.

Pekar, S. & Haddad, C.R. (2011) Trophic strategy of ant-eating *Mexcala elegans* (Araneae: Salticidae): looking for evidence of evolution of prey-specialization. *Journal of Arachnology*, **39**, 133–138.

Podgaiski, L.R., Joner, F., Lavorel, S., Moretti, M., Ibanez, S., Mendonc, M. D. & Pillar, V.D. (2013) Spider trait assembly patterns and resilience under fire-induced vegetation change in South Brazilian grasslands. *PLoS One*, **8**, 1–11.

Pryke, J.S. & Samways, M.J. (2012) Importance of using many taxa and having adequate controls for monitoring impacts of fire for arthropod conservation. *Journal of Insect Conservation*, **16**, 177–185.

Queiroz, M. & Gasnier, T.R. (2017) Strong negative effect of diurnal rainfall on nocturnal activity of a wandering spider in Central Amazonia. *International Journal of Tropical Biology*, **65**, 1152–1160.

R Core Team. (2019) R: a language and environment for statistical computing. Version 2.15.1. R Foundation for Statistical Computing, Vienna, Austria.

Rampai, R.I. (2017) *Effects of different grazing systems on ecosystem functions in Lesotho and Iceland.* Final project. United Nations University Land Restoration Training Programme, Reykjavik, Iceland.

Reid, R.S. & Ellis, J.E. (1995) Impacts of pastoralists on woodlands in South Turkana, Kenya: livestock-mediated tree recruitment. *Ecological Applications*, **5**, 978–992.

Riechert, S.E. & Gillespie, R.G. (1986) *Habitat choice and utilization in web-building spiders.* In : Spiders: Webs, Behavior, and Evolution. Shear, W. A (ed). Stanford University Press, Stanford, California, United States of America.

Roberson, E.J., Chips, M.J., Carson, W.P. & Rooney, T.P. (2016) Deer herbivory reduces web-building spider abundance by simplifying forest vegetation structure. *PeerJ*, **4**, 1–15.

Rodrigues, E., Mendonca, M. & Ott, R. (2009) Spider diversity in a rice agroecosystem and adjacent areas in southern Brazil. *Revista Colombiana de Entomología*, **8**, 89-97.

Rodrigues, E., Mendonca, M. & Costa-Schimidt, L.E. (2014) Spider diversity responds strongly to edge effects but weakly to vegetation structure in riparian forests of Southern Brazil. *Arthropod Plant Interactions*, **8**, 123–133.

Rodriguez-Artigas, S.M., Ballester, R. & Corronca, J.A. (2016) Factors that influence the beta-diversity of spider communities in northwestern Argentinean grasslands. *PeerJ*, **4**, 19-46.

Russell-Smith, A. (1981) Seasonal activity and diversity of the ground-living spiders in two African savanna habitats. *Bulletill of the British Arachnological Society*, **5**, 145–154.

Russell-Smith, A. (2002) A comparison of the diversity and composition of ground active spiders in Mkomazi game reserve,Tanzania and Etosha National Park, Namibia. *Journal of Arachnology*, **30**, 383–388.

Savely, H.E. (1939) Ecological relations of certain animals in dead pine and oak logs. *Ecological Monographs*, **9**, 323–385.

Savory, A. (1983) The Savory Grazing Method or Holistic Resource Management. *Rangelands*, **5**, 155–159.

Savory, A. & Parsons, S.D. (1980) The Savory Grazing Method. *Rangelands*, **2**, 234–237.

Savory, A. (2013) Response to request for information on the science and methodology underpinning holistic management and holistic planned grazing, online at http://www.savoryinstitute.com, accessed on the 12th January 2015.

Schwerdt, L., De Villalobos, A.E. & Miles, F.P. (2018) Spiders as potential bioindicators of mountain grasslands health: the Argentine tarantula *Grammostola vachoni* (Araneae, Theraphosidae). *Wildlife Research*, **45**, 64–71.

Sibanda, P., Sebata, A., Mufandaedza, E. & Mawanza, M. (2016) Effect of short-duration overnight cattle kraaling on grass production in a southern African savanna. *African Journal of Range and Forage Science*, **33**, 217–223.

Silva, G.O. & Ott, R. (2017) Iheringia after cattle removal in grazed grassland. *Iheringia Serie Zoologia*, **107**, 1–8.

Silva, E.B., Franco, J.C., Vasconcelos, T. & Branco, M. (2010) Effect of ground cover vegetation on the abundance and diversity of beneficial arthropods in citrus orchards. *Bulletin of Entomological Research*, **100**, 489–499.

Stelfox, J.B. (1986) Effects of livestock enclosures (bomas) on the vegetation of the Athi Plains, Kenya. *African Journal of Ecology*, **24**, 41–45.

Turnbull, A.L. (1973) Ecology of the true spiders (Araneomorphae) 6051. *Annual Reviews Entomology*, **18**, 305–348.

Ulyshen, M.D. & Hanula, J.L. (2007) Litter-dwelling arthropod abundance peaks near coarse woody debris in Loblolly pine forests of the southeastern United States. *Florida Entomologists*, **92**, 163–164.

Veblen, K.E. & Porensky, L.M. (2019) Thresholds are in the eye of the beholder: plants and wildlife respond differently to short-term cattle corrals. *Ecological Applications*, **29**, 1–15.

Wang, Y., Naumann, U., Wright, S.T. & Warton, D.I. (2012) mvabund –an R package for model-based analysis of multivariate abundance data. *Methods in Ecology & Evolution*, **3**, 471–474.

Weeks, R.D. & Holtzer, T.O. (2000) Habitat and season in structuring ground-dwelling spider (Araneae) communities in a shortgrass steppe ecosystem. *Enviromental Entomology*, **29**, 1164–1172.

Weyman, G.S., Sunderland, K.D. & Jepson, P.C. (2002) A review of the evolution and mechanisms of ballooning by spiders inhabiting arable farmland. *Ethology Ecology and Evolution*, **14**, 307–326.

Whitmore, C., Slotow, R., Crouch, T. E. & Dippenaar-Schoeman, A.S. (2002) Diversity of spiders (Araneae) in a savanna reserve, northern province, South Africa. *Journal of Arachnology*, **30**, 344-356.

Whittingham, M. J., Stephens, P.A., Richard, B. & Freckleton, R.P. (2006) Why do we still use stepwise modelling in ecology and behaviour ? *Journal of Animal Ecology*, **75**, 1182–1189.

Widhe, M., Johansson, J., Hedhammar, M. & Rising, A. (2011) Current progress and limitations of spider silk for biomedical applications. *Wiley Online Library*, **97**, 468–478.

Wise, D.H. (2006). Cannibalism, food limitation, intraspecific competition,and the regulation of spider populations. *Annual Review of Entomology*, **51**, 441-65.

Work, T.T., Buddle, C.M., Korinus, L.M. & Spence, J.R. (2002) Pitfall trap size and capture of three taxa of litter-dwelling arthropods: implications for biodiversity studies. *Enviromental Entomology,* **31**, 438–448.

World Spider Catalog. (2020) World Spider Catalog Version 20.0 Natural History Museum Bern, online at https;//wsc.nmbe.ch, accessed on the 22nd January 2020.

Zuur, A.F. & Ieno, E.N. (2016) A protocol for conducting and presenting results of regression-type analyses. *Methods in Ecology and Evolution*, **7**, 636–645.

Zuur, A.F., Ieno, E.N. & Elphick, C.S. (2010) A protocol for data exploration to avoid common statistical problems. *Methods in Ecology and Evolution*, **1**, 3–14.

Site		Herd 1		Herd 6		Herd 7		Herd Nguni	
1	No kraaling	**S** 19.69107	E 0 29.36696	**S** 19.64025	E 0 29.28051	**S** 19.64025	E 0 29.28051	S 19.65937	E.0.29.30470
2	During cattle occupation	S.19.67682	E 0 29.35800	S 19.62839	E 0 29.26832	S 19.62839	E 0 29.26832	S 19.66359	E.0.29.30591
3	Cattle removed 2 weeks	S.19.70195	E 0 29.37732	S.19.63600	E 0 29.27242	S.19.63600	E 0 29.27242	S 19.65645	E.0.29.30718
4	Cattle removed 4 weeks	S. 19.704	E 0 29.38647	S. 19.64098	E 0 29.28668	S. 19.64098	E 0 29.28668	S 19.64968	E.0.29.31193
5	Cattle removed 6 weeks	S.19.70825	E 0 29.38339	S.19.64497	E 0 29.28898	S.19.64497	E 0 29.28898	S 19.64270	E.0.29.31764
6	Cattle removed 8 weeks	S 19.72186	E 0 29.39313	S 19.63866	E 0 29.30556	S 19.63866	E 0 29.30556	S 19.63724	E.0.29.32519
7	Cattle removed 10 weeks	S 19.71991	E 0 20.39707	S 19. 64994	E 0 29.29533	S 19. 64994	E 0 29.29533	S 19.63164	E.0.29.32266
8	Cattle removed 12 weeks	S 19. 71873	E 0 29.40114	S 19.64422	E 0 29.30052	S 19.64422	E 0 29.30052	S 19.63172	E.0.29.32062
9	Cattle removed 14 weeks	S 19. 73152	E 0 29.39768	S 19.64205	E 0 29.30202	S 19.64205	E 0 29.30202	S 19.692457	E.0.29.32191
10	Cattle removed 16 weeks	S 19.74315	E 0 29.42289	S 19.62868	E 0 29.31593	S 19.62868	E 0 29.31593	S 19.63012	E.0.29.33309
11	Cattle removed 18 weeks	S 19.73836	E 0 29.42373	S 19.62694	E 0 29.31413	S 19.62694	E 0 29.31413	S 19.63012	E.0.29.33611
12	Cattle removed 20 weeks	S 19. 74 376	E0 29. 42631	S 19. 62601	E0 29.31075	S 19. 62601	E0 29.31075	S 19.63163	E.0.29.33611
13	Cattle removed 22 weeks	S 19.74297	E0 29.41851	S 19.63155	E0 29.30695	S 19.63155	E0 29.30695	S 19.637227	E.0.29.33861
14	Cattle removed 24 weeks	S 19.74234	E0 29.40565	S 19.62930	E0 29.30505	S 19.62930	E0 29.30505	S 19.63723	E.0.29.33862
15	Cattle removed 26 weeks	S 19.74806	E 029.39758	S 19.62251	E 029.30556	S 19.62251	E 029.30556	S 19.63806	E.0.29.34593
16	Cattle removed 28 weeks	S 19. 75067	E 0 29. 40096	S 19.61911	E 0 29.30643	S 19.61911	E 0 29.30643	S 19.64313	E.0.29.35061
17	Cattle removed 30 weeks	S 19.74418	E 0 29.40181	S 19.61567	E 0 29.30366	S 19.61567	E 0 29.30366	S 19.65482	E.0.29.35157
18	Cattle removed 32 weeks	S 19. 74141	E0 29.39508	S 19.60712	E0 29.29185	S 19.60712	E0 29.29185	S 19.64731	E.0.29.34930
19	Cattle removed 34 weeks	S 19.74192	E0 29.39731	S 19.61065	E0 29.29114	S 19.61065	E0 29.29114	S 19.64973	E.0.29.34956
20	Cattle removed 36 weeks	S 19.74323	E 0 29.39994	S 19.62635	E 0 29.26821	S 19.62635	E 0 29.26821	S 19.66277	E.0.29.33269
21	Cattle removed 38 weeks	S 19.73297	E0 29.39868	S 19.61976	E0 29.27555	S 19.61976	E0 29.27555	S 19.65349	E.0.29.32700
22	Cattle removed 40 weeks	S 19.71975	E0 29.38755	S 19.61897	E0 29.27895	S 19.61897	E0 29.27895	S 19.64792	E.0.29.33011

Appendix 3.1 Geographic co-ordinates of each of the twenty-two sites sampled in each herd at the Debshan Ranch, Shangani, Zimbabwe in November 2017 and March 2018.

Appendix 3. 2 Checklist of spiders caught during the early sampling season (November 2017) and the late sampling season (March 2018) on 88 previously kraaled sites and their surrounding vegetation at Debshan Ranch, Shangani; G= inside kraals, C= surrounding vegetation, a= early sampling season, b= late sampling season.

Family/species	C a	b	C Total	G a	b	G Total	Grand Total
Agelenidae	**1**		**1**				**1**
Agelena sp.	1		1				1
Ammoxenidae	**3**	**5**	**8**	**5**	**4**	**9**	**17**
Ammoxenus daedalus Dippenaar & Meyer, 1980		4	4		3	3	7
Ammoxenus sp.	3	1	4	5	1	6	10
Araneidae	**3**	**2**	**5**	**1**		**1**	**6**
Caerostris sp.	1		1				1
Neoscona hirta C.L. Koch, 1844	1		1				1
Neoscona sp.	1		1				1
Pararaneus spectator Karsch, 1885				1		1	1
Trichonephila inaurata (Walckenaer, 1841)		2	2				2
Barychelidae	**1**		**1**				**1**
Sipalolasma sp.	1		1				1
Caponiidae		**1**	**1**				**1**
Caponia sp.		1	1				1
Corinnidae	**2**		**2**	**7**		**7**	**9**
Copa flavoplumosa Simon, 1886	2		2	7		7	9
Ctenidae	**2**	**1**	**3**	**1**	**1**	**2**	**5**
Afroneutria velox Blackwall, 1865	1		1				1
Ctenus sp.	1	1	2	1	1	2	4

Cyrtaucheniidae	1		1			1	1
Ancylotrypha sp.	1		1				1
Cheiracanthiidae	2	1	3		1	1	4
Cheiracanthium furculatum Karsch, 1879					1	1	1
Cheiracanthium minshullae Lotz, 2007	1		1				1
Cheiracanthium sp.	1	1	2				2
Gnaphosidae	38	8	46	37	13	50	96
Asemesthes paynteri Tucker, 1923	29	1	30	28	3	31	61
Asemesthes sp.	1	1	2		3	3	5
Camilina sp.	2		2		1	1	3
Drassodes sp.	1		1		1	1	2
Gnaphosidae sp.1				1		1	1
Ibala minshullae FitzPatrick, 2009					1	1	1
Ibala sp.					1	1	1
Nomisia varia Tucker, 1923	2		2	6		6	8
Scotophaeus sp.				1		1	1
Trephopoda aplanita Tucker, 1923		1	1				1
Trephopoda parvipalpa Tucker, 1923		1	1				1
Xerophaeus vickermani Tucker, 1923		1	1				1
Zelotes bastardi Simon, 1896	1		1				1
Zelotes brennanorum FitzPatrick, 2007					1	1	1
Zelotes frenchi Tucker, 1923				1		1	1
Zelotes tuckeri Roewer, 1951	1		1				1
Zelotes sp 5.	1	1	2			1	3
Zelotes sp 6.					1	1	1
Zelotes sp.		2	2		1	1	3

Taxon							
Hersiliidae	**1**		**1**				**1**
Hersilia sericea Pocock, 1898	1		1				1
Liocranidae	**3**		**3**	**2**		**2**	**5**
Rhaeboctesis secundus Tucker, 1920	1		1	2		2	3
Rhaeboctesis trinotatus Tucker, 1920	2		2				2
Lycosidae	**109**	**45**	**154**	**95**	**64**	**159**	**313**
Allocosa faberrima Simon, 1910	1		1				1
Allocosa lawrencei Roewer, 1951	2	3	5				5
Allocosa schoenlandi Pocock,1900	5		5	6	1	7	12
Allocosa umtalica Purcell,1903	47	5	52	47	2	49	101
Allocosa sp.	18	26	44	10	42	52	96
Amblyothele ecologica Russell-Smith,Jocque&Alderweireldt,2009		1	1				1
Evippomma plumipes Lessert,1936	1		1				1
Evippomma squamulatum Simon,1898	3		3				3
Evippomma sp.	2		2				2
Lycosidae sp 1	1		1				1
Pardosa crassipalpis Purcell,1903	3	1	4	6	5	11	15
Pardosa leipoldti Purcell,1903	1		1				1
Pardosa manubriata Simon,1898	16	6	22	17	7	24	46
Pardosa sp 4		1	1		1	1	2
Pardosa sp 5					1	1	1
Pardosa sp.	3		3	4	2	6	9
Proevippa albiventris Simon,1898	2		2	2	1	3	5
Proevippa fascicularis Purcell,1903	1	2	3				3
Schizocosa darlingi Pocock,1898	1		1	1		1	2
Trabea purcelli Roewer,1951				1		1	1
Trabea sp 2	1		1				1

Trabea sp.				1		1	1
Zenonina albocaudata Lawrence,1952				1		1	1
Zenonina sp 1	1	1	2				2
Migidae		**1**	**1**				**1**
Moggridgea sp 1		1	1				1
Oxyopidae	**9**	**8**	**17**	**9**	**3**	**12**	**29**
Oxyopes dumonti Vinson,1863				1		1	1
Oxyopes hoggi Lessert,1915		1	1				1
Oxyopes sp 3	2	2	4	3		3	7
Oxyopes sp 4	1		1				1
Oxyopes sp 5	3		3				3
Oxyopes sp 6	1		1	1		1	2
Oxyopes sp.	2	5	7	4	3	7	14
Philodromidae	**2**	**2**	**4**	**3**	**1**	**4**	**8**
Hirriusa variegata Simon,1895					1	1	1
Suemus punctatus Lawrence,1938				1		1	1
Thanatus dorsilineatus Jezequeli,1964	2	1	3				3
Thanatus sp 1		1	1	1		1	2
Tibellus minor Lessert,1919				1		1	1
Pisauridae	**7**	**1**	**8**	**4**	**1**	**5**	**13**
Euprosthenopsis armata Strand,1913				1		1	1
Maypacius roeweri Blandin,1975	1		1	1		1	2
Maypacius sp.	6		6	1	1	2	8
Perenethis simoni Lessert,1916		1	1				1
Rothus sp.				1		1	1
Prodidomidae	**2**		**2**	**1**	**1**	**2**	**4**

Theuma parva Purcell,1907				1		1	1
Theuma sp.	2		2		1	1	3
Salticidae	**35**	**18**	**53**	**22**	**13**	**35**	**88**
Euophrys purcelli Peckham & Peckham,1903					1	1	1
Heliophanus sp 1		1	1				1
Heliophanus sp.				1		1	1
Hyllus brevitarsis Simon,1902				1		1	1
Langona bethae Wesolowska & Cumming, 2011		3	3		3	3	6
Pellenes bulawayoensis Wesolowska,2000					1	1	1
Pellenes tharinae Wesolowska,2006	1		1		1	1	2
Pellenes sp.					1	1	1
Salticidae sp 12				1		1	1
Stenaelurillus guttiger Simon,1901	5	8	13	1	4	5	18
Stenaelurillus termitophagus (Wesolowska & Cumming,1999)	27	5	32	19	1	20	52
Stenaelurillus sp.	1		1				1
Thyene sp.	1	1	2				2
Segestriidae				**1**		**1**	**1**
Ariadna corticola Lawrence,1952				1		1	1
Selenopidae					**1**	**1**	**1**
Selenops kruegeri Lawrence,1940					1	1	1
Sparassidae	**4**	**2**	**6**	**1**	**1**	**2**	**8**
Olios correvoni Lessert,1921	1	1	2	1		1	3
Olios freyi Lessert,1929	1		1				1
Olios sp.	1		1				1
Panaretella minor Lessert,1919		1	1		1	1	2
Panaretella sp.	1		1				1

Theridiidae		1	1				1
Lactrodectus sp.		1	1				1
Thomisidae	1		1				1
Monases sp.	1		1				1
Zodariidae	11	2	13	3	2	5	18
Capheris fitzsimonsi Lawrence,1936	1		1				1
Diores magicus Jocque&Dippenaar-Schoeman,1992	1	1	2		1	1	3
Diores n.sp.	4		4	1	1	2	6
Diores sp.				1		1	1
Hermippus loricatus Simon,1893		1	1				1
Hermippus sp.	1		1				1
Hermippus tenebrosus Jocque,1986	2		2	1		1	3
Ranops caprivi Jocque,1991	2		2				2
Grand Total	237	98	335	193	106	299	634

Appendix 3.3 Mean and standard deviation for each of the seven vegetation structure variables measured inside previously kraaled inclusions and their surrounding areas at Debshan Ranch conditional on time since kraal removal

Time since kraal removal	Bare ground	Ground vegetation cover	Litter cover	Small rocks/pebbles cover	Coarse woody debris	Cow dung cover	Mean grass height (cm)
Less than 2 months	49.08±8.83	49.29±15.56	17.23±11.48	1.58±1.20	20.55±11.25	76.00±87.76	54.53±16.85
2 months	25.12±11.80	53.83±17.42	15.65±9.93	0.99±0.71	29.21±13.66	58.36±69.02	56.08±26.79
4 months	26.09±10.22	51.69±13.85	15.42±8.02	1.96±3.17	22.64±8.64	74.21±82.56	66.16±17.39
6 months	21.26±8.99	63.06±14.69	10.63±5.72	0.80±1.10	17.91±15.78	66.82±65.33	74.38±35.41
8 months	27.40±16.98	61.56±18.77	7.31±5.31	0.75±1.05	19.50±11.72	39.43±46.85	79.53±34.50
10 months	32.69±17.32	55.43±20.33	7.79±5.91	0.48±0.78	30.08±20.75	23.42±32.74	85.13±42.48

Chapter 4

Testing the efficacy of the South African National Survey of Arachnida sampling protocol within Khami World Heritage

4.1 Abstract

Since the initial first design of sampling protocols for invertebrates, utilisation of standardized protocols has been a widely recommended concept. One such protocol is the South African National Survey of Arachnida, sampling protocol that yields high levels of coverage for most sites and the methods utilised in the protocol yield an impressive diversity of arachnids. Despite, its successful utilisation within South Africa it has however not yet be utilised in Zimbabwe. As a result of the limited knowledge on species richness of spiders that is available within Zimbabwe the protocol was selected to sample spider diversity within one of Zimbabwe protected areas that had not been sampled before. The purpose of this study was to establish a database of spider fauna at the Khami World Heritage site in Bulawayo. Furthermore, species richness of spider assemblages caught by six methods in three seasons (summer, winter and spring) was estimated using coverage-based rarefaction methods amongst four selected biotopes. In each biotope ten transects were demarcated and six sampling methods, pitfall traps, beating, sweep-netting, litter sifting, day hand collecting and night hand collecting were utilised. A total of 133 species in 110 genera and 27 families were collected. The riparian woodland had the highest species richness compared to mixed woodland, *Vachellia* and Mopane woodland, the summer period caught the greatest diversity with winter recording the least diversity. Night and day hand collecting had the greatest observed species richness, with adult individuals. The mixed woodland required the least number of samples in order to collect 50% of the spider assemblages. The optimal sampling protocol suggested a combination of almost half of the sweep netting and night hand collecting and four samples of pitfall traps, with two samples of day hand collecting method required in order to collect at least 80% of the species. Standardised sampling protocols are essential as they enable comparison of studies with optimization enabling the collection of the maximum diversity within an area with less effort especially when human resources and time are restricted.

4.2 Introduction

Species richness is one of the oldest, simplest and commonly used measures of diversity that has been utilised by most community ecologists and conservation biologists (Gotelli & Colwell 2001). Unfortunately, observed species richness is a downward biased estimator (Colwell *et al.* 2012) that is affected by sample size and its exclusion of the relative abundance of species (Chao *et al*. 2014). Attempts to address these issues include utilisation of standardised comparisons of species richness after interpolation with rarefaction (Tipper 1979, Chao *et al.* 2014, Hsieh & Chao 2017) to a mutual level of abundance (Gotelli & Colwell 2001, 2011) sampling effort (Colwell *et al.* 2004, 2012) and sampling completeness (Chao & Jost 2012). Alternatively, non-parametric asymptotic richness estimators may also be utilised (Gotelli & Colwell 2011, Chao & Lin 2012, Chao & Chiu 2016) as they attempt to reduce undersampling bias, and the majority are valid for almost all species abundance distributions (Chao & Chiu 2016). The asymptotic approach is based on species richness estimators that aim to estimate the asymptote of a species accumulation curve, which is later utilised as a species richness estimate that can then be used to compare assemblages (Chao & Chiu 2016).

Accurate and comparable data is required to describe, monitor and assess drivers of biodiversity. In conservation planning species richness provides the foundation for decision making (Magurran 2004). Despite the efficiency of *ad hoc*, non-standardized approaches in compiling species lists (Gordon & Newton 2006) standardised protocols are preferred, as these provide reliable estimations of true species richness when compared to the utilization of estimators (Malumbres-Olarte *et al.* 2016). Further, they are also useful in the comparison of even undersampled areas with incomplete species lists. A standardised sampling protocol is one which enables comparability of data when it is applied to sites of the same biotope, whereas an optimised protocol seeks to distribute the number of samples among methods in order to obtain the maximum possible species and species assemblages with minimum effort (Malumbres-Olarte *et al.* 2016). According, to Cardoso *et al.* (2009) standardised, optimised and *ad hoc* sampling all have different benefits when generating species data. However, in order to improve monitoring of biodiversity there should be standardized and regular repeated measurements of each biome and biota, of which Teder *et al*. (2007) argues as lacking in most countries.

The improvement of sampling protocols has been one of the most critical priorities of arthropod research since the first design by Coddington *et al*. (1991). Several studies have modified and applied the Coddington's (1991) sampling protocol in both tropical and temperate forests (Coddington *et al.* 1996, Silva & Coddington 1996, Dobyns 1997, Toti *et al.* 2000, Scharff *et al.* 2003, Muelelwa *et al.* 2010, Haddad & Dippenaar-Schoeman 2015). Nevertheless, standardised sampling protocols have not only been developed for spiders but also for other taxa as well (Agosti *et al.* 2000, Hering *et al.* 2004, VanSwaay *et al.* 2015). According to Jocqué *et al*. (2013) the species richness of arachnids known in most African countries is equivalent to the amount of research that has been done in that particular country. For example, South Africa has the highest described spider richness in Africa (Jocqué *et al.* 2013, Dippenaar-Schoeman *et al.* 2015) mainly as a result of the South African National Survey Arachnida (SANSA) project that enhanced the development of the SANSA sampling protocol (Muelelwa *et al.* 2010, Haddad & Dippenaar-Schoeman 2015, Malumbres-Olarte *et al.* 2016) from preceding standardized protocols (Muelelwa *et al.* 2010). Similarly, within the Mediterranean region as a result of the tremendous effort of research on spiders, another standardized field protocol for spiders known as Conservation Oriented Biodiversity Rapid Assessment (COBRA) was also developed (Cardoso 2009) in that region. Because of similar climatic conditions, with South Africa and also considering that both Zimbabwe and parts of South Africa fall within the sub-tropical region, the SANSA sampling protocol was selected to be utilised within the Khami World Heritage Site.

In Zimbabwe significant efforts have been made to develop all-inclusive and elaborate biodiversity inventories and monitoring programmes for most organism's such as mammals (Dunham *et al.* 2009, Jammes *et al.* 2009, Dunham & du Toit 2013), reptiles (Zisadza-Gandiwa *et al.* 2013, Sai *et al.* 2016), birds (Bird Life Zimbabwe 2010), plants (Mapaura & Timberlake 2004), however, the least documented are the arthropods in particular spiders where few inventories or monitoring programmes has been implemented leading to development of a few individual checklists (FitzPatrick 2001, Wesolowska & Cumming 2011) conducted by mostly private researchers and museum taxonomists. However, none of the previous inventories utilised the SANSA sampling protocol, the latter was developed specifically for the SANSA project and has been tested within a savanna biome and has yielded impressive diversity of arachnids of the Ophathe Game Reserve, South Africa, despite sampling having been done within one season.

Efforts to consolidate knowledge on species checklists have been initiated at a globally level by various projects, one such example includes, the Global Biodiversity Information Facility (GBIF) (www.gbif.org). Despite, the effort of placing all known published records in such databases, the knowledge of all species will still be relatively poor (Cardoso 2009). For example, in Zimbabwe the current status on the knowledge of spider fauna, is 349 species of spiders from published records (Jocqué *et al.* 2013) however FitzPatrick (2001) lists 150 000 specimen lots that still awaits identification. Thus in order to improve the knowledge on biodiversity not only should the collection of published records be compiled, together with the identification of specimens within museums but other activities that include, carrying out of inventories and monitoring programs can also be initiated. Inventories are quite significant in the description of flora and fauna of a selected area and monitoring permits the comprehension of trends of selected parameters (Inventory and Monitoring Studies 2008).

The lack of baseline information on spider fauna is one of the reasons why invertebrates are often neglected in conservation programs (Cardoso *et al.* 2011). Therefore, in order to improve baseline information on spider diversity in Zimbabwe, one of the lesser studied countries in Africa (Jocqué *et al.* 2013) this study utilized the SANSA sampling protocol, as it yields high level of coverage for most sites and methods (Haddad & Dippenaar-Schoeman 2015) to: (i) generate a semi-quantitative database for the Khami World Heritage Site; (ii) determine the abundance and species richness of spider assemblages in each biotope, season and caught using different methods; (iii) determine sampling completeness of each sample and estimate species richness and (iv) explore the implications of the combinations of sampling methods that will be optimal in the estimation of species richness.

4.3 Materials and Methods

4.3.1 Study area

The study was carried out at the Khami World Heritage Site (20° 09' S 28° 22' E) (Fig. 4.1) in three seasons spring (March 2017); winter (July 2017) and summer (November 2017). The Khami World Heritage site is the second largest cultural site in Zimbabwe after Great Zimbabwe (Sinamai 2019) with an extensive complex of dry-walled sites, and is situated 22 km west of Bulawayo with a total coverage of 108 hectares (Hubbard *et al.* 2017). It lies to the west bank of the Khami Dam (Musarandega 2015) built during 1928-1929 (Mukwende 2016). It is also

surrounded by the Dlodlo and Green Cables cattle ranching farms to the west and the Department of Water Works Bulawayo City Council Staff Compound to the east (Hubbard *et al.* 2017). The property has been subjected to some degradation due to veld fires, effects of tourism and encroaching vegetation (Musarandega 2015). The Khami World Heritage Site lies at an elevation of approximately 1299 m a.s.l.
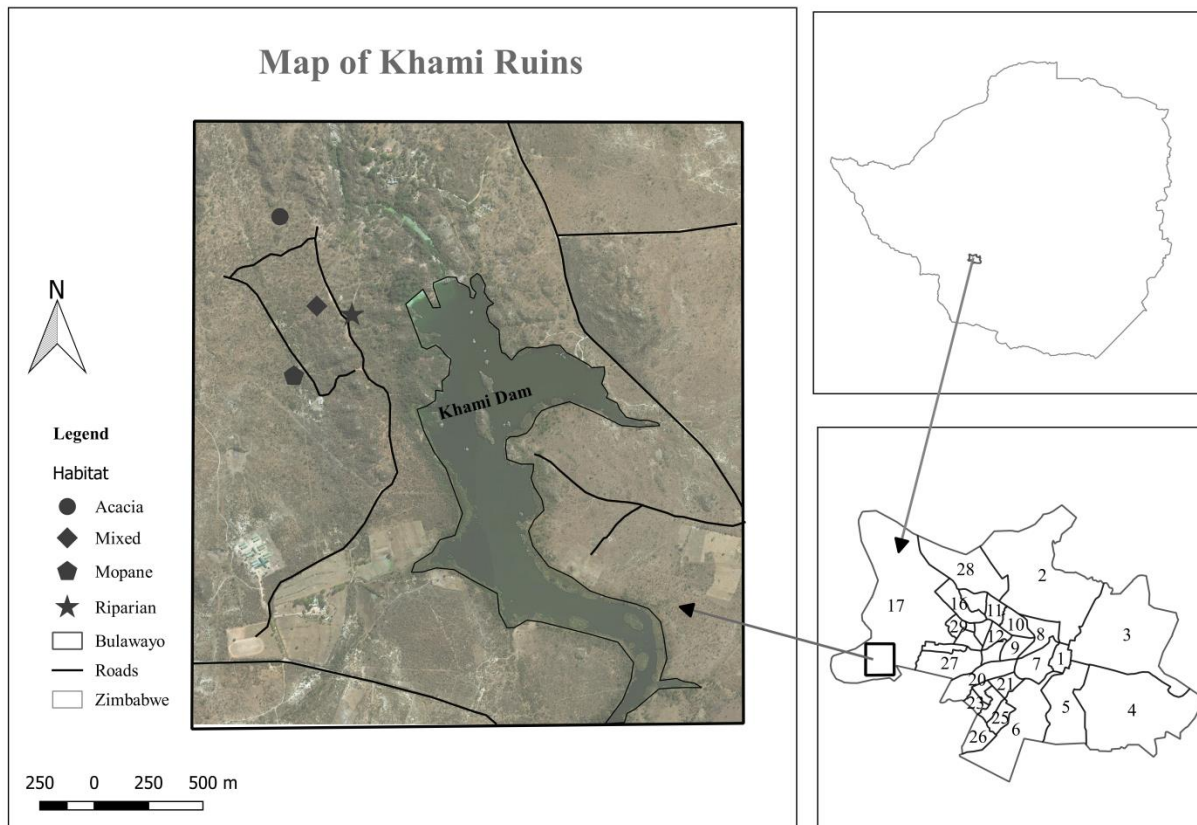
Fig. 4.1 Location of Khami World Heritage Site within Zimbabwe and the distribution of the spider sampling sites within the world heritage site.

The Khami World Heritage Site have an annual average rainfall of 567 mm, with most precipitation falling in December, with an average of 131 mm, while the driest month is July, with an average of 0 mm. Rainfall is subject to periodic seasonal droughts and dry spells during the rainy season (Cousins 1992). It is also characterised by an average annual temperature of 19 ℃. The hottest month is October, with an average of 22.4 ℃, and the coldest month is June, with an average of 13.4 ℃. The terrain of the Khami World Heritage Site is generally flat, with a number of scattered granite outcrops (Hubbard *et al.* 2017). The region is characterised by generally greyish-brown sandy loams that arise from granitic parental rocks (Hubbard *et al.*

2017). The climate supports a natural vegetation of open woodland dominated by mopane woodland and shrub (Arconsult 1998). There are very few wild animals that are commonly seen at Khami World Heritage Site, but those that have been observed include baboons (*Papio ursinus*), vervet monkeys (*Chlorocebus pygerythrus*), common duiker (*Sylvicapra grimmia*), bushpig (*Potamochoerus larvatus*), common warthog (*Phacochoerus africanus*), common genet (*Genetta genetta*), serval (*Leptailurus serval*), black-backed jackal (*Canis mesomelas*), impala (*Aepyceros melampus*) and the occasionally leopard (*Panthera pardus*). The bird life is prolific, and at least 150 different species are present due to the wide range of biotopes (Hubbard *et al.* 2017). Activities that can be done at Khami World Heritage Site include nature walking, bird watching, picnicking and fishing (Bulawayo Publicity Association 2020).

4.3.2 Study design

The study utilised four selected biotopes that were classified according to the dominant vegetation type (Fig. 4.2). These were selected as there were considered to be representative of the area under study (Haddad & Dippenaar-Schoeman 2015).

4.3.2.1 Biotope 1: Mopane woodland (Fig. 4.2a) is normally found at low elevations below 900 m and where the climatic conditions are dry and hot, with rainfall below 600mm. The woodland develops best on deep-well drained soils with clay-rich substrates (Mufandaedza 2002) characterised by sodic and alluvial soils (Mapaure 1994). It is dominated by *Colophospermum mopane*, and where mixed stands occur, associated species include *Senegalia* spp, and other shrub species such as *Combretum* spp and *Kirkia* spp (Mufandaedza 2002). Where mopane occurs grasses are often excluded with low species diversity resulting from the resultant woodland or shrubland (Mapaure 1994).

4.3.2.2 Biotope 2: Riparian woodland (Fig. 4.2b) is normally found in swampy depressions along edges of streams and rivers that remain underwater continuously for a long period throughout the rainy season (Little *et al.* 2015). Structurally, this biotope naturally has a combination of sparse and closed canopy, associated with rich humus soil and a deep layer of leaf litter. The most common trees include *Terminalia* spp and the Cape Fig. *Ficus capensis* and the swamp couch, *Hemarthria altissima*, occurs in the swampy areas associated with the Khami river (Arconsult 1998).

4.3.2.3 Biotope 3: Mixed woodland (Fig. 4.2c) comprises a wide variety of trees, shrubs and herbaceous plants, typical of this rocky terrain, such as, *Cassia abbreviata, Carissa edulis* and *Azanza garckeana,* large sourplum, *Ximenia caffra* and red-leaved rock fig (*Ficus ingens*) occur within the mixed bushveld, amongst the granite rocks and riverine area (Arconsult 1998).

4.3.2.4 Biotope 4: *Vachellia* woodland (Fig. 4.2d) is normally found in dry areas and grows on eutrophic soils that established from base-rich geological formations, mainly along river valleys, low lying areas. Depending on the soil type, they are dominated by various *Vachellia* and *Senegalia* species. For example, Kalahari sands are dominated by *Vachellia erioloba*, black clays by *Vachellia nilotica* and on alluvial sands *Vachellia karroo* is usually associated with *Vachellia tortilis.* Because of their soil enriching properties, *Vachellia* woodlands are often associated with a rich grass understory. Common grass species include *Hematrium altissima, Eragrostis rigidior and Panicum coloratum* (Shumba 2001).
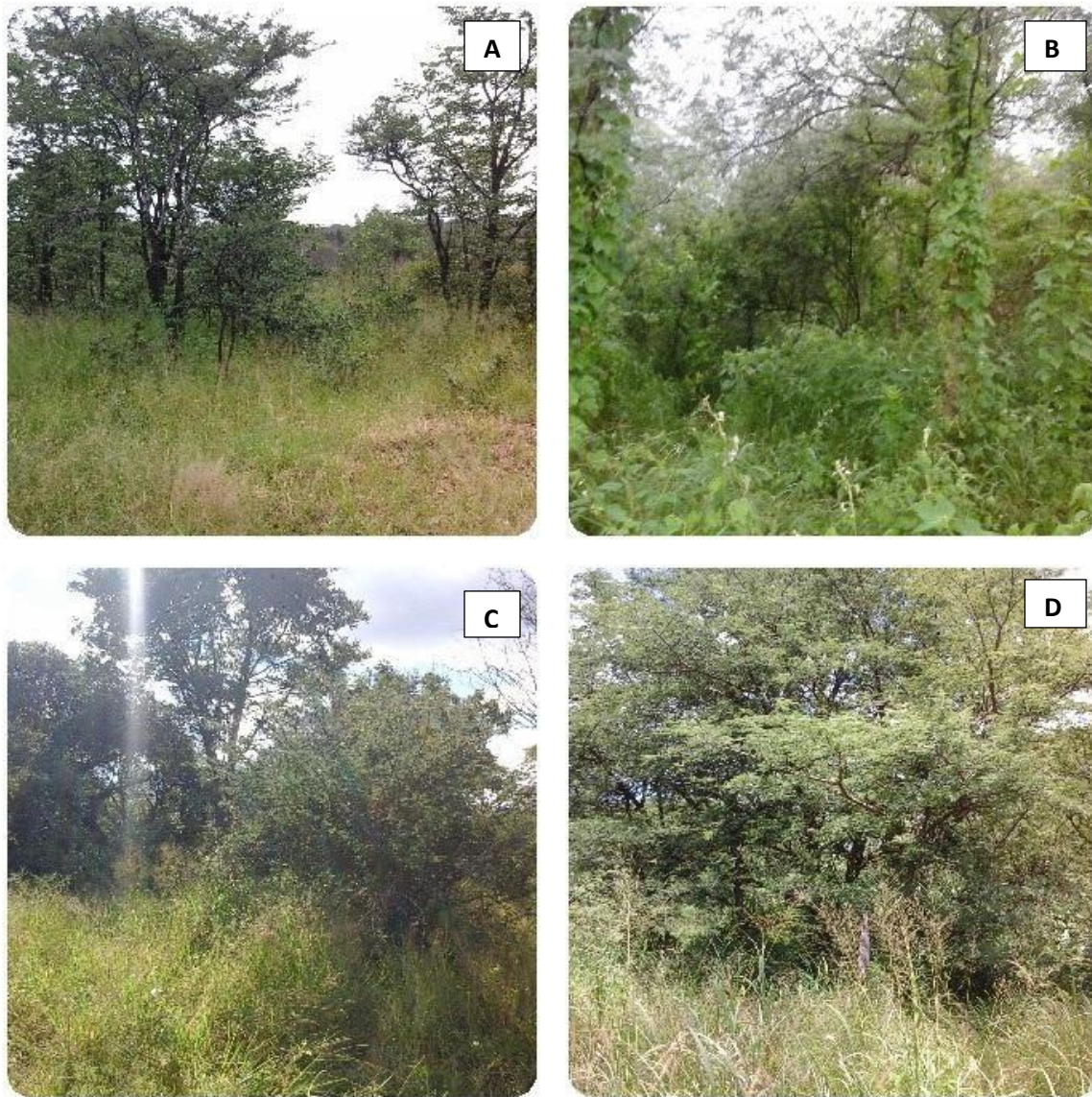
Figs 4.2 Biotopes where spider sampling was done during March, July and November 2017 using the South African National Survey of Arachnida protocol at the Khami World Heritage Site, Zimbabwe: (A) Mopane woodland, (B) Riparian woodland, (C) Mixed woodland and (D) *Vachellia w*oodland. Photographs: Sicelo Sebata.

In each biotope, ten transects measuring 10 x 50 m were positioned within a plot of 0.5 hectare (Fig. 4.3). The transects were set out parallel to each other, and the same transects were sampled on three occasions, March, July and November 2017.
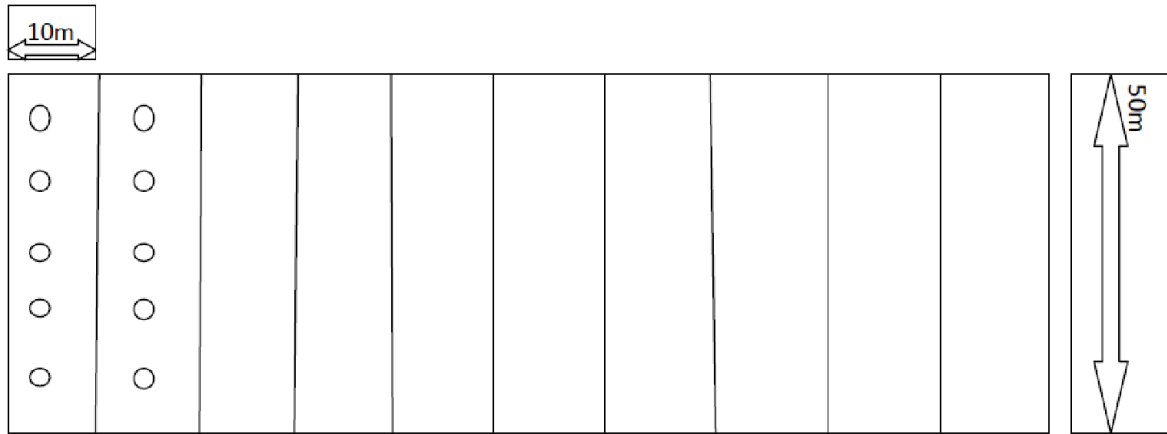
Fig. 4.3 Placement of 10 transects within each biotope used to sample spider diversity at the Khami World Heritage Site, Zimbabwe showing pitfall traps placed 10m apart.

4.3.3 Sampling of spiders

To sample spiders, six of the seven sampling techniques proposed by the SANSA sampling protocol (Haddad & Dippenaar-Schoeman 2015) were utilized. The Winkler traps were not utilised in this study, as they produced a very low number of specimens per sample and did not contribute any taxa to overall diversity that were not sampled by other methods within the Ophathe Game Reserve, KwaZulu-Natal, South Africa (Haddad & Dippenaar-Schoeman 2015). Sampling was done once in each of the three selected seasons using the methods detailed below. Collecting was done in the morning between 07:00 to 11:00 and at night between 20:00 and 22:00 (night collecting only) by a four-person survey team comprised of two technical assistants, one staff member of the Natural History Museums, and the student.

4.3.3.1 Pitfalls: Fifty pitfalls, placed 10 m apart, were kept open for four days in each biotope. Pitfalls consisted of glass cups 14 cm deep and 9 cm wide at the mouth, placed inside a plastic PVC pipe and buried to their rims in the ground (Fig. 4.4A). The glass cups were filled with 100 ml of propylene glycol. At the end of the collecting period the contents were passed through a sieve to eliminate excess sand and the material emptied into suitable containers for sorting in the laboratory.

4.3.3.2 Leaf litter: Ten litter sifting samples (two filled 5 litre containers constituting one sample) were collected from each transect. The material was sifted over a white sheet (Fig. 4.4B), after which spiders were collected with an aspirator and preserved in 70% ethanol. Each sample was

stored as a separate unit.

4.3.3.3 Beating: Five hundred beats, divided into 10 x 50 beat samples, were taken from each site by hitting spiders from shrubs, tall herbal vegetation, bushes and small trees with a 1 m long stick, onto a white beating sheet (50 cm x 50 cm). Spiders were captured with an aspirator (Fig. 4.4 C) and put into separate bottles for each sample.

4.3.3.4 Sweep nets: Five hundred sweeps, divided into 10 x 50 sweep samples, were taken from each site. This involved using a sweep net (50 cm in diameter with a 75cm handle) that was swung back and forth across the grass for a few paces per sampling point along each transect until 50 sweeps had been completed (Fig. 4.4 D). The sweep net handle was rotated so that the net stayed shut between sweeps, and each sweep covered an arc of almost 180° through the vegetation (Coddington *et al.* 1996). After 25 sweeps, samples were emptied on a plain sheet of cloth and all visible invertebrates were captured with an aspirator. Each sample of 50 sweeps was kept separately in 70 % ethanol.

4.3.3.5 Day hand collecting: Each of the four team members carried out two hours of hand collecting during the day, split into 30-minute subsamples (16 per site), which included active searching of the ground (Fig. 4.4E), shrubs, rocks, logs, bark and stones for spiders. Specimens were collected using either by hand or using an aspirator.

4.3.3.6 Night hand collecting: Each of the four team members carried out two hours of hand collecting during the night, split into 30-minute subsamples (16 per site), which included active searching of the ground (Fig. 4.4F), shrubs, rocks, logs, bark and stones for spiders, with the aid of a headlamp. Specimens were collected using either by hand or using an aspirator.

Fig. 4.4 Sampling methods utilised to sample spider diversity at the Khami World Heritage Site, Zimbabwe during March, July and November 2017 using the South African National Survey of Arachnida protocol: (A) Pitfall traps, (B) Litter sifting, (C) Beating sheets, (D) Sweep nets, (E) Day hand collecting and (F) Night hand collecting. Photographs: Rorisang Sebata.

4.3.4 Identification of specimens

All of the collected specimens were sorted to morphospecies and to species level where possible in the laboratory and stored in vials in 70% ethanol. The student performed preliminary identifications, which were confirmed by Dr. M. J FitzPatrick and Professor C. Haddad. All adult specimens were deposited in the Arachnid collection of the Natural History Museum of Zimbabwe, in Bulawayo (NMZ).

4.3.5 Data Analysis

Estimated species richness and completeness was calculated for two data sets, one including and the other excluding juveniles. Juveniles have been shown to be useful surrogates (Foord *et al.* 2013) and their inclusion in samples enables reliable estimates of species richness (Jiménez-

Valverde & Lobo 2006). To estimate and compare species richness within the Khami World Heritage site, a non-asymptotic approach was utilised, using coverage-based rarefaction and extrapolation (Chao & Jost 2012, Chao *et al.* 2014). Hill numbers are the effective current diversity measure of choice (Chao *et al.* 2014), as rarefaction and extrapolation tends to make fair comparisons amongst incomplete samples (Chao *et al.* 2014).

The iNEXT package in R 3.2.3 software (R project for Statistical Computing, http://www.r-project.org) was utilised to obtain estimates of species richness for all biotopes, methods and seasons. These were utilised to compare diversity amongst samples using rarefaction, with the aid of coverage-based rarefaction curves, that standardized all samples to an equal coverage (Chao *et al.* 2014); 500 bootstrap replicates were utilised to estimate the 95% confidence intervals. First, coverage based rarefaction curves for each sample was constructed. Then the curve with the lowest final sample coverage was identified and all the other curves that consisted of higher coverage were rarefied down to the coverage of the lowest final coverage. Then on each of the other curves, points that had the same final coverage were located and then utilised to identify the species richness that corresponded to that point. According to Chao & Jost (2012) the set of species richness obtained in this way are based on equal coverage of samples and can be legitimately compared with each other.

In order to identify the combination of samples that maximises the amount of species caught an algorithm (function "optim.alpha") included in the R package (Cardoso *et al.* 2018) was utilised to run 1 000 simulations of all selected sites. A sampling completeness of 50% was selected as a target for all sites as it has been applied in most tropical forests (Cardoso 2009, Cardoso *et al.* 2009) and has been considered practical for most tropical regions due to its ability to obtain relatively high species diversity (Malumbres-Olarte *et al.* 2016). Summer has been shown to be the best optimal period for collection spiders especially within sub-tropical regions (Whitmore *et al.* 2002, Muelelwa *et al.* 2010), which has been considered to be attributable to the maturation of sub adults during winter that attain adulthood during summer when they mate (Muelelwa *et al.* 2010). Similarly, in this study the greatest diversity was collected during the summer period as compared to the other seasons therefore it was selected for utilisation in the optimisation of the sampling protocol.

4.4 RESULTS

4.4.1 Community composition

The inventory produced a total of 744 actively collected samples (248 per sampled month) and 600 pitfall traps in total (200 per sampled month), which altogether yielded 6508 spider individuals. Of these, 1386 specimens (662 males and 724 females) were adults that represented 133 species, 110 genera and 27 families (Appendix 4.1). The dominant families collected were the Araneidae (26 spp.), followed by Oxyopidae (19 spp.), Thomisidae (17 spp.) and Theridiidae (7 spp.) (Table 4.1). However, the most abundant species collected was *Stenaelurillus termitophagus* (Wesolowska & Cumming, 1999) (n = 145), as it accounted for 12.4% of all adults. *Argyrodes zonatus* (Walckenaer, 1841) (n = 66) was the next most abundant species (2% of whole assemblage). The most species rich genus was *Oxyopes* Latreille, 1804 (Oxyopidae, 13 spp.), followed by *Araneus* Clerck, 1757 (Araneidae, 5 spp.). All species are new records for the Khami World Heritage Site, as there had been no prior sampling. Two species are possibly new, based on identifications of specialists (Appendix 4.1).

4.4.2 Biotope changes

Sample coverage, which is a measure of sample completeness, showed values above 0.88 for all biotopes for both adult and total morphospecies composition. This implies that at least 88% of all individuals in the Khami World Heritage Site spider community belonged to the species that were caught. This means that in all biotopes, using the least coverage recorded in the mopane woodland there is a 12% chance that the next individual sampled at the Khami World Heritage Site would be a species that had not been sampled before. For example, when considering the adult species richness recorded within the Khami World Heritage Site (133) that value is slightly less than the greatest Chao 1 estimator estimated for the Mopane woodland (152). For all biotopes, coverage for the adult morphospecies was quite similar, ranging between 88% and 92%, the mixed woodland recorded the highest while the mopane woodland recorded the least coverage. When juvenile morphospecies were included, coverage values increased slightly for all biotopes by a mean value of 9.01% (Table 4.2).

Table 4.1 Summary table of adult abundance and species richness for all spiders recorded in March, July and November 2017 for Khami World Heritage Site, Zimbabwe. BT = beating, HD = day hand collecting, HN = night hand collecting LL= leaf litter, PT = pitfall traps and SW = sweep nets.

	ADULT ABUNDANCE							ADULT SPECIES RICHNESS						
	BT	HD	HN	LL	PT	SW	TOT %	BT	HD	HN	LL	PT	SW	TOT (n)
Ammoxenidae	0	0	0	0	36	0	2.60	0	0	0	0	1	0	1
Araneidae	20	185	519	14	1	10	54.04	8	16	20	4	1	8	26
Cheiracanthiidae	3	1	11	0	0	0	1.08	2	1	2	0	0	0	2
Corinnidae	0	3	0	0	2	0	0.36	0	2	0	0	1	0	3
Ctenidae	0	0	0	0	1	0	0.07	0	0	0	0	1	0	1
Gnaphosidae	0	18	4	17	13	0	3.75	0	9	4	11	7	0	19
Hersiliidae	0	1	1	0	0	0	0.14	0	1	1	0	0	0	1
Liocranidae	0	0	0	0	1	0	0.07	0	0	0	0	1	0	1
Lycosidae	0	12	2	2	37	0	3.82	0	5	1	2	9	0	10
Mimetidae	0	0	11	0	0	0	0.79	0	0	2	0	0	0	2
Oxyopidae	12	10	32	2	32	12	7.22	8	6	13	2	4	7	17
Philodromidae	0	4	1	0	1	11	1.23	0	2	1	0	1	1	3
Salticidae	44	42	17	9	24	23	11.47	6	12	8	6	6	7	17
Sparassidae	0	0	4	0	0	0	0.29	0	0	2	0	0	0	2
Tetragnathidae	0	2	10	1	0	2	1.08	0	2	1	1	0	2	3
Theridiidae	2	39	35	19	2	6	7.43	2	6	5	2	1	2	7
Thomisidae	6	6	9	0	1	9	2.24	6	4	6	0	1	8	17
Trachelidae	1	0	0	0	0	0	0.07	1	0	0	0	0	0	1
Zodariidae	0	4	2	2	23	0	2.24	0	3	1	1	4	0	5

Table 4.2 Summary table of actual and estimated species richness for all spiders recorded in March, July and November 2017 for Khami World Heritage Site. Biotope: MxW = mixed woodland, MW = mopane woodland, RW = riparian woodland, VW = *Vachellia* woodland; Sampling method: BT = beating, HD = day hand collecting, HN = night hand collecting LL= leaf litter, PT = pitfall traps and SW = sweep nets; Seasons: SU = summer, WI = winter and SP = spring, imm = immature.

Indicator	Biotope				Sampling method						Seasons		
	MxW	MW	RW	VW	BT	HD	HN	LL	PT	SW	SU	WI	SP
Adult abundance (n)	407	341	328	310	88	327	658	66	174	73	770	207	409
Total abundance (incl.imm)	1793	1747	1500	1468	1184	1178	2758	318	427	643	2610	2486	1412
Adult richness (S obs)	72	74	81	72	33	69	67	28	37	35	82	53	70
Genera richness (incl.imm)	69	76	80	78	56	67	63	46	38	42	82	82	69
Adult singletons (F $_1$)	32	40	36	30	22	37	25	20	18	22	29	26	29
Adult doubletons (F $_2$)	12	9	14	12	6	13	14	2	5	8	15	11	14
Genera singletons (F1)	14	13	21	17	21	13	9	17	17	8	20	19	15
Genera doubletons(F2)	8	4	11	6	6	10	9	4	4	8	10	10	9
Chao 1 richness (incl.imm)	83	96	98	102	86	74	67	73	65	45	99	98	80
Chao 1 richness (adults)	110	152	123	106	66	117	87	91	63	61	109.99	83.57	99.96
Coverage (%) (incl.imm)	99.16	99.20	98.60	98.84	98.23	98.90	99.67	94.66	96.02	98.76	99.23	99.24	98.94
Coverage (%) adults	92.15	88.25	89.05	90.32	75.16	88.64	96.21	69.79	89.69	70.16	96.23	87.49	92.91
EstimateD (adults)	55	74	78	65	24	21	13	28	13	35	42	53	50

The observed adult species richness was highest in the riparian woodland as compared to the other biotopes. When juveniles were added, genera richness was still the highest in the riparian woodland with mixed woodland recording the least richness (Table 4.2). Similarly, coverage based rarefaction estimated species richness also recorded the highest in the riparian woodland with the mixed woodland recording the least species richness (Table 4.2 & Fig. 4.5). The same trend was also true for Shannon diversity (q = 1) and Simpsons diversity (q =2), with the riparian woodland showing higher diversity, despite the overlap of the confidence intervals (Fig. 4.5). In contrast, the Chao 1 estimator indicated the highest species richness in the mopane woodland with the *Vachellia* woodland recording the least for adults, when juveniles were included *Vachellia* woodland was estimated to record the highest genera richness whist the mixed woodland had the least genera richness. The largest number of adult individuals was collected in the mixed woodland (n = 407) with the lowest being recorded in the *Vachellia* woodland (n = 310), a similar trend was observed when juveniles were also included with the mixed woodland still recording the largest number of caught spiders (n = 1793) and the least recorded in the *Vachellia* woodland (n = 1468) (Table 4.2).

Fig. 4.5 Coverage-based rarefaction (solid lines) and extrapolation (dashed lines) sampling curves with 95 % confidence intervals (shaded areas, based on a bootstrap method with 200 replications) comparing spider species richness between the four sampled biotopes during the three season at Khami World Heritage Site, Zimbabwe. Hill numbers of order 0, 1 and 2 : q = 0 =:species richness; q= 1: Shannon diversity; q = 2: Inverse of Simpsons concentration.

4.4.3 Seasonality

Sample completeness for each season ranged between 87.49 % and 96.23 % for adult species and between 98.94% and 99.24% for assemblages that included juvenile's (Table 4.2). Sample coverage based on the pooled data of the four biotopes was above 87 % in all the three seasons. This implies that 87% of all the species caught within the Khami World Heritage Site community belonged to the species that were recorded in all the three seasons, therefore indicating adequate sampling of the spider communities during each season. The observed adult species richness was highest during summer with winter recording the lowest. When juveniles were added the summer and winter period recorded similar observed adult species richness which was higher than the least observed richness during spring (Table 4.2).

Coverage based rarefaction estimated species richness indicated that winter recorded the highest species richness with summer with the least species richness at a coverage of 87.49% for all seasons (Table 4.2 & Fig. 4.6). The same trend was also true for Shannon diversity (q = 1) with the winter period recording the highest diversity while summer had the least, however the

163

Simpsons diversity (q =2), estimated the highest similar diversity for both summer and winter with the spring recording the least diversity. Chao1 species richness estimator also estimated summer as having the highest species diversity, for both adult and total morphospecies composition with winter having the least estimated adult species richness while spring had the least estimated total genera richness (Table 4.2). The largest number of adult individuals was collected during summer (n = 770) with the lowest being recorded during winter (n = 207), however, when juveniles were added spring recorded the lowest number of caught spider individuals (n = 1412) while summer retained the highest number of collected spider individuals (n = 2610).
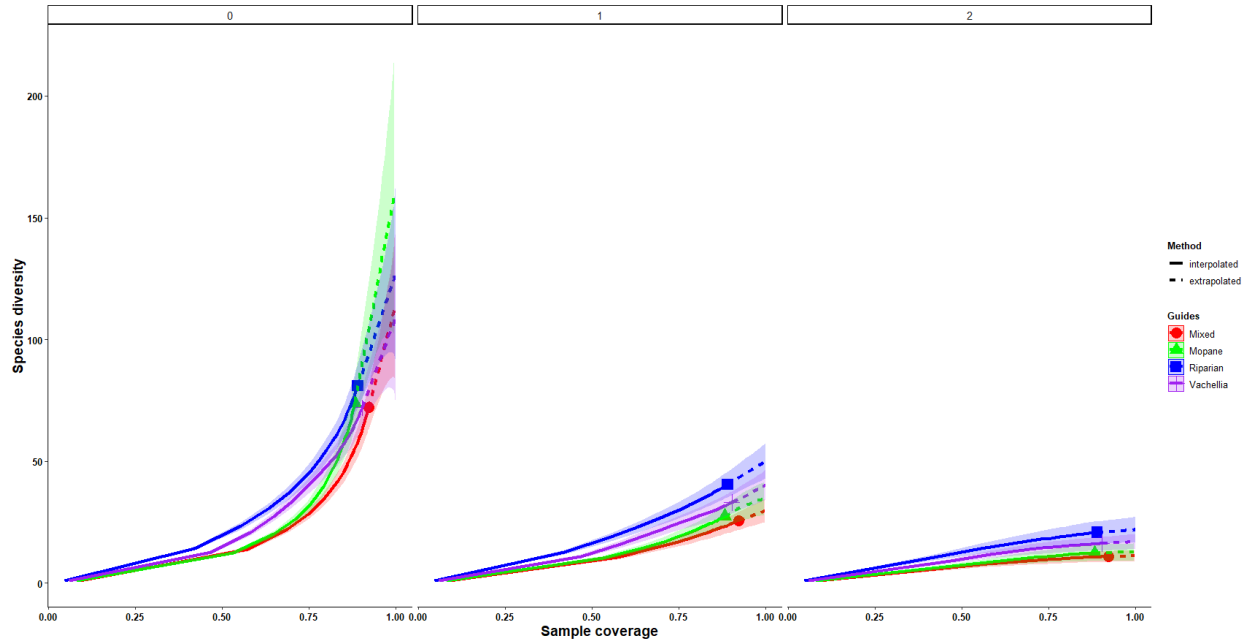

Fig. 4.6 Coverage- based rarefaction (solid lines) and extrapolation (dashed lines) sampling curves with 95 % confidence intervals (shaded areas, based on a bootstrap method with 200 replications) comparing spider species richness between the three sampled seasons at Khami World Heritage Site, Zimbabwe. Hill numbers of order 0, 1 and 2 : q = 0 =:species richness; q= 1: Shannon diversity; q = 2: Inverse of Simpsons concentration.

4.4.4 Sampling methods

Sample coverage of the night hand collecting had the highest sample completeness of (96.21%) when adults were considered, followed by pitfall traps (89.69%) and day hand collecting (88.64 %), with leaf litter recording the least coverage of (69.79%). When juveniles were included, sample coverage for all methods increased, with all values above 94% (Table 4.1). This validates that a satisfactory and representative samples were collected using all six methods that were

selected for this study, as suggested by the SANSA sampling protocol. Day hand collecting and night hand collecting yielded the greatest observed species richness for both adult spiders, as well as the assemblages that included juveniles. Coverage based rarefaction estimated species richness indicated that sweep netting recorded the highest species richness with leaf litter with the second highest species richness; night hand collecting and pitfall traps had the least species richness at a coverage of 69.79% for all methods (Table 4.2 & Fig. 4.7).

The same trend was also true for Shannon diversity (q = 1) and Simpsons diversity (q =2), with sweep netting recording the highest diversity while day hand collecting had the least diversity, despite the overlap of the confidence intervals (4.6). In terms of abundance, the largest number of caught individuals was collected by night hand collecting for both adults (n = 658) and assemblage including juveniles (n = 2758) with litter sifting having caught the least individuals also for both adults (n =66) and when juveniles have been included (n = 318). In contrast, the Chao 1 estimator indicated the highest species richness having been caught by day hand collecting for adults while when juveniles were included beating sheets were estimated to have the highest diversity, while the least diversity was estimated by sweep netting for both adults and assemblage including juveniles.
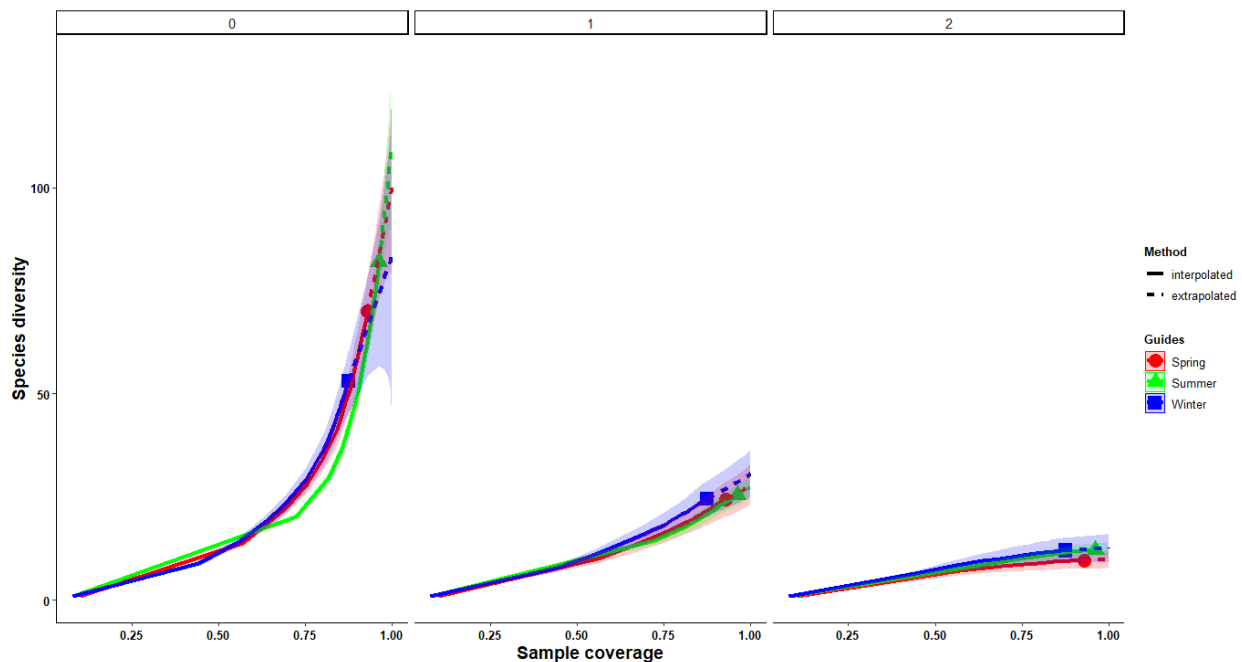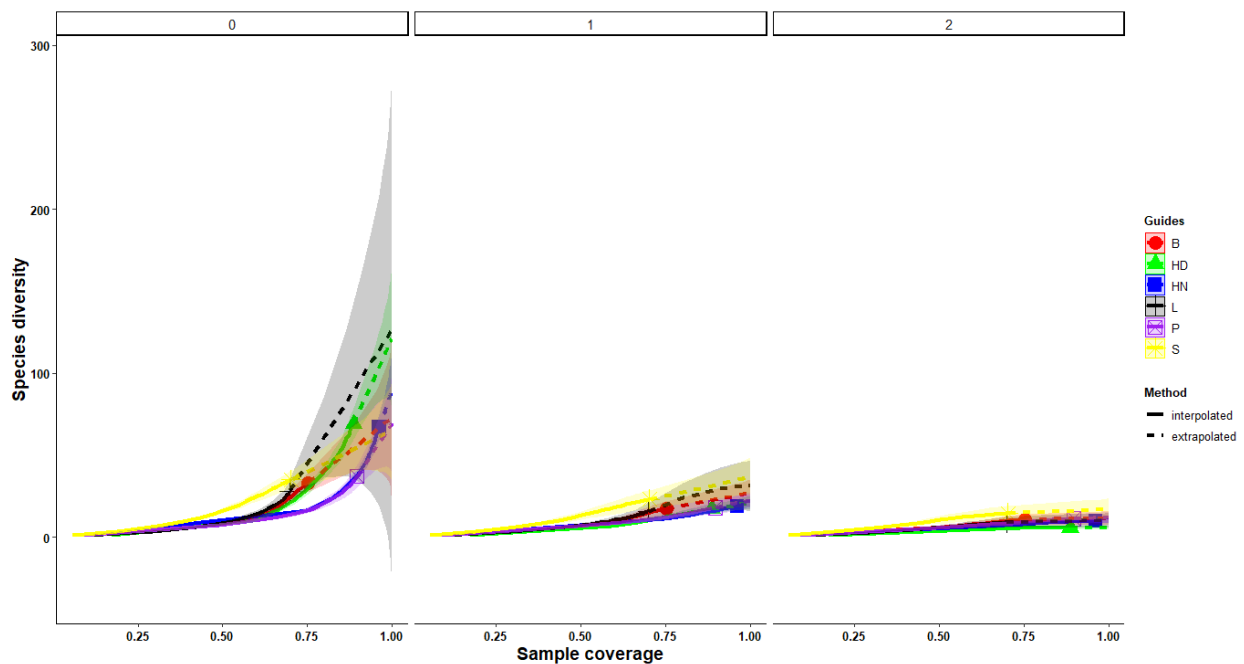

Fig. 4.7 Coverage- based rarefaction (solid lines) and extrapolation (dashed lines) sampling curves with 95 % confidence intervals (shaded areas, based on a bootstrap method with 200 replications) comparing spider species richness between the six sampling methods utilised at Khami World Heritage Site, Zimbabwe. Hill numbers of order

0, 1 and 2: q = 0 =: species richness; q= 1: Shannon diversity; q = 2: Inverse of Simpsons concentration. Abbreviations: B= beating sheets, HD= day hand collecting, HN= night hand collecting, L = litter sifting, P = pitfall trapping and S= sweep netting.

4.4.5 Optimization of the protocol

In order to sample 50% and 80% of all species with minimum effort, the mixed woodland and the riparian woodland required the least number of samples whereas mopane woodland required the most number of samples (Table 4.3). This result is also articulated by the randomized species accumulation curves that show steeper curves for the mixed and riparian woodland suggesting that the optimisation procedure was most effective in these biotopes with the mopane woodland with the less stepper curve (Fig. 4.8).
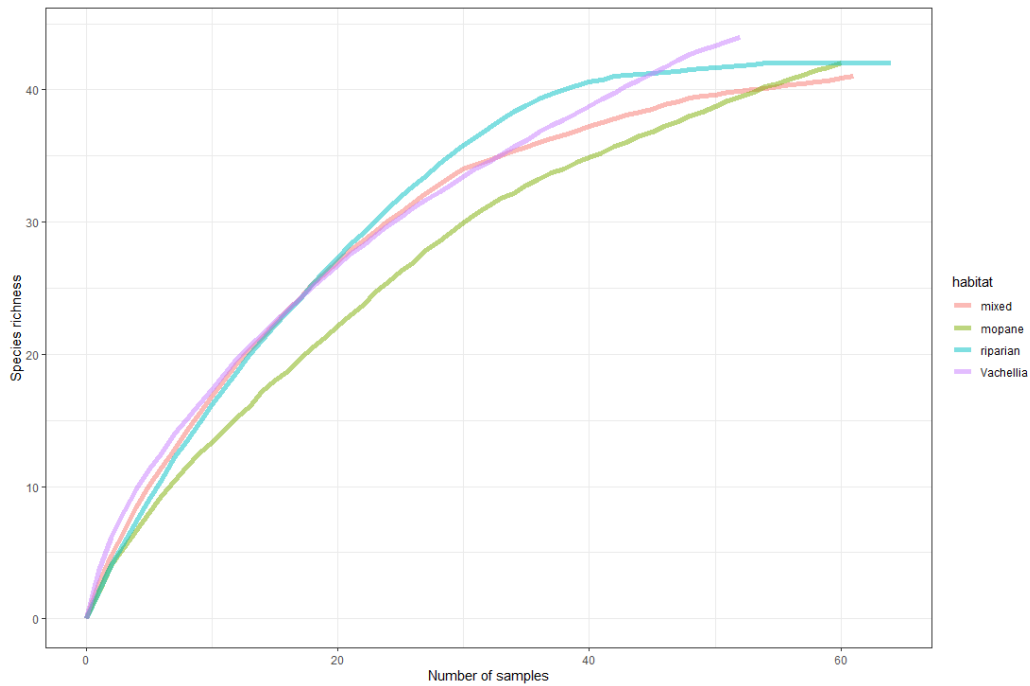
Fig. 4.8 Randomised spider species accumulation curves of the four sampled biotopes sampled during the summer season at Khami World Heritage Site, Zimbabwe.

In general, the most optimal protocols in all the four biotopes were biased towards night hand collecting, pitfall traps and sweep netting (Table 4.3), with night hand collecting sampling (19 species), pitfall traps (12 species) and sweep nets (6 species) exclusively. Despite the collection of exclusive species by beating sheets (n = 7), litter sifting (n = 7) and day hand collecting (n= 10) there were redundant in the optimal protocol for sampling at least 50 % of the species in the mixed woodland and riparian woodland, however in order to sample 80% of all species day hand collecting was required in the mixed woodland with the riparian woodland needing an extra

sample of beating sheets. Beating sheets and litter sifting were however necessary within the *Vachellia* woodland in order to capture at least 50% of all the species with the day hand collecting method being also required when capturing 80% of all species. The mopane woodland optimal protocol required a combination of all the sampling methods in order to capture at least 50% and 80% of all the spider species.

Table 4.3 Optimal combination of samples that maximises the amount of spider species caught per method for each biotope given 50% and 80% of sampling completeness for species sampled during the summer season at Khami World Heritage Site, Zimbabwe .

	Mixed woodland		Mopane woodland		Riparian woodland		*Vachellia* woodland	
Sampling completeness	50 %	80%	50%	80%	50%	80%	50 %	80%
Number of samples of the optimal protocol	15	32	22	48	16	32	16	36
Number of collected species	43	43	47	47	46	46	46	46
Beating sheets	0	0	2	6	0	1	3	4
Day hand collecting	0	2	9	13	0	0	0	6
Litter sifting	0	0	2	6	0	0	1	1
Night hand collecting	5	12	2	7	7	13	7	14
Pitfall traps	4	10	1	8	5	9	3	7
Sweep netting	6	8	6	8	4	9	2	4

4.5 DISCUSSION

4.5.1. Community composition

This study endeavored to utilise a standardized sampling protocol known as the South African National Survey of Arachnida sampling protocol to estimate spider diversity in Zimbabwe in a manner that may be comparable with studies in other regions of the world. Results indicate that the most diverse families collected from the Khami World Heritage Site are also widely distributed in South Africa, particularly within the savanna biome (Foord *et al.* 2011, Dippenaar-Schoeman *et al.* 2015). Similarly, the number of adults caught in this study (1386) is fairly comparable to the 1328 adult spiders collected from two protected areas in the South African Savannah biome over a period of 14 days in two sampling seasons (early and late summer) using a standardized protocol (Muelelwa *et al.* 2010) that was designed based on the Coddington protocol (Coddington *et al.* 1991). However, it was above the collected 966 adult spider individuals in the Ophathe Game Reserve in a single week in spring, by Haddad & Dippenaar-Schoeman (2015) using the SANSA sampling protocol. According to Duffey (1962) and Breymer (1966) adult individuals in most surveys rarely exceeds 40 % of the natural population in most invertebrates, in this study the number of adult individuals collected was 21% of the total collected individuals.

In contrast, species richness in this study (133) was slightly lower compared to studies that have also utilised standardized protocols. For example, Muelelwa *et al.* (2010) found 186 species using six sampling methods in two sampling seasons (early and late summer) and Haddad & Dippenaar-Schoeman (2015) recorded 197 species using seven sampling methods within a single sampling season. Compared to the above studies, spider species richness within Khami World Heritage Site can be regarded as relatively low, considering that the spider fauna of Khami World Heritage Site was collected within three sampling seasons which was more than utilised by previous studies. In addition in this study an additional method of night collecting was also used which previous studies did not implement due to logistical aspects (Haddad & Dippenaar-Schoeman 2015). However, according to Vedel *et al.* (2015) night collecting within the Neotropics has been considered not to be relevant as a method in sampling protocols as day sampling has been considered to be sufficient in obtaining spider estimates.

According to Chao and Lee (1992), an estimated coverage value should be at least 50%. The sample coverage for all sites, seasons and methods was higher than 94% when all spiders were considered and above 69% for adult spiders, and therefore can be considered as almost complete, with the majority of the species pool having being sampled. Thus, this survey may be considered as a representative snap-shot of the true diversity (Muelelwa *et al.* 2010). Unlike most species richness estimates that perform well with lower inventory completeness estimates (Toti *et al.* 2000), Chao1 richness estimate performs very well with high completeness values (Cardoso *et al.* 2008), such as the > 90% coverage obtained in this study, and has been used as reference for diversity studies of spiders (Coddington *et al.* 2009) in tropical communities. This result has also been consistent with similar tropical studies (Cardoso *et al.* 2008), despite the high singleton count for adult spiders (40%) obtained in the Mopane woodland that was way above the average for most surveys (32%; Coddington *et al.* 2009) which may be indicative of undersampling (Coddington *et al.* 2009). Nevertheless, by adding the juveniles, which have been shown to be useful surrogates that can be used to estimate species richness (Foord *et al.* 2013), the quality of the inventory was improved.

4.5.2 Biotope changes

The riparian woodland recorded the highest species richness compared to the other biotopes due to the characteristic gradients offered by the microhabitats found within riparian areas such as soil moisture, light availability, relative humidity, and temperatures of air and soil (Ramey & Richardson 2017). Studies have shown that the density of riparian spider along a stream is greatly affected by prey availability (i.e. emerging aquatic insects) (Marczak & Richardson 2007). For example, in a study conducted within the coastal temperate rainforests in south-western British Columbia the overall density of riparian spiders was significantly and positively correlated with aquatic insect abundances (Marczak & Richardson 2007). However, for the wolf spiders (Lycosidae) their abundance and diversity is relatively mostly correlated to relative soil moisture and low air temperature more than prey availability (Wenninger & Fagan 2000). Surprisingly, mopane woodland had the second highest species richness which was higher than that recorded in the mixed woodland, considering that mopane woodland is dominated by a single tree species (*Colophospermum mopane*) while the mixed woodland comprises a combination of different trees such as woody species, shrubs and herbs which would be most

likely to retain greater diversity due to the variable micro habitats found within the mixed woodland, despite the barren biotope (floristically) found within the mopane woodland, it still recorded higher species richness even compared to that of the *Vachellia* woodland.

4.5.3 Seasonality

Spider species richness at the Khami World Heritage Site was the highest during the summer sampling period, these results are consistent with previous studies conducted within the sub-tropical regions of South Africa (Whitmore *et al*. 2002, Foord *et al*. 2013). This may be attributed to the growth phenology of spiders (Haddad & Louw 2006), where most juveniles normally mature through the winter, are adults by early summer and die during late summer, and are overtaken by an outbreak of juveniles by early spring. However, according to Whitmore *et al*. (2002) a true biotope type effect can be obtained by sampling of the same biotope at different times of the year than simply in one season. When juvenile data was added however, the winter sampling period recorded a surprisingly higher abundance of spiders than specimens collected during spring. This could be attributed to the very heavy rainfall period that was experienced during spring, which might have reduced the individuals caught during this period, as spider activity has been recorded to be affected by rainfall and overwintering (e.g. Lensing *et al.* 2005, Queiroz & Gasnier 2017). Thus, as much as seasonality is an important consideration, when sampling spider assemblages, comparability of studies requires the utilisation of standardized and optimized protocols in inventory surveys that seek to characterise and/or monitor invertebrate species (Cardoso *et al.* 2007).

4.5.4 Sampling methods

Contrary to the results of Haddad & Dippenaar-Schoeman (2015), which also utilised the SANSA sampling protocol within savanna biotopes, night hand collecting recorded the highest abundance and richness in this study. However, beating did not perform as expected, as it is usually known to catch the highest diversity in many inventories (e.g. Muelelwa *et al.* 2010, Haddad & Dippenaar-Schoeman 2015). This difference could be attributed to two factors. Firstly, Haddad & Dippenaar-Schoeman (2015) only managed to do night collecting in one of the four selected biotopes, mainly due to safety concerns at night of encountering potentially dangerous animals. This was not an issue in our study, as the Khami World Heritage Site rarely has any dangerous wildlife. Secondly, the collecting was done for four hours by the lead

researcher, which is equivalent to a sampling effort of two individuals where each member collects for 2 hours (Haddad & Dippenaar-Schoeman 2015). In this study, four individuals collected for two hours in each selected biotope. Thus, the variability in the sampling effort is likely responsible for the differing results. However, when coverage based estimators were utilised sweeping, beatings and litter sifting were more efficient methods in estimating spider species richness, as a result when selecting sampling methods for inclusion in inventories beating, sweeping and litter sampling should also be considered. In support of the results in this study, elsewhere night sampling has been shown to record more species and specimens than day (Dobyns 1997, Cardoso *et al.* 2008) and hence samples collected at night and during the day have been suggested to be considered as different methods altogether (Cardoso 2009). With a contrary idea, Vedel *et al.* (2015) argued against this by stating that day time and night does not affect diversity, thus suggesting that collecting during one period will be sufficient to estimate spider diversity. However, coverage-based rarefaction curves and estimators in this study revealed a difference between day and night hand collecting, indicating that the methods are complimentary and together contribute towards a greater representation of the fauna than they would individually.

4.5.5 Optimization of the protocol

Results suggests that a protocol combining almost half of the samples being sweep netting and night hand collecting with some samples of pitfall traps would ensure the collection of at least 50% of the spider diversity within the mixed woodland (most optimal biotope that required the least number of samples) of the Khami World Heritage Site, in order to collect 80% of the spider fauna an additional method (day hand collecting) was also required. This result is also consistent with the standardization and optimization study on arthropod inventory inventories for Iberian spiders (Cardoso 2009) that also reported higher productivity in samples of both night hand collecting and night ground collection. Despite the belief, that pitfalls are not particularly an efficient sampling method (Cardoso *et al.* 2009, Patrick & Hansen 2013), they were selected by the algorithm in almost all the sampled biotopes, indicating the importance of the species that are normally caught by pitfall traps in the determination of overall spider diversity of Khami World Heritage Site. The biotope that did not perform very well was the Mopane woodland which required almost the requirement of the inclusion of all the sampling methods in order to obtain

50% of the species population, this seems coherent to the few microhabitats that are possibly found within the mopane woodland as it is dominated by one species the (*Colophospermum mopane*) with limited understory grasses (Mufandaedza 2002). The mixed woodland required the least number of samples in order to sample 50% of the spider species population, which may also be due to the vast diverse microhabitats that are found in this biotope that ranges from shrubs, woody tree species and herbs including a rich understory of grasses thereby enabling 50% of the population to be caught with minimum effort using only three selected methods.

Despite the importance of beating which has been reported to also include a wide range of morphospecies (i.e. especially when juveniles are included) (Haddad & Dippenaar-Schoeman 2015), it was not selected by the algorithm in the mixed woodland biotope for both the 50% and the 80% threshold of sampled spider species. It has been argued that such methods should simply be ignored in sampling protocols rather investing time to other productive methods (Cardoso *et al.* 2008). However, Muelelwa *et al.* (2010) argues that this might not be a great idea, but rather suggested that such methods should be utilised by experienced collectors. Nevertheless, as much as each method is essential in a sampling protocol to ensure that all microhabitats are sampled, optimisation ensures that at least a reasonable species diversity can be estimated using the minimum combination of sampling methods (Malumbres-Olarte *et al.* 2016), taking into consideration efforts to reduce costs but still retaining the quality of the data collected.

In conclusion, this study was a first attempt to utilise a standardised sampling protocol in Zimbabwe to estimate species richness within a protected area. Results of the study yielded 133 species with two possible new species, which will likely contribute in enhancing the knowledge of spider fauna of Zimbabwe. As the sampling was the first in Khami World Heritage Site new distribution records of species will also contribute to spider knowledge in Zimbabwe. An attempt to also optimize the sampling protocol was also done whereby suggestions of a combination of almost half of the sweep netting and night hand collecting and four samples of pitfall traps were recommended as necessary, and should be included in a sampling protocol that will ensure the collection of at least 50% of the spider species composition, especially when human resources and time are restricted. The SANSA sampling protocol is a productive standardised sampling protocol that can also be utilised to sample spider assemblages in other regions of the country, which will go a long way in ensuring the inclusion of neglected invertebrates in biodiversity assessments of Zimbabwe in the future.

4.6 References

Agosti, D., Majer, J., Alonso, E. & Schultz, T.R. (2000) *Ants: standard methods for measuring and monitoring biodiversity.* Smithsonian Institution Press.Washington and London, United States of America.

Arconsult. (1998) Management Plan for Khami World Heritage Site. Volume 1: Condition Survey. Volume 2: Management Plan. Harare: National Museums and Monuments of Zimbabwe, Harare, Zimbabwe.

Bird Life Zimbabwe. (2010) *Important bird areas in Zimbabwe: national status and trends* 2009. Genesis Advertising and Design, Harare, Zimbabwe.

Breymer, A. (1966) Relations between wandering spiders and other epigeic predatory. *Arthropoda Ekologia Polska*, **14**, 7–27.

Bulawayo Publicity Association. (2020) Bulawayo Publicity Association. KoBulawayo, online at https://www.bulawayopublicity.com, accessed on the 16th January 2020.

Cardoso, P. (2009) Standardization and optimization of arthropod inventories-the case of Iberian spiders. *Biodiversity and Conservation*, **18**, 3949–3962.

Cardoso, P., Crespo, L.C., Carvalho, R., Rufino, A.C. & Henriques, S.S. (2009). Ad-hoc vs. standardized and optimized arthropod diversity sampling. *Diversity*, **1**, 36–51.

Cardoso, P., Erwin, T.L., Borges, P.A.V. & New, T.R. (2011) The seven impediments in invertebrate conservation and how to overcome them. *Biological Conservation*, **144**, 1–11.

Cardoso, P., Rigal, F. & Carvalho, J.C. (2018) BAT: Biodiversity Assessment Tools. R package version 1.6.0, online at https://CRAN.R-project.org/package=BAT, accessed on 24th February 2018.

Cardoso, P., Scharff, N., Gaspar, C., Henriques, S.S., Carvalho, R., Castro, P.H., Schmidt, J.B., Silva, I., Szuts, T., Castro, A. & Crespo, L.C. (2008b) Rapid biodiversity assessment of spiders (Araneae) using semi-quantitative sampling : a case study in a Mediterranean forest. *Insect Conservation and Diversity*, **1**, 71–84.

Cardoso, P., Silva, I., De Oliveira, N.G. & Serrano, A.R. (2007) Seasonality of spiders (Araneae) in Mediterranean ecosystems and its implications in the optimum. *Ecological Entomology*, **32**, 516–526.

Chao, A. & Chiu, C.H. (2016) *Nonparametric estimation and comparison of species richness.* In eLS. John Wiley & Sons, Ltd., Chichester, United Kingdom.

Chao, A., Gotelli, N.J., Hsieh, T.C., Sander, E.L., Ma, K.H., Colwell, R.K. & Ellison, A. M. (2014) Rarefaction and extrapolation with Hill numbers: a framework for sampling and estimation in species diversity studies. *Ecological Monographs*, **84**, 45–67.

Chao, A. & Jost, L. (2012) Coverage-based rarefaction and extrapolation : standardizing samples by completeness rather than size. *Ecology*, **93**, 533–2547.

Chao, A. & Lee, S.M. (1992) Estimating the number of classes via sample coverage. *Journal of the American Statistical Association*, **87**, 210–217.

Chao, A. & Lin, C. (2012) Nonparametric lower bounds for species richness and shared species richness under sampling without replacement. *Biometrics*, **68**, 912–921.

Coddington, J.A., Agnarsson, I., Miller, J.A., Kuntner, M. & Hormiga, G. (2009) Undersampling bias : the null hypothesis for singleton species in tropical arthropod surveys. *Journal of Animal Ecology*, **78**, 573–584.

Coddington, J.A., Griswold, C.E., Davila, D.S., Peftaranda, E. & Larcher, S.F. (1991) *Designing and testing sampling protocols to estimate biodiversity in tropical ecosystems.* In Dudley, E.C (ed) the Unity of Evolutionary Biology: Proceedings of the Fourth International Congress of Systematic and Evolutionary Biology, Dioscorides Press, Portland, Oregon.

Coddington, J., Young, L.H. & Coyle, F.A. (1996) Estimating spider species richness in a southern appalachian cove hardwood forest. *Journal of Arachnology*, **24**, 111–128.

Colwell, R.K., Chao, A., Gotelli, N.J., Lin, S., Mao, C.X., Chazdon, R.L. & Longino, J. T. (2012) Models and estimators linking individual-based and sample-based rarefaction, extrapolation and comparison of assemblages. *Journal of Plant Ecology*, **5**, 3–21.

Colwell, R.K., Mao, C.M. & Chang, J. (2004) Interpolating, extrapolating and comparing incidence-based species accumulation curves. *Ecology*, **85**, 2717–2727.

Cousins, B. (1992) Managing communal rangeland in Zimbabwe: experinces and lessons. *Journal of Experimental Psychology: General*, **136**, 23–42.

Dippenaar-Schoeman, A.S., Haddad, C.R., Foord, S.H., Lyle, R., Lotz, L.N. & Marais, P. (2015) South African National Survey of Arachnida (SANSA): review of current knowledge, constraints and future needs for documenting spider diversity (Arachnida : Araneae). *Transactions of the Royal Society of South Africa*, **70**, 245–275.

Dobyns, J.R. (1997) Effects of sampling intensity on the collection of spider (Araneae) species and the estimation of species richness. *Enviromental Entomology*, **26**, 150–162.

Duffey, E. (1962) A population study of spiders in limestone grassland. *Journal of Animal Ecology*, **31**, 571–599.

Dunham, K. & Du Toit, A. (2013) Using citizen based survey data to determine densities of large mammals : A case study from Mana Pools National Park, Zimbabwe. *African Journal of Ecology*, **51**, 431–440.

Dunham, K., Van Der Westhuizen, E., Van Der Westhuizen, H.F. & Gandiwa, E. (2009) *Aerial survey of elephants and other large herbivores in Gonarezhou National Park (Zimbabwe) Zinave National Park (Mozambique) and surroundings.* Gonarezhou Conservation project : Parks and Wild Management Authrority and Frankfurt Zoological Society, Harare, Zimbabwe.

FitzPatrick, M.J. (2001) The spider fauna of Zimbabwe (Arachnida: Araneae). *Arnoldia Zimbabwe*, **10**, 177–187.

Foord, S.H., Dippenaar-Schoeman, A.S. & Haddad, C.R. (2011) The faunistic diversity of spiders (Arachnida : Araneae) of the Savanna Biome in South Africa. *Transactions of the Royal Society of South Africa*, **66**, 170–201.

Foord, S.H., Dippenaar-Schoeman, A.S. & Stam, E.M. (2013) Surrogates of spider diversity, leveraging the conservation of a poorly known group in the Savanna Biome of South Africa. *Biological Conservation*, **161**, 203–212.

Gordon, J.E. & Newton, A.C. (2006) Efficient floristic inventory for the assessment of tropical tree diversity: a comparative test of four alternative approaches. *Forest Ecology and Management*, **237**, 564–573.

Gotelli, N.J. & Colwell, R.K. (2001) Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. *Ecology Letters*, **4**, 379–391.

Gotelli, N.J. & Colwell, R. K. (2011) Estimating species richness. *Biological Diversity: Frontiers in Measurement and Assessment*, **12**, 39– 54.

Haddad, C.R. & Dippenaar-Schoeman, A.S. (2015) Diversity of non-acarine arachnids of the Ophathe Game Reserve, South Africa : Testing a rapid sampling protocol. *Koedoe*, **57**, 1–15.

Haddad, C.R. & Louw, S.M. (2006) Phenology, ethology and fecundity of *Heliophanus pistaciae* Wesolowska (Araneae: Salticidae), an agrobiont jumping spider in South African pistachio orchards. *African Plant Protection*, **12**, 1–11.

Hering, D., Moog, O., Sandin, L. & Verdonschot, P.F.M. (2004) Overview and application of the AQEM assessment system. *Hydrobiologica,* **516**, 1–20.

Hsieh, T.C. & Chao, A. (20170 Rarefaction and extrapolation: making fair comparison of abundance-sensitive phylogenetic diversity among multiple assemblages. *Systematic Biology*, **66**, 100–111.

Hubbard, P., Barret, R., Mukwende, T., Nyoni, L., Zihore, T., FitzPatrick, M. & Khumalo, S. (2017) Khami World Heritage Site: draft site managemnt plan 2013-2017, Harare, Zimbabwe.

Morrison, M., Block, W., Strickland, M., Collier, B. & Peterson, M. (2008) *Wildlife Study Design*. 2nd Editition. Springer, New York, United State of America.

Jammes, C., Valeix, S., Bourgarel, M., Murindagomo, F. & Fritz, H. (2009) Seasonal density estimates of common large herbivores in Hwange National Park. *African Journal of Ecology*, **47**, 804–808.

Jimenez-Valverde, A. & Lobo, J.M. (2006) Establishing reliable spider (Araneae , Araneidae and Thomisidae) assemblage sampling protocols : estimation of species richness, seasonal coverage and contribution of juvenile data to species richness and composition. *Acta Oecologica*, **30**, 21–32.

Jocque, R., Alderweireldt, M. & Dippenaar-Schoeman, A.S. (2013) *Biodiversity: an African perspective.* In Penney, D. (ed). Spider research in the 21st century: trends and perspectives. Siri Scientific Press, Manchester, England.

Lensing, J., Todd, S. & Wise, D. (2005) The impact of altered precipitation on spatial stratification and activity-densities of springtails (Collembola) and spiders (Araneae). *Ecological Entomology*, **30**, 194–200.

Cross, J., Perrin, P., & Little, D. (2010). The classification of native woodlands in Ireland and its application to native woodland management. Native Woodland Information Note 6, Woodlands of Ireland, Dublin.

Magurran, A.E. 2004. Measuring biological diversity. Blackwell Publishing, Malden, Massachusetts.

Malumbres-Olarte, J., Scharff, N., Pape, T., Coddington, J.A., Nikolaj, S., Thomas, P., Coddington, J.A. & Cardoso, P. (2016) Gauging megadiversity with optimized and standardized sampling protocols: a case for tropical forest spiders. *Ecology and Evolution*,

7, 494-506.

Mapaura, A. & Timberlake, J. (2004) *A checklists of Zimbabwean vascular plants.* Southern African Botanical diversity Network Report No.33. Sabonet, Pretoria and Harare.

Mapaure, I. (1994) The distribution of *Colophospermum mopane* (Leguminosae-Caesalpinioid eae) in Africa. *Kirkia*, **15**, 1–5.

Marczak, L.B. & Richardson, J.S. (2007) Spiders and subsidies : results from the riparian zone of a coastal temperate rainforest. *Journal of Animal Ecology*, **76**, 687–694.

Muelelwa, M.I., Foord, S.H. & Stam, E.M. (2010) Towards a standardized and optimized protocol for rapid biodiversity assessments : spider species richness and assemblage composition in two savanna vegetation types. *African Zoology,* **45**, 273–290.

Mufandaedza, E. (2002) *Tropical secondary forest management in Africa: reality and perspectives Zimbabwe Country Paper.* Workshop on tropical secondary forest management in Africa, Nairobi, Kenya

Mukwende, T., (2016) An Archaeological Study of the Zimbabwe Culture Capital of Khami, South-western Zimbabwe. Unpublished PhD thesis, University of Cape Town, South Africa.

Musarandega, L. (2015) *Butterfly species diversity and abundance at the Khami World Heritage Site.* MSc dissertation. Bindura State Univerity, Bindura, Zimbabwe.

Nhamo, A. (2019) Memory and Cultural Landscape at Khami World Heritage Site, Zimbabwe: an un-inherited past, conservation and management of Archaeological Sites, **21**, 140-142.

Patrick, L.B. & Hansen, A. (2013) Comparing ramp and pitfall traps for capturing wandering spiders. *Journal of Arachnology*, **41**, 404–406.

Queiroz, M. & Gasnier, T.R. (2017) Strong negative effect of diurnal rainfall on nocturnal activity of a wandering spider in Central Amazonia. *International Journal of Tropical Biology*, **65**, 1152–1160.

Ramey, T.L. & Richardson, J. (2017) Terrestrial invertebrates in the riparian zone: mechanisms underlying their unique diversity. *BioScience*, **67**, 808–819.

Sai, M., Utete, B., Chinoitezvi, E., Moyo, G.H. & Gandiwa, E. (2016) A survey of the abundance, population structure and distribution of Nile Crocodiles *Crocodylus niloticus* using day ground surveys in Sengwa wildlife research. *Herpetological Conservation and Biology*, **11**, 426–433.

Scharff, N., Coddington, J.A., Griswold, C.E., Hormiga, G. & Bjorn, P.D.P. (2003) When to quit? estimating spider species richness in a Northern European deciduous forest. *Journal of Arachnology*, **31**, 246–273.

Shumba EM. 2001. *Biodiversity and planning support programme Zimbabwe case study.* Paper prepared for an international workshop on 'Integration of Biodiversity in National Forestry Planning Programme', 13–16 August 2001, CIFOR Headquarters, Bogor, Indonesia

Silva, D.A. & Coddington, J.A. (1996) *Spiders of Pakitza (Madre de Dios, Peru): species richness and notes on community structure.* Edited by Wilson, D.E & Sandoval, A. The Biodiversity of Southern Peru Smithsonian Institution, Washington, DC, United States of America.

Teder, T., Moora, M., Roosaluste, E., Zobel, K., Meelis, P.A., Koljalg, U. & Zobel, M. (2007) Monitoring of biological diversity: a common-ground approach. *Essays*, **21**, 313–317.

Tipper, J.C. (1979) Rarefaction and rarefiction-the use and abuse of a method in paleoecology. *Paleobiology*, **5**, 423–434.

Toti, D.S., Coyle, F.A. & Miller, J.A. (2000) A structured inventory of appalachian grass bald and heath bald spider assemblages and a test of species richness estimator perfomance. *Journal of Arachnology*, **28**, 329–345.

Van Swaay, C., Regan, E., Ling, M., Bozhinovska, E., Fernandez, M., Marini-Filho, O. J., Huertas, B., Phon, C.K., Korosi, A., Meerman, J., Peer, G., Uehara-Prado, M., Safian, S., Sam, L., Shuey, J., Taron, D., Terblanche, R. & Underhill, L. (2015) *Guidelines for standardised global butterfly monitoring.* Group on Earth Observations Biodiversity Observation Network, Leipzig, Germany. GEO BON Technical Series 1.

Vedel, V., Cerdan, A., Martinez, Q., Baraloto, C., Petitclerc, F., Orivel, J. & Fortunel, C. (2015) Day-time vs. night-time sampling does not affect estimates of spider diversity across a land use gradient in the Neotropics. *Journal of Arachnolgy*, **43**, 413–416.

Wenninger, E.J. & Fagan, W.F. (2000) Effect of river flow manipulation on wolf spider assemblages at three desert riparian sites. *Journal of Arachnology*, **28**, 115–122.

Wesolowska, W. & Cumming, M.S. (2011) New species and records of jumping spiders (Araneae: Salticidae) from Sengwa Wildlife Research Area in Zimbabwe. *Journal of Afrotropical Zoology*, **7**, 75–104.

Whitmore, C., Slotow, R., Crouch, T.E. & Dippenaar-Schoeman, A.S. (2002) Diversity of spiders (Araneae) in a savanna reserve, northern province, South Africa. *Journal of Arachnology*, **30**, 344-356.

Zisadza-Gandiwa, P., Gandiwa, E., Jakarasi, J., Van Der Westhuizen, H. & Muvengwi, J. (2013) Abundance, distribution and population trends of Nile crocodile (*Crocodylus niloticus*) in Gonarezhou National park, Zimbabwe. *Water South Africa*, **39**, 165–170.

Appendix 4.1 Species composition of spiders collected in mixed woodland (MxW), mopane woodland (MW), riparian woodland (RW) and Vachellia woodland (VW) biotopes at the Khami World Heritage site during three seasons: summer, winter and spring in 2017.

Family/species	MxW	MW	RW	VW	Grand Total
Ammoxenidae	**8**	**34**	**4**		**46**
Ammoxenus daedalus Dippenaar & Meyer, 1980	7	33	4		44
Ammoxenus sp. 2	1	1			2
Araneidae	**657**	**758**	**575**	**554**	**2544**
Araneidae sp. 1	1		1		2
Araneidae sp. 2	3	13	10	2	28
Araneidae sp. 3	2	2	2		6
Araneidae sp. 4		1			1
Araneidae sp. 5		1		1	2
Araneidae sp. 6	13	40	10	6	69
Araneus apricus Karsch, 1884	3		5	2	10
Araneus holzapfelae Lessert, 1936				1	1
Araneus sp. 3	1				1
Araneus sp. 4		2		1	3
Araneus sp. 5	4		1	4	9
Araneus sp. 6	130	123	189	132	574
Argiope australis Walckenaer, 1805	8	5	3	8	24
Argiope sp. 2	4	6	3	2	15
Caerostris corticosa Pocock, 1902	1		1		2
Caerostris sexcuspidata Fabricius, 1793		4		1	5
Caerostris sp. 3	17	15	12	9	53
Cyclosa sp.		3			3
Cyphalonotus larvatus Simon, 1881		9	4	6	19
Cyphalonotus sp. 2	4	13	4	14	35
Cyrtophora citricola Forsskål, 1775	2	1			3
Cyrtophora sp. 2	10	8	5	13	36
Eustala anastera Walckenaer, 1841	2				2
Gasteracantha sp.				1	1
Hypsacantha sp.				10	10
Hypsosinga sp.	95	74	120	29	318
Isoxya sp.	1	1		1	3
Larinia sp. 1			1		1

	C1	C2	C3	C4	Total
Larinia sp. 2		3	1	2	6
Larinioides sp.			2	6	8
Lipocrea sp.			1		1
Nemoscolus cotti Lessert, 1933		2	3	2	7
Nemoscolus vigintipunctatus Simon, 1897			3		3
Nemoscolus sp. 3	1	6	8	8	23
Neoscona hirta C. L. Koch, 1844	5	3	1	15	24
Neoscona triangula mensamontella (Strand, 1907)	20	290	11	57	378
Neoscona sp. 3	173	89	85	112	459
Pararaneus spectator Karsch, 1885	36	22	22	81	161
Pararaneus sp. 2	1	1	6	1	9
Singa sp.				1	1
Trichonephila inaurata (Walckenaer, 1841)	105	9	55	22	191
Trichonephila senegalensis (Walckenaer, 1841)	15	11	6	4	36
Trichonephila sp. 3		1			1
Caponiidae			**1**	**1**	**2**
Caponia sp.			1	1	2
Cheiracanthiidae	**36**	**27**	**28**	**63**	**154**
Cheiracanthium furculatum (Karsch, 1879)	1	2	2	5	10
Cheiracanthium schenkeli (Caporiacco, 1949)	1	4		1	6
Cheiracanthium sp. 3	34	21	26	57	138
Clubionidae	**4**	**4**	**3**	**3**	**14**
Clubiona sp.	4	4	3	3	14
Corinnidae		**1**	**4**		**5**
Copa flavoplumosa Simon, 1886			2		2
Corinnomma semiglabrum (Simon, 1896)		1			1
Merenius alberti Lessert, 1923			2		2
Ctenidae	**2**	**3**	**1**	**1**	**7**
Afroneutria n. sp.			1		1
Ctenus sp.	2	3		1	6
Cyrtaucheniidae	**2**	**1**	**2**	**2**	**7**
Ancylotrypa sp.	2		2	1	5
Homostola sp.		1		1	2
Gnaphosidae	**32**	**36**	**25**	**52**	**145**
Asemesthes lineatus Purcell, 1908	1	1			2
Asemesthes paynteri Tucker, 1923	5	3	1	3	12
Asemesthes sp. 3	2	4	1	6	13

Taxon					
Camillina cordifera Tullgren, 1910			1		1
Camillina maun Platnick & Murphy, 1987	2	2	2	2	8
Camillina sp. 3		1			1
Drassodes solitarius Purcell, 1907			7		7
Drassodes splendens Tucker, 1923	2		3	3	8
Drassodes sp. 3		4		13	17
Eilica cincta Simon, 1893		3			3
Ibala minshullae (FitzPatrick & Sebata 2018).		3			3
Nomisia varia Tucker, 1923			1	1	2
Nomisia sp. 2				4	4
Pterotricha sp. 1				2	2
Pterotricha sp. 2	2			2	4
Scotophaeus relegatus Purcell, 1907			1		1
Theuma fusca Purcell, 1907			1	1	2
Theuma sp. 2		1	1		2
Trephopoda parvipalpa Tucker, 1923	1				1
Xerophaeus aurariarum Purcell, 1907			1		1
Xerophaeus vickermani Tucker, 1923				1	1
Xerophaeus sp. 3	12	3	2	5	22
Zelotes brennanorum FitzPatrick, 2007				2	2
Zelotes gooldi Purcell, 1907		1			1
Zelotes tuckeri Roewer, 1951	1				1
Zelotes sp. 4	4	10	3	6	23
Hersiliidae	**1**	**1**	**1**	**2**	**5**
Hersilia sericea Pocock, 1898				1	1
Hersilia sp. 2	1		1	1	3
Tyrotama sp.		1			1
Liocranidae			**1**		**1**
Rhaeboctesis secundus Tucker, 1920			1		1
Lycosidae	**30**	**51**	**58**	**55**	**194**
Allocosa lawrencei Roewer, 1951		1	1	1	3
Allocosa schoenlandi Pocock, 1900		2			2
Allocosa umtalica Purcell, 1903	9	10	5	3	27
Allocosa sp. 4	20	34	38	37	129
Amblyothele ecologica Russell-Smith, Jocqué & Alderweireldt, 2009		1			1
Pardosa crassipalpis Purcell, 1903		1	3	4	8
Pardosa lawrencei Roewer, 1959			3		3

Pardosa manubriata Simon, 1898	1	1		3	5
Pardosa umtalica Purcell, 1903				1	1
Pardosa sp. 5	1	1		3	5
Proevippa fascicularis Purcell, 1903			1	1	2
Schizocosa darlingi Pocock, 1898			1		1
Trabea sp.			1	4	5
Zenonina sp.			1		1
Mimetidae	**4**	**1**	**4**	**6**	**15**
Mimetidae sp. 1	1				1
Mimetidae sp. 2				5	5
Mimetus cornutus Lawrence, 1947	3	1	4	1	9
Nemesiidae	**1**				**1**
Lepthercus sp.	1				1
Oxyopidae	**386**	**194**	**239**	**262**	**1081**
Hamataliwa rostrifrons Lawrence, 1928	1	1	3	1	6
Hamataliwa rufocaligata Simon, 1898		1			1
Hamataliwa tuberculata Chamberlin, 1925	1				1
Hamataliwa sp. 4	4	1		3	8
Oxyopes angulitarsus Lessert, 1915			1		1
Oxyopes bothai Lessert, 1915	1		2		3
Oxyopes chapini Lessert, 1927	1		1		2
Oxyopes dumonti Vinson, 1863	2			2	4
Oxyopes falconeri Lessert, 1915		1		2	3
Oxyopes flavipalpis Lucas, 1858	2	2			4
Oxyopes hoggi Lessert, 1915	12	4	6	4	26
Oxyopes jacksoni Lessert, 1915			3		3
Oxyopes russoi Caporiacco, 1940	5	1			6
Oxyopes sp. 10	5	4	6	1	16
Oxyopes sp. 11	2			1	3
Oxyopes sp. 12		3	1		4
Oxyopes sp. 13				2	2
Oxyopes sp. 14	324	159	196	230	909
Peucetia transvaalica Simon, 1896	1		1		2
Peucetia sp. 2	25	17	19	16	77
Philodromidae	**17**	**23**	**25**	**29**	**94**
Philodromidae sp. 1				1	1
Philodromus sp.		5	7	14	26

Thanatus dorsilineatus Jézéquel, 1964	1		1		2
Tibellus minor Lessert, 1919	3	3	5	4	15
Tibellus sp. 2	13	15	12	10	50
Pisauridae	**69**	**34**	**70**	**95**	**268**
Charminus sp.				1	1
Maypacius sp.	69	34	70	94	267
Salticidae	**245**	**262**	**222**	**151**	**880**
Evarcha prosimilis Wesolowska & Cumming, 2008	2	2	1	3	8
Evarcha zimbabwensis Wesolowska & Cumming, 2008	4	15	6	4	29
Heliophanus lesserti Wesolowska, 1986	27	3	26	5	61
Heliophanus orchestra Simon, 1886		2			2
Heliophanus sp. 3		1			1
Heliophanus sp. 4	3		1		4
Heliophanus sp. 5	22	13	13	5	53
Hyllus argyrotoxus Simon, 1902		1			1
Hyllus brevitarsis Simon, 1902	7	3	2	3	15
Langona bethae Wesolowska & Cumming, 2011	5	2	2	4	13
Pellenes bulawayoensis Wesolowska, 2000	2	10	6		18
Pellenes tharinae Wesolowska, 2006		9			9
Pellenes sp. 3	15	22	8	6	51
Pseudicius squamatus Haddad & Wesolowska, 2013			2		2
Stenaelurillus guttiger Simon, 1901	3	25	13	8	49
Stenaelurillus termitophagus (Wesołowska & Cumming, 1999)	48	27	50	32	157
Stenaelurillus sp. 3	1	1			2
Thyene inflata Gerstäcker, 1873)	1	1	2	2	6
Thyene natalii Peckham & Peckham, 1903	3	1	6	1	11
Thyene thyeniodes (Lessert, 1925)	1	1	1	1	4
Thyene sp. 4	93	115	80	73	361
Thyenula sp.	8	8	3	4	23
Selenopidae	**1**	**5**	**5**	**11**	**22**
Selenops sp.	1	5	5	11	22
Sparassidae	**4**	**2**	**4**	**2**	**12**
Olios correvoni Lessert, 1921		1	2		3
Olios sjostedti Lessert, 1921				1	1
Olios sp. 3	4	1	2	1	8
Tetragnathidae	**14**	**35**	**42**	**55**	**146**
Meta sp.		1			1

Tetragnatha demissa L. Koch, 1872	1	4		27	32
Tetragnatha sp. 2	13	29	42	28	112
Tetragnathidae sp.1		1			1
Theridiidae	**167**	**128**	**87**	**42**	**424**
Argyrodes zonatus Walckenaer, 1841	47	54	5	4	110
Argyrodes sp. 2	56	26	11	1	93
Enoplognatha molesta O. Pickard-Cambridge, 1904		1	1		2
Enoplognatha sp. 2		1			1
Episinus marignaci Lessert, 1933	1			1	2
Episinus sp. 2		3	3	1	7
Euryopis episinoides Walckenaer, 1847	10	1	13	8	32
Euryopis sp. 2	5	2			7
Histagonia sp.		2			2
Latrodectus sp.		1	1		2
Rhomphaea sp.	14	1	4	2	21
Theridiidae sp.3		1		1	2
Theridion sp. 1			1		1
Theridion sp. 2	35	34	48	24	141
Theridula sp.		1			1
Thomisidae	**98**	**143**	**86**	**71**	**398**
Ansiae tuckeri Lessert, 1919		1			1
Ansiea sp. 2		1	1		2
Camaricus sp.			1		1
Cynathea sp.	1				1
Diaea puncta Karsch, 1884	3	1	2	1	7
Diaea sp. 2	10	5	9	8	32
Dieta sp.	1				1
Firmicus sp. 1				1	1
Firmicus sp. 2	1				1
Holopelus sp.		3			3
Misumena sp.	1		1		2
Misumenops rubrodecoratus Millot, 1942		3		2	5
Misumenops sp. 2	2	7		5	14
Monaeses austrinus Simon, 1910			1		1
Monaeses gibbus Dippenaar-Schoeman, 1984	1		1		2
Monaeses quadrituberculus Lawrence, 1927		1			1
Monaeses sp. 4	51	57	25	24	157

Species					
Oxytate concolor Caporiacco, 1947		1			1
Oxytate sp. 2				1	1
Pactactes sp.				1	1
Parabomis sp.	2	7			9
Runcinia flavida Simon, 1881	5	17	8	4	34
Simorcus cotti Lessert, 1936		2			2
Simorcus sp. 2		1			1
Synema sp. 1	1				1
Synema sp. 2	12	4	8	3	27
Thomisidae sp. 1	1				1
Thomisus congoensis Comellini, 1957	1		1		2
Thomisus granulatus Karsch, 1880	1	6	5	3	15
Thomisus scrupeus Simon, 1886			2		2
Thomisus stenningi Pocock, 1900				1	1
Thomisus sp. 5	1	20	18	6	45
Tmarus comellinii Garcia-Neto, 1989	2	3			5
Tmarus sp. 2	3	3	2	5	13
Xysticus havilandi Lawrence, 1942				3	3
Xysticus sp. 2				2	2
Trachelidae			1		**1**
Fuchiba acquilonia Haddad & Lyle, 2008			1		1
Uloboridae	**2**				**2**
Uloborus sp.	2				2
Zodariidae	**14**	**7**	**10**	**9**	**40**
Capheris abrupta Jocqué, 2009	9		1	6	16
Diores magicus Jocqué & Dippenaar-Schoeman, 1992	1	1	5		7
Diores n.sp	1	4	1	2	8
Diores sp. 3	1	2	2		5
Diores sp. 4	1		1		2
Hermippus tenebrosus Jocqué, 1986	1			1	2
Grand Total	**1794**	**1750**	**1498**	**1466**	**6508**

Chapter 5

General Discussion, Concluding remarks and recommendations

5.1 General discussion

As an endeavour to sustainably utilise rangelands that are prone to land degradation due to poor management practises, several grazing system have been utilised within African countries over the past decades (Rampai 2017). Currently, three grazing systems are in use in Zimbabwe, of which the communal grazing system is by far the most damaging system when compared to small-scale farming systems and large scale-systems (Gusha *et al.* 2017). The continuous consumption of plants without a rest period leads to stress in plants, which may eventually lead to death due to the lack of fences to restrict foraging. In order to improve utilisation of rangelands, ranch managers and local Zimbabwean communities have since advocated the utilisation of livestock with increased stocking rates (Savory 1983), also known as holistic planned grazing, the latter which has been the main focus of this project. This study significantly advances knowledge of spider biodiversity and ecology in a largely unstudied biogeographically area. It makes a good contribution to functional knowledge of spiders and their ecology, in particular with reference to how changes in numbers over space and time occur within spider assemblages as a result of holistic planned grazing, which may then be used to assess and predict pasture health. This is an important topic that supports the idea of a unified sampling protocol for Zimbabwe for use in grazing systems. The study was also dedicated to rigorous testing of sampling protocols, which will allow for increased future sampling efficiencies as it varies across habitats. In summary, the study shows that short duration kraaling changes the spider community composition both inside and around previously kraaled inclusions and their control sites through changing vegetation patterns and that a limited set of standardized sampling method could deliver valuable faunistic and potentially monitoring data for nature conservation in a protected area.

5.1.1 Vegetation structure changes over sampling periods

Disturbances such as grazing have been shown to affect ecosystems (Diaz *et al.* 2007) by reducing vegetation complexity and structure (Malumbres-Olarte *et al.* 2013), and also through indirectly altering micro-climatic conditions at the ground level (Bell *et al.* 2001). For example,

grazing results in biotopes that have more open spaces and usually have less structurally complex microhabitats as a result of the grazing and trampling (Huruba *et al.* 2018). The model that best explained change in mean grass height (cm) and percentage grass cover around previously kraaled inclusions and their control sites, was that which included time since kraal removal compared to the other models (Table 2.2), with mean grass height (cm) decreasing continously from the period since cattle occupation until 6 months after kraal removal, with 10 months after kraal removal recording the highest mean grass height (cm) and percentage grass cover. This result was mainly attributed to seasonality effects that were also affected by rainfall patterns, as the period of 10 months since kraal removal also coincided with the summer period that also receives the highest rainfall in Zimbabwe. This, in turn, contributed to the growth surge of grasses that was experienced in that sampling period. However, the highest mean grass height (cm) inside previously kraaled inclusions was observed at 10 months after kraal removal most probably as a result of nutrient deposition within the previously kraaled sites that also promoted an upsurge growth of the vegetation structure.

5.1.2 Effect of previously kraaled inclusions on spider assemblages

The influence of short duration kraaling has been well documented, with previous work focusing on the importance of kraals as nutrient reserves that can be used in agriculture (Kangalawe *et al.* 2008), utilisation of kraals to improve plant diversity (Sibanda *et al.* 2016, Huruba *et al.* 2018), soil aeration, water penetration and seed germination (Neely & Butterfield 2004), utilisation of previously kraaled sites by wildlife (Huruba *et al.* 2017), and influence of short duration grazing on invertebrates (Chikorowondo *et al.* 2018, Muvengwi *et al.* 2018). In this study, the response of spider fauna to short duration kraaling was shown to vary at different spatial scales. Around previously kraaled sites only the web builder's genera richness (Table 2.3) was significantly and negatively correlated to short duration kraaling, while inside previously kraaled sites only the ground-dwelling abundance was significantly lower inside the kraals as compared to the surrounding vegetation (Fig. 3.5). The results obtained around previously kraaled sites may be attributed to the biology of the spiders caught by the pitfall traps and sweep nets. Foliage-dwellers normally caught by sweep nets depend mostly on vegetation cover. However, pitfall traps that capture mostly ground-active invertebrates are normally not associated with vegetation cover (Warui *et al.* 2005). Therefore, vegetation associated spiders and web builders would

respond better with regards to distubances such as short duration kraaling as compared to the ground dwelling spiders caught in this study, which did not show any differences between the treatments between spider genera richness. However, the lower spider abundance obtained inside previously kraaled sites was most likely as a result of the trampling action of the livestock during cattle occupation, which in turn reduced the microhabitat required by the ground dwelling spiders when resting during the day (Haddad *et al.* 2013). It has been suggested that spiders mostly oviposit or overwinter in ungrazed areas to avoid unfavourable climatic conditions, thus implying higher abundances in such areas (De Keer & Maelfait 1988), as observed in the surrounding vegetation of the previously kraaled inclusions.

5.1.3 Effect of geographic distances around previously kraaled inclusions on spider fauna

Spider assemblages have been reported to vary with geographic distance (Whittaker 1956, 1960, Carvalho *et al.* 2011), due to the lack of similarity that occurs between two distances. This may be due to several factors that include high dispersal ability of spider assemblages (Jimenez-Valverde *et al.* 2010, Rodriguez-Artigus *et al*. 2017) and vegetation complexity (Jimenez-Valverde *et al.* 2010), amongst other factors. In this study only the ground dwelling genera composition responded to distance from kraal (Table 2.4). This was most probably as a result of the lower dispersal abilty of the ground wanderers, which was different to that of plant dwellers and the web builders, whose dispersal ability is dependent on plant vegetation. However, the lack of effect of distance from kraal in all the functional groups' genera richness and abundance was not considered to be a result of low sampling effort, as the latter has been indicated to reduce similarity between assemblages because of rare species being omitted in inventories. Owing to the high sampling coverage that was above 98% in both the unkraaled and kraaled sites, insufficient sampling was not considered as a contributing factor. Rather, the results reflect the true lack of geographic distance effects on both the genera richness and abundance in all the functional groups.

5.1.4 Time since kraal removal effects

The contrasting lifecycles in different spider species usually result in variable peaks of species abundances throughout the year. The latter has been considered to be one of the main causes of seasonal differences observed in spider diversity (Muelelwa *et al*. 2010), which was also a contributory factor in this study too. Around previously kraaled sites, mean grass height (cm)

(Fig. 2.9 and Table 2.2), percentage grass cover (Fig. 2.10 and Table 2.2), ground dwelling genera richness and abundance (Table 2.3), web builders abundance and genera richness (Table 2.3), and genera composition of all the three functional groups (Table 2.4) varied with time since kraal removal. Only the plant dwelling abundance and genera richness (Table 2.3) did not respond to time since kraal removal (Table 2.4). Despite the significant effect of time since kraal removal on the genera composition of plant wanderers, they showed less clear distinction of spider assemblages when compared to compositions of ground dwellers and web builder's species (Fig. 2.12). Mean grass height (cm) (Fig. 3.3 and Table 3.3) and ground wanderers' abundance (Fig. 3.5) and genera richness (Fig. 3.4) also varied with time since kraal removal inside the kraals.

The temporal scale change of mean grass height (cm) and percentage grass cover around previously kraaled sites was mostly probably as a result of both the issue of seasonality and rainfall patterns at Debshan Ranch. The mean grass height (cm) was highest 10 months after kraal removal, coinciding with the summer period that receives high rainfall, which would most likely have contributed to the growth surge of the vegetation. However, when the seasonality factor was excluded from the design, previously kraaled sites still recorded the highest mean grass height (cm) at 10 months since kraal removal. This result therefore is mostly likely to be as a result of the cow dung deposits and urine in previously kraaled inclusion that promoted growth of the grasses with time since kraal removal. This study suggests that seasonality masked the effects of time since kraal removal when sampling was done in the six sampling periods around previously kraaled sites. However, when sampling was done within one season inside previously kraaled sites a better reflection of the kraaling effect on spider fauna was obtained.

5.1.5 Spider diversity at Khami World Heritage Site

The lack of information and species distribution of spider fauna is a great hindrance to conservation management (Oliveira *et al.* 2016). Thus, in order to complete the initial stage of developing plans, implementation of inventories or at least partial inventories is required (Whitmore *et al.* 2002). Chapter four presented spider fauna of a virgin protected area, the Khami World Heritage Site, where four biotopes (mixed bushveld, mopane woodland, riparian woodland and the *Vachellia* woodland) were sampled using six sampling methods (sweep netting, beating, litter sifting, night hand collecting and day hand collecting and pitfall traps). A

checklist was also included for the Khami World Heritage Site (Appendix 4.1), which contributes to the knowledge of spider fauna of Zimbabwe through the discovery of two possibly new species that still need to be described, as well as contributing to new distribution records of spider assemblages. When compared to other regional studies that also utilised standardised sampling protocols, e.g. work by Haddad and Dippenaar (2015) and Muelelwa *et al.* (2010), the spider species richness recorded at the Khami World Heritage Site can be regarded as relatively very low, particularly considering that sampling was done in three sampling seasons. This was more than the aforementioned studies, and also because night collecting was comprehensively utilized, which was not the case in the other studies. Apart from estimating species richness, the study also sought to optimize the South African National survey of Arachnida sampling protocol, by suggesting the combination of sampling methods that were most likely sample at least 50% of the species richness with minimum effort. This is mostly useful when human resources and time is restricted. Overall, the results suggested a combination of almost half of the sweeping and night hand collecting, with four samples of pitfall traps, carried out in the mixed woodland which was the biotope that required the least number of samples, that would result in the collection of least 50% of the species assemblage, with an additional two samples of day hand collecting per biotope that would result in the collection of at least 80% of the spider species composition.

5.2 Conclusions and future research

As with previous research in Africa and worldwide, this study revealed that grazing affects spider assemblages due to changes in vegetation structure modified by holistic planned grazing and trampling. The study endorses web builders as fairly good indicators of change and quality (Gibson *et al.* 1992, Robertson *et al.* 2016). They were sensitive to short duration kraaling around previously kraaled sites, but to mean grass height (cm), time since kraal removal and leaf litter complexity at genera level, with ground dwelling species composition responding to distance from kraal. The latter was attributed to low dispersal ability by the ground wanderers, thus leading to low similarity between the distances. The change in mean grass height (cm) and time since kraal removal was attributed to the seasonality effects, which are also affected by changes in rainfall patterns that affected vegetation growth, as well as spider diversity, genera richness and abundance, owing to their different life cycles.

Since the ground dwelling spiders explained the greatest variation between the utilised functional groups around previously kraaled inclusions they were therefore an adequate functional group that could be utilised to also assess the response of spider assemblages to holistic planned grazing inside the inclusions. Furthermore 14 of the 25 genera of ground dwelling responded contrastingly to vegetation structure variables, hence aiding in indicating their level of sensitivity to disturbance. The lack of significant differences between the inside and the outside of the kraals were attributed to the dispersal ability of the ground wanderers, especially after disturbance (Mrzljak & Wiegleb 2000; Weyman *et al.* 2002), as well as the high number of generalist spiders that are normally found in open and disturbed areas (Mallis & Hurd 2005), which was a possible indication of the long grazing history of the ranch. However, in order to ascertain such results, further research in the future is required that will determine the presence of such generalists in all the woodlands located within the Ranch. For example, by sampling in the Mopane woodlands, the notion that generalists were located only within the miombo woodlands that were selected in this study, for the purposes of homogeneity would be eliminated.

However, one of the major problems with utilising spider diversity at a level of the guild is the lack of sensitivity of each individual species that might not be detected, as there is an assumption that all species within the same functional groups will most likely respond in a similar manner to anthropogenic changes (Lawton *et al.* 1988), which might not be true. Future work should therefore seek to identify species-specific changes that may occur as a result of holistic planned grazing, which was not possible in this study because of the small data sets of the adult individuals. Therefore, utilisation of genera richness enabled the addition of juveniles, which have been shown to be useful surrogates of species richness (Foord *et al.* 2013). Overall, the results of this study indicate that the response of spider assemblages is not always the same. For example, around previously kraaled sites when analysis was done using genera richness and abundance, the web builder's genera richness and abundance responded to short duration kraaling whereas when similar analysis was done at the genera composition resolution there was no significant response to short duration by the web builder's genera composition. This indicates that the different levels of resolution of spider assemblages are complimentary, and together contribute towards a better understanding of spider responses to anthropogenic changes than they would individually.

5.3 Recommendations

- Currently in Zimbabwe, conservation approaches take into account vertebrates and plants, with the belief that protection efforts that are given to both vertebrates and the plants will profit arthropods such as spiders as well. However, this method usually does not provide for the protection of threatened and rare spider species (Lovell *et al.* 2009). Application of protection efforts for invertebrates becomes more complicated, as spiders have been reported to respond differently to disturbance, with some species declining while others increase. Ranch managers may decide to protect spider communities by targeting species of concern, whereby species-specific management plans are implemented. However, such information requires more than a single inventory, but rather long-term monitoring of spider assemblages within a region, in order to come up with a complete database of all possible spiders in each biotope, which will later contribute to the recognition of the rare, endemic and threatened species found within that region. In order to prioritise protection efforts for spiders, individual species need to be ranked according to the impacts of grazing and trampling of the holistic management approach, with protection efforts also being distributed accordingly.

- This study was conducted in a mixed cattle and wildlife ranch within a large-scale commercial scheme that has herds of approximately 350 livestock (Huruba *et al.* 2018), however a similar comparative study can also be done in other parts of Zimbabwe, such as the Dimbangombe Ranch that is managed at communal level, which will also have different herd sizes, due to the flexibility of the framework of holistic management practises. This will advance the information on the response of arachnids to holistic planned grazing, as well as improve distribution records of spider assemblages within Zimbabwe.

- Results in this study suggests that a least a minimum of not less than 6 months resting period is recommended around inclusions of previously kraaled sites while a period of not less than 4 months is essential inside the inclusions, in order to ensure an increase in the spider diversity that was initially lowered by short duration kraaling. Similarly, short duration kraaling utilised at Debshan, with a

short retention period of 7 days within the kraals, has also been shown to be suitable for plant re-establishment within a period of less than ten months after kraal removal. Additionally, long-term monitoring of previously kraaled sites that go beyond 10 months since kraal removal is recommended, which will enable the identification of the time necessary for the initial species assemblage to re-establish following short duration kraaling effects.

- The retention of the kraaled and unkraaled sites within Debshan Ranch is recommended, as it enables the maintenance of a mosaic of different vegetation heights that range from tall and short grasses. This is essential for the retention of diverse functional groups, thus maximizing overall spider diversity.

- As spider fauna need to also be included in future biodiversity reports in Zimbabwe, there is a need for further surveys that should be conducted in other protected areas of the country that do not have any distribution records of arachnids. Preferably, these surveys can also be done using a standardised sampling protocol, such as the SANSA protocol, which will enhance comparisons of such surveys. Since this study only managed to sample once in each of the selected seasons in the Khami World Heritage Site, long term monitoring of each biotope is recommended in the future. This will aid in understanding the biology and phenology of the different species of arachnids within the protected area, thereby enhancing the status of spider knowledge within Zimbabwe.