Soil biota as bioindicators of levels of erosion and fire disturbances in Afromontane grassland areas within the Golden Gate Highlands

By

Sylvia Shalomé van der Merwe

Submitted in fulfilment of the requirements in respect of the Doctoral Degree in Entomology in the Department of Zoology and Entomology in the Faculty of Natural and Agricultural Sciences at the University of the Free State

Promoter: Dr Vaughn R. Swart

Co-promoters: Dr Emile Bredenhand and Prof Charles R. Haddad

Department of Zoology and Entomology

Faculty of Natural and Agricultural Sciences

January 2020

Declaration

I, Sylvia Shalomé van der Merwe, declare that the thesis or publishable manuscripts that I herewith submit for the Doctoral Degree in Entomology at the University of the Free State, is my independent work, and that I have not previously submitted it for a qualification at another institution of higher education.

Sylvia Shalomé van der Merwe

Table of Contents

i. Abstract

Soil erosion and wildfires are serious problems throughout the terrestrial ecosystems of the world. The Golden Gate Highlands National Park (GGHNP) of South Africa experiences high incidences of erosion and wildfires due to its orographic nature and expansive grassland habitats. Various conservation strategies are employed by Park Management to lessen the effects of these environmental factors, including instating regular prescribed burn regimes for grasslands and construction of rehabilitation structures in eroded localities. While the effects of fire and erosion are investigated across a variety of habitats and fauna, little attention is given to their effects on soil-dwelling arthropods. The overall objective was to determine the state of soil-dwelling arthropods across eroded and currently rehabilitated localities, as well as their responses to fire regimes, in the GGHNP.

Chapter 2 aimed at determining the impact of prescribed burning on soil-dwelling arthropods in an Afromontane grassland habitat, by comparing assemblage patterns and responses of species richness and diversity between a single burnt and non-burnt locality. Soil arthropod assemblages were more species rich and abundant in the burnt site, with a higher number of species only observed in the burnt site overall. The study hints at fires creating a preferable niche for soil arthropods adapted to frequent fires in the fire-prone landscape.

Chapter 3 attempted to identify possible indicators of erosion in the GGHNP, and to determine differences in soil-dwelling arthropod assemblages found in non-rehabilitated and rehabilitated eroded sites. IndVal results indicated a single strong indicator species, the mite *Speleorchestes meyerae* Theron and Ryke, 1969, relevant to non-rehabilitated sites, suggesting that soil arthropods show potential use in grading changes in soils of the GGHNP. Statistical modelling identified phosphorus as having a significant negative correlation on species richness in both rehabilitated and non-rehabilitated eroded sites. These results form a basis for future investigations of erosion in the GGHNP, while also indicating that soil mineralogy in conjunction with soil arthropod richness may provide sufficient usability in monitoring strategies of erosion.

In Chapter 4, the changes in soil arthropod assemblages in areas with implemented erosion rehabilitation, namely rock-wall gabions near dirt roads in the GGHNP, was examined. Results showed that arthropod species richness significantly interacted with increased sediment build-up in the rehabilitated sites as well as with sub-site position around the gabions. The findings of this study suggests that, if compared to an older rehabilitated site, rock-wall gabions could have an indirect effect on soil-dwelling arthropods, possibly through the resulting sediment build-up over the long-term.

In Chapter 5, the renewal of deteriorated erosion rehabilitation structures provided a unique opportunity to study the effect of major restructuring of a site and subsequent implementation of alternative rehabilitation structures on soil-dwelling arthropods. Both soil arthropod species richness and diversity was higher after the restructuring of the site, possibly due to reallocation of species from the soil surrounding the flattened area. The results suggest that restructuring caused no significant changes on soil arthropod assemblages in this single site over a nine-month period, but is not conclusive as to the effects that a major disturbance, such as land reformation and renewed rehabilitation implementation, may have over the long-term.

Fire-treated and non-rehabilitated eroded sites show a surprising attribute in that these habitats support a greater soil arthropod species richness and abundance than natural sites. These sites show potential as important environments that act as unique niches in the GGHNP, vital to supporting soil arthropod diversity in the soil environments of the park. Interestingly, themes discussed highlight the importance of fire and erosion in the GGHNP as natural ecosystem phenomena, and the association of soil arthropods to these areas.

Keywords: Conservation; Erosion; Erosion Rehabilitation; Fire; Golden Gate Highlands National Park; Grassland; Indicators; Mineralogy; Soil Arthropods; Soil Biota

ii. Acknowledgements

- Thank you to my supervisors, Dr Vaughn Swart, Dr Emile Bredenhand and Prof Charles Haddad for all their support, supervision and input during the course of this project.
- This work was financially supported by the Afromontane Research Unit (ARU), University of the Free State (Grantholder: Emile Bredenhand), and the National Research Foundation (NRF) (Grant Number: SFH180528335602).
- Thank you to South African National Parks (SANParks) Scientific Services for access to the Golden Gate Highlands National Park (Permit no.: VDMES1387, Appendix 1), as well as the University of the Free State's Ethical Committee for permission to conduct this project (UFS-HSD2017/0074, Appendix 2).
- Thank you to Mr Jan-Andries Neetling, Dr Charlene Janion-Scheepers, Dr Mikhail Potapov and Miss Hannelene Badenhorst for assistance in identification of arthropod type material. A big thank you to Dr Daryl Codron for providing advice and assistance with statistical analyses.
- Thank you to the support staff at the Department of Zoology and Entomology at the University of the Free State, Bloemfontein Campus, for administrative and technical support.
- Thank you to my colleagues, friends and family for having faith in my capabilities.
- A special thank you to Mr Jason Lee Botham, love of my life and best friend, for assisting in many aspects of field sampling and providing laboratory support.
- Last, but definitely not least, the Lord is good, and I thank Him for carrying me through it all.

Chapter 1:

General Introduction

1. General Introduction

The Golden Gate Highlands National Park (GGHNP) in the eastern Free State Province is affected by many factors of disturbance, exacerbated by increased tourism numbers and vehicular traffic through the park (SANParks, 2013). The park is more well known by tourists for its beautiful sandstone cliffs and historically for its rock paintings. However, in recent years, scientists have started investigating terrains in the GGHNP from an ecological perspective, finding mechanisms to increase conservation success, and ultimately promote sustainability in the area. This has given rise to multi-disciplinary research projects in the area, with scientists from both an ecological and economical field applying knowledge to promote protective practices while engaging the local community and attempting to alleviate the pressures that the people endure (SANParks personnel, personal communication, 2019[1]). A key topic that is not well known in this area is the benefits of promoting soil health through establishment of pioneer plants and through the conservation of beneficial fauna. As many types of disturbances affect the GGHNP, it is imperative that investigations into beneficial fauna, in this case soil fauna, be carried out to identify the effect and possible indicators of said disturbances.

Controlled- and wildfires are known to affect a number of soil faunal assemblages (Rice, 1932; Lawrence, 1966; Vogl, 1973; Sutherland and Dickman, 1999; Engstrom, 2010; Pringle et al., 2015). However, little is known in regards to the effect of prescribed fires on soil faunal assemblages. As a whole, studies into erosion in the park are also limited to the causes, nature, and effects of erosion on the landscape (Moon and Munro-Perry, 1988; Brady, 1993; Grab et al., 2011). This leaves an information gap on the soil arthropods which may be associated with erosion areas. In addition, monitoring of soils in erosion sites, and their

[1] D. Nariandas, verbal communication, Senior Section Range/Conservation Manager - Golden Gate Highlands National Park, 22 August 2019.

associated soil biota assemblages after erosion mitigation or rehabilitation, is often not considered and studied.

1.1 History of the GGHNP

The formation of the park was first discussed in 1962, incorporating the farms *Gladstone*, *Wilgerhof*, *Golden Gate*, *Glen Reenen* and *Wodehouse* (van Rensburg, 1968) on the western sectors of the present park's borders, and was offered to the National Parks Board for the development of the first national park in the Free State Province, South Africa. This initial core area of 1792 ha formerly became the Golden Gate Highlands National Park in 1963 (SANParks, 2013). Over the years, the park increased to 11 630 ha and was recognised as a significant biodiversity and tourism spot (Rademeyer and van Zyl, 2014).

In 2004, it was announced that the Qwaqwa National Park, situated adjacent to the eastern border of the park, was to be incorporated into the GGHNP, in an effort to transform the park into a more impactful environmental management unit. The incorporation was finalised in 2008, increasing the park to 32 758 ha and enhancing the biodiversity value of the park (SANParks, 2009). The Qwaqwa National Park was formed in 1991, consisting of former farmlands on which the agricultural labourers and farmers remained after the park was proclaimed (Rademeyer and van Zyl, 2014). Conflict in the park arose, stemming from the inhabitants who regularly grazed their livestock on rented land, which now formed part of the park, with residents showing displeasure over not being involved in discussions about the establishment of the park (Slater, 2002). Tension continued long after the amalgamation of the two parks, with farmers still allowing agricultural livestock to graze in the protected area (Rademeyer and van Zyl, 2014). Today, several livestock flocks are seen throughout the GGHNP, with many animals aggregating in areas already under heavy grazing stresses. In addition, daily movement of these flocks are believed to increase levels of erosion in affected areas, especially near water sources and overgrazed patches (SANParks, 2013).

In an attempt to engage the community and reinforce the park as a valuable commodity and natural resource, South African National Parks (SANParks) Management made it an active objective to inform, engage, and employ the community in conservation directives (SANParks, 2013). The park has had a level of success in past endeavours to re-engage the surrounding community in an initiative to educate and inform on conservation management practices. The Expanded Public Works Programme (EPWP), together with the Working for Water (WfW) initiative, created funded projects to both educate the community and provide income relief through temporary work for the unemployed (SANParks, 2019). As part of the SANParks management plan, the programme plays an integral role in terms of social investment into the neighbouring community by the national park, while at the same time directly addressing biodiversity management and strategic infrastructure development initiatives (SANParks, 2019). Projects include mitigation of many ecosystem threats, including divisions for fire prevention and control, erosion control and rehabilitation, and removal of alien invasive plants (SANParks, 2019). From these projects, erosion areas today have implemented erosion rehabilitation structures, specifically near the perennial river areas running through the park.

Studies on biodiversity, ecology and the effectiveness of conservation practices in national parks are vital to the monitoring and maintenance of their natural ecosystems (McGeoch et al., 2011; Gerlach et al., 2013; Muhumuza and Balkwill, 2013). Some national parks in Southern Africa have had several groups of faunal and floral assemblages check-listed and studied (e.g. Kruger National Park: Obermeijer, 1937; Brynard, 1961; Pienaar, 1963a,b; Lawrence, 1964; Pienaar, 1964; Lawrence, 1967a,b; Pienaar, 1967; Pienaar, 1968; Pienaar, 1969a,b; Pienaar, 1970; Pienaar, 1972; Rautenbach et al., 1979; Jacobsen and Pienaar, 1983; MacDonald and Gertenbach, 1988; Trollope, 1990; Oosthuizen, 1991; Boomker, 1994; Clark and Samways, 1996; Foxcroft and Hoffmann, 2000; Smith et al., 2000; Dippenaar-Schoeman and Leroy, 2003; Foxcroft et al., 2003; Redfern et al., 2003; Foxcroft et al., 2008). The GGHNP is an exception to this rule, with few faunal groups being documented and studied in the park and seemingly restricted to herpetofauna (Bates, 1991; Bates, 1997), oribatid mites (Hugo-Coetzee, 2014), tetranychid mites (Meyer, 1970), opilionids (Lotz, 2002), beetles (Louw, 1988), and various mammals, birds and aquatic life (De Graaff and Penzhorn, 1976; Rautenbach, 1976; van Hoven and Boomker, 1981; van der Walt and van Zyl, 1982; Earlé and Lawson, 1988; Reilly et al., 1990; Hutsebaut et al., 1992; Novellie and Knight, 1994; De Swardt and van Niekerk, 1996; Avenant, 1997; Russell and Skelton, 2005). This excludes studies into soil arthropod ecology, as some studies are available for the geomorphology and soils of the GGHNP (Groenewald, 1986; Grab et al., 2011; Telfer et al., 2012), but no studies are available looking into large groups of soil arthropod species associated with ecosystems in the park.

1.2 Soil-dwelling arthropods and the environment

Soils have played a significant role in the development of Earth ecosystems, with a strong link between the soils and the evolution of life (Wall et al., 2012). The role that soils and their biodiversity play in supporting terrestrial environments has been justified by identifying the ecosystem services that these faunal groups carry out (Ritz and van der Putten, 2012). Soil harbours a wide variety of organisms, with many of them remaining understudied or unidentified. However, soil meso- and macrofauna play strong roles in nutrient cycling, food web dynamics and disease oppression (Wardle et al., 2004; Wurst et al., 2012; Bardgett and van der Putten, 2014), as well as can be linked to monitoring factors ultimately contributing to human health (Wall et al., 2015). Soil arthropods, falling under meso- and macrofaunal classification, are integral to the functioning of ecosystems, with a variable abundance and diversity of soil faunal populations recorded in different landscapes (Madson, 2003; e.g. Lavelle, 1996; Huhta et al., 1998; Vanbergen et al., 2007; Wall et al., 2012). In addition, many of these groups may display a level of sensitivity to environmental change. For example, soil mites (Acari) have been focused on as biomonitoring indicators due to their displayed sensitivity to various types of soil disturbances (Gulvik, 2007).

The use of soil-dwelling arthropod groups as bioindicators of soil status has been debated and studied through recent years (e.g. Cortet et al., 1999; Dunger and Voigtländer, 2009; Neto et al., 2012; Yan et al., 2012), with many findings identifying these groups as strong indicators of environmental stress. Despite this, soil arthropod groups in South Africa still remain poorly studied, with few investigations addressing this gap in knowledge (Janion-Scheepers et al., 2016). This especially proves to be the case in national parks, with studies in the GGHNP limited only to certain taxa.

1.3 Controlled fires in national parks

Fires were initially thought to be devastating occurrences, drastically altering environments with undesirable effects, and greatly affecting fauna and flora (Kozlowski and Ahlgren, 1974). This was particularly true for environments that were suddenly experiencing increased irregular burnings across the world. However, it was not long until research in other types of ecosystems started uncovering the mechanisms in which fire plays a role with regards to ecosystem restoration and turnover (van Wilgen et al., 1994; Holmes and Richardson, 1999; Hirsch et al., 2001; Allen et al., 2002; Govender et al., 2006). For example, the fynbos ecosystem, unique among Mediterranean-type ecosystems in its nature, both in terms of biodiversity and management techniques needed, relies on variable degrees of fire to maintain its diversity and control encroachment by alien plant species (van Wilgen et al., 1994; van Wilgen, 2013). Controlled application of fire may promote restoration of ecosystems. Rangelands and prairies in North America are prime examples of this, where a combination of applied burning and limiting grazing control promotes the restoration and productivity of their ecosystems (Fuhlendorf and Engle, 2004).

Prescribed burnings are intentionally set fires under the control of a monitoring team for purposes of grassland and forest management, farming, landscape restoration and/or greenhouse gas abatement (Brose and van Lear, 1998; Hirsch et al., 2001; Hutchinson et al., 2005; Piñol et al., 2005). It is widely accepted that prescribed burnings may mitigate larger burn risk and may reduce the intensity and magnitude of larger runaway wildfires by greatly reducing the accumulation of burnable biomass (Gill and McCarthy, 1998; Neary et al., 2005; Arkle and Pilliod, 2010). SANParks management, together with SANParks Scientific Services researchers, monitor the burnable biomass over time and initiate planned prescribed burns based on the highest recorded biomass (SANParks personnel, personal communication, 2019[2]). These burnings greatly affect the intensity of future wildfires, limiting the degree of fire damage. In certain cases, park landscapes would have to be burned on a yearly basis, specifically during the drier months of June and July. Nonetheless, fires are a natural occurrence in the GGHNP, and form a vital part of regulating its ecosystems (SANParks, 2013).

Fire and its effects on fauna and fire application have been strongly debated over the years (Kozlowski and Ahlgren, 1974; Neary et al., 2005; Piñol et al., 2005), but with few studies done on the effects that fire may have on soil-dwelling arthropods in South African grasslands. This brings up a rather surprising gap in knowledge, as fires affect soils in various

[2] D. Nariandas, verbal communication, Senior Section Range/Conservation Manager - Golden Gate Highlands National Park, 22 August 2019.

ways depending on its intensity and the nature of burnable biomass in an area (Neary et al., 2005). These can include soil properties such as soil temperature and moisture, organic matter content, and mineralogy (Kozlowski and Ahlgren, 1974; Debano and Conrad, 1978; González-Pérez et al., 2004). It is assumed that, as the nature of soils may be altered during and after the application of fire, so may the nature of soil biota assemblages be affected by such change. As prescribed fires are initiated regularly in the park, it is vital that we study the effect of fire on soil-dwelling assemblages.

1.4 Soil erosion in South Africa, rehabilitation and mitigation of erosion losses

There are a number of environmental issues that play a significant role as causative factors of environmental problems in South African landscapes, ranging from different levels of land degradation to water resource threats (e.g. Scott et al., 1998; Smith et al., 2010; Seutloali and Beckedahl, 2015). Soil erosion, for example, is regarded as one of the most significant environmental problems causing the degradation of many types of ecosystems (Pimentel and Kounang, 1998; Meadows, 2003; Pimentel et al., 2004; Durán Zuazo and Rodríguez Pleguezuelo, 2008). It is strongly suggested that over 70% of South African landscapes are threatened by the effects of soil erosion to varying degrees (Le Roux et al., 2007), with many arguing the level of detrimental effects (i.e. loss of soil nutrients, desertification, and deteriation of soil quality) that erosion has on soils (e.g. van Dissel and de Graaff, 1998; Meadows and Hoffman, 2002; Le Roux et al., 2007; Le Roux, 2008; Compton et al., 2010). Although soil erosion is a topic of importance in South Africa, many erosion rehabilitation methods, specifically in natural areas, concentrate more on prevention of siltation and sedimentation of nearby water bodies (SANParks, 2013), rather than the actual effects on soils. Furthermore, monitoring of factors that may actively contribute to soil degradation overall is mainly focussed on agricultural or mining settings (e.g. Carrick and Krüger, 2007; Hoffman et al., 2014; Lal, 2015), and not necessarily on landscapes in protected areas.

Soil erosion is a natural occurrence which drives the creation of new landscape types and the formation of mountainous areas (Mhangara et al., 2012). In steep landscapes with heavy rainfall, erosion is an expected phenomenon with higher rates of erosion recorded in areas with more erodible soils. However, the rate at which erosion takes place in these areas is a growing concern, especially with the ever-present threat of global climate change shifting rainfall frequency, intensity, and patterns, thereby affecting the intensity of resulting soil erosion (Monlar and England, 1990; Nearing et al., 2004). The GGHNP is a national park known for its high altitude mountain formations, with deeply eroded sandstone outcroppings and cliffs alongside large expanses of natural grassland hills and valleys (SANParks, 2019). Sandstone in the GGHNP is known to produce large areas of shallow sandy soils with very low fertility that is highly susceptible to erosion losses (Roberts, 1969). This makes the GGHNP a site of interest when investigating erosion, as many aspects of erosion can be investigated in a number of site types. Despite this, little to no known studies have been conducted on erosion and the effects of erosion rehabilitation on soil communities in the GGHNP.

For many years, South African studies and reviews into land degradation and soil erosion have been centralised around agroecosystem studies and the direct effect of erosion on soil properties and soil loss, as well as plant growth (e.g. Lal, 1995; Le Roux et al., 2007; Mhangara et al., 2012). However, other studies have brought emphasis to the the actual role of soil biota in soil health (Orgiazzi and Panagos, 2018). In order to understand the communities of soil biota and their responses in varying terrains, community structure and change under different conditions need to be investigated.

1.5 General overview of thesis

This thesis is presented in the form of four research chapters, dealing with interlinked topics regarding soil-dwelling arthropods in various disturbed sites. Each chapter deals with soil faunal assemblages in regards to their presence, abundance and species richness in burnt, eroded and rehabilitated sites over variable lengths of time dependent on the nature of the sites. From this data, certain chapters deal with the overall effect of erosion disturbances on individual soil species, alongside species groups, to identify possible bioindicators for eroded sites in the GGHNP. This thesis broadly aims to investigate soil-dwelling arthropods in the GGHNP as indicators of soil status, while looking into their possible use in conservation management strategies.

Chapter 2: Effect of prescribed fire on soil arthropod assemblages in an Afromontane grassland landscape

While grassland wildfires and prescribed burning regimes are a constant occurrence in the mountainous landscape of the GGHNP, its impact on soil-dwelling arthropods is unclear. Thus, Chapter 2 aimed to determine the effect of an annual prescribed burning regime on soil-dwelling arthropod assemblages in a montane grassland landscape in comparison to an unburnt locality (B/NB, Fig. 1.1). To determine such effects, assemblage compositions, species richness, diversity and abundance were evaluated over a 12-month period, post-burning. This chapter also attempted to correlate assemblage diversity response to recorded soil mineralogy and environmental factors, to determine how soil arthropod assemblages are impacted by soil moisture and mineralogy. This chapter contributes to a large gap in knowledge in literature on the consequences of fire on soil arthropod abundance and diversity, relevant to the GGHNP.

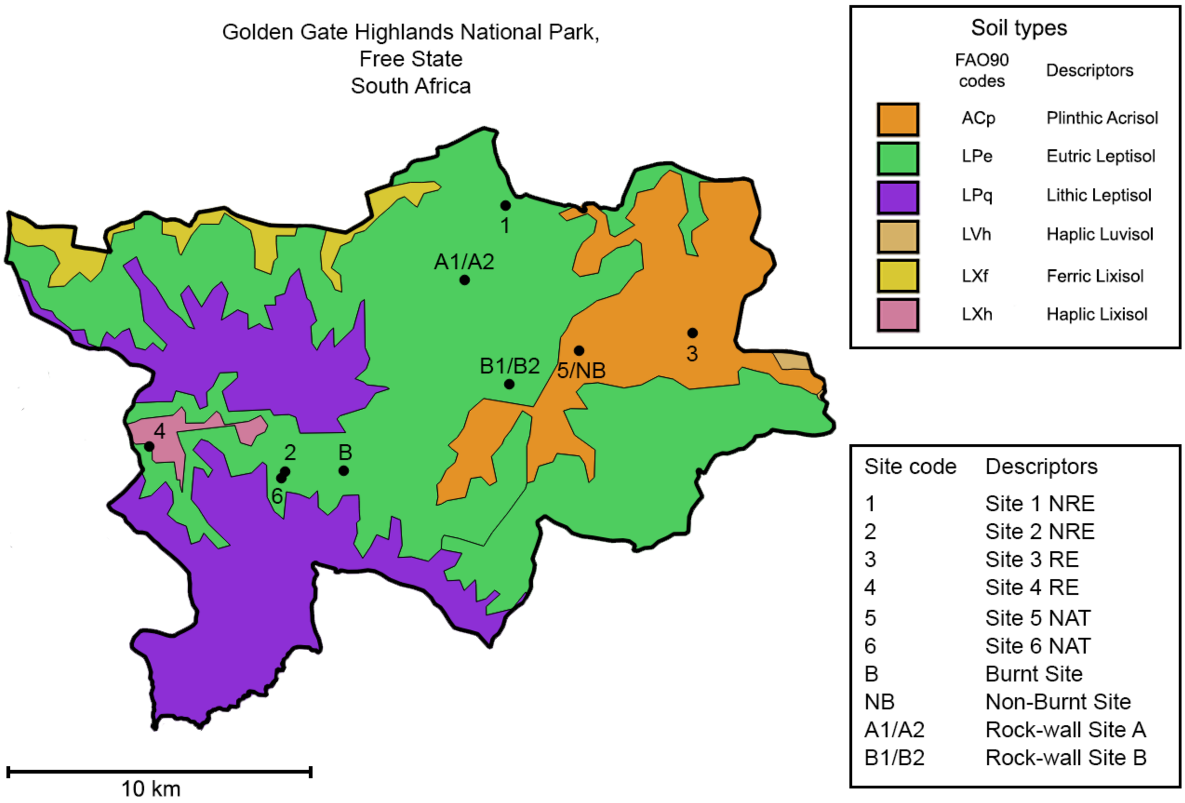

Fig. 1.1: Map of the Golden Gate Highlands National Park (GGHNP), showing sites investigated during this thesis, as well as described soil types, in accordance to FAO90 major groups and soil type codes and descriptors, within the park. NRE - Non-rehabilitated eroded site; RE - rehabilitated eroded site; NAT - Non-eroded undisturbed site. Soil types, codes and soil mapping zones obtained, with full permission, from SANParks Scientific Services in South Africa. GGHNP map processed with QGIS, version 2.18.15.

Chapter 3: Soil-dwelling arthropods as indicators of erosion in a South African grassland habitat

This chapter addressed the relatively unexplored topic of soil arthropods in eroded sites, in an attempt to identify possible soil-dwelling arthropod indicators of soil erosion. Additionally, the difference between non-rehabilitated and rehabilitated eroded site soil arthropod assemblages in the GGHNP was investigated (Site 1–6, Fig 1.1). Using species data from the study sites over a 24-month period, diversity indices values, abundances and significance test results were compared to identify significant differences between assemblages of each defined site type. In addition, the study attempted to test for significant interaction between soil arthropod species richness and major soil minerals, with best fit tests done for each generated model. This chapter serves as a starting point for bioindication studies in the GGHNP, as results suggest that soil arthropods, together with soil mineralogy, can be used as a method for defining erosion sites in the area.

Chapter 4: Effect of gabions on soil biota in eroded sites of a South African grassland habitat

Chapter 4 continues on the topic of soil arthropods in eroded sites. However, the study puts more emphasis on assemblages in sites with constructed gabion placement (Sites A1/A2 and B1/B2, Fig. 1.1). The specific aim of this investigation was to identify changes in soil arthropod assemblages in areas with implemented erosion rehabilitation, namely gabions near dirt roads, in the GGHNP. Species richness, diversity and interaction between soil arthropod groups and the site types were analysed for significant differences between rehabilitated and non-rehabilitated eroded sites. More importantly, this chapter briefly discusses the effects of sedimentation on arthropod species richness, highlighting the need for monitoring of soil deposition and associated biota after the implementation of rehabilitation structures, in order to monitor and maintain soil functions.

Chapter 5: The effect of renewal of previously implemented erosion rehabilitation methods on soil biota in a South African grassland habitat

The incidental restructuring of a previously rehabilitated eroded site in the GGHNP (Site 3, Fig. 1.1) provided a unique opportunity to assess the direct impact of renewed rehabilitation structures on soil-dwelling arthropod communities. Chapter 5 briefly investigates the effect of major landscape reformation and construction of alternative erosion mitigation structures, before and after implementation. As the site was treated as a stand-alone occurrence, responses of soil arthropod abundance and species richness were analysed over the time series. The chapter places emphasis on the need for monitoring of erosion mitigation methods and their subsequent effects on soil biota.

Chapter 6: General Discussion

In the final chapter, the results obtained from the four investigative studies are discussed, with emphasis on the implications of the most significant findings. Some insight into the overarching themes of the results are given based on conservation strategies already in place in the park, and recommendations regarding conservation management of eroded sites and fire regimes in the GGHNP are provided.

1.6 References

Allen, C.D., Savage, M., Falk, D.A., Suckling, K.F., Swetnam, T.W., Schulke, T., Stacey, P.B., Morgan, P., Hoffman, M., Klingel, J.T., 2002. Ecological restoration of southwestern Ponderosa Pine ecosystems: a broad perspective. Ecol. Appl. 12, 1418–1433.

Arkle, R.S., Pilliod, D.S., 2010. Prescribed fires as ecological surrogates for wildfires: a stream and riparian perspective. For. Ecol. Manag. 259, 893–903.

Avenant, N.L., 1997. Mammals recorded in the Qwaqwa National Park (1994-1995). Koedoe 40, 31–40.

Bardgett, R.D., van der Putten, W.H. 2014. Belowground biodiversity and ecosystem functioning. Nature 515, 505–511.

Bates, M.F., 1991. A provisional check list of the reptiles and amphibians of Golden Gate Highlands National Park. Koedoe 34, 153–155.

Bates, M.F., 1997. Herpetofauna of the nature reserves and national parks of the Free State province of South Africa. Afr. J. Herpetol. 46, 13–29.

Boomker, J., 1994. Parasites of South African freshwater fish. VI. Nematode parasites of some fish species in the Kruger National Park. Onderstepoort J. Vet. Res. 61, 35–43.

Brady, H.M., 1993. An investigation into the nature of gully erosion at Golden Gate Highlands National Park. MSc Dissertation, University of Natal, Pietermaritzburg.

Brose, P.H., van Lear, D.H., 1998. Responses of hardwood advance regeneration to seasonal prescribed fires in oak-dominated shelterwood stands. Can. J. For. Res. 28, 331–339.

Brynard, A.M., 1961. A supplementary check-list of plants recorded in the Kruger National Park. Koedoe 4, 110–116.

Carrick, P.J., Krüger, R., 2007. Restoring degraded landscapes in lowland Namaqualand: lessons from the mining experience and from regional ecological dynamics. J. Arid Environ. 70, 767–781.

Clark, T.E., Samways, M.J., 1996. Dragonflies (Odonata) as indicators of biotope quality in the Kruger National Park, South Africa. J. Appl. Ecol. 33, 1001–1012.

Compton, J.S., Herbert, C.T., Hoffman, M.T., Schneider, R.R., Stuut, J-B., 2010. A tenfold increase in the Orange River mean Holocene mud flux: implications for soil erosion in South Africa. Holocene, 115–122.

Cortet, J., Gomot-De Vauflery, A., Poinsot-Balaguer, N., Gomot, L., Texier, C., Cluzeau, D., 1999. The use of invertebrate soil fauna in monitoring pollutant effects. Eur. J. Soil Biol. 35, 115–134.

De Graaff, G., Penzhorn, B.L., 1976. The re-introduction of springbok *Antidorcas marsupialis* into South African National Parks - a documentation. Koedoe 19, 75–82.

De Swardt, D.H., van Niekerk, D.J., 1996. An annotated check list of the birds of Qwaqwa National Park. Koedoe 39, 89–106.

Debano, L.E., Conrad, C.E., 1978. The effect of fire on nutrients in a Chaparral ecosystem. Ecology 59, 489–197.

Dippenaar-Schoeman, A.S., Leroy, A., 2003. A check list of the spiders of the Kruger National Park, South Africa (Arachnida: Araneae). Koedoe 46, 91–100.

Dunger, W., Voigtländer, K., 2009. Soil fauna (Lumbricidae, Collembola, Diplopoda and Chilopoda) as indicators of soil eco-subsystem development in post-mining sites of eastern Germany- a review. Environ. Sci. 81, 1–51.

Durán Zuazo, V.H., Rodríguez Pleguezuelo, C.R., 2008. Soil-erosion and runoff prevention by plant covers. A review. Agron. Sustain. Dev. 28, 65–86.

Earlé, R.A., Lawson, A.B., 1988. An annotated check list of the birds of the Golden Gate Highlands National Park. Koedoe 31, 227–243.

Engstrom, R.T., 2010. First-order fire effects on animals: review and recommendations. Fire Ecol. 6, 115–130.

Foxcroft, L.C., Henderson, L., Nichols, G.R., Martin, B.W., 2003. A revised list of alien plants for the Kruger National Park. Koedoe 46: 21-44.

Foxcroft, L.C., Hoffmann, J.H., 2000. Dispersal of *Dactylopius opuntiae* Cockerell (Homoptera: Dactylopidae), a biological control agent of *Opuntia stricta* (Haworth.) Haworth. (Cactaceae) in the Kruger National Park. Koedoe 43, 1–5.

Foxcroft, L.C., Richardson, D.M., Wilson, J.R.U., 2008. Ornamental plants as invasive aliens: problems and solutions in Kruger National Park, South Africa. Environ. Manag. 40, 32–51.

Fuhlendorf, S.D., Engle, D.M., 2004. Application of the fire-grazing interaction to restore a shifting mosaic on tallgrass prairie. J. Appl. Ecol. 41, 604–614.

Gerlach, J., Samways, M., Pryke, J. 2013. Terrestrial invertebrates as bioindicators: an overview of available taxonomic groups. J. Insect Conserv. 17, 831–850.

Gill, A.M., McCarthy, M.A., 1998. Intervals between prescribed fires in Australia: what intrinsic variation should apply? Biol. Conserv. 85, 161–169.

González-Pérez, J.A., González-Vila, F.J., Almendros, G., Knicker, H., 2004. The Effect of fire on soil organic matter — a review. Environ. Int. 30, 855–870.

Govender, N., Trollope, S.W., van Wilgen, B.W., 2006. The effect of fire season, fire frequency, rainfall and management on fire intensity in savannah vegetation in South Africa. J. Appl. Ecol. 43, 748–758.

Grab, S.W., Goudie, A.S., Viles, H.A., Webb, N., 2011. Sandstone geomorphology of the Golden Gate Highlands National Park, South Africa, in a global context. Koedoe 53, 1–14.

Groenewald, G.H., 1986. Geology of the Golden Gate Highlands National Park. Koedoe 29, 165–181.

Gulvik, M., 2007. Mites (Acari) as indicators of soil biodiversity and land use monitoring: a review. Pol. J. Ecol. 55, 415–440.

Hirsch, K., Kafka, V., Tymstra, C., McAlpine, R., Hawkes, B., Stegehuis, H., Quintilio, S., Gauthier, S., Peck, K., 2001. Fire-smart forest management: a pragmatic approach to sustainable forest management in fire-dominated ecosystems. For. Chron. 77, 357–364.

Hoffmann, T., Todd, S., Ntshona, Z., Turner, S., 2014. Land Degradation in South Africa. University of Cape Town, South Africa.

Holmes, P.M., Richardson, D.M., 1999. Protocols for restoration based on recruitment dynamics, community structure, and ecosystem function: perspectives from South African fynbos. Restor. Ecol. 7, 215–230.

Hugo-Coetzee, E.A., 2014. New Oppiidae (Acari: Oribatida) from the Golden Gate Highlands National Park in South Africa. Zootaxa 3884, 533–552.

Huhta, V., Persson, T., Setälä, H., 1998. Functional implications of soil fauna diversity in boreal forests. Appl. Soil Ecol. 10, 277–288.

Hutchinson, T.F., Sutherland, E.K., Yaussy, D.A., 2005. Effects of repeated prescribed fires on the structure, composition, and regeneration of mixed-oak forests in Ohio. For. Ecol. Manag. 218, 210–228.

Hutsebaut, M.S., Kok, O.B., Daneel, J-H., 1992. Additions to the bird list of the Golden Gate Highlands National Park. Koedoe 35, 99–100.

Jacobsen, N.H.G., Pienaar, U. De V., 1983. New reptile records from the Kruger National Park. Koedoe 26, 135–144.

Janion-Scheepers, C., Measey, J., Braschler, B., Chown, S.L., Coetzee, L., Colville, J.F., Dames, J., Davies, A.B., Davies, S., Davis, A., Dippenaar-Schoeman, A.S., Duffy, G.A., Fourie, D., Griffiths, C., Haddad, C.R., Hamer, M., Herbert, D.G., Hugo-Coetzee, E.A., Jacobs, A., Jansen van Rensburg, C., Lamani, S., Lotz, L.N., Louw, S. vdM., Lyle, R., Malan, A.P., Marais, M., Neetling, J-A., Nxele, T.C., Plisko, D.J., Prendini, L., Rink, A.N., Swart, A., Theron, P., Truter, M., Ueckermann, E., Uys, V.M., Villet, M.H., Willows-Munro, S., Wilson, J.R.U. 2016. Soil biota in a megadiverse country: current knowledge and future research directions in South Africa. Pedobiologia 59, 129–174.

Kozlowski, T.T., Ahlgren, C.E., 1974. Fire and Ecosystems. Academic Press Inc., London.

Lal, R., 1995. Erosion-crop productivity relationships for soils of Africa. Soil Sci. Soc. Am. J. 59, 661–667.

Lal, R., 2015. Restoring soil quality to mitigate soil degradation. Sustainability 7, 5875–5895.

Lavelle, P., 1996. Diversity of soil fauna and ecosystem function. Biol. Int. 33, 3–16.

Lawrence, G.E., 1966. Ecology of vertebrate animals in relation to Chaparral fire in the Sierra Nevada foothills. Ecology 47, 278–291.

Lawrence, R.F., 1964. The Solifugae, Scorpions and Pedipalpi of the Kruger National Park. Koedoe 30–39.

Lawrence, R.F., 1967a. The pseudoscorpions (false-scorpions) of the Kruger National Park. Koedoe 10, 87–91.

Lawrence, R.F., 1967b. Supplementary list of the Solifugae, Scorpions and Pedipalpi of the Kruger National Park. Koedoe 10, 82–86.

Le Roux, J.J., Morgenthal, T.L., Malherbe, J., Pretorius, D.J., Sumner, P.D., 2008. Water erosion prediction at a national scale for South Africa. Water SA 34, 305–314.

Le Roux, J.J., Newby, T.S., Sumner, P.D., 2007. Monitoring soil erosion in South Africa at a regional scale: review and recommendations. S. Afr. J. Sci. 103, 329–335.

Lotz, L.N., 2002. Researches of the National Museum. Navors. nas. Mus. 18, 173–183.

Louw, S., 1988. Arboreal Coleoptera associated with *Leucosidea sericea* (Rosaceae) at the Golden Gate Highlands National Park. Koedoe 31, 53–70.

MacDonald, I.A.W., Gertenbach, W.P.D., 1988. A list of alien plants in the Kruger National Park. Koedoe 31, 137–150.

Madson, S.L., 2003. Response of Soil Microarthropods and Microclimatic Conditions to two-age Regeneration in Hardwood Forests in the Southern Appalachians. MSc Dissertation, University of Georgia, Athens.

McGeoch, M.A., Dopolo, M., Novellie, P., Hendriks, H., Freitag, S., Ferreira, S., Grant, R., Kruger, J., Bezuidenhout, H., Randall, R.M., Vermeulen, W., Kraaij, T., Russell, I.A., Knight, M.H., Holness, S., Oosthuizen, A., 2011. A strategic framework for biodiversity monitoring in South African National Parks. Koedoe 53. https://doi.org/10.4102/koedoe.v53i2.991.

Meadows, M.E., 2003. Soil erosion in the Swartland, Western Cape Province, South Africa: implications of past and present policy and practice. Environ. Sci. Policy 6, 17–28.

Meadows, M.E., Hoffman, M.T., 2002. The nature, extent and causes of land degradation in South Africa: legacy of the past, lessons for the future? Area 34, 428–437.

Meyer, M.K.P., 1970. South African Acari II check-list of mites in our parks (part I). Koedoe 13, 29–35.

Mhangara, P., Kakembo, V., Lim, K.J., 2012. Soil erosion risk assessment of the Keiskamma catchment, South Africa using GIS and remote sensing. Environ. Earth Sci. 65, 2087–2102.

Molnar. P., England, P., 1990. Late Cenozoic uplift of mountain ranges and global climate change: chicken or egg? Nature 346, 29–34.

Moon, B.P., Munro-Perry, P.M., 1988. Slope development on the Clarens sandstone formation in the Northeastern Orange Free State. S. Afr. Geogr. J. 70, 57–68.

Muhumuza, M., Balkwill, K., 2013. Factors affecting the success of conserving biodiversity in National Parks: a review of case studies from Africa. Int. J. Biodivers. https://dx.doi.org/10.1155/2013/798101.

Nearing, M.A., Pruski, F.F., O'Neal, M.R., 2004. Expected climate change impacts on soil erosion rates: a review. J. Soil. Water Conserv. 59, 43–50.

Neary, D.G., Ryan, K.C., DeBano, L.F., 2005. Wildland Fire on Ecosystems: Effects of Fire on Soil and Water. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden.

Neto, F.V.C., Correia, M.E.F., Pereira, G.H.A., Pereira, M.G., Leles, P.S.S., 2012. Soil fauna as an indicator of soil quality in forest stands, pasture and secondary forest. R. Bras. Ci. Solo. 36, 1407–1417.

Novellie, P.A., Knight, M., 1994. Repatriation and translocation of ungulates into South African national parks: an assessment of past attempts. Koedoe 37, 115–119.

Obermeijer, A.A., 1937. A preliminary list of the plants found in the Kruger National Park. Ann. Transvaal Mus. 17, 185–227.

Oosthuizen, J.H., 1991. An annotated check list of the leeches (Annelida: Hirudinea) of the Kruger National Park with a key to the species. Koedoe 34, 25–38.

Orgiazzi, A., Panagos, P., 2018. Soil biodiversity and soil erosion: it is time to get married. Global Ecol. Biogeogr. 27, 1155–1167.

Pienaar, U. De V., 1963a. The large mammals of the Kruger National Park — their distribution and present-day status. Koedoe 6, 1–10.

Pienaar, U. De V., 1963b. The Zoogeography and systematic list of Amphibia in the Kruger National Park. Koedoe 6, 76–82.

Pienaar, U. De V., 1964. The small mammals of the Kruger National Park — a systematic list and zoogeography. Koedoe 7, 1–25.

Pienaar, U. De V., 1967. A supplementary check-list of birds recorded in the Kruger National Park. Koedoe 10, 92–100.

Pienaar, U. De V., 1968. The freshwater fishes of the Kruger National Park. Koedoe 11, 1–20.

Pienaar, U. De V., 1969a. Observation on developmental biology, growth and some aspects of the population ecology of African Buffalo (*Syncerus caffer caffer* Sparrman). Koedoe 12, 29–47.

Pienaar, U. De V., 1969b. Predator-prey relationships amongst the larger mammals of the Kruger National Park. Koedoe 12, 108–118.

Pienaar, U. De V., 1970. A note on the occurrence of bat-eared fox *Otocyon megalotis megalotis* (Desmarest) in the Kruger National Park. Koedoe 13, 23–27.

Pienaar, U. De V., 1972. A new bat record for the Kruger National Park. Koedoe 15, 91–93.

Pimentel, D., Harvey, C., Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M., Crist, S., Shpritz, L., Fitton, L., Saffouri, R., Blair, R., 2004. Environmental and economic costs of soil erosion and conservation benefits. Science 267, 1117–1123.

Pimentel, D., Kounang, N., 1998. Ecology of soil erosion in ecosystems. Ecosystems 1, 416–426.

Piñol, J., Beven, K., Viegas, D.X., 2005. Modelling the effect of fire-exclusion and prescribed fire on wildfire size in Mediterranean ecosystems. Ecol. Model. 183, 397–409.

Pringle, R.M., Kimuya, D.M., Sensinig, R.L., Palmer, T.M., Riginos, C., Veblen, K.E., Young, T.P., 2015. Synergistic effects of fire and elephants on arboreal animals in an African savanna. J. Anim. Ecol. 84, 1637–1645.

Rademeyer, C., van Zyl, W., 2014. Golden jubilee for Golden Gate — a concise history of Golden Gate Highlands National Park, 1963 to 2013. Mediterr. J. Soc. Sci. 5, 1169–1177.

Rautenbach, I.L., 1976. A survey of the mammals occurring in the Golden Gate Highlands National Park. Koedoe 19, 133–144.

Rautenbach, I.L., de Graaff, G., Pienaar, U. De V., 1979. Records of seven small mammal species (Insectivora, Chiroptera) new to the Kruger National Park. Koedoe 22, 81–87.

Redfern, J.V., Grant, R., Biggs, H., Getz, W.M., 2003. Surface-water constraints on herbivore foraging in the Kruger National Park, South Africa. Ecology 84, 2092–2107.

Reilly, B.K., Theron, G.K., Bothma, J. Du P., 1990. Food preferences of oribi *Ourebia ourebi* in the Golden Gate Highland National Park. Koedoe 33, 55–61.

Rice, L.A. 1932. The effect of fire on the Prairie animal communities. Ecology 13, 392–401.

Ritz, K., van der Putten, H., 2012. Section 1 — The Living soil and Ecosystem Services: Introduction. In: Wall, D.H., Bardgett, R.D., Behan-Pelletier, V., Herrick, J.E., Jones, H., Rits, K., Six, J., Strong, D.R., van der Putten, W.H. (Eds.), Soil Ecology and Ecosystem Services. Oxford University Press, Oxford. pp. 5–6.

Roberts, B.R., 1969. The Vegetation of the Golden Gate Highlands National Park. Koedoe 12, 15–28.

Russell, I.A., Skelton, P.H., 2005. Freshwater fishes of Golden Gate Highlands National Park. Koedoe 48, 87–94.

Scott, D.F., Versfeld, D.B., Lesch, W., 1998. Erosion and sediment yield in relation to afforestation and fire in the mountains of the Western Cape province, South Africa. S. Afr. Geogr. J. 80, 52–59.

Seutloali, K.E., Beckedahl, H.R., 2015. Understanding the factors influencing rill erosion on roadcuts in the south eastern region of South Africa. Solid Earth 6, 633–641.

Slater, R., 2002. Between a rock and a hard place: contested livelihoods in Qwaqwa National Park in South Africa. Geogr. J. 168, 116–129.

Smith, A.M., Mather, A.A., Bundy, S.C., Cooper, J.A.G., Guastella, L.A., Ramsay, P.J., Theron, A., 2010. Contrasting styles of swell-driven coastal erosion: examples from KwaZulu-Natal, South Africa. Geol. Mag.: 1–14.

Smith, K.L., DeVos, V., Bryden, H., Price, L.B., Hugh-Jones, M.E., Keim, P., 2000. *Bacillus anthracis* diversity in Kruger National Park. J. Clin. Microbiol. 38, 3780–3784.

South African National Parks (SANParks), 2009. South African National Parks Annual Report 2008/9. SANParks.

South African National Parks (SANParks), 2013. Golden Gate Highlands National Park, Park Management Plan for the period 2013-2023. SANParks.

South African National Parks (SANParks), 2019. Expanded Public Work Programmes (EPWP). Accessed at: https://www.sanparks.org/about/connecting_to_society/epwp.php, on 28/11/2019, 15:31.

Sutherland, E.F., Dickman, C.R., 1999. Mechanisms of recovery after fire by rodents in the Australian environment: a review. Wildlife Res. 26, 405–419.

Telfer, M.W., Thomas, Z.A., Breman, E., 2012. Sand ramps in the Golden Gate Highlands National Park, South Africa: evidence of periglacial Aeolian activity during the last glacial. Palaeogeogr. Palaeoclimatol. Palaeoecol. 313–314, 59–69.

Trollope, W.S.W., 1990. Development of a technique for assessing veld condition in the Kruger National Park using key grass species. J. Grassl. Soc. South. Afr. 7, 46–51.

van der Walt, P.T., van Zyl, L.J., 1982. Die invloed van veldbrand op bewegingspatrone van wildsbokke in die Golden Gate-Hoogland Nasionale Park. Koedoe 25, 1–11.

van Dissel, S.C., de Graaff, J., 1998. Differences between farmers and scientists in the perception of soil erosion: a South African case study. Agris report, Indigenous Knowledge and Development monitor – Indigenous Knowledge Worldwide (IKDM).

van Hoven, W., Boomker, E.A., 1981. Feed utilization and digestion in the black wildebeest (*Connochaetes gnou*, Zimmerman, 1780) in the Golden Gate Highlands National Park. S. Afr. Tydskr. Natuurnav. 11, 35–40.

van Rensburg, A., 1968. Golden Gate, die geskiedenis van twee plase wat 'n Nasionale Park geword het. Koedoe 11, 83–137.

van Wilgen, B.W., 2013. Fire management in species-rich Cape fynbos shrublands. Front Ecol. Environ. 11, e35–e44.

van Wilgen, B.W., Richardson, D.M., Seydack, A.H.W., 1994. Managing fynbos for biodiversity: constraints and options in a fire-prone environment. S. Afr. J. Sci. 90, 322–329.

Vanbergen, A.J., Watt, A.D., Mitchell, R., Truscott, A-M., Palmer, S.C.F., Ivits, E., Eggleton, P., Jones, T.H., Sousa, J.P., 2007. Scale-specific correlations between habitat heterogeneity and soil fauna diversity along a landscape structure gradient. Oecologia 153, 713–725.

Vogl, R.J., 1973. Effects of fire on the plants and animals of a Florida wetland. Am. Midl. Nat. 89, 334–347.

Wall, D.H., Bardgett, R.D., Behan-Pelletier, V., Herrick, J.E., Jones, H., Rits, K., Six, J., Strong, D.R., van der Putten, W.H., 2012. Soil Ecology and Ecosystem Services. Oxford University Press, Oxford.

Wall, D.H., Nielsen, U.N., Six, J. 2015. Soil biodicersity and human health. Nature 528, https://doi.org/10.1038/nature15744.

Wardle, D.A., Bardgett, R.D., Klironomos, J.N., Setälä, H., van der Putten, W.H., Wall, D.H. 2004. Ecological linkage between aboveground and belowground biota. Science 304, 1629–1633.

Wurst, S., De Deyn, G.B., Orwin, K., 2012. Section 1.2: Soil Biodiversity and Function. In: Wall, D.H., Bardgett, R.D., Behan-Pelletier, V., Herrick, J.E., Jones, H., Rits, K., Six, J., Strong, D.R., van der Putten, W.H. (Eds.), Soil Ecology and Ecosystem Services. Oxford University Press, Oxford. pp. 28–44.

Yan, S., Singh, A.N., Fu, S., Liao, C., Wang, S., Li, Y., Cui, Y., Hu, L., 2012. A soil fauna index for assessing soil quality. Soil Biol. Biochem. 47, 158–165.

Chapter 2:

Effect of prescribed fire on soil arthropod assemblages in an Afromontane grassland landscape

Sylvia S. van der Merwe[a], Vaughn R. Swart[a], Emile Bredenhand[b], Charles R. Haddad[a]

[a] University of the Free State, Bloemfontein Campus, Bloemfontein, South Africa

[b] University of the Free State, Qwaqwa Campus, Phuthaditjhaba, South Africa.

Abstract

Prescribed burnings in protected areas are performed on the basis of maintaining the natural, cultural, and biodiversity components of ecosystems. Many studies investigating the effects of fire on arthropods and their surroundings tend to focus on the surface-active ground-dwelling fauna, while the impacts it has on biota occurring below-ground are often rarely studied. This study focussed on the effect of a prescribed fire on the soil-dwelling arthropods in a montane grassland habitat in the Golden Gate Highlands National Park, South Africa. Soil samples were obtained monthly from June 2017 to June 2018 from a scheduled burnt and non-burnt site. Overall, 595 soil arthropod individuals were sampled over a 12-month post-fire period, representing 38 families and 67 morphospecies. Soil arthropod abundance and species richness decreased considerably directly post-fire in the burnt site. Overall, species diversity was higher in the non-burnt site ($H = 3.04$) compared to the burnt site ($H = 2.34$), but species richness and abundance were consistently higher in the burnt site post-fire. Differences in trophic structure for each site was observed, with increased predator abundance in the burnt site. Assemblages between the two treatment areas were significantly different (ANOSIM global $R = 0.164$, $p = 0.011$), with ordination showing less variation among burnt assemblages. There was significant correlation between changes in soil mineralogy and soil arthropod diversity in response to fire. However, the data does not imply that these changes affected soil arthropod assemblages in this area detrimentally.

Keywords: Abundance, Burning, Management, Mineralogy, Post-fire, Richness

2.1 Introduction

Regular fires are a natural part of ecological function in many parts of the world, and have the potential to affect a variety of faunal components in an environment (Hermann et al., 1998; Roberts, 2000; Koponen, 2005; Collins et al., 2007; Kral et al., 2017). Fires typically lead to shifts and changes in environmental conditions, species dispersal and diversity, biomass, and overall ecosystem function (Moretti et al., 2006). However, the full effect of these fires is heavily dependent on factors such as the burn severity, the duration of fire events, and the intensity of fires (Coutinho, 1990; Certini, 2005; Keeley, 2009; Pivello et al., 2010). Similarly, the effects of fire on the microbial, meso- and macrofaunal soil life are also dependent on the fire severity and environmental conditions experienced post-fire (Ritz and Young, 2004; Matiax-Solera et al., 2009; Pivello et al., 2010; Dooley and Treseder, 2012; Dove and Hart, 2017).

The idea of replicating natural disturbances for conservation management stems from past observations of faunal and floral responses to disturbance, indicating that they are adapted to cope with large-scale disturbances such as wildfires (Buddle et al., 2005). Prescribed fires have been used over the years as an important conservation management tool (Harper et al., 2000; Apigian et al., 2006; Pryke and Samways, 2012), and are used as a conservation management tactic. This is done mainly to promote vegetation regrowth and prevent more intense fires from sweeping through fire-prone areas, which would normally cause unintended damage and faunal casualties (Trollope, 1993; Ramos-Neto and Pivello, 2000).

The Golden Gate Highlands National Park (GGHNP) in the eastern Free State Province, South Africa, is an example of this, as the grassland terrain is prone to annual sweeping wildfires, and thus regulatory prescribed fires are applied to areas with notable amounts of burnable biomass. Park Management regularly uses the informed advice of South African

National Parks (SANParks) Scientific Services to determine the area, time and control for each prescribed burning. However, even well-controlled prescribed fires can have a highly complex effect, dependent on the nature of a site (New et al., 2010). Areas that are disturbed without allowing for sufficient recovery before the next disturbance may prove detrimental to the continued persistence of certain arthropod groups (Buddle et al., 2005; Moretti et al., 2006), particularly in fire-intolerant assemblages (Moretti et al., 2004). However, studies on the effect of less frequent prescribed fires suggest that controlled burnings could promote species turnover (Siemann et al., 1997; Buddle et al., 2000; Buddle et al., 2005; Ferrenberg et al., 2006). Other investigations have reported a varied effect on arthropod biodiversity on affected areas after 12-months post-fire, suggesting that the effect of fire is dependent on the tolerance of species found in investigated areas (Swengel, 2001; Camann et al., 2008; Pryke and Samways, 2012; Haddad et al., 2015). The impact of fire on soil faunal groups in South Africa have been investigated to some extent, showing varying responses in different faunal groups (e.g. Parr & Chown, 2001; Hugo-Coetzee & Avanant, 2011; Janion-Scheepers et al., 2016), but few have been done on the larger soil faunal populations in the GGHNP.

Soil invertebrates provide vital ecosystem services at both plot and landscape scales, but are largely overlooked when considering their functions in an ecosystem (Brussaard et al., 1997; Lavelle et al., 2006). In practice, the effect of fires on soil macrofaunal groups should focus on the impacts on the soil and surrounding vegetation cover, rather than on the direct death of the fauna (Sgardelis et al., 1995; Gongalsky and Persson, 2013). The growing recognition that classifying species based on their functional feeding groups rather than only their higher taxonomic identity (Kaiser et al., 2009; Buschke and Seaman, 2011) is a welcome approach to studying ecosystem recovery and stability on the scale of ecosystem, landscape and biome (Moretti et al., 2006). It is thus vital to not only investigate the pattern of assemblage recovery over time after fires, but also consider how trophic structure changes. Despite the significant role that soil-dwelling arthropods play in ecosystem function, conservation methods in the GGHNP do not necessarily prioritise or monitor the effects of conservation practices on these soil animals. Soil arthropod species assemblages, specifically in the GGHNP, remain poorly investigated in respect to their responses to ecological change such as applied fires, and thus the effect of fire on these arthropod groups are not clear.

This study aimed to address the effect of a prescribed fire on soil-dwelling arthropods in a grassland site of the GGHNP in the Free State Province, South Africa, for advising Park Management in conservation strategies and future analysis. Considering that this was a small-scale investigation focussing on one South African National Park, the hypotheses aligned to answer preliminary questions relevant to the information needed to make informed conservation monitoring decisions in the park. Furthermore, the implications of the findings are briefly discussed. It was hypothesised that (1) soil arthropod species abundance and species richness would initially decrease due to mortality caused by burning (Ahlgren, 1974); (2) changes in soil properties and attributes would impact soil arthropod diversity due to heat effects, as well as increased pyrogenic organic matter deposition post-burning (Kozlowski and Ahlgren, 1974; Knicker, 2007; Bird et al., 2015), and (3) soil arthropod taxa and functional feeding groups would show differential responses to fire, with some taxa showing more resilience to prescribed fire (Malmström et al., 2008; Gongalsky and Persson, 2013; Pressler et al., 2019).

2.2 Materials and methods

2.2.1 Study area and period

The study was conducted in the GGHNP, located in the Eastern parts of the Free State Province, South Africa, which borders with Lesotho. It covers an area of approximately 340 km^2 and comprises many deeply eroded sandstone outcrops and cliffs, alongside large expanses of undisturbed grassland hills and valleys. The Park is situated in the Rooiberge of the Free State Province, in the foothills of the Maluti mountain range, with the park's highest peak being the Ribbokkop at 2 829 m a.s.l.. The GGHNP comprises a rich highveld and montane grassland flora, with more than 60 grass species identified within the park area (Roberts, 1969). These grasses belong to the vegetative units of Eastern Free State Sandy, Northern Drakensberg Highland and Lesotho Highland Basalt grasslands (Mucina et al., 2006). Soils are highly variable in the GGHNP, with several different groups of soil types described in the park (SANParks personnel, personal communication, 2019[1]; Supplementary material, Chapter 2, Fig. S2.1). It is currently characterised as the only grassland national park in South Africa. The GGHNP falls in the summer rainfall region of central South Africa, with rainfall averaging 760 mm annually. Summers are generally hot, with daily peak temperatures reaching between 30–38°C, while winters are cold, with minimum temperatures frequently between -10–0°C.

The study focussed on the upper layers of the soil in two grassland areas located near the main road which passes through the park. For the purpose of this study, soil samples were taken once every four weeks from two different areas: 1) an open grassland unaffected by scheduled fires carried out by the South African National Parks (SANParks) Authorities (S 28°28.833', E 28°43.280'); and 2) an open grassland affected by a scheduled and prescribed burning carried out by the SANParks Authorities as per conservation regulations (S 28°31.213', E 28°38.362'). Sampling commenced one month before the scheduled burning of July 2017, to gather baseline conditions for both sites, and continued until June 2018. Two sites per site type were selected based primarily on similarities in the composition of grass species, to ensure comparability of the results.

[1] C. Simms, electronic communication, GIS/RS Analyst - South African National Parks, 18 September 2019.

2.2.2 Soil biota sampling

In order to determine the relevant soil macro- and mesofaunal groups to monitor the health of the investigative sites, a soil arthropod survey was conducted to identify the species in each area and observe occurrence patterns over time. Soil samples were taken from the field and extracted through a Berlese-Tullgren funnel (Triplehorn and Johnson, 2005; Badenhorst, 2016) to isolate arthropod specimens over a course of twelve months, to avoid overlaps with the next year's prescribed burnings. An individual soil sample was defined as a soil mass of between 400 and 500 g each from the top 10 cm of soil and within a 10 cm radius of a chosen sampling spot. Ten soil samples were taken in a random pattern, at least 5 m apart, in each site every month to determine changes over time. Each sample was then transported to the University of the Free State, Bloemfontein, South Africa, and placed into individual Berlese-Tullgren funnels with connected storage bottles containing 70% ethyl alcohol (Tullgren, 1918; Triplehorn and Johnson, 2005; Badenhorst, 2016). Extractions proceeded for a period of seven days to allow for sufficient soil arthropod extraction.

Arthropods were sorted according to order, family and morphospecies, with special consideration given to groups of Collembola (springtails), soil mites (Mesostigmata, Prostigmata, and Oribatida) and other Insecta. Major soil arthropod groups used as indicators in previous studies (Burbidge et al., 1992; Ruf, 1998; Kimberling et al., 2001; Blakely et al., 2002; Gulvik, 2007; Philpott et al., 2010), were separated and analysed to establish possible groups of interest for the area (Formicidae, Collembola, oribatid, mesostigmatid and prostigmatid mites). In addition, all species were allocated to functional feeding groups (mycophages, phytophages, saprophages, omnivores, bacteriophages and predators) to monitor changes in trophic structure at each site (Bardgett and Cook, 1998; Brussaard, 1998; Triplehorn and Johnson, 2005; Badenhorst, 2016). All Collembola type material was stored at the Iziko South African Museum, Cape Town.

2.2.3 Soil analysis

Additional soil samples were taken from each study site for mineralogical analysis in order to track changes in mineralogy over time. Three soil samples of between 100 and 200 g were taken at a 10 cm depth and at a 10 cm radius of a chosen sampling spot in each site every four months during the study period. Samples were processed and analysed using X-ray Fluorescence (XRF) major element analysis (SiO_2, Al_2O_3, CaO, K_2O, TiO_2, Fe_2O_3, MgO, MnO, P_2O_5, Na_2O, % Organic Matter based on Loss on Ignition (LOI)), carried out by the Department of Geology, University of the Free State. In addition, soil temperature and moisture were recorded from the topsoil layer (0–10 cm) using a handheld SMT-100 moisture and temperature probe. Rainfall, per month, was also recorded over the post-burning period of the study.

2.2.4 Statistical analysis

To determine whether controlled burning has a significant effect on soil biota assemblages, alpha diversities using Shannon-Wiener diversity indices and Chao1 estimators were calculated per treatment site, for overall data, using EstimateS version 9.1.0 (Colwell, 2013). Non-metric Multidimensional Scaling (3D) ordination using the Bray-Curtis similarity index was used to determine the patterns of similarity in soil arthropod assemblages between the burnt and non-burnt sites over the 12-month post-burning period using the Vegan package in R, version 3.5.3 (Oksanen et al., 2019).

Analysis of similarity (ANOSIM) with Bray-Curtis similarity index was also used to determine differences between soil biota assemblages of each treatment site using Paleontological Statistics (PAST), version 3.25 (Hammer et al., 2001). This analysis was performed using log-transformed abundance data (log10(n+1)) with 9999 permutations.

Additionally, a diversity t-test was conducted using the species abundances of one-month before prescribed burning was scheduled, and of the final month post-burn (twelfth month) to determine whether the burnt site assemblages had recovered to a similar level as seen in the pre-burn month. A similar test was conducted on the same criteria for the non-burnt site. This diversity t-test compared the Shannon-Wiener diversities of the two months, utilising the species abundances present in each, by using a t-test described by Poole (1974). The Shannon indices from this test include a bias correction term.

In order to quantify the possible effects of changes in soil mineralogy post-fire on soil biota assemblages, Linear Mixed Effects modelling was conducted on rank Shannon-Wiener diversity indices from the sampled months in which soil samples were taken, with time as a random factor and treatment and subsequent soil mineralogy (11 major minerals) as fixed effects. Similar modelling was performed using soil temperature and soil moisture. Each analysis was conducted using the LmerTest package in R, version 3.5.3 (Kuznetsova et al., 2017).

2.3 Results

2.3.1. Faunistic composition

Overall, 595 soil arthropod individuals were sampled over the 12-month post-fire period, representing 67 morphospecies and 38 families. Of the two treatments, the burnt site displayed the highest species richness and abundance over time (Table 2.1). Despite this, the non-burnt site retained a higher diversity than the burnt site. More than half of the species collected from each treatment were unique to that specific treatment (Table 2.1). Representative soil biota groups, sampled one month before the commencement of the scheduled fire in July 2017, decreased sharply after the burning, with both abundance and richness declining after the fire incident (Fig. 2.1). Of interest, an influx of species individuals of a single taxon caused a spike in abundance, observed in November, specifically an increase in a predatory mite of the family Dermanyssidae.

Table 2.1: Total number of individuals, observed species richness (S_{obs}) and calculated species diversity (Shannon H and S_{Chao1}), from pooled overall totals of all species sampled over the 12-month post-fire period for the burnt and non-burnt sites. Number of unique species for each site is indicated from overall data.

	Individuals	S_{obs}	Shannon H	S_{Chao1}	Unique species
Burnt	440	48	2.34	71.75	27
Non-burnt	155	39	3.04	50	19
Total	595	67	-	-	-

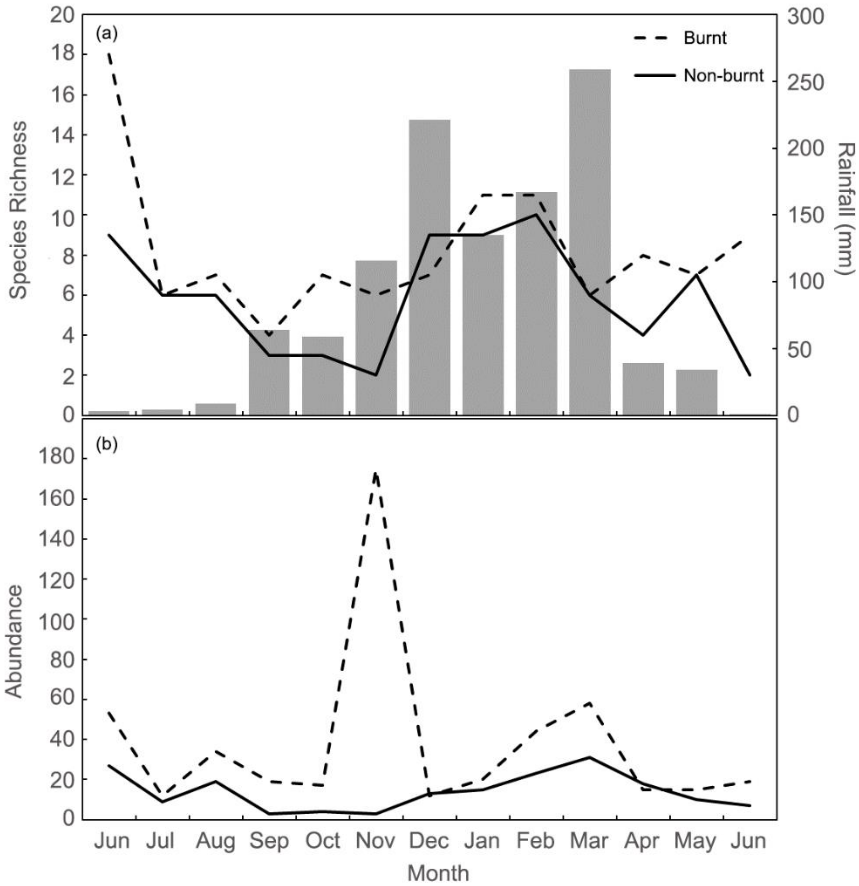

Fig. 2.1: Observed (a) species richness with monthly rainfall, and (b) abundance of burnt and non-burnt soil arthropod assemblages over the one-month pre-fire and 12-month post-fire period (burning occurred during the month of July).

Approximately 77.31% of all soil biota sampled was represented by Acari groups. Among these, predatory mites were the most abundant within the burnt area, making up 62.73% of all soil biota overall. Trophic groups in the non-burnt site were more evenly distributed, with the most dominant feeding groups being mycophages (21.29%), predators (16.77%), and bacteriophages (15.48%) (Table 2.2).

Table 2.2: Total trophic abundance and species richness, from pooled overall totals of each site over a 12-month post-fire treatment, for the burnt and non-burnt sites.

Trophic abundance	Burnt	%	Non-burnt	%	Total	%
Bacteriophagous	4	0.90	24	15.48	28	4.71
Mycophagous	63	14.32	33	21.29	96	16.13
Omnivorous	45	10.23	52	33.55	97	16.30
Phytophagous	9	2.05	9	5.81	18	3.03
Predaceous	276	62.73	26	16.77	302	50.76
Saprophagous	30	6.82	9	5.81	39	6.55
Varied	13	2.95	2	1.29	15	2.52
Total	440	100	155	100	595	100
Trophic species richness						
Bacteriophagous	1	2.08	2	5.41	2	2.99
Mycophagous	5	10.42	9	24.32	10	14.93
Omnivorous	11	22.92	5	13.51	11	16.42
Phytophagous	7	14.58	5	13.51	13	19.40
Predaceous	11	22.92	12	32.43	17	25.37
Saprophagous	8	16.66	4	5.41	9	13.43
Varied	5	10.42	2	5.41	5	7.46
Total	48	100	39	100	67	100

Although numbers of ants (Formicidae) sampled in the burnt and non-burnt sites were similar, their proportional abundance was nearly three times higher in the non-burnt site, making up 33.55% of the arthropod abundance; their species richness was also noticeably higher in the burnt site (Table 2.3). Collembola proportional abundance was similarly low in both treatments (<4%), but both their actual abundance and species richness were more than double in the burnt than non-burnt site. Oribatid and mesostigmatid mites were more abundant, proportionally abundant, and species rich in the burnt site, while prostigmatid mites were clearly negatively affected by the fire, being the most species rich and the second most abundant group in the non-burnt site (Table 2.3).

Table 2.3: Total number of major soil arthropod group abundance and observed species richness (S_{obs}), from pooled overall totals sampled during the 12-month post-fire period.

	Burnt				Non-burnt			
	Abundance	%	S_{obs}	%	Abundance	%	S_{obs}	%
Collembola	15	3.41	8	16.67	6	3.87	3	7.69
Hymenoptera - Formicidae	45	10.23	11	22.92	52	33.55	7	17.95
Acari - mites								
Prostigmatid mites	31	7.04	8	16.67	46	29.68	14	35.90
Mesostigmatid mites	227	51.59	9	18.75	10	6.45	4	10.26
Oribatid mites	117	26.59	7	14.58	29	18.71	5	12.82
Psocoptera	0	0.00	0	0.00	2	1.29	1	2.56
Thysanoptera	1	0.23	1	2.08	3	1.94	2	5.13
Coleoptera	2	0.45	2	4.17	4	2.58	2	5.13
Diptera	1	0.23	1	2.08	0	0.00	0	0.00
Hymenoptera - other	1	0.23	1	2.08	0	0.00	0	0.00
Hemiptera	0	0.00	0	0.00	3	1.94	1	2.56

2.3.2. Impact on arthropod assemblages

Soil biota assemblages between the two treatment sites were found to be significantly different (ANOSIM global R = 0.164, p = 0.011). Ordination of the two treatments showed a large overlap in species assemblages over the post-fire period, but the monthly assemblages in the burnt site showed less variation, indicating a possible shift towards more resilient species groups within the burnt site (Fig. 2.2).

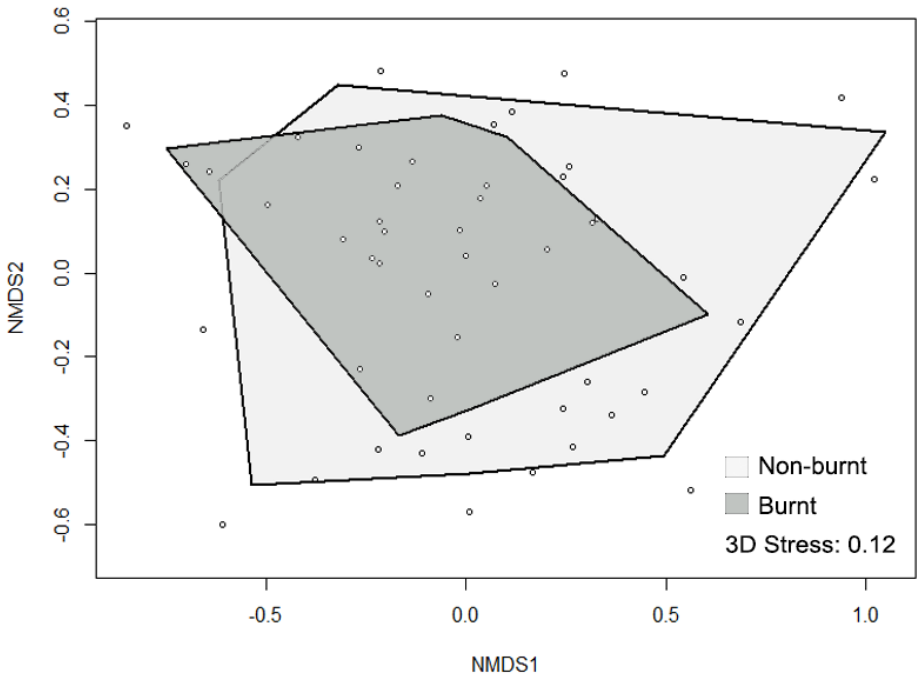

Fig. 2.2: Non-metric Multidimensional Scaling (3D) ordination of soil arthropod assemblages sampled from the burnt and non-burnt sites over the 12-month post-fire period.

Comparisons of the effects of recorded soil mineralogy on arthropod diversity showed that mineralogical changes in the burnt site had a significant impact, while variables in the non-burnt site were entirely non-significant (Table 2.4; Supplementary material, Chapter 2, Table S2.1). In the non-burnt site, soil moisture ($t = 1.455$, $p = 0.152$) and temperature ($t = -1.364$, $p = 0.179$) did not affect soil arthropod diversity (Supplementary material, Chapter 2, Fig. S2.2). The burnt site, while showing no significance against soil moisture ($t = -0.324$, $p = 0.747$), did, however, indicate a significant effect of soil temperature on soil arthropod diversity ($t = -2.138$, $p = 0.037$). The diversity t-test showed a significant difference between the Shannon-Wiener diversities of the one-month pre-fire and twelfth-month post-fire ($t = 3.556$, $p = 0.001$), with the twelfth month possessing a lower diversity. However, results of the non-burnt site also showed significance between the two months ($t = -3.100$, $p = 0.016$) with the twelfth month also possessing a lower diversity. This suggests that the assemblage changes observed could have been attributed more to annual differences in assemblage structure rather than to the effect of the burning.

Table 2.4: Resulting p-values of general linear mixed effect models testing links between soil arthropod diversity and changes in soil mineralogy of the burnt and non-burnt sites.

	p-values	
Soil mineralogy	Burnt	Non-burnt
SiO_2	0.025*	0.567
Al_2O_3	0.359	0.839
CaO	0.009**	0.346
K_2O	0.973	0.341
TiO_2	0.959	0.835
Fe_2O_3	0.006**	0.595
MgO	0.016*	0.977
MnO	0.031*	0.089
P_2O_5	0.046*	0.813
Na_2O	0.127	0.857
% Organic Matter (LOI)	0.048*	0.837

p-value significance : 0.001 '***' 0.01 '**' 0.05 '*'

2.4 Discussion

2.4.1 Effect of fire on soil arthropod abundance and species richness

Many studies investigating the effect of fires on arthropods and their surroundings tend to focus on the surface-active ground-dwelling fauna (Richardson and Holliday, 1982; Wikars and Schimmel, 2001; Hanula and Wade, 2003; Saint-Germain et al., 2005; Frizzo et al., 2012; Haddad et al., 2015), while the impacts it has on biota occurring below-ground are often overlooked (Malmström et al., 2008). This is most likely due to surface-active arthropods affected at the time of burning, whereas soil arthropods are widely affected by the resulting soil changes post-fire (McCullough et al., 1998). Nonetheless, soil-dwelling biota have often been considered an important indicator of habitat disturbance and recovery, as they have been found to show responses to changing environmental conditions (Paoletti et al., 1991; Behan-Pelletier, 1999; Nakamura et al., 2007; Santorufo et al., 2012). During our study, soil arthropod species richness and abundance of the burnt site were noted to decrease dramatically immediately post-fire, consistent with results from other fire treatment studies (Swengel, 2001; Coleman and Rieske, 2006; Malmström et al., 2008). However, this observation may have been more attributed to seasonal change as these assemblages move and adapt to the drier environment.

Fires are a natural occurrence during dry winter months in the mountainous grasslands of the GGHNP, with prescribed fire regimes implemented on an annual basis (Rademeyer and van Zyl, 2014). In certain instances, grassland areas of the park are not burnt, to act as protection areas for wildlife (SANParks personnel, personal communication, 2019[2]). These unburned refugia often contain a greater diversity and abundance of soil biota (Gongalsky and Zaitsev, 2016), acting as colonising populations of soil arthropod communities within an area

[2] D. Nariandas, verbal communication, Senior Section Range/Conservation Manager - Golden Gate Highlands National Park, 22 August 2019.

(Zaitsev et al., 2014). However, while a greater diversity in the non-burnt site was seen during this study, a higher abundance was not. The opposite was seen in the burnt site. This was an interesting occurrence as usual observed patterns of fire disturbance see a lower abundance of soil biota in burned localities (e.g. Kim and Jung, 2008; Malmström et al., 2008; Lisa et al., 2015). Callaham et al. (2003) noted the response of native earthworms in frequently burned prairie, suggesting an adaptation to the warmer soil conditions, and a response to improved grass growth, post-fire. Therefore, soil arthropod assemblages in the GGHNP may be adapted to survive frequent fires in the area, displaying a relatively fast population recovery rate, increasing abundance in the burnt site. It can also be assumed that soil arthropod assemblages in regularly burnt sites benefit from the effects of fires due to the timing of prescribed burnings and the effect of these low intensity fires on soil structure (Pressler et al., 2019).

Recorded species richness and abundance of the burnt site was observed to differ between the one-month pre-burning and twelfth-month post-burning, with observed richness and abundance of the twelfth-month being lower. It could be considered that an adequate recovery point was not necessarily reached at the time of the next scheduled burning, suggesting that sufficient recovery time was not provided in regards to soil arthropod assemblages during this study. This is in contrast to Haddad et al. (2015), which found spiders recovered to suggested pre-fire levels a year after burning. This suggests that there is a taxon-determined effect dependent on tolerance and dispersal ability of soil-dwelling arthropods. Disturbance is a primary factor contributing to spatial and temporal heterogeneity of communities (Walker, 2012). In certain cases, small changes to the initial state can lead to changes in population densities, affecting initial conditions (e.g. abundance of certain species) causing a continual reset (Maaß et al., 2014). This, in turn, contributes to a rendering of the assembly process, which remains variable in regards to specific species and their formation of assemblages. However, with comparisons between the burnt and non-burnt sites regarding the diversity t-test showing similar trends, it could be argued that lower abundances and species richness were more attributed to annual assemblage changes rather than the effect of fire. While sufficient data for determining the driving factors of soil arthropod community structure in the park was not obtained during this study, it could be hypothesised that changes in soil-dwelling arthropod species richness and abundance in the GGHNP was not solely dependent on environmental fluctuation. Factors such as habitat gradients, dispersal ability of the species groups and competition trade-offs could explain the variable changes, as studies have found certain soil arthropods to show responses corresponding more to their stochastic and deterministic community determinants (Ellwood et al., 2009; Caruso et al., 2012; Ingimarsdóttir et al., 2012; Ferrenberg et al., 2016).

2.4.2 Soil attribute changes on soil arthropod diversity

Significance results showed a possible relationship between soil arthropod diversity and soil temperature over the post-fire period within the burnt, but not non-burnt, site. While previous studies have found that soil moisture correlates strongly with changes in soil arthropod assemblages (O'Lear and Blair, 1999; Chikoski et al., 2006), the same was not true in our study. A possible reason for this could be that soil arthropod assemblages present in these moist alpine grasslands are adapted to the seasonal fluctuations of soil moisture experienced throughout the year, and that prescribed fires did not drastically affect soil moisture levels after burning within the site (Supplementary material, Chapter 2, Fig. S2.2).

Annual prescribed burning regimes have a varied impact on soil conditions compared to biennial burning intervals and their long-term counterparts (Pivello et al., 2010; Alcañiz et al., 2018). During our study the percentage of soil organic matter had an effect on soil arthropod diversity over time, suggesting that fires may alter assemblages as pyrogenic carbon deposition increases with each fire. The addition of pyrogenic organic matter may alter the quality of organic matter already in the soil, resulting in an indirect effect to euedaphic soil communities (Knicker, 2007; Bird et al., 2015). Changes in soil mineralogy, because of burning, can be long-term, and changes brought about by the fires can alter an ecosystem (Klopatek, 1987; Knicker et al., 2006; Knicker, 2007). Significant effects of soil mineralogy on arthropod diversity in the burnt site may support the theory that soils within the prescribed burnt site may have been altered by yearly treatments, subsequently affecting the nature of assemblages that are able to survive under the altered conditions.

Although certain soil minerals were indicated to be of significance when correlating the effects of soil conditions on soil arthropod diversity in the burnt site, it is important to note that the observations of this study was taken from a single treated site and an untreated site with no comparably similar replicates to the study sites. The need for replicated studies on prescribed fires in contrasting vegetation units in the GGHNP is necessary to better understand the role of soil arthropods in the park, as it is unclear how assemblages may change under contrasting fire regimes. This study can, however, be used as a starting point for soil arthropod studies in the GGHNP.

2.4.3 Response of functional feeding groups and soil arthropod taxa

Fire impacts faunal groups to varying degrees based on their lifestyle and behaviour (Harper et al., 2000). Results from our study showed differences between burnt and non-burnt soil assemblages, with large variations occurring within trophic groups. The effect of fire often alters vegetation coverage and subsequent food availability within an area, leading to shifts in feeding guilds (Moretti et al., 2006; Caut et al., 2013). The results of the current study supported this, as burnt assemblages showed a distinct shift, with an uneven spread of trophic groups occurring post-fire compared to the non-burnt site. The adaptive feeding behaviour of most soil arthropods may account for the resilience of some soil arthropod species after a fire (Pressler et al., 2019). However, as no studies are available on soil arthropods in the GGHNP in respect to their interactions in a food web context, the pressures that play a role in the patterns of these soil feeding groups remains unclear.

While previous studies have shown fire to cause a decrease in abundances and diversity of predatory, bacteriophagous and mycophagous soil biota (Ajee, 1993; Hanula and Wade, 2003), a sharp increase in predatory abundance occurred five-months post-fire during our study. McSorley (1993) observed similar occurrences, in which omnivorous nematodes and predators increased in a single site post-fire, while mycophagous nematodes declined, suggesting responses of functional feeding groups to fire are site-specific. Additionally, other groups may have fluctuated during the month before sampling due to regenerating vegetative growth, leading to the subsequent increase in predators (Kim and Jung, 2008; Evans, 2017). Alternatively, the presence of these individuals are arguably attributed more to phenology rather than the fluctuations of other groups, as mite population numbers are often dependent on the season of rainfall and temperature fluctuations (Yaninek et al., 1989; Badejo, 1990; Hugo-Coetzee and Avenant, 2011). The predatory influx may have also been attributed to the asynchronous recovery time of both predator and prey populations after the fire treatment (Malmström et al., 2009).

As previous studies have concentrated on specific arthropod groups when looking at disturbance effects, it is important to analyse the observational data in light of past findings, to highlight certain patterns found for particular soil arthropod groups:

2.4.3.1 Acari (mites)

Oribatid mites are regarded as slow to adjust to environmental change, due to their low metabolic rates, slow development and low fecundity, thus making them sensitive to change (Behan-Pelletier, 1999). Previous studies have shown oribatid mites to respond negatively to prescribed fires (Barratt et al., 2006; Camann et al., 2008). This was contrary in our study, with markedly higher oribatid abundance, suggesting that alpine grassland species may be able to survive and thrive in the altered conditions brought about by fires. Similar results were observed by Hugo-Coetzee and Avenant (2011), with little effect of fire on most species of mites recorded four months after the treatment. Oribatid mites are typically more armoured than other soil-dwelling taxa, resulting in increased protection against factors, such as heightened temperatures, during fires (Malmström et al., 2008).

Prostigmatid mites have been found to show a stronger tolerance to unfavourable conditions (Gergóc and Hufnagel, 2009), which would explain their prominence during this study overall. However, the lower number of prostigmatid mites found within the burnt site could suggest that the altered soil conditions negatively affected this group, but made it more favourable to oribatid and mesostigmatid mites. This corresponds with previous research showing a positive relationship between increased grassland productivity, brought about by prescribed burning, and the abundance and diversity of soil arthropods (Clapperton et al., 2002). Similarly, it can be hypothesised that prescribed fires not only benefit grassland management by serving as firebreaks, but also support soil arthropod groups that would not normally thrive under undisturbed conditions in the GGHNP.

2.4.3.2 Collembola (springtails)

Collembola are readily characterised as being sensitive to harsher conditions, with varying responses to disturbances between family groups (Barbercheck et al., 2009). The soil conditions of the study sites within the GGHNP may not have favoured collembolan groups within the area generally, explaining their low abundance, specifically within the sampled areas. However, these conditions could explain the presence of the hardier Collembola groups present within the burnt site based on their character states (Huebner et al., 2012). This was especially true for members of the families Tullbergidae and Isotomidae, with notably streamline bodies, and in the case of Tullbergidae, little pigmentation and harder bodies (Dunger and Shlitt, 2011). Alternatively, these groups may be more prominent in areas with higher soil moisture levels (Convey et al., 2003) and could be used in assessing other sites within the park.

2.4.3.3 Hymenoptera - Formicidae (ants)

Ants are effective indicators in ecological studies due to their varied behaviour and wide range of habitats (Parr and Chown, 2001; Ribas et al., 2012). Opportunistic groups tend to respond at a fast rate to frequently disturbed sites, and abundances are usually dependent on the nature of the disturbance (Anderson, 1991; King et al., 1998; Matsuda et al., 2011; Ribas et al., 2012; Arnan et al., 2013). Omnivory is common among soil arthropod communities, allowing for a variety of prey options being available, increasing survival despite limitations to specific food resources post-fire (Pressler et al., 2019). A large abundance and species richness of formicid groups found over the 12-month post-fire period in the burnt site could suggest that these arthropods acted upon plant regrowth over time with subsequent change in conditions and larger availability of food sources. This hints at the possible use of ants in the GGHNP for long-term monitoring of the effects of fire disturbance extent and frequency.

The absence of other insect groups was not unmerited, as soil niches are not usually inhabited by other groups for the entirety of their life cycle. The rare occurrence of beetles and thrips in the soil was incidental, as these individuals are present in small numbers and only for a short part of their development (Stork and Eggleton, 1992; Pfiffner and Luka, 2000). In light of this, it can be suggested that only Collembola, Formicidae and Acari groups should be investigated when monitoring soil arthropod activity for the use of bioindication in GGHNP grasslands. Nevertheless, soil arthropod groups seem to vary in their responses in the GGHNP soil. However, as this study was conducted on a single site in grasslands of the GGHNP, and little is known about the behavioural and physiological responses of these animals, the causative factors of their observed changes are still unclear. The study results are highly specific to the investigated areas in the GGHNP, and further investigation on the impact of fire, specifically on functional feeding groups and soil arthropod taxon responses, is necessary to draw any conclusive findings.

2.4.4 Implications of findings

Fire is an important factor in shaping environments and ecological processes (Certini, 2005; Vermeire et al., 2005; Burton et al., 2008; Williams and Bradstock, 2008). The consideration of fire regimes with regard to grassland arthropod assemblages is often nonfocal (Fuhlendorf et al., 2009; Joern and Laws, 2012), as the use of prescribed burning regimes has been recommended mainly for the purpose of increasing vegetative growth and reducing the impact of runaway fires in protected and agricultural areas (Trollope, 1993; Lechmere-Oertel, 2014).

Prescribed fires in the GGHNP are performed on the recommendation of SANParks-associated ecologists, based on each area’s vegetative composition, available burnable biomass and burning season. Burning frequencies are dependent on risk assessments and, in the case of the burnt study site, the need for fire breaks near infrastructure, and are therefore different for each burning site within the park (SANParks personnel, personal communication, 2019[3]). In this study, the percentage of soil organic matter had an effect on soil arthropod diversity over time, suggesting that fires may alter assemblages as pyrogenic carbon deposition increases with each fire. However, the differences in soil mineralogy between sites could be the result of different factors (e.g. site location, soil type differences, etc.), and may not necessarily be attributed fully to the effect of fire. It is possible that, given that assemblages showed increased abundance and diversity in the burnt site, fire could promote soil populations, possibly over time.

Although prescribed burnings within this area are performed upon the basis of maintaining natural, cultural and biodiversity components of the ecosystem (SANParks, 2013), monitoring of fire regimes in differing habitats of protected areas should be considered, as the full effect of repeated burnings upon soil arthropods and their functions within GGHNP grasslands is still unclear. Long-term monitoring of the effect of prescribed burning on soil arthropod assemblages becomes necessary in this regard.

2.5 Conclusion

The results of this study supported the hypotheses put forward in that soil arthropod abundance and species richness initially decreased immediately post-fire within the burnt site. Certain soil minerals, as well as soil temperature, showed significant interaction with soil arthropod diversity post-fire within the burnt site, however, soil moisture showed no significant effect. Despite this, it is highly likely that soil type and locality were more influential in affecting soil arthropod diversity than soil mineral change. Although species groups between

[3] D. Nariandas, verbal communication, Senior Section Range/Conservation Manager - Golden Gate Highlands National Park, 22 August 2019.

the burnt and non-burnt sites overlapped, with similarities in assemblages being evident, taxon groups did not display equal tolerance to fire effects. The maintained decreased diversity and less variation in assemblage structure of the burnt site indicates this area as being possibly comprised of more individuals from resilient arthropod groups such as mites. The direct disturbance of fire may favour opportunistic feeders such as formicid groups, and groups of soil-dwelling arthropods could give insight into patterns of recovery, as a medium to trace changes in burnt environments if monitored over a long-term period within the GGHNP. The results are highly specific to the GGHNP and could differ for other areas in South Africa. Nevertheless, we can recommend that a longer survey be required in order to gauge the full effects of yearly prescribed burning on changes in soil-dwelling assemblages within montane grasslands of the GGHNP over a number of years.

2.6 References

Ajee, J.K., 1993. Fire Ecology of Pacific Northwest Forests. Island Press, Washington.

Alcañiz, M., Outeiro, L., Francos, M., Úbeda, X., 2018. Effects of prescribed fires on soil properties, a review. Sci. Total Environ. 613–614, 944–957.

Ahlgren, I.F., 1974. The Effect of Fire on Soil Organisms. In: Kozlowski, T.T., Ahlgren, C.E. (Eds.) Fire and Ecosystems. Academic Press, London, pp. 47–72.

Anderson, A.N., 1991. Responses of ground–foraging ant communities to three experimental fire regimes in a savanna forest of tropical Australia. Biotropica 23, 575–585.

Apigian, K.O., Dahlsten, D.L., Stephens, S.L., 2006. Fire and fire surrogate treatment effects on leaf litter arthropods in a western Sierra Nevada mixed-conifer forest. Forest Ecol. Manag. 221, 110–122.

Arnan, X., Cerdá, X., Rodrigo, A., Retana, J., 2013. Response of ant functional composition to fire. Ecography 36, 1182–1192.

Badejo, M.A., 1990. Seasonal abundance of soil mites (Acarina) in two contrasting environments. Biotropica 22, 382–390.

Badenhorst, H., 2016. Mesofaunal Assemblages in Soil of selected Crops under diverse Cultivation Practices in Central South Africa, with Notes on Collembola Occurrence and Interactions. MSc dissertation. University of the Free State, Bloemfontein.

Barbercheck, M.E., Neher, D.A., Anas, O., El–Allaf, S.M., Weicht, T.R., 2009. Response of soil invertebrates to disturbance across three resource regions in North Carolina. Environ. Monit. Assess. 152, 283–298.

Bardgett, R.D., Cook, R., 1998. Functional aspects of soil animal diversity in agricultural grasslands. Appl. Soil Ecol. 10, 263–276.

Barratt, B.I.P., Tozer, P.A., Wiedemer, R.L., Ferguson, C.M., Johnstone, P.D., 2006. Effect of fire on microarthropods in New Zealand indigenous grassland. Rangeland Ecol. Manag. 59, 383–391.

Behan-Pelletier, V.M., 1999. Oribatid mite biodiversity in agroecosystems, role for bioindication. Agr. Ecosyst. Environ. 74, 411–423.

Bird, M.I., Wynn, J.G., Saiz, G., Wurster, C.M., McBeath, A., 2015. The Pyrogenic Carbon Cycle. Annu. Rev. Earth Planet. Sci. 43, 273–298.

Blakely, J.K., Neher, D.A., Spongberg, A.L., 2002. Soil invertebrates and microbial communities, and decomposition as indicators of polycyclic aromatic hydrocarbon contamination. Appl. Soil Ecol. 21, 71–88.

Brussaard, L., 1998. Soil fauna, guilds, functional groups and ecosystem processes. Appl. Soil Ecol. 9, 123–135.

Brussaard, L., Behan–Pelletier, V.M., Bignell, D.E., Brown, V.K., Didden, W., Folgarait, P., Fragoso, C., Freckman, D.W., Gupta, V.V.S.R., Hatorri, T., Hawksworth, D.L., Klopatek, C., Lavelle, P., Malloch, D.W., Rusek, J., Söderström, B., Tiedje, J.M., Virginia, R.A., 1997. Biodiversity and ecosystem functioning in soil. Swed. Acad. Sci. 26, 563–570.

Buddle, C.M., Langor, D.W., Pohl, G.R., Spence, J.R., 2005. Arthropod responses to harvesting and wildlife: implications for emulation of natural disturbance in forest management. Biol. Cons. 128, 346–357.

Buddle, C.M., Spence, J.R., Langor, D.W., 2000. Succession of boreal forest spider assemblages following wildfire and harvesting. Ecography 23, 424–436.

Burbidge, A.H., Leicester, K., McDavitt, S., Majer, J.D., 1992. Ants as indicators of disturbance at Yanchep National Park, Western Australia. J. R. Soc. West. Aust. 75, 89–95.

Buschke, F.T., Seaman, M.T., 2011. Functional feeding groups as a taxonomic surrogate for a grassland arthropod assemblage. Afr. Invert. 52, 217–228.

Callaham, M.A., Blair, J.M., Todd, T.C., Kitchen, D.J., Whiles, M.R., 2003. Macroinvertebrates in North American tallgrass prairie soils: effects of fire, mowing, and fertilization on density and biomass. Soil Biol. Biochem. 35, 1079–1093.

Camann, M.A., Gillette, N.E., Lamoncha, K.L., Mori, S.R., 2008. Response of forest soil Acari to prescribed fire following stand structure manipulation in the southern Cascade Range. Can. J. For. Res. 38, 956–968.

Caruso, T., Taormina, M., Migliorini, M., 2012. Relative role of deterministic and stochastic determinants of soil animal community: a spatially explicit analysis of oribatid mites. J. Anim. Ecol. 81, 214–221.

Caut, S., Jowers, M.J., Arnan, X., Pearce-Duvet, J., Rodrigo, A., Cerda, X., Boulay, R.R., 2013. The effect of fire on ant trophic assemblage and sex allocation. Ecol. Evol. 4, 35–49.

Certini, G., 2005. Effects of fire on properties of forest soils, a review. Oecologia 143, 1–10.

Chikoski, J.M., Ferguson, S., Meyer, L., 2006. Effects of water addition on soil arthropods and soil characteristics in a precipitation-limited environment. Acta Oecol. 30, 203–211.

Clapperton, M.J., Kanashiro, D.A., Behan–Pelletier, V.M., 2002. Changes in abundance and diversity of microarthropods associated with Fescue Prairie grazing regimes. Pedobiologia 46, 496–511.

Coleman, T.W., Rieske, L.K., 2006. Arthropod response to prescription burning at the soil-litter interface in oak-pine forests. Forest Ecol. Manag. 233, 52–60.

Collins, B.M., Kelly, M., van Wagtendonk, J.W., Stephens, S.L., 2007. Spatial patterns of large natural fires in Sierra Nevada wilderness areas. Landsc. Ecol. 22, 545–557.

Colwell, R.K., 2013. EstimateS, statistical estimation of species richness and shared species from samples. URL: https://viceroy.eeb.uconn.edu/EstimateS.

Convey, P., Block, W., Peat, H.J., 2003. Soil arthropods as indicators of water stress in Antartic terrestrial habitats? Glob. Change Biol. 9, 1718–1730.

Coutinho, L.M., 1990. Fire in the ecology of Brazilian cerrado. In: Goldamer J.G. (Ed.) Fire in the Tropical Biota - Ecosystem Processes and Global Challenges. Springer-Verlag, Berlin, pp. 81–105.

Dooley, S.R., Treseder, K.K., 2012. The effect of fire on microbial biomass: a meta-analysis of field studies. Biogeochemistry 109, 49–61.

Dove, N.C., Hart, S.C., 2017. Fire reduces fungal species richness and in situ mycorrhizal colonization: a meta-analysis. Fire Ecol. 13, 37–65.

Ellwood, M.D.F., Manica, A., Foster, W.A., 2009. Stochastic and deterministic processes jointly structure tropical arthropod communities. Ecol. Lett. 12, 277–284.

Evans, T.R., 2017. Management Implications for Invertebrate Assemblages in the Midwest American Agricultural Landscape. PhD thesis. Leiden University, Leiden.

Ferrenberg, S., Martinez, A.S., Faist, A.M., 2016. Aboveground and belowground arthropods experience different relative influences of stochastic versus deterministic community assembly processes following disturbance. PeerJ 4, e2545. https://doi.org/10.7717/peerj.2545.

Ferrenberg, S.M., Schwilk, D.W., Knapp, E.E., Groth, E., Keeley, J.E., 2006. Fire decreases arthropod abundance but increases diversity, early and late season prescribed fire effects in a Sierra Nevada mixed-conifer forest. Fire Ecol. 2, 79–102.

Frizzo, T.L.M., Campos, R.I., Vasconcelos, H.L., 2012. Contrasting effect of fire on arboreal and ground-dwelling ant communities of a Neotropical savanna. Biotropica 44, 254–261.

Gergóc, V., Hufnagel, L., 2009. Application of oribatid mites as indicators (review). Appl. Ecol. Environ. Res 7, 79–98.

Gongalsky, K.B., Persson, T., 2013. Recovery of soil macrofauna after wildfires in boreal forests. Soil Biol. Biochem. 57, 182–191.

Gongalsky, K.B., Zaitsev, A.S., 2016. The role of spatial heterogeneity of the environment in soil fauna recovery after fires. Dokl. Earth Sci. 471, 1265–1268.

Gulvik, M., 2007. Mites (Acari) as indicators of soil biodiversity and land use monitoring, a review. Pol. J. Ecol. 55, 415–440.

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. Afr. Zool. 4, 281–292.

Hammer, Ø., Harper, D.A.T., Ryan, P.D., 2001. PAST, Palaeontological statistics software package for education and data analysis. Palaeontol. Electron. 4, 1–9.

Hanula, J.L., Wade, D.D., 2003. Influence of long-term dormant-season burning and fire exclusion on ground-dwelling arthropod populations in longleaf pine flatwoods ecosystems. Forest Ecol. Manag. 175, 163–184.

Harper, M.G., Dietrich, C.H., Larimore, R.L., Tessene, P.A., 2000. Effects of prescribed fire on prairie arthropods, an enclosure study. Nat. Areas J. 20, 325–335.

Hermann, S.M., van Hook, T., Flowers, R.W., Brennan, L.A., Glitzenstein, J.S., Streng, D.R., Walker, J.L., Myers, R.L., 1998. Fire and biodiversity, studies of vegetation and arthropods. T. N. Am. Wildl. Nat. Res. 63, 304–401.

Huebner, K., Lindo, Z., Lechowicz, M.J., 2012. Post-fire succession of collembolan communities in a northern hardwood forest. Eur. J. Soil Biol. 48, 59–65.

Hugo-Coetzee, E.A., Avenant, N.L., 2011. The effect of fire on soil oribatid mites (Acari: Oribatida) in a South African grassland. Zoosymposia 6, 210–220.

Ingimarsdóttir, M., Caruso, T., Ripa, J., Magnúsdóttir, O.B., Migliorini, M., Hedlund, K., 2012. Primary assembly of soil communities: disentangling the effect of dispersal and local environment. Oecologia 170, 745–754.

Janion-Scheepers, C., Bengsston, J., Leinaas, H.P., Deharveng, L., Chown, S.L. 2016. The response of springtails to fire in the fynbos of the Western Cape, South Africa. Appl. Soil Ecol. 108, 165–175.

Kaiser, W., Avenant, N., Haddad, C.R., 2009. Assessing the ecological integrity of a grassland ecosystem, refining the SAGraSS method. Afr. J. Ecol. 47, 308–317.

Keeley, J.E., 2009. Fire intensity, fire severity and burn severity, a brief review. Int. J. Wildland Fire 18, 116–126.

Kim, J.W., Jung, C., 2008. Abundance of soil microarthropods associated with forest fire severity in Samchoek, Korea. J. Asia-Pac. Entomol. 11, 77–81.

Kimberling, D.N., Karr, J.R., Fore, L.S., 2001. Measuring human disturbance using terrestrial invertebrates in shrub-steppe of eastern Washington (USA). Ecol. Indic. 1, 63–81.

King, J.R., Andersen, A.N., Cutter, A.D., 1998. Ants as bioindicators of habitat disturbance, validation of the functional group model for Australia's humid tropics. Biodivers. Conserv. 7, 1627–1638.

Klopatek, J. M., 1987. Nitrogen mineralization and nitrification in mineral soils of pinyon-juniper ecosystems. Soil Sci. Soc. Am. J. 51, 453–457.

Knicker, H., 2007. How does fire affect the nature and stability of soil organic nitrogen and carbon? A review. Biogeochemistry 85, 91–118.

Knicker, H., Almendros, G., González-Villa, F.J., González-Pérez, J.A., Polvillo, O., 2006. Characteristic alterations of quantity and quality of soil organic matter caused by forest fires in continental Mediterranean ecosystems, a solid-state 13C NMR study. Eur. J. Soil Sci. 57, 558–569.

Koponen, S., 2005. Early succession of a boreal spider community after forest fire. J. Arachnol. 33, 230–235.

Kozlowski, T.T., Ahlgren, C.E., 1974. Fire and Ecosystems. Academic Press, London.

Kral, K.C., Limb, R.F., Harmon, J.P., Hovick, T.J., 2017. Arthropods and fire, previous research shaping future conservation. Rangeland Ecol. Manag. 70, 589–598.

Kuznetsova, A., Brockhoff, P.B., Christensen, R.H.B., 2017. lmerTest Package, Tests in Linear Mixed Effects Models. J. Stat. Softw. 82, 1–26.

Lavelle, P., Decaëns, T., Aubert, M., Barot, S., Blouin, M., Bureau, F., Margerie, P., Mora, P., Rossi, J.P., 2006. Soil invertebrates and ecosystem services. Eur. J. Soil Biol. 42, S3–S15.

Lisa, C., Paffetti, D., Nocentini, S., Marchi, E., Bottalico, F., Fiorentini, S., Travaglini, D., 2015. Impact of wildfire on the edaphic microarthropod community in a Pinus pinaster forest in central Italy. iForest 8, 874–883.

Maaß, S., Migliorini, M., Rillig, M.C., Caruso, T., 2014. Disturbance, neutral theory, and patterns of beta diversity in soil communities. Ecol. Evol. 4, 4766–4774.

Malmström, A., Persson, T., Ahlström, K., 2008. Effects of fire intensity on survival and recovery of soil microarthropods after a clearcut burning. Can. J. Forest Res. 38, 2465–2475.

Malström, A., Persson, T., Ahlström, K., Gongalsky, K.B., Bengtsson, J., 2009. Dynamics of soil meso- and macrofauna during a 5-year period after clear-cut burning in a boreal forest. Appl. Soil Ecol. 43, 61–74.

Matiax-Solera, J., Guerrero, C., García-Orenes, F., Bárcenas, G.M., Torres, M.P., 2009. Forest fire effects on soil microbiology. In: Cerdà, A., Robichaud, P.R. (Eds.) Fire Effects on Soil and Restoration Strategies. CRC Press, Boca Raton, pp. 133–176.

Matsuda, T., Turkschak, G., Brehme, C., Rochester, C., Mitrovich, M., Fisher, R., 2011. Effects of large-scale wildfires on ground foraging ants (Hymenoptera, Formicidae) in Southern California. Environ. Entomol. 40, 204–216.

McCullough, D.G., Werner, R.A., Neumann, D., 1998. Fire and insects in Northern and boreal ecosystems of Northern America. Annu. Rev. Entomol. 43, 107–127.

McSorley, R., 1993. Short-term effects of fire on the nematode community in a pine forest. Pedobiologia 37, 39–48.

Moretti, M., Duelli, P., Obrist, M.K., 2006. Biodiversity and resilience of arthropod communities after fire disturbance in temperate forests. Oecologia 149, 312–327.

Moretti, M., Martin, K., Duelli, P., 2004. Arthropod biodiversity after forest fires, winners and losers in the winter fire regime of the southern Alps. Ecography 27, 173–186.

Mucina, L., Hoare, D.B., Lötter, M.C., Du Preez, P.J., Rutherford, M.C., Scott-Shaw, C.R., Bredenkamp, G.J., Powrie, L.W., Scott, L., Camp, K.G.T., Cilliers, S.S., Bezuidenhout, H., Mostert, T.H., Siebert, S.J., Winter, P.J.D., Burrows, J.E., Dobson, L., Ward, R.A., Stalmans, M., Oliver, E.G.H., Siebert, F., Schmidt, E., Kobisi, K., Kose, L., 2006. Grassland biome. In: Mucina, L., Rutherford, M.C. (Eds.) The Vegetation of South Africa, Lesotho and Swaziland. SANBI, Pretoria, pp. 349–437.

Nakamura, A., Catterall, C.P., House, A.P.N., Kitching, R.L. Burwell, C.J., 2007. The use of ants and other soil and litter arthropods as bio–indicators of the impacts of rainforest clearing and subsequent land use. J. Insect. Conserv. 11, 177–186.

New, T.R., Sands, A.L., Greenslade, P., Neville, P.J., York, A., Collett, N.G., 2010. Planned fires and invertebrate conservation in south east Australia. J. Insect Conserv. 14, 567–574.

O'Lear, H.A., Blair, J.M., 1999. Responses of soil microarthropods to changes in soil water availability in tallgrass prairie. Biol. Fert. Soils 29, 207–217.

Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson, G.L., Solymos, P., Stevens, M.H.H., Szoecs, E., Wagner, H.,

2019. Vegan, Community Ecology Package. R package version 2.5–5. https://CRANS.R–project.org/package=vegan.

Paoletti, M.G., Favretto, M.R., Stinner, B.R., Purrington, F.F., Bater, J.E., 1991. Invertebrates as bioindicators of soil use. Agr. Ecosyst. Environ. 34, 341–362.

Parr, C.L., Chown, S.L., 2001. Inventory and bioindicator sampling: testing pitfall and Winkler methods with ants in a South African savanna. J. Insect Conserv. 5, 27–36.

Pfiffner, L., Luka, H., 2000. Overwintering of arthropods in soils of arable fields and adjacent semi-natural habitats. Agr. Ecosyst. Environ. 78, 215–222.

Philpott, S.M., Perfecto, I., Armbrecht, I., Parr, C.L., 2010. Ant diversity and function in disturbed and changing habitats. In: Lach, L., Parr, C., Abbott, K. (Eds.) Ant Ecology. Oxford University Press, Oxford, pp. 137–156.

Pivello, V., Oliveras, I., Miranda, H.S., Haridasan, M., Sato, M.N., 2010. Effect of fires on soil nutrient availability in an open savanna in Central Brazil. Plant Soil 337, 111–123.

Poole, R.W., 1974. An Introduction to Quantitative Ecology. McGraw-Hill, New York.

Pressler, Y., Moore, J.C., Cotrufo, M.F., 2019. Belowground community responses to fire: meta-analysis reveals contrasting responses of soil microorganisms and mesofauna. Oikos 128, 309–327.

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

Rademeyer, C., van Zyl, W., 2014. Golden jubilee for Golden Gate - a concise history of Golden Gate Highlands National Park, 1963 to 2013. Mediterr. J. Soc. Sci. 5, 1169–1177.

Ramos-Neto, M.B., Pivello, V.R., 2000. Lighting fires in the Brazilian Savannah National Park, rethinking management strategies. Environ. Manag. 26, 675–684.

Ribas, C.R., Campos, R.B.F., Schmidt, F.A., Solar, R.R.C., 2012. Ants as indicators in Brazil, a review with suggestions to improve the use of ants in environmental monitoring programs. Psyche 2012, 1–23. https://doi.org/10.1155/2012/636749.

Richardson, R.J., Holliday, N.J., 1982. Occurrence of carabid beetles (Coleoptera: Carabidae) in a boreal forest damaged by fire. Can. Entomol. 113, 509–514.

Ritz, K., Young, I.M., 2004. Interactions between soil structure and fungi. Mycology 18, 52–59.

Roberts, B.R., 1969. The vegetation of the Golden Gate Highlands National Park. Koedoe 12, 15–28.

Roberts, S.J., 2000. Tropical Fire Ecology. Prog. Phys. Geog. 24, 281–288.

Ruf, A., 1998. A maturity index for predatory soil mites (Mesostigmata, Gamasina) as an indicator of environmental impacts of pollution on forest soils. Appl. Soil Ecol. 9, 447–452.

Saint-Germain, M., Larrivée, M, Drapeau, P., Fahrig, L., Buddle, C.M., 2005. Short-term response of ground beetles (Coleoptera, Carabidae) to fire and logging in a spruce-dominated boreal landscape. Forest Ecol. Manag. 212, 118–126.

Santorufo, L., van Gestel, C.A.M., Rocco, A., Maisto, G., 2012. Soil invertebrates as bioindicators of urban soil quality. Environ. Pollut. 161, 57–63.

Sgardelis, S.P., Pantis, J.D., Argyropoulou, M.D., Stamou, G.P., 1995. Effects of Fire on soil macroinvertebrates in a Mediterranean phryganic ecosystem. Int. J. Wildland Fire 5, 113–121.

Siemann, E., Haarstad, J., Tilman, D., 1997. Short-term and long-term effects of burning on oak savannah arthropods. Am. Midl. Nat. 137, 349–361.

Stork, N.E, Eggleton, P., 1992. Invertebrates as determinants as indicators of soil quality. Am. J. Altern. Agric. 7, 38–47.

Swengel, A., 2001. A literature review of insect responses to fire, compared to other conservation managements of open habitats. Biodivers. Conserv. 10, 1141–1169.

Triplehorn, C.A., Johnson, N.F., 2005. Borror and DeLong's Introduction to the Study of Insects. 7th Edition. Thompson Brooks/Cole, Belmont.

Trollope, W.S.W., 1993. First regime of the Kruger National Park for the period 1980-1992. Koedoe 36, 45–52.

Tullgren, A. 1918. Ein sehr einfacher Ausleseapparat fur terricole Tierformen. Z. Angew. Entomol. 4, 149–150.

Walker, L.R., 2012. The Biology of Disturbance Ecology. Oxford University Press, New York.

Wikars, L.O., Schimmel, J., 2001. Immediate effects of fire-severity on soil invertebrates in cut and uncut pine forests. Forest Ecol. Manag. 141, 189–200.

Yaninek, J.S., Herren, H.R., Gutierrez, A.P., 1989. Dynamics of *Mononychellus tanajoa* (Acari: Tetranychidae) in Africa, seasonal factors affecting phenology and abundance. Environ. Entomol. 18, 625–632.

Zaitsev, A.S., Gongalsky, K.B., Persson, T., Bengtsson, J., 2014. Connectivity of litter islands remaining after a fire and non-burnt forest determines the recovery of soil fauna. App. Soil Ecol. 83, 101–108.

Chapter 3:

Soil-dwelling arthropods as indicators of erosion in a South African grassland habitat

Sylvia S. van der Merwe[a], Vaughn R. Swart[a], Emile Bredenhand[b], Charles R. Haddad[a]

[a] University of the Free State, Bloemfontein Campus, Bloemfontein, South Africa.

[b] University of the Free State, Qwaqwa Campus, Phuthaditjhaba, South Africa.

Published in Pedobiologia, Elsevier. In press.

Abstract

Soil erosion is a natural process that drives the formation of lowland terrains in mountainous regions. Its role in ecosystem degradation has been strongly debated, due to the significant effect it has on a range of land types. Soil arthropods are effective indicators of several disturbance types. However, little is known about arthropods in eroded sites in South Africa. The aim of this study was to identify possible indicators of erosion in the Golden Gate Highlands National Park, South Africa, and to determine differences in soil-dwelling arthropod assemblages found in non-rehabilitated and rehabilitated sites over a 24-month period. A total of 5661 soil arthropods were sampled during the study, with the highest species richness recorded in non-rehabilitated eroded sites. A higher number of unique arthropod species were observed from the non-rehabilitated sites compared to other site types. Overlaps in assemblage structure between site types differed, depending on site locality. IndVal results indicated a single strong indicator, the mite *Speleorchestes meyerae* Theron and Ryke, 1969, for the non-rehabilitated eroded sites. Linear model analyses of site richness with mineralogy identified the significance of phosphorus (in the form of P_2O_5) in eroded sites. Given these results, soil mineralogy in conjunction with soil arthropod richness of eroded sites could be used to investigate the effect of erosion and form a basis for future study in soil biodiversity and function.

Keywords: Biodiversity, IndVal, Mineralogy, Rehabilitation, Species Richness, Soil Biota

3.1. Introduction

Soil erosion is a natural occurrence that drives the creation of new landscape types and the formation of mountainous areas (Mhangara et al., 2012). However, soil erosion is a serious problem throughout the world’s terrestrial ecosystems, due to the increased rate of erosion during recent years (Pimentel and Kounang, 1998; Pimentel et al., 2004; Durán Zuazo and Rodríguez Pleguezuelo, 2008). Soil erosion reduces the quality of soils, in turn, decreasing the productivity of agricultural, natural, and forest ecosystems (Pimentel, 2006).

In South Africa, soil erosion is one of the most significant environmental problems causing ecosystem degradation (Meadows, 2003; Le Roux et al., 2007, Dlamini et al., 2011). The levels and severity of erosion in South African landscapes are dependent on soil composition, as well as the sources of erosion, such as intensity and frequency of rainfall, anthropogenic influences such as overgrazing and agricultural practices, and wind speeds (Meadows, 2003; Le Roux et al., 2007). Research strongly suggests that over 70% of South African landscapes are affected by soil erosion to varying degrees (Garland et al., 2000; Le Roux et al., 2008). Soil erosion by water flow is a major environmental issue that threatens the ecological function of terrestrial and aquatic systems (Mhangara et al., 2012). The Golden Gate Highlands National Park (GGHNP) in South Africa is well-known for its sandstone cliffs and formations across the western side of the park. This type of sandstone produces a shallow sandy soil with very low fertility (Roberts, 1966) that is notoriously susceptible to erosion losses (Roberts, 1969; SANParks, 2013). The park comprises many smaller rivulets running through the landscape, which form part of the water bodies found within the area. Although rivers around the GGHNP are situated in the upper catchment area, South African National Parks (SANParks) Authorities are concerned that finer soils could be carried into the aquatic sectors of the park during the heavier rainfall season, and measures are taken to prevent excessive deposition into these aquatic areas (SANParks, 2013).

Edaphic biota play a vital role in soil functions, which include nutrient element cycling, organic matter breakdown, and humus formation (Menta et al., 2011). In addition, soil arthropods also affect the aeration, porosity, infiltration and distribution of organic matter in soils, subsequently modifying soil structure and improving its fertility. As a result, below-ground diversity is essential for the function of above-ground ecosystems (van Straalen, 2004). However, soil arthropods are not readily considered when looking at erosion, despite their below-ground endemism and role in major ecosystem functions (Orgiazzi and Panagos, 2018).

Grasslands have been indicated as important habitats in the protection of soil biodiversity (Menta et al., 2011), highlighting the importance of monitoring grassland health in conjunction with soil biota assemblages. Soil arthropods are useful indicators of a number of disturbance types in varying terrains, due to their rapid responses to changes in an environment (Paoletti et al., 1991; Stork and Eggleton, 1992; Kremen et al., 1993; Convey et al., 2003; Gulvik, 2007; Nakamura et al., 2007). It is vital that scientists identify bioindicators in grasslands in order to monitor areas that may be at risk of deterioration or, at least, changes that may alter the nature of ecosystems. The GGHNP is currently the only national park in the Free State Province (SANParks, 2013), and may play a significant role in the conservation of soil biota groups in the area. However, soil biota assemblages in this area are poorly understood and very few studies have been done on soil-dwelling arthropods as indicators within protected areas in South Africa.

The main aim of this study was to identify possible soil-dwelling arthropod indicators for soil erosion and the differences between non-eroded, non-rehabilitated and rehabilitated sites in the GGHNP. Differences in soil arthropod assemblages of the eastern and western parts of the park are also briefly discussed.

3.2. Materials and Methods

3.2.1. Study area and period

The study was carried out in the GGHNP, located in the eastern parts of the Free State Province, South Africa, bordering with Lesotho to the south (Fig. 3.1). It covers an area of 340 km^2 and features several deeply eroded sandstone outcroppings and cliffs, alongside large expanses of undisturbed grassland hills and valleys. The park comprises small streams that flow through sections of the landscape, with temporary water bodies forming during peak rainfall seasons. Erosion rehabilitation methods, such as rock walling and silt netting, are often evident near roadsides and water bodies in the park. Terrain largely differs between the eastern and western parts of the park, with flatter grasslands in the east and more highland montane grasslands in the west.

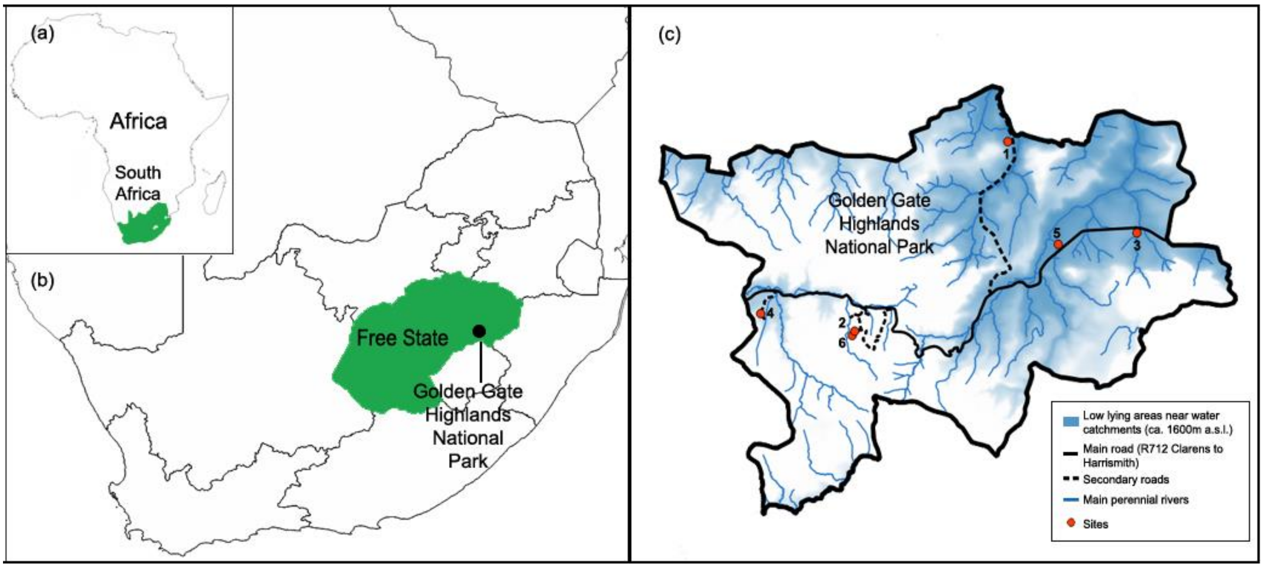

Fig. 3.1: Location of study areas: (a) country of South Africa, (b) proximity of the Golden Gate Highlands National Park in the Free State Province, and (c) locality of site 1–6, with major perennial rivers and roads shown. (Altitudinal map obtained from Web GIS (http://www.webgis.com/srtm3.html), river map obtained, with permission, from SANParks Scientific Services in South Africa, and sites, roads and river lines processed with QGIS, version 2.18.15.

Sites were selected to incorporate a variety of site types (non-rehabilitated eroded, rehabilitated eroded, and non-eroded undisturbed) from each of the sections of the park (Supplementary material, Chapter 3, Fig. S3.1). Each rehabilitated eroded site was chosen according to similar erosion rehabilitation methods, namely silt netting and rock-wall gabions. All erosion sites selected during this study were sites affected by sheet and gully erosion through natural processes such as rainfall. Areas were specifically characterised as follows:

3.2.1.1 Non-rehabilitated eroded sites (NRE)

Site 1 (28°25.834’S, 28°41.832’E, elevation: 1614 m) was located in the north-eastern part of the park (Fig. 3.1c), with large levels of furrow erosion near an east-facing slope of a hill (Supplementary material, Chapter 3, Fig S3.1 - 1). Soils were characterised as fine sandy-clay soils, with high compaction zones around furrow areas. Daily disturbances brought about by large groups of livestock were frequent in and around the area. Site 2 (28°31.291’S, 28°37.138’E, elevation: 2061 m) was a high altitude site located within the western peaks of the park, with shallow sandy soils interspersed with shale and rocky deposits (Supplementary material, Chapter 3, Fig S3.1 - 2). Shallow erosion occurred downslope, with a small area of soil erosion. Although wildlife occurred around the site on some occasions, levels of disturbance were limited. Main grasses surrounding the sites included *Eragrostis chloromelas* and *E. viscosa.*

3.2.1.2 Rehabilitated eroded sites (RE)

Site 3 (28°25.284’S, 28°45.477’E, elevation: 1731 m; Supplementary material, Chapter 3, Fig S3.1 - 3) was a lower altitude site in the eastern sector near the border of the park. This site included rehabilitation structures over five years old at the time of commencement of the study (SANParks personnel, personal communication, 2016[1]). Soils were initially heavy clay soil with highly compacted sandy soils around the site. However, soils within furrows and gullies were relatively low compacted. The site was levelled in an attempt to implement new

[1] T. Nsibande, verbal communication, Working for Ecosystems Park Ranger - Golden Gate Highlands National Park, 21 September 2016.

rehabilitation structures during July 2018, incorporating sandy soils from around the eroded furrow into the area, and renewed rehabilitation methods were implemented in August 2018 (Supplementary material, Chapter 3, Fig. S3.2). Site 4 (28°30.730'S, 28°34.414'E, elevation: 1875 m; Supplementary material, Chapter 3, Fig S3.1 - 4) was a progressed eroded site on the far western part of the park, with fine sandy soils deposited over a vast area. This site included rehabilitation structures implemented over seven years before the commencement of this study, with interval implementation due to the large area of the site (SANParks personnel, personal communication, 2016[2]). The site was mainly rehabilitated using rock walling and strategic placement of silt netting, with various levels of implementation across furrows down a slope. *E. chloromelas* and *E. viscosa* grasses generally surrounded both sites.

3.2.1.3 Non-eroded undisturbed sites (NAT)

Site 5 (28°28.833'S, 28°43.280'E, elevation: 1721 m; Supplementary material, Chapter 3, Fig S3.1 - 5) was characterised as a relatively flat grassland on the eastern side of the park. Frequent grazing and movement of wildlife, i.e. ungulates such as zebra, black wildebeest, springbuck, eland (SANParks, 2012), were observed in and around the area, mostly during early morning and late afternoon. The area had highly compacted sandy soils with low moisture levels. Site 6 (28°31.321'S, 28°37.118'E, elevation: 2064 m; Supplementary material, Chapter 3, Fig S3.1 - 6) was a high altitude natural sandy grassland on a slope in the western sector of the park, within the Blesbok loop drive trail. Main grasses in both sites include *E. chloromelas* and *E. viscosa*, interspersed by tufts of *Themeda triandra*. Criteria for these sites were mainly based on similar ratios of these grasses present at both sites.

[2] T. Nsibande, verbal communication, Working for Ecosystems Park Ranger - Golden Gate Highlands National Park, 21 September 2016.

3.2.2 Arthropod survey and sampling

A soil arthropod survey was conducted to observe occurrence and abundance overall to establish the relevant soil faunal groups, identify possible indicators of erosion, and determine the effect of erosion rehabilitation on these assemblages. Soil samples were taken from the six sites characterised under the three treatment types once every month over the course of 24 months to incorporate eight seasons of sampling. A total of ten samples per site were taken each month and set through arthropod extraction. A single soil sample was characterised as a soil quantity of between 400 and 500 g taken within the top 10 cm of soil and within a 10 cm radius of a chosen sampling spot. A total of five incorporated samples were taken in a linear design along each eroded site type on each date, with two sub-samples taken within the sampling point pooled together to form an incorporated sample. Each sampling sequence was conducted from the highest point in the eroded furrows to the lowest level of soil deposition. Non-eroded undisturbed sites were sampled in a similar technique along their terrains.

All samples were transported to the Department of Zoology and Entomology, University of the Free State, South Africa, and placed into individual Berlese-Tullgren funnels with connected storage bottles containing 70% ethyl alcohol (Tullgren, 1918; Triplehorn and Johnson, 2005; Badenhorst, 2016) for a period of seven days, to allow for sufficient soil arthropod extraction. Samples were sorted to arthropod order, family and morphospecies level, with special consideration given to groups of mites (Oribatida, Mesostigmata and Prostigmata), Collembola and other insects. Species that were identified as indicators of each site type by Indicator Analysis were identified to genus level and species, if possible, with the assistance of specialists and literature (Theron and Ryke, 1969; Olivier and Theron, 1989; Triplehorn and Johnson, 2005; Dunger and Schlitt, 2011; Hugo-Coetzee, 2013; Badenhorst, 2016; Slingsby, 2017). All Collembola specimens were stored at the Iziko South African Museum, Cape Town.

3.2.3 Soil analysis

Soil samples were taken from each study site in the eastern section of the park for mineralogy analysis, in order to correlate soil minerals (SiO_2, Al_2O_3, CaO, K_2O, TiO_2, Fe_2O_3, MgO, MnO, P_2O_5, Na_2O, % Organic Matter based on Loss on Ignition (LOI)) with soil-dwelling arthropod richness. Three soil samples were taken within a 10 cm depth and 10 cm radius of a chosen sampling spot every three months for a full seasonal cycle (July 2017 – June 2018) in one non-rehabilitated eroded site (Site 1), one rehabilitated eroded site (Site 3) and a non-eroded undisturbed site (Site 5). Sites on the eastern sections of the park were concentrated on due to the higher richness and diversity sampled from these areas. Samples were transported to the University of the Free State and analysed by the Department of Geology, using X-ray Fluorescence (XRF) major element analysis. In addition, soil temperature and soil moisture per month was recorded over the study period using a handheld SMT-100 moisture and temperature probe meter.

3.2.4 Statistical analysis

In order to establish possible indicator groups for each site type, three analyses of association were run on species data obtained over the 24-month period. These included Dufrêne-Legendre Indicator Species analysis (IndVal), multi-level pattern analysis, and Pearson's phi coefficient of association.

IndVal was first proposed by Dufrêne and Legendre (1997) as an applicable stopping rule for clustering and determines indicator values through the fidelity and specificity of species in a given habitat. The multi-level pattern analysis is an extension of the original IndVal to investigate indicator species of individual sites, as well as their combinations (De Cáceres et al., 2010). In addition, several other indices are used to analyse the ecological preferences of species (De Cáceres and Legendre, 2009). Pearson's phi coefficient of association has been used in this regard to determine association in the form of fidelity, which differs from the fidelity used in the IndVal, between habitat type and species (Chytrý et al., 2002; De Cáceres, 2013). This measure is performed between two binary vectors, with species in the form of presence-absence data, to derive an ecological correlation (De Cáceres, 2013).

Each site was clustered into site groups (non-rehabilitated eroded, rehabilitated eroded, non-eroded undisturbed), and species abundance was tested across sites to establish possible groups of interest. Only genera falling under an indicator value of 30.0 were fully omitted from results, in addition to morphospecies that could not be identified to genus level. Strong indicators were considered as species scoring above 70% as a subjective benchmark value suggested by McGeogh et al. (2002). All association analyses were performed using the 'multipatt' function of the Indicspecies package in R, version 3.5.3 (De Cáceres and Legendre, 2009).

It has been advised that caution should be taken when reporting results on indicator species analyses (De Cáceres et al., 2010). Due to multiple testing issues associated with the use of several species for indicator analyses, corrections for multiple testing are strongly advised. As a result, all p-values obtained during association analyses by 'Bonferroni' correction were adjusted with the 'p.adjust' function in R, version 3.5.3 (R Core Team, 2019). Only species indicated to be significant after p-values were adjusted were reported.

Non-metric Multidimensional Scaling (3D) (NMDS) ordinations were conducted between site assemblages on the eastern and western sides of the park to identify how similar sites are within the terrains, with special consideration for overlaps between each site type. Rare species (species with an abundance of less than 0.2% over the 24-month period) were omitted from the analysis to reduce noise and effect of outliers in isolated sites. Unique and shared species were mapped to further express how assemblages may differ between site types overall. Analysis of Similarity (ANOSIM) using Bray-Curtis similarity index was conducted on soil arthropod assemblages for each site across the park. Species diversity was quantified by means of the Shannon-Wiener diversity index and mapped according to sites, site type and site location (east or west) to observe overall differences between soil arthropod diversity of each site. Shannon-Wiener diversity values, ANOSIM and NMDS were calculated using Paleontological Statistics (PAST), version 3.25 (Hammer et al., 2001), and diversity ranges were mapped using the GGPLOT2 package in R, version 3.5.3 (Wickham, 2016).

In order to identify a possible best fit correlation between soil arthropods and soil attribute data obtained from the areas investigated, Linear Modelling was performed between each tested soil mineral (11 minerals), together with soil temperature and soil moisture, and the obtained species richness of the eastern sectors of the park. A log transformation on species richness was applied to normalise the data. An Akaike Information Criterion (AIC) estimator was calculated for each model to identify the best correlation from the linear models and subsequently investigate possible compounds that could be used to indicate erosion, in conjunction with soil arthropods. Linear models were performed using the LME4 package in R, version 3.5.3 (Bates et al., 2015).

3.3 Results

A total of 5661 individuals were sampled during the 24-month study period, representing 13 orders and 62 families. The non-rehabilitated eroded sites had the highest species richness over the study period, with the lowest species richness sampled from the non-eroded undisturbed sites and the rehabilitated site located on the western side of the park (Fig. 3.2).

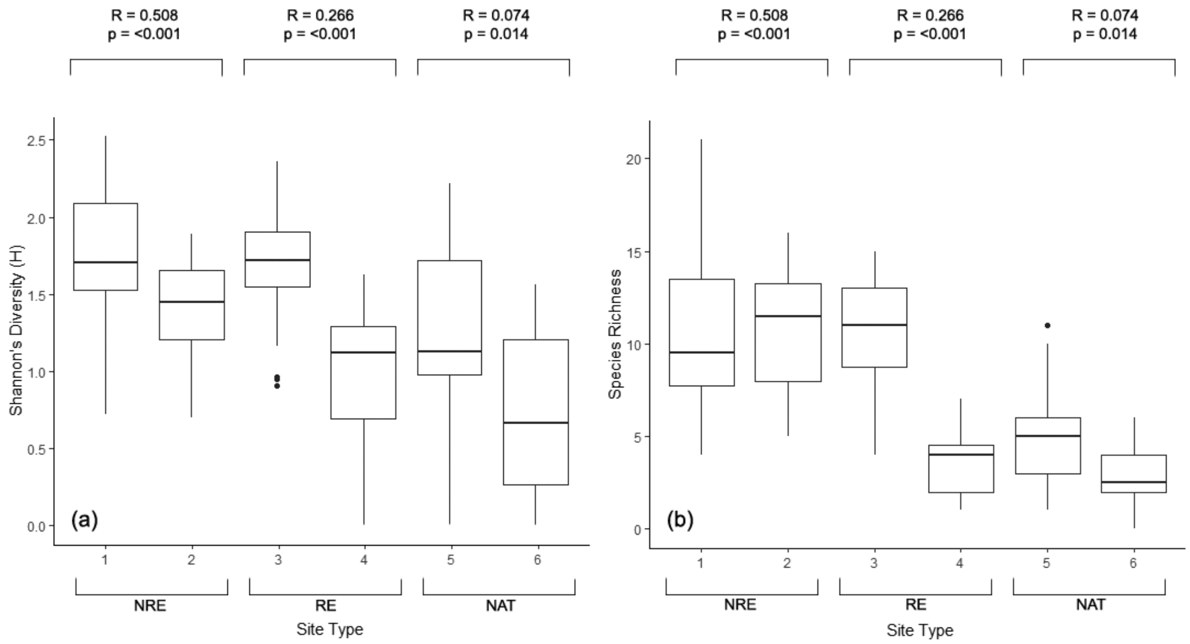

Fig. 3.2: Box plot analysis on soil arthropods sampled, showing (a) Shannon-Wiener diversity indices values, and (b) Species richness values per site and site type. P-values indicated were obtained by ANOSIM per site type, with associated R values. NRE - Non-rehabilitated eroded site; RE - rehabilitated eroded site; NAT - Non-eroded undisturbed site.

Of interest was the higher species diversity sampled from site 3. However, this could be explained by the flattening of the site halfway through the study, changing the site structure and displacing soil arthropods from the areas surrounding the site (Supplementary material, Chapter 3, Fig. S3.2).

Generally, diversity was lower for each treatment type in the west compared to the eastern sectors of the park. ANOSIM results for overall comparisons showed sampled soil biota assemblages were significantly different, but with a lower R-value (ANOSIM, Global: $R = 0.323$, $p < 0.001$) (Table 3.1). R-values of ANOSIM results between each site showed the highest similarity in assemblages between the non-eroded undisturbed sites, with variable similarity between sites from rehabilitated and non-rehabilitated eroded sites (Table 3.1).

Table 3.1: Post-hoc R- and p-values calculated from ANOSIM analysis of soil arthropod assemblages associated with sampled sites. R-values are given in the lower triangle of the matrix, and p-values are given in the upper triangle. Sites 1 and 2 - Non-rehabilitated eroded sites; Sites 3 and 4 - Rehabilitated eroded sites; Sites 5 and 6 - Non-eroded undisturbed sites

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Site 1		<0.001	<0.001	<0.001	<0.001	<0.001
Site 2	0.508		<0.001	<0.001	<0.001	<0.001
Site 3	0.387	0.637		<0.001	<0.001	<0.001
Site 4	0.188	0.424	0.266		0.012	<0.001
Site 5	0.266	0.501	0.429	0.074		0.014
Site 6	0.373	0.501	0.405	0.158	0.074	

The ordination results showed an overlap between sites of each part of the park, with differing patterns observed between eastern and western sector sites (Fig. 3.3a–c). Site type comparisons indicated a higher number of unique species in non-rehabilitated sites, with a higher number of shared species between rehabilitated and non-rehabilitated sites (Fig. 3.3d).

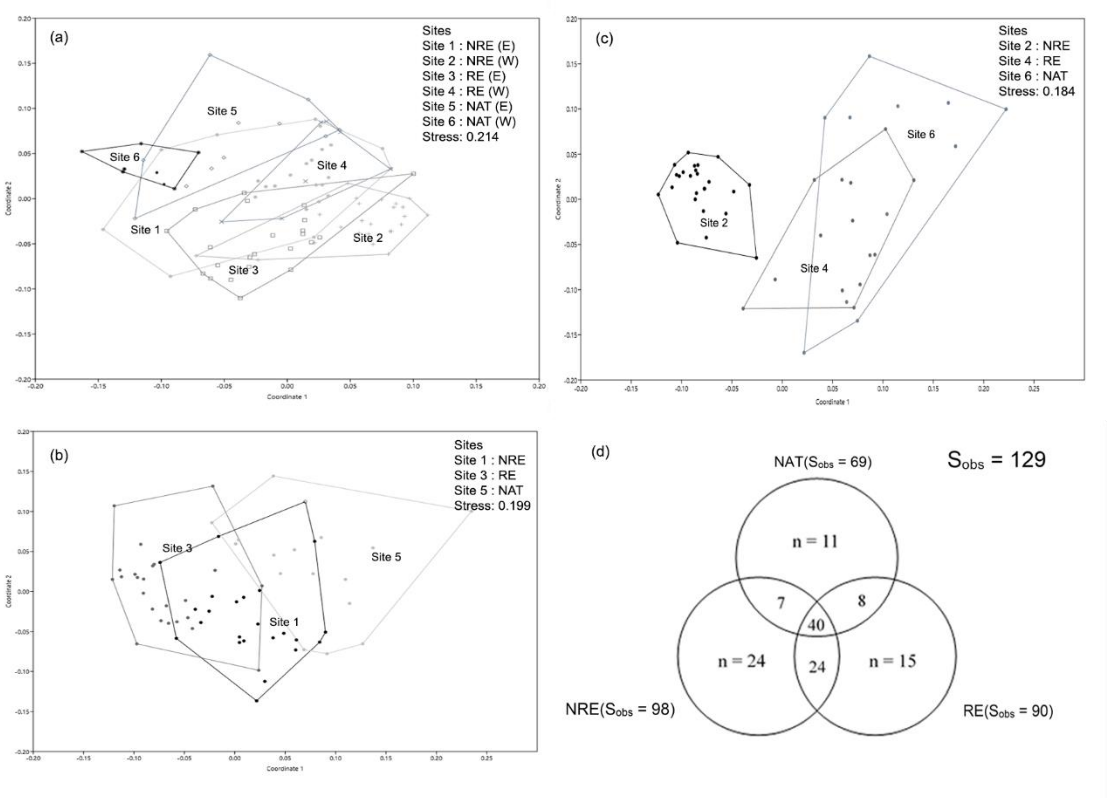

Fig. 3.3: Non-metric Multidimensional Scaling (3D) ordination, representing (a) soil-dwelling arthropod assemblages from all sites, with indication of sites on eastern and western parts of the park, (b) soil dwelling arthropods of sites on the eastern sector of the GGHNP, and (c) soil-dwelling arthropods of sites on the western sectors of the GGHNP. (d) indicates overlap in species from each site group, with unique species shown. NRE - Non-rehabilitated eroded site; RE - rehabilitated eroded site; NAT - Non-eroded undisturbed site.

Ordination over all sites showed overlaps between assemblages of each area, with the most similar species assemblages occurring in site 6. Site 1 shared the greatest similarity with most other assemblages, with the largest area of overlap (Fig. 3.3a). Non-rehabilitated erosion assemblages of sites in the eastern sectors of the park showed more similarity to natural assemblages in comparison to rehabilitated (Fig. 3.3b). Clear separation between the eroded site type assemblages of the western part of the park was observed (Fig. 3.3c), suggesting a difference in ecosystem types, despite the large number of shared species between rehabilitated and non-rehabilitated eroded sites. Overall, a larger number of unique species was observed in the non-rehabilitated eroded sites in comparison to the others. R-values of the ANOSIM between sites of each sector of the park indicated a stronger similarity between non-rehabilitated and non-eroded undisturbed sites within eastern parts of the park ($R = 0.266$, $p < 0.001$) (Table 3.2). Western assemblages showed a different pattern, with undisturbed and rehabilitated assemblages showing highest similarity ($R = 0.158$, $p < 0.001$).

Table 3.2: Post-hoc R- and p-values calculated from ANOSIM analysis of soil arthropod assemblages associated with sampled sites in the eastern and western sections of the Park. R-values are given in the lower triangle of the matrices, and p-values are given in the upper triangle.

	Eastern Section				**Western Section**		
	Site 1	Site 3	Site 5		Site 2	Site 4	Site 6
Site 1		<0.001	<0.001	Site 2		<0.001	<0.001
Site 3	0.387		<0.001	Site 4	0.424		<0.001
Site 5	0.266	0.429		Site 6	0.501	0.158	

The IndVal analysis identified a strong indicator for non-rehabilitated eroded sites (Table 3.3), with a single mite species, *Speleorchestes meyerae* Theron and Ryke, 1969, identified as a strong indicator in the site type. Collembolan groups were identified as possible indicators in rehabilitated eroded sites, with both species possessing physical characters that suggest individuals with a more resilient lifestyle (i.e. slim, harder bodies). A single ant species, *Solenopsis geminata* (Fabricius, 1804), was identified as a possible indicator of non-eroded undisturbed sites within the GGHNP. Only one strong indicator, *S. meyerae*, was identified for the investigated sites (over 70.0% IndVal value).

Table 3.3: Results of original IndVal, multi-level pattern analysis and Pearson's phi coefficient analysis for each site type or site combination. Only major soil arthropod groups are shown, with the omission of species not identified at least up to genus level. IndVal1 - Dufrêne and Legendre IndVal analysis; IndVal2 - Multi-level pattern analysis; IndVal3 - Pearson's phi coefficient.

Order	Family	Genera/species	Site type	IndVal1 (%)	Site type	IndVal2 (%)	Site type	IndVal3 (%)
Trombidiformes	Nanorchestidae	*Speleorchestes meyerae*	1	79.4	1	89.1	1	47.5
Trombidiformes	Tydeidae	*Pronematus* sp.	1	59.3	1	77.0	1	53.6
Oribatida	Nanhermanniidae	*Nanhermannia* sp.	1	48.2	1	69.4	1	47.2
Trombidiformes	Bdellidae	*Bdellodes* sp.	1	39.7	1	63.0	1	49.3
Oribatida	Aleurodamaeidae	*Aleurodamaeus salvadordali*	—	—	1	50.3	1	38.6
Collembola	Tullbergiidae	*Mesaphorura* sp.	2	42.4	2	65.1	1+2	38.1
Trombidiformes	Cunaxidae	*Cunaxa* sp.	—	—	2	43.9	—	—
Collembola	Entomobryidae	*Seira* sp.	—	—	1+2	51.7	—	—
Collembola	Isotomidae	*Proisotoma davidi*	2	37.9	1+2	78.8	1+2	44.2
Hymenoptera	Formicidae	*Solenopsis geminata*	3	36.2	—	—	3	43.1

Site type 1 - Non-rehabilitated eroded; Site type 2 - Rehabilitated eroded; Site type 3 - Non-eroded undisturbed

Multi-level pattern analysis shows a slightly different result, displaying higher indicator values for listed species (Table 3.3). A new strong indicator was shown in this analysis, with high indicator values shown for both *S. meyerae* and *Pronematus* sp. Canestrini, 1886, with indicator values above 70%. Non-rehabilitated and rehabilitated sites indicated an additional species each as being significant for the site types in this analysis. However, the loss of *S. geminata* as an indicator of undisturbed sites was noted. The combination of rehabilitated and non-rehabilitated site types possessed two species as potential indicators, both members of the order Collembola.

Results of Pearson's phi correlation provided comparable results to the previous analyses. However, indicator values and the strength of indicators varied for each site type, with different species weighted as stronger indicators per type (Table 3.3). In comparison to the previous two analyses, *Pronematus* sp. had the highest ecological preference for non-rehabilitated eroded sites. However, all indicator values were lower than 70%. Additionally, *S. geminata* indicated high ecological preference for undisturbed sites, while two collembolan species were indicated as having preference for both rehabilitated and non-rehabilitated site types.

Linear Models showed a best fit correlation with phosphorus and species richness, with the lowest AIC indicated for this model (Supplementary material, Chapter 3, Table S3.1 and S3.2). It is important to note the significant negative correlation values obtained for both non-rehabilitated and rehabilitated eroded sites, suggesting that increased phosphorus levels could be used in conjunction with soil arthropod richness as a possible indicator for levels of soil erosion.

3.4 Discussion

3.4.1 Comparison of erosion site types versus non-eroded sites

Soil erosion within protected areas is a major concern in regards to conservation strategy and land degradation prevention, as it vastly threatens natural environments (Rahman et al., 2009). Although soil erosion is a particular concern around the various streams running through the park, the results suggest that erosion sites provide a differing niche for species not typically found in the natural undisturbed grasslands. We can suggest that eroded sites promote the overall population heterogeneity of the GGHNP, ultimately preserving the diverse ecological processes that these organisms carry out in their respective habitats. The high number of unique species, as well as shared species found in the non-rehabilitated erosion site type, strongly suggests that erosion sites in the GGHNP support a greater diversity of soil arthropod species. This may differ for areas experiencing accelerated erosion due to anthropogenic influences, and thus future studies should examine eroded sites affected by different erosion sources.

The findings suggest that rehabilitation somewhat breaks the pattern of increased species richness and diversity in eroded sites, possibly over a long period of time, subsequently affecting soil arthropod species richness and diversity. This could possibly be due to sediment build-up around rehabilitation structures, such as rock-wall gabions and silt nets, over a long time series (van der Merwe et al., unpublished data). This is especially evident in Site 4, with low species richness recorded several years after rehabilitation structures were implemented. Surprisingly, soil arthropod assemblages in the non-eroded sites displayed very low species richness throughout the study. The reason for the low species richness observed in non-eroded undisturbed sites in this study is not clear, but could possibly be attributed to differences in soil quality, such as nutrient compounds, and simply ecosystem preference of individual species (Coleman et al., 2004). Additionally, rarefaction curves suggest that more sampling is necessary to ensure complete coverage of species present in these localities (Supplementary material, Chapter 3, Fig. 3.3)

3.4.2 Soil arthropods as indicators of erosion in the park

When assessing the indicators identified by the association analyses, a varied pattern of species was seen. The initial IndVal analysis shows a general difference in feeding groups within each site type. *S. meyerae* is a prostigmatid mite species with an algae/bacteriophagous feeding behaviour (Badenhorst, 2016), found in large numbers in the non-rehabilitated eroded sites of the park. This differs greatly from *Mesaphorura* sp. Börner, 1901 and *Proisotoma davidi* Barra, 2001, which are broadly regarded as general saprophages (Ponge, 2000; Castaño-Meneses et al., 2004; Badenhorst, 2016), and were identified as possible, but relatively weak indicators of the rehabilitated eroded sites. The identification of *S. geminata* as a possible indicator of the sampled non-eroded undisturbed sites in the park suggests that ant assemblages could be used in the area as an indicator of ecosystem health of these montane grasslands. *Solenopsis* Westwood, 1840 have often been used as indicators of disturbance in a variety of landscapes (Tschinkel, 1988; Peck et al., 1998; Vasconcelos, 1999; Graham et al., 2008). However, as our study only dealt with soil-dwelling arthropods, more extensive research is needed to establish a complete list of indicators, which incorporates both soil-dwelling and surface-active arthropod assemblages for a more well-rounded view.

Multi-level pattern analysis displayed a slightly different pattern of association, with similar species to the IndVal indicated with stronger statistical values, and the addition of certain species that were not initially indicated. This suggests that differences in tested indicators depend on the testing method, making it vital to consider multiple analyses with different variables to adequately identify species associated with erosion (Siddig et al., 2016). This is particularly seen with the Pearson's phi coefficient as different species are identified as stronger indicators in terms of ecological preference. Although these groups closely resemble the species groups identified by the IndVal, the differences in indicator values of each group are considered with caution when attempting to determine indicator species, as species responses may be measured differently (Zettler et al., 2013).

Although species identified as significant in rehabilitated eroded and non-eroded sites had an indicator value of less than 70.0, it is important to consider these groups in light of their functions and their relative abundance in the soils of the GGHNP. It is possible that these species groups could still be considered and analysed under a combined grouped analysis, which would take additional parameters into consideration, such as functional feeding and more specific ecosystem limiters (De Cáceres et al., 2010). It is important to consider that little is still known about the specific feeding habits of a large group of soil arthropods, and how these arthropods may change their feeding preferences in different conditions (Rusek, 1998; Briones et al., 1999; Castaño-Meneses et al., 2004). This concretes the need for further study regarding soil biota in the GGHNP and their specific lifestyles, in order to determine whether a combined species approach would be suitable for the area. Nonetheless, the possible indicators identified in this study could be used as a suitable starting point for monitoring responses of soil arthropods in grassland national parks in South Africa.

Soil-dwelling biota are shown to respond to changes in phosphorus levels in soil (Kevan et al., 1995; Scheu and Schaefer, 1998; Coleman et al., 2004; Sayer et al., 2010; Ashford et al., 2013). A significant correlation between eroded sites and increased levels of phosphorus in our study was found, suggesting that phosphorus could be used in conjunction with soil arthropod species richness in grading the effects of erosion. Although a correlation was found between phosphorus levels and non-rehabilitated and rehabilitated eroded arthropod assemblages, it could be argued that other factors need to be taken into consideration depending on the nature of each study site.

3.4.3 Differences in western and eastern soil assemblages of the GGHNP

It is important to explain why each sector was analysed separately between the eastern and western parts of the park. The GGHNP was formerly comprised of only the western part of the park, with a focus on conservation. The eastern section, formally known as the Qwaqwa National Park, focused on the preservation of cultural traditions and practices and was mainly used as grazing lands for livestock (Rademeyer and van Zyl, 2014). The parks announced a merger in 2004 and the amalgamation was finalised in 2008, incorporating the various ecosystem types into a single national park (Ministry of Environmental Affairs and Tourism, 2004). Terrains between the two sectors are notably different (western sectors are situated in a high elevation, mountainous terrain, whereas eastern sectors are situated at a lower elevation and along a relatively flatter landscape), which could explain the differences between soil arthropod assemblages of the eastern and western parts of the park. In addition, soil type differences between each section could also explain the higher species richness and species diversity found within the eastern section. Elevation may also play a role, as studies have found that lower altitude sites could yield higher levels of certain soil arthropod groups than higher altitudes (Jing et al., 2005; Illig et al., 2010). The results of our study suggest that the areas of higher altitude in the western sections of the park yielded lower arthropod assemblage numbers, specifically in the higher altitude non-eroded undisturbed site (Site 6) and the rehabilitated eroded site (Site 4). It could be argued that the high altitude sloping nature of site 6, and the degradation of site 4, may have contributed to the lower abundance and species richness sampled from these two sites. In light of this, the nature of each site, coupled with the differences between the eastern and western parts of the park, justified the analysis of sites within each sector as separate groups.

3.5 Conclusion

Overall, the results show that there are definite differences between arthropods in undisturbed and eroded sites, with variable similarities between rehabilitated eroded, non-rehabilitated eroded and non-eroded undisturbed sites between the eastern and western parts of the park. A single strong indicator was identified, with the strongest indication for non-rehabilitated eroded sites. Although other species were shown as weaker indicators, a combined species approach could be suggested to better represent possible indicators for the GGHNP. Differences in species richness and diversity show that sites corresponded with site proximity in the park, and suggest that data from each site should be analysed according to area in conjunction with site type.

Although this investigation was performed specifically for application in the GGHNP, the study shows great potential for extensive study into soil-dwelling arthropods as indicators of disturbance in other types of landscapes, as well as their role as ecosystem drivers in South African landscapes. Studying these arthropod groups on a taxon-specific basis could give insight into the individual roles these groups play in eroded sites, for use in conservation strategy planning.

3.6 References

Ashford, O.S., Foster, W.A., Turner, B.L., Sayer, E.J., Sutcliffe, L., Tanner, E.V.J., 2013. Litter manipulation and the soil arthropod community in a lowland tropical rainforest. Soil Biol. Biochem. 62, 5–12.

Badenhorst, H., 2016. Mesofaunal Assemblages in Soil of selected Crops under Diverse Cultivation Practices in Central South Africa, with Notes on Collembola Occurrence and Interactions. MSc dissertation, University of the Free State, Bloemfontein.

Bates, D., Maechler, M., Bolker, B., Walker, S., 2015. Fitting linear mixed-effects models using lme4. J. Stat. Softw. 67, 1–48.

Briones, M.J.I., Ineson, P., Sleep, D., 1999. Use of δ13C to determine food selection in collembolan species. Soil Bio. Biochem. 31, 937–940.

Castaño-Meneses, G., Palacios-Vargas, J.G., Cutz-Pool, L.Q., 2004. Feeding habits of Collembola and their ecological niche. Ser. Zool. 75, 135–142.

Chytrý, M., Tichý, L., Holt, J., Botta-Dukát, Z., 2002. Determination of diagnostic species with statistical fidelity measures. J. Veg. Sci. 13, 79–90.

Coleman, D.C., Crossley Jr., D.A., Hendrix, P.F., 2004. Fundamentals of Soil Ecology 2nd Edition. Elsevier Academic Press, California.

Convey, P., Block, W., Peat, H.J., 2003. Soil arthropods as indicators of water stress in Antartic terrestrial habitats? Global Change Biol. 9, 1718–1730.

De Cáceres, M., 2013. How to use the indicspecies package (ver. 1.7.1). Website: https://cran.r-project.org/web/packages/indicspecies/vignettes/indicspeciesTutorial.pdf.

De Cáceres, M., Legendre, P., 2009. Associations between species and groups of sites: indices and statistical inference. Ecology 90, 3566–3574.

De Cáceres, M., Legendre, P., Moretti, M., 2010. Improving indicator species analysis by combining groups of sites. Oikos 119, 1674–1684.

Dlamini, P., Orchard, C., Jewitt, G., Lorentz, S., Titshall, L., Chaplot, V., 2011. Controlling factors of sheet erosion under degraded grasslands in the sloping lands of KwaZulu-Natal, South Africa. Agric. Water Manag. 98, 1711–1718.

Dufrêne, M., Legendre, P., 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecol. Monogr. 67, 345–366.

Dunger, W., Schlitt, B., 2011. Synopses on Paleartic Collembola. Museum of Natural History, Görlitz.

Durán Zuazo, V.H., Rodríguez Pleguezuelo, C.R., 2008. Soil-erosion and runoff prevention by plant covers. A review. Agron. Sustain. Dev. 28, 65–86.

Garland, G.G., Hoffman, M.T., Todd, S., 2000. Soil degradation. In: Hoffman, M.T., Todd, S., Ntshona, Z. and Turner, S. (Eds.), A National Review of Land Degradation in South Africa. South African National Biodiversity Institute, Pretoria, South Africa. pp 69–107.

Graham, J.H., Krzysik, A.J., Kovacic, D.A., Duda, J.J., Freeman, D.C., Emlen, J.M., Zak, J.C., Long, W.R., Wallace, M.P., Chamberlin-Graham, C., Nutter, J.P., Balbach, H.E., 2008. Ant community composition across a gradient of disturbed military landscapes at Fort Benning, Georgia. Southeast. Nat. 7, 429–448.

Gulvik, M., 2007. Mites (Acari) as indicators of soil biodiversity and land use monitoring, a review. Pol. J. Ecol. 55, 415–440.

Hammer, Ø., Harper, D.A.T., Ryan, P.D., 2001. PAST: Palaeontological statistics software package for education and data analysis. Palaeontol. Electron. 4, 1–9.

Hugo-Coetzee, E., 2013. New species of *Aleurodamaeus* Grandjean, 1954 (Oribatida: Aleurodamaeidae) from South Africa. Zootaxa 3670, 531–556.

Illig, J., Norton, R.A., Scheu, S., Maraun, M., 2010. Density and community structure of soil- and bark-dwelling microarthropods along an altitudinal gradient in a tropical montane rainforest. Exp. Appl. Acarol. 52, 49–62.

Jing, S., Solhøy, T., Huifu, W., Vollan, T.I., Rumei, X., 2005. Differences in soil arthropod communities along a high altitude gradient at Shergyla Mountain, Tibet, China. Arct. Antarct. Alp. Res. 37, 261–266.

Kevan, P.G., Forbes, B.C., Kevan, S.M., Behan-Pelletier, V., 1995. Vehicle tracks on high arctic tundra: their effects on soil, vegetation, and soil arthropods. J. Appl. Ecol. 32, 655–667.

Kremen, C., Colwell, R.K., Erwin, T.L., Murphy, D.D., Noss, R.F., Sanjayan, M.A., 1993. Terrestrial arthropod assemblages: their use in conservation planning. Conserv. Bio. 7, 796–808.

Le Roux, J.J., Morgenthal, T.L., Malherbe, J., Pretorius, D.J., Sumner, P.D., 2008. Water erosion prediction at a national scale for South Africa. Water SA 34, 305–314.

Le Roux, J.J., Newby, T.S., Sumner, P.D., 2007. Monitoring soil erosion in South Africa at a regional scale: review and recommendations. S. Afr. J. Sci. 103, 329–335.

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. J Appl Ecol 39, 661–672

Meadows, M.E., 2003. Soil erosion in the Swartland, Western Cape Province, South Africa: implications of past and present policy and practice. Environ. Sci. Policy 6, 17–28.

Menta, C., Leoni, A., Gardi, C., Conti, F.D., 2011. Are grasslands important habitats for soil microarthropod conservation? Biodivers. Conserv. 20, 1073–1087.

Mhangara, P., Kakembo, V., Lim, K.J., 2012. Soil erosion risk assessment of the Keiskamma catchment, South Africa using GIS and remote sensing. Environ. Earth Sci. 65, 2087–2102.

Ministry of Environmental Affairs and Tourism, 2004. Qwaqwa National Park incorporated with the Golden Gate Highlands National Park. Public Statement, South African Government. Accessed at: http://www.info.gov.za/speeches/2004/04031813461001.htm, on 06/08/2019, 15:36.

Nakamura, A., Catterall, C.P., House, A.P.N., Kitching, R.L., Burwell. C.J., 2007. The use of ants and other soil and litter arthropods as bio-indicators of the impacts of rainforest clearing and subsequent land use. J. Insect Conserv. 11, 177–186.

Olivier, P.A.S., Theron, P.D., 1989. A new species of *Speleorchestes* (Nanorchestidae: Prostigmata) from a savanna biotope in South Africa. S. Afr. J. Zool. 24, 356–360.

Orgiazzi, A., Panagos, P., 2018. Soil biodiversity and soil erosion: it is time to get married. Global Ecol. Biogeogr. 27, 1155–1167.

Paoletti, M.G., Favretto, M.R., Stinner, B.R., Purrington, F.F., Bater, J.E., 1991. Invertebrates as bioindicators of soil use. Agr. Ecosyst. Environ. 34, 341–362.

Peck, S.L., Mcquaid, B., Campbell, C.L., 1998. Using ant species (Hymenoptera: Formicidae) as a biological indicator of agroecosystem condition. Environ. Entomol. 27, 1102–1110.

Pimentel, D., 2006. Soil erosion: a food and environmental threat. Environ. Dev. Sustain. 8, 119–137.

Pimentel, D., Harvey, C., Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M., Crist, S., Shpritz, L., Fitton, L., Saffouri, R., Blair, R., 2004. Environmental and economic costs of soil erosion and conservation benefits. Science 267, 1117–1123.

Pimentel, D., Kounang, N., 1998. Ecology of soil erosion in ecosystems. Ecosystems 1, 416–426.

Ponge, J., 2000. Vertical distribution of Collembola (Hexapoda) and their food resources in organic horizons of beech forests. Biol. Fert. Soils 32, 508–522.

R Core Team, 2019. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL: http://www.R-project.org/.

Rademeyer, C., van Zyl, W., 2014. Golden jubilee for Golden Gate - a concise history of Golden Gate Highlands National Park, 1963 to 2013. Mediterr. J. Soc. Sci. 5, 1169–1177.

Rahman, M.R. Shi, Z.H., Chongfa, C., 2009. Soil erosion hazard evaluation – an integrated use of remote sensing, GIS and statistical approaches with biophysical parameters towards management strategies. Ecol. Model. 220, 1724–1734.

Roberts, B.R., 1966. Conservation of the Vegetation of the Orange Free State. 6th A.E.T.A.T. Symposium, Uppsala.

Roberts, B.R., 1969. The Vegetation of the Golden Gate Highlands National Park. Koedoe 12, 15–28.

Rusek, J., 1998. Biodiversity of Collembola and their functional role in the ecosystem. Biodivers. Conserv. 7, 1207–1219.

Sayer, E.J., Sutcliffe, L.M.E., Ross, R.I.C., Tanner, E.V.J., 2010. Arthropod abundance and diversity in a lowland tropical forest in Panama: the role of habitat space vs. nutrient concentrations. Biotropica 42, 194–200.

Scheu, S., Schaefer, M., 1998. Bottom-up control of the soil macrofauna community in a beechwood on limestone: manipulation of food resources. Ecology 79, 1573–1585.

Siddig, A.A.H., Ellison, A.M., Ochs, A., Villar-Leeman, C., 2016. How do ecologists select and use indicator species to monitor ecological change? Insight from 14 years of publication in Ecological Indicators. Ecol. Indic. 60, 223–230.

Slingsby, P., 2017. Ants of Southern Africa: The Ant Book for All. Slingsby Maps, Muizenberg.

South African National Parks (SANParks), 2013. Golden Gate Highlands National Park, Park Management Plan for the period 2013-2023. SANParks.

Stork, N.E, Eggleton, P., 1992. Invertebrates as determinants as indicators of soil quality. Am. J. Altern. Agric. 7, 38–47.

Theron, P.D., Ryke, P.A., 1969. The family Nanorchestidae Grandjean (Acari: Prostigmata) with descriptions of new species from South African soils. J. Ent. Soc. Sth. Afr. 32, 31–60.

Triplehorn, C.A., Johnson, N.F., 2005. Borror and DeLong's Introduction to the Study of Insects 7th Edition. Thompson Brooks/Cole, Belmont.

Tschinkel, W.R., 1988. Distribution of the fire ants *Solenopsis invicta* and *S.geminata* (Hymenoptera: Formicidae) in Northern Florida in relation to habitat and disturbance. Ann. Entomol. Soc. Am. 81, 76–81.

Tullgren, A. 1918. Ein sehr einfacher Ausleseapparat fur terricole Tierformen. Z. Angew. Entomol. 4, 149–150.

van Straalen, N., 2004. Chapter 6: The use of soil invertebrates in ecological surveys of contaminated soils. In: Doelman, P., Eijsackers, H.J.P. (Eds.), Vital Soil: Function, Value and Properties. Elsevier Science and Technology, Oxford.

Vasconcelos, H.L., 1999. Effects of forest disturbance on the structure of ground-foraging ant communities in central Amazonia. Biodivers. Conserv. 8, 409–420.

Wickham, H., 2016. ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag, New York.

Zettler, M.L., Proffitt, C.E., Darr, A., Degraer, S., Devriese, L., Greathead, C., Kotta, J., Magni, P., Martin, G., Reiss, H., Speybroeck, J., Tagliapietra, D., van Hoey, G., Ysebaert, T., 2013. On the myths of indicator species: issues and further consideration in the use of static concepts for ecological applications. PLOS ONE 8, e78219. https://doi.org/10.1371/journal.pone.0078219.

Chapter 4:

Effect of gabions on soil biota in eroded sites of a South African grassland habitat

Sylvia S. van der Merwe[a], Vaughn R. Swart[a], Emile Bredenhand[b], Charles R. Haddad[a]

[a] University of the Free State, Bloemfontein Campus, Bloemfontein, South Africa

[b] University of the Free State, Qwaqwa Campus, Phuthaditjhaba, South Africa.

Paper submitted to Soil Research, CSIRO Publishing.

Abstract

Soil erosion and subsequent soil degradation is an ever-increasing concern worldwide. Rehabilitation methods implemented to combat soil erosion are focussed on the prevention of siltation of surrounding water resources. However, few studies have looked into the effects of implemented rehabilitation structures on soil biota assemblages. The aim of this study was to identify changes in soil arthropod assemblages in areas with implemented erosion rehabilitation in the Golden Gate Highlands National Park. In addition, this study also investigated the effect of sediment build-up around gabion walls on soil arthropod richness. Abundance and species richness data showed no significant differences between rehabilitated and non-rehabilitated site types. Ordination results showed no specific patterns between assemblages and sub-site type, with sub-sites indicating a relatively high similarity on average. Additionally, association analyses indicated no ecological correlation of species for any of the site types. Sediment build-up was noted to have a significant effect on soil arthropod richness. The results of this study suggest that gabions do not significantly affect or change soil assemblages in the rehabilitated areas. However, it remains a question whether the use of rock-wall gabions causes any long-term detriment or benefit to soil arthropod assemblages in eroded sites. As such, regular monitoring of both soil biota and the surrounding terrain may be necessary to gauge the progression and effects of eroded areas and their rehabilitation methods in terrestrial terrains.

Keywords: Arthropods; Erosion Rehabilitation; Sediment; Species Richness

4.1 Introduction

Soils are a vital component in monitoring the quality of landscapes and can effectively be used in assessing the health of an ecosystem (Gulvik, 2007; Rousseau et al., 2013). Soil erosion of landmasses is widespread and is proven to affect all the natural and human-controlled ecosystems, including forestry and agricultural land areas. The effect of soil erosion is ranked as one of the most severe environmental problems facing the world's landscapes, as it leads to degradation of soils (Pimental et al., 1995; Pimental and Kounang, 1998; Pimental, 2006; Durán Zuazo and Rodgríguez Pleguezuelo, 2008; Pimental and Burgess, 2013). The ripple effect of degradation of soils is thought to result in long-term changes to ecosystems, extending to above-ground processes, permanently altering the nature of landscapes and ultimately restructuring natural ecosystems as a whole (De Deyn et al., 2003; Hooper et al., 2012; Mhangara et al., 2012; Lal, 2015).

Soil erosion in natural areas shapes landscapes and gradually forms new niches, especially in steeply sloped areas such as high altitude mountains (Harden, 2001). The rate of erosion is often directly linked to rainfall in these areas, being more severe in sites with more erodible soils (Le Roux et al., 2007; Mhangara et al., 2012). The Golden Gate Highlands National Park (GGHNP) is an Afromontane grassland park known for its sandstone cliffs and formations. However, the type of sandstone produces shallow sandy soil with exceptionally low fertility (Roberts, 1966). South African National Parks (SANParks) officials are concerned about the rate at which erosion may occur in the GGHNP, due to the steep nature of sloping in the park and the levels of dispersive soils, which are susceptible to erosion during peak rainfall seasons in each area (SANParks, 2013). In light of this, Park Management engages in erosion rehabilitation methods to reduce the level of silt deposition into water bodies and near infrastructure and pathing. However, soil arthropod assemblages are not necessarily considered when planning management interventions against erosion. According to Orgiazzi and Panagos (2018), there is a need to consider soil arthropod assemblages when studying the effects of erosion, as these assemblages often prove to be essential to an area's natural ecosystem processes. The loss of soil arthropods can prove highly detrimental to ecosystem health (Hunt and Wall, 2002; Culliney, 2013), possibly causing a higher failure rate in implemented conservation strategies.

Erosion control or rehabilitation is a fundamental approach to protect water quality and prevent excessive siltation around infrastructure (SANParks, 2013). Soil stabilisation practices, such as rock-wall gabions and silt netting, is implemented to allow for plant growth and subsequent restoration of natural grasses to reduce future siltation. Rock- or stone-wall gabions are primarily used to control sheet erosion on flatter flow areas around roads (Rivas, 2006; SANParks, 2009). However, the direct effect of these walls on soil arthropods is poorly understood. Soil erosion sediment has generally been investigated in relation to its effects on aquatic fauna and plants (Trombulak and Frissell, 2000; Gleason et al., 2003; Ekholm and Lehtoranta, 2012). Very few studies have specifically investigated soil biodiversity in erosion sediment in terrestrial terrains (Groffman and Bohlen, 1999), with no records for soil arthropods in erosion areas available for South African terrains. The importance of soil arthropods as role-players in ecosystem functions has been studied in various terrains (Paoletti et al., 1991; Kremen et al., 1993; Menta et al., 2011). However, little is known about the effect of erosion rehabilitation methods on soil arthropods. The investigation into the effect of gabion implementation and subsequent sediment build-up around these walls on soil arthropods in the GGHNP could give insight into the long-term effects of gabion rehabilitation on overall soil assemblages, and may be used in conservation strategies to plan for possible changes in vital soil faunal groups.

The aim of this study was to identify changes in soil arthropod assemblages in areas with implemented erosion rehabilitation, namely gabions near dirt roads, in the GGHNP. In addition, the study also monitored sediment build-up behind and in front of gabion walls over two rainfall seasons to investigate the effect of sediment build-up on soil arthropod assemblages.

4.2 Materials and Methods

4.2.1 Study area and sites

The study was conducted in the northern parts of the GGHNP, located in the south-eastern Free State Province of South Africa. The national park is located between the towns of Clarens and Phuthaditjhaba, at the foothills of the Maluti Mountains in the Rooiberg Range. The park comprises both flatter grasslands of the former Qwaqwa National Park (incorporated into the GGHNP in 2008) in the east, and the steeper grasslands of the old GGHNP in the west, extending approximately 32 690 ha (SANParks, 2013).

Each site was selected according to similar rock-wall gabion implementations near dirt roads to assess their impact on soil arthropod assemblages (Supplementary material, Chapter 4, Fig. S4.1). Overall, two sites were selected and divided into two subsets: one subset to represent soil arthropods in direct proximity of a rock-walled (gabion) eroded area (A1 and B1), with areas for sampling chosen behind and in front of the gabion. The other subset was located at most 15 m away from the rock-walled site as a comparable point (A2 and B2) to assess assemblages in a non-rehabilitated eroded area. Gabions were implemented mainly to prevent sediment displacement around the roads running between the central and northern sections of the park, mainly used by farmland residents and park rangers. Both gabions were built using unweathered, solid gabion rock pieces placed into a woven mesh basket (aperture of 50 mm), and laid in a dug trench of 0.2 m lined with a fibre netting blanket. Each wall had a height of 0.7 m and a width of 0.5 m. Length of the gabions differed between each sub-site, with A1 constructed 10 m and B1 15 m in length.

Site A (28°27.313’S, 28°40.851’E, elevation: 1704 m) was characterised as a flatter eroded site about 6 m away from a dirt road in the northern sectors of the park. The site was situated next to a stream running through a deep crevice between the hills that flowed through the area during most parts of the year. Livestock were often noted in and around the site, grazing on the steeper sections around the eroded area. A single gabion was constructed facing the roadside, and the slope running down to the stream, in July 2015 (SANParks personnel, personal communication, 2016[1]). Site B (28°29.444'S, 28°41.760'E, elevation: 1851 m) was characterised as a sloped eroded site adjacent to a dirt road running around the mountain terrain. Unlike site A, site B was situated far from any water bodies and domestic grazing influences. However, indigenous antelope and other wildlife were observed in and around the area during the study. Construction of a gabion facing the sloping road was done during September 2015 (SANParks personnel, personal communication, 2016[1]).

The vegetative units in the park were identified as Eastern Free State Sandy, Northern Drakensberg Highland and Lesotho Highland Basalt grasslands (Mucina et al., 2006). Both areas were classified under the Eastern Free State Sandy grassland vegetative unit. These areas were surrounded by grass species comprising *Eragrostis chloromelas* and *E. viscosa* grasses, interspersed by several tufts of *Themeda triandra* (van Oudtshoorn, 2014). Further erosion around the rehabilitated sites was assumed to form during the years after the implementation of gabions, based on map imaging over the years after implementation (Supplementary material, Chapter 4, Fig. S4.2 and S4.3).

4.2.2 Soil arthropod sampling

At each sample point, five soil samples of 300–400 g each were taken from the 0–10 cm soil layer, and pooled into a single sample each month. Sampling in rock-walled sub-sites was done behind and in front of the gabion, whereas non-rehabilitated sub-sites were sampled

[1] T. Nsibande, verbal communication, Working for Ecosystems Park Ranger - Golden Gate Highlands National Park, 21 September 2016.

at the highest point closest to the source of erosion (source) and the lowest part down the eroded area (base), both within the eroded area. Sampling position was selected by measuring gabion sites from the rock-walls to the source and base of the erosion sites, and similar distances were used for sampling their paired non-rehabilitated sub-sites. Each sub-site was sampled monthly from December 2017 to March 2019. Soil samples were placed in Berlese-Tullgren funnels and extracted for seven days into 70% ethyl alcohol (Tullgren, 1918; Triplehorn and Johnson, 2005; Badenhorst, 2016). Samples were sorted under a light microscope and arthropods identified to morphospecies level using specialist assistance and associated literature (Theron and Ryke, 1969; Triplehorn and Johnson, 2005; Dunger and Schlitt, 2011; Badenhorst, 2016; Slingsby, 2017).

4.2.3 Sediment build-up

In order to monitor sediment build-up in gabion sub-sites, each rock-wall was equipped with eight measuring rulers (in mm), with four behind and four in front of the gabion. Rulers were positioned at four fixed points in a straight line, parallel to the gabion, at 0.5 m intervals along a 2 m section to prevent error caused by soil displacement during the initial installation. Each ruler extended from the base of the rock-wall to its top. Measurements were recorded monthly during soil sampling. All measurements of sediment build-up behind and in front of the gabion for each sub-site were averaged.

4.2.4 Statistical analysis

Each sample was quantified per site, with observed species richness, Shannon-Wiener diversity indices and Chao1 estimators calculated for species data to analyse differences. Box plot analyses were done using soil arthropod abundances and species richness to identify patterns in mean differences between each sub-site. Diversity indices and Chao1 estimators were calculated using EstimateS, version 9.1.0 (Colwell, 2013), and plots were constructed using ggplot2 in R, version 3.5.3 (Wickham, 2016).

In order to further investigate the difference between species assemblages of rehabilitated and non-rehabilitated eroded sites, and establish possible ecological preferences, association of species assemblages and the site types was investigated using a multilevel pattern analysis (‘multipatt’) function. A Dufrêne-Legendre Indicator Species analysis (IndVal) on species data obtained over the 16-month period was also conducted to assign an indicator value to each species. Each site was clustered into the two sub-site types (rehabilitated and non-rehabilitated), and species abundance was tested across sites to establish differences in species. The original IndVal method is used by the ‘multipatt’ function, with an extension of the original concept of Dufrêne and Legendre (1997), to investigate indicator species of individual sites and their combinations, as described by De Cáceres et al. (2010). Pearson’s phi coefficient of association was used to determine the association between species and the sub-site types (Chytrý et al., 2002), measured between two binary vectors in the form of presence and absence data to produce a correlation (De Cáceres, 2013). In order to correct for testing errors that may be associated with indicator values (De Cáceres et al., 2010), a ‘Bonferroni’ correction was applied to p-values using the ‘p.adjust’ function in R, version 3.5.3 (R Core Team, 2019). Only species indicated as significant after applying p-value corrections were considered as significant for association. All IndVal tests with multilevel pattern analyses were done using the Indicspecies package in R, version 3.5.3 (De Cáceres and Legendre, 2009).

Non-metric Two-dimensional Scaling ordination, using a Bray-Curtis similarity index, was done to identify any close similarities between soil arthropod assemblages of each sub-site and each site. In addition, cluster analysis using the Bray-Curtis similarity matrix, with paired group algorithms, was performed to identify sub-sites and sites with highest similarities in soil arthropod assemblages. Ordinations and cluster dendrogram analyses were done using Paleontological Statistics (PAST), version 3.16 (Hammer et al., 2001).

In order to test for significance between soil arthropod assemblage richness and abundance and factors of each sub-site, Generalised Linear Models (GLM) were done on assemblage data, with sub-site sampling position and rehabilitation method as factors, using Poisson error structures. In addition, a Permutational Multivariate Analysis of Variance (PERMANOVA) was processed to determine differences between sub-site assemblage compositions. Pairwise tests between each sub-site combination were done to test for significant differences in assemblage clusters overall, with a False Discovery Rate ('fdr') correction applied to obtained p-values to reduce false-positive readings.

Significance between soil arthropod richness and sediment build-up in rock-walled sites was tested using GLM, with sediment height and sub-site sampling position as factors, using Poisson error structures. All linear models were processed using the LME4 package (Bates et al., 2015), and PERMANOVA analyses were done using the Vegan package in R, version 3.5.3 (Oksanen et al., 2019).

4.3 Results

A total of 2490 individual arthropods were sampled over the 16-month period, representing 82 morphospecies, 48 families and 13 orders. Sites A1 and A2 possessed the highest observed species richness, diversity and abundance overall (Table 4.1, Fig. 4.1a–b). Species richness was generally lower in both B sites, with a lower species richness recorded in the sub-sites behind the gabion and at the base of the non-rehabilitated erosion site (Fig. 4.1a–b).

Table 4.1: Total observed species richness and diversity indices for each site. Sobs - observed species richness; H - Shannon-Wiener Diversity index; SChao1 ± SD - species richness estimate with 1 standard deviation

Site	Individuals	S_{obs}	H	S_{Chao1} ± SD
A1	667	53	2.47	66.60 ± 8.74
A2	774	54	2.84	76.67 ± 14.9
B1	421	43	2.17	52.55 ± 6.55
B2	628	38	1.92	47.75 ± 7.19
Total	2490	80	-	-

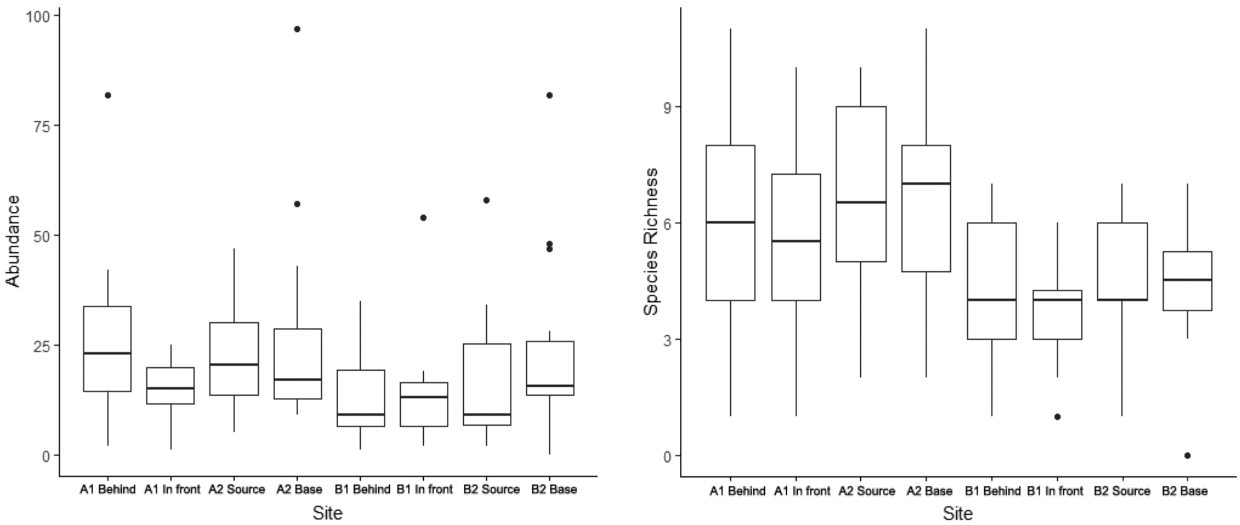

Fig. 4.1: Boxplots of (a) soil arthropod abundance, and (b) species richness per sub-site, with dots representing outlier species groups.

Association analysis using the multi-level pattern analysis function showed that only a single species, namely the mite *Pronematus* sp. Canestrini 1886, tested significant for one sub-site type once p-value correction was applied (Table 4.2). Tests for association showed no ecological correlation of species for any of the site types, with no species being identified as significant.

Table 4.2: Significance results of association analysis, showing indicator value (Stat) as derived by 'multipatt' function, and adjusted p-values ('p-adjust') using 'Bonferroni' correction

Species	Site type	Stat	p-value	p-adjusted
Pronematus sp.	Non-rehabilitated	0.635	0.016	**0.047**
Bdellodes sp.	Non-rehabilitated	0.530	0.048	0.143
Nanhermannia sp.	Non-rehabilitated	0.499	0.039	0.116

Scaling ordination showed no specific patterns between assemblages and sub-site type, with cluster analysis showing that most sites had similar assemblages (above 0.600) (Figure 4.2a–b). When comparing overall assemblages of the rehabilitated and non-rehabilitated sites, no visible pairing was seen, with most site assemblages being similar (above 0.500) (Figure 4.2c–d).

GLM results for species richness of assemblages showed no significant difference between the rehabilitated and non-rehabilitated eroded sites ($F_{1,124} = 2.305$; $p = 0.132$), with no interaction between sub-site position and presence of gabions ($F_{1,124} = 0.067$; $p = 0.796$). In addition, abundance showed no significant difference between soil assemblages and sampling position ($F_{1,124} = 0.049$; $p = 0.824$), as well as presence of gabions ($F_{1,124} = 3.052$; $p = 0.083$) and interaction between ($F_{1,124} = 3.331$; $p = 0.070$). PERMANOVA results showed no significant difference between assemblages of the different sub-sites (global $F_{3,124} = 0.748$; $p = 0.619$), with only two pairings showing significant differences in the post-hoc analysis (Supplementary material, Chapter 4, Table S4.1).

GLM results for sediment build-up showed a significant interaction between soil arthropod species richness and increasing sediment depth ($F_{1,61} = 4.631$; $p = 0.035$), as well as with sub-site position in front of the gabion ($F_{1,61} = 10.041$; $p = 0.002$) (Supplementary material, Chapter 4, Fig. S4.4).

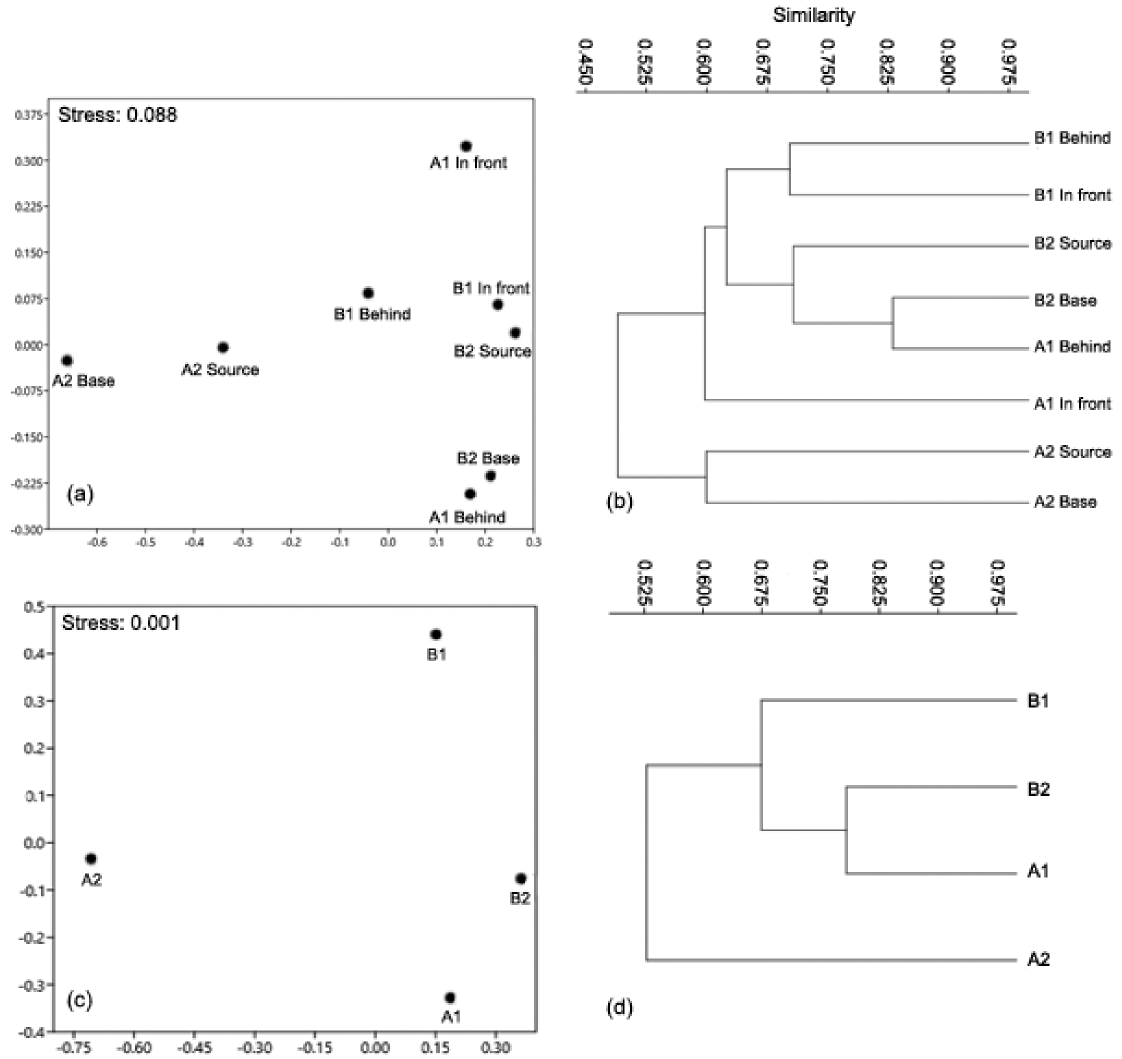

Fig. 4.2: Non-metric Two-dimensional Scaling ordination with Bray-Curtis similarity clustering using paired group algorithms, showing (a and b) pairing between sub-sites, and (c and d) pairing between overall sites

4.4 Discussion

4.4.1 Changes in soil arthropod assemblages in rehabilitated and non-rehabilitated eroded sites

The results of this study suggest that gabions do not significantly affect or change soil assemblages in the rehabilitated areas compared to the non-rehabilitated eroded sites. The lack of ecological correlation of species also hints at the striking similarity of soil arthropod assemblages between the investigated sites. It remains a question whether these implemented rehabilitation methods affect assemblages over a longer period. However, it is of interest to see that over the 16-month investigative period, assemblages showed no difference between the two site types, approximately 28 and 26 months after the establishment of the gabions at sites A and B, respectively. Results from a seven-year old rehabilitated site sampled during the same time suggest that assemblages could change drastically over longer periods of time, with decreased soil arthropod abundance and species richness found in highly sedimented rehabilitated sites (van der Merwe et al., in press). This emphasises the need for continuous monitoring of soil arthropod assemblages after implementing erosion rehabilitation methods, in order to assess changes in assemblage structure and function over time.

4.4.2 Effect of sediment build-up on soil arthropods

The effects of sediment caused by soil erosion are usually examined in respect to rivers and water bodies, due to the detrimental effect of siltation on these environments (de Vente and Poesen, 2005; Jain and Kothyari, 2009). The results of our study take a different approach to examining the effects of sediment build-up around erosion sites, mainly around the implemented gabions. A significant interaction was identified between soil arthropod richness and sediment depth, suggesting that there may be a possible link between sediment accumulation around gabions and species richness in these sites. This coincides with similar data found in the GGHNP in older rehabilitated sites sampled during the same time (van der Merwe et al., in press). Although the other area had a combination of rock-wall gabions and silt netting implemented, it is important to note the possible long-term effects of silt build-up on soil arthropods in rehabilitated areas, which may have an effect on their ecosystem functions. It is of ever-increasing importance to consider soil arthropod assemblages when studying the effects of erosion, as losses of the functions brought about by these biota groups could have a far-reaching detrimental effect on landscapes (Orgiazzi and Panagos, 2018). It is recommended that, even after rehabilitation method implementation, regular monitoring of both soil biota and the surrounding terrain should be done to gauge the progression and effects of sediment build-up in terrestrial terrains.

4.4.3 Conservation implications

Zoning in the GGHNP is done by means of analysing and mapping areas of sensitivity by their value as biophysical, heritage, and scenic resources (SANParks, 2013). These zones include protected and sensitive ecological areas, such as areas vulnerable to erosion losses and sites near water catchment areas. Implementation of soil erosion rehabilitation in the GGHNP is concentrated around the larger effects of sedimentation, mainly near water catchment areas, and therefore management strategies generally are focused on the prevention of siltation in aquatic areas and not on the direct effects on soil and its biota.

Constructed rock-filled gabions have been recommended for use to combat erosion losses in terrains worldwide (Nsor et al., 2013; Krishnan and Chowdary, 2016; Okorafor et al., 2017; Lee et al., 2019; Liu et al., 2019). The practices put in place to prevent erosion losses are mainly a response to concerns that erosion could largely affect the transport capacity of river systems, promoting land and water pollution, and subsequently jeopardising larger socio-economic objectives (Mihai et al., 2015). In South Africa, the use of gabions, along with other conservation strategies, has been found to promote restoration and sustainability of certain terrain types (e.g. Brent and Mulder, 2005; Blignaut et al., 2011; Nsor and Gambiza, 2013). In addition, many efforts to combat erosion are coupled with economic incentives and poverty alleviation strategies in SANParks-based terrains such as the GGHNP (SANParks, 2013). In light of this, the construction of gabions in the GGHNP to combat erosion and subsequent water system contamination have been counted as both an ecological and economic benefit in management strategies for the park. Our findings suggest that gabions did not cause significant differences in soil arthropod assemblages between the investigated rehabilitated and non-rehabilitated eroded sites during the 16-month study. However, further study is necessary to gauge whether gabions may have an effect over a longer term as sediment build-up increases.

Most research regarding erosion losses and sedimentation has been concentrated on water conservation and agricultural lands (Poesen et al., 1994; Kosmas et al., 1997; Pimental and Kounang, 1998; van Oost et al., 2000; Borselli et al., 2001; Benda et al., 2004; van Maren, 2007), with little emphasis on the soil assemblages associated with rehabilitated sites. Our study starts to address the aspect of the effects of gabion implementation on soil arthropods, which conservation science does not necessarily consider in conservation management strategies. The similarities between rehabilitated and non-rehabilitated soil arthropod assemblages, coupled with the significant interaction between species richness and sediment depth around gabions, suggests that further monitoring of these sites is needed to establish the changes in soil biota groups over time as the environment changes with increased silt build-up and establishment of vegetation.

It is of interest to note that our results indicated that further sampling is still necessary to accurately identify the state of soil arthropod populations in these areas, based on Chao1 values. It is possible that further intervention will be necessary after longer periods since gabion implementation in order to combat the effects of excessive sediment build-up, which could involve implementing newer rehabilitation strategies to further promote vegetation growth in deteriorated sites. Seed and fertilizer amendments performed alongside other erosion rehabilitation methods may strengthen the desired effects of rehabilitation by promoting pioneer plant growth, encouraging water retention to promote long-term plant establishment, and ultimately promote the conservation and establishment of beneficial organisms in the affected areas (Rivas, 2006). The benefits of implementing grass growth on rock-wall gabions have been noted to maintain vegetative growth, as root systems can remain firmly attached to the gabion mesh and underlying rock fill (Freeman and Fischenich, 2000; Matić, 2009). This not only increases silt retention ability and reduces hydrological flow in these eroded sites, but also can increase soil arthropod diversity and species richness over time due to the association of certain arthropod groups with plant root systems (Coleman et al., 2004; Wall et al., 2012). The use of endemic vegetation species in rehabilitation strategies may also ensure a higher success rate in erosion control, as these plants establish faster and more effectively (Xu et al., 2006). It is recommendeded that conservation strategies involving rock-wall gabions in the GGHNP be combined with other erosion control methods which may further promote plant growth.

4.5 Conclusion

The results show that no differences were found between soil arthropod abundance and species richness of the rehabilitated and non-rehabilitated sites. This was also true for the investigated sub-sites, showing similarities between assemblages on both sides of the gabion walls. However, findings did show that sediment build-up around constructed rehabilitation structures affected soil arthropod species richness significantly. The study provides a basis for studying the effects of erosion rehabilitation methods in light of soil biota in the GGHNP, and hints at the importance of surveying these assemblages in eroded areas as a part of a restoration regime.

4.6 References

Badenhorst, H., 2016. Mesofaunal Assemblages in Soil of selected Crops under Diverse Cultivation Practices in Central South Africa, with Notes on Collembola Occurrence and Interactions. MSc dissertation, University of the Free State, Bloemfontein.

Bates, D., Maechler, M., Bolker, B., Walker, S., 2015. Fitting linear mixed-effects models using lme4. J. Stat. Softw. 67, 1–48.

Benda, L., Andras, K., Miller, D., Bigelow, P., 2004. Confluence effects in rivers: interactions of basin scale, network geometry, and disturbance regime. Water Resour. Res. 40, 1–15.

Blignaut, J., van Ierland, J., Xivuri, T., van Aarde, R., Aronson, J., 2011. The ARISE project in South Africa. In: Egan, D., Hjerpe, E.E., Abrams, J. (Eds.), Human Dimensions of Ecological Restoration: Integrating Science, Nature and Culture. Island Press, Washington. pp. 207–219.

Borselli, L., Torri, D., Poesen, J., Sanchis, P.S., 2001 Effects of water quality on infiltration, runoff and interrill erosion processes during simulated rainfall. Earth Surf. Process. Landf. 26, 329–342.

Brent, A.C., Mulder, J., 2005. Selection of sustainable rural agriculture projects in South Africa: case studies in the LandCare programme. Engineering Management Conference Proceedings, IEEE International, Volume 1. https://doi.org/10.1109/IEMC.2005.1559136.

Chytrý, M., Tichý, L., Holt, J., Botta-Dukát, Z., 2002. Determination of diagnostic species with statistical fidelity measures. J. Veg. Sci. 13, 79–90.

Coleman, D.C., Crossley Jr., D.A., Hendrix, P.F., 2004. Fundamentals of Soil Ecology 2nd Edition. Elsevier Academic Press, California.

Colwell, R.K., 2013. EstimateS: statistical estimation of species richness and shared species from samples. http://viceroy.eeb.uconn.edu/EstimateS.

Culliney, T.W., 2013. Role of arthropods in maintaining soil fertility. Agriculture 3, 629–659.

De Cáceres, M., 2013. How to use the indicspecies package (ver. 1.7.1). https://cran.r-project.org/web/packages/indicspecies/vignettes/indicspeciesTutorial.pdf.

De Cáceres, M., Legendre, P., 2009 Associations between species and groups of sites: indices and statistical inference. Ecology 90, 3566–3574.

De Cáceres, M., Legendre, P., Moretti, M., 2010. Improving indicator species analysis by combining groups of sites. Oikos 119, 1674–1684.

De Deyn, G.B., Raaijmakers, C.E., Rik Zoomer, H., Berg, M.P., de Ruiter, P.C., Verhoef, H.A., Martijn Bezemer, T., van der Putten, W.H., 2003. Soil invertebrate fauna enhances grassland succession and diversity. Nature 422, 711–713.

De Vente, J., Poesen, J., 2005. Predicting soil erosion and sediment yield at the basin scale: scale issues and semi-quantative models. Earth-Sci. Rev. 71, 95–125.

Dufrêne, M., Legendre, P., 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecol. Monogr. 67, 345–366.

Dunger, W., Schlitt, B., 2011. Synopses on Paleartic Collembola. Museum of Natural History, Görlitz.

Durán Zuazo, V.H., Rodríguez Pleguezuelo, C.R., 2008. Soil-erosion and runoff prevention by plant covers. A review. Agron. Sustain. Dev. 28, 65–86.

Ekholm, P., Lehtoranta, J., 2012. Does control of soil erosion inhibit aquatic eutrophication? J. Environ. Manag. 93, 140–146.

Freeman, G.E., Fischenich, J.C., 2000. Gabions for Streambank Erosion Control. EMRRP Technical Notes Collection (ERDC TN-EMRRP-SR-22). US Army Engineer Research and Development Center, Vicksburg, Mississippi.

Gleason, R.A., Euliss Jr., N.H., Hubbard, D.E., Duffy, W.G., 2003. Effects of sediment load on emergence of aquatic invertebrates and plants from wetland soil egg and seed banks. Wetlands 23, 26–34.

Groffman, P.M., Bohlen, P.J., 1999. Soil and sediment biodiversity: cross-system comparisons and large-scale effects. Bioscience 49, 139–148.

Gulvik, M., 2007. Mites (Acari) as indicators of soil biodiversity and land use monitoring, a review. Pol. J. Ecol. 55, 415–440.

Hammer, Ø., Harper, D.A.T., Ryan, P.D., 2001. PAST: Palaeontological statistics software package for education and data analysis. Paleontol. Electron. 4, 1–9.

Harden, C.P., 2001. Soil erosion and sustainable mountain development: experiments, observations, and recommendations from the Ecuadorian Andes. Mt. Res. Dev. 21, 77–83.

Hooper, D.U., Adair, E.C., Cardinale, B.J., Byrnes, J.E.K., Hungate, B.A., Matulich, K.L., Gonzalez, A., Duffy, J.E., Gamfeldt, L., O'Conner, M.I., 2012. A global synthesis reveals biodiversity loss as a major driver of ecosystem change. Nature 486, 105–109.

Hunt, H.W., Wall, D.H., 2002. Modelling the effects of loss of soil biodiversity on ecosystem function. Glob. Change Biol. 8, 33–50.

Jain, M.K., Kothyari, U.C., 2000. Estimation of soil erosion and sediment yield using GIS. Hydrol. Sci. J. 45, 771–786.

Kosmas, C., Danalatos, N., Cammeraat, L.H., Chabart, M., Diamantopoulos, J., Farand, R., Gutierrez, L., Jacob, A., Marques, H., Martinez-Fernandez, J., Mizara, A., Moustakas, N., Nicolau, J.M., Oliveros, C., Pinna, G., Pudda, R., Puigdefabregas, J., Roxo, M., Simao, A., Stamou, G., Tomasi, N., Usai, D., Vacca, A., 1997. The effect of land use on runoff and soil erosion rates under Mediterranean conditions. Catena 29, 45–59.

Kremen, C., Colwell, R.K., Erwin, T.L., Murphy, D.D., Noss, R.F., Sanjayan, M.A., 1993. Terrestrial arthropod assemblages: their use in conservation planning. Conserv. Biol. 7, 796–808.

Krishnan, L., Chowdary, K.R., 2016. Adoption of eco-friendly conservation practices in the Nilgiris district of western Ghats. Int. J. Res. Appl. Nat. Soc. Sci. 7–16.

Lal, R., 2015. Restoring soil quality to mitigate soil degradation. Sustainability 7, 5875–5895.

Le Roux, J.J., Newby, T.S., Sumner, P.D., 2007. Monitoring soil erosion in South Africa at a regional scale: review and recommendations. SAEON Rev. 103, 329–335.

Lee, J-Y., Raza, M., Kwon, K.D., 2019. Land use and land cover changes in the Haean Basin of Korea: impacts on soil erosion. Episodes 42, 17–32.

Liu, X., Li, H., Zhang, S., Cruse, R.M., Zhang, X., 2019 Gully erosion control practices in Northeast China: a review. Sustainability 11, 1–16.

Matić, V., 2009. Use of gabions and vegetation in erosion-control works. Arch. Biol. Sci. Belgr. 61, 317–322.

Menta, C., Leoni, A., Gardi, C., Conti, F.D., 2011. Are grasslands important habitats for soil microarthropod conservation? Biodivers. Conserv. 20, 1073–1087.

Mhangara, P., Kakembo, V., Lim, K.J., 2012. Soil erosion risk assessment of the Keiskamma catchment, South Africa using GIS and remote sensing. Environ. Earth Sci. 65, 2087–2102.

Mihai, V., Dîrja, M., Hoble, A., 2015. Prevention of rain erosion on sloping terrains. ProEnvironment 8, 331–339.

Mucina, L., Hoare, D.B., Lötter, M.C., Du Preez, P.J., Rutherford, M.C., Scott-Shaw, C.R., Bredenkamp, G.J., Powrie, L.W., Scott, L., Camp, K.G.T., Cilliers, S.S., Bezuidenhout, H., Mostert, T.H., Siebert, S.J., Winter, P.J.D., Burrows, J.E., Dobson, L., Ward, R.A., Stalmans, M., Oliver, E.G.H., Siebert, F., Schmidt, E., Kobisi, K., Kose, L., 2006. Grassland biome. In: Mucina, L., Rutherford, M.C., (Eds), The vegetation of South Africa, Lesotho and Swaziland. SANBI, Pretoria. pp. 349–437.

Nsor, C.A., Gambiza, J., 2013. Land use changes and their impacts on the vegetation Kromme river peat basin, South Africa. Sci. J. Environ. Eng. Res. 166, 1–12.

Okorafor, O.O., Oluwakunmi, A.C., Jonathan, A.A., 2017. Soil erosion in South eastern Nigeria: a review. Sci. Res. J. 5, 30–37.

Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson, G.L., Solymos, P., Stevens, M.H.H., Szoecs, E., Wagner, H., 2019. Vegan, Community Ecology Package. R package version 2.5–5. https://CRANS.R–project.org/package=vegan.

Orgiazzi, A., Panagos, P., 2018. Soil biodiversity and soil erosion: it is time to get married. Glob. Ecol. Biogeogr. 27, 1155–1167.

Paoletti, M.G., Favretto, M.R., Stinner, B.R., Purrington, F.F., Bater, J.E., 1991. Invertebrates as bioindicators of soil use. Agric. Ecosyst. Environ. 34, 341–362.

Pimental, D., 2006. Soil erosion: a food and environmental threat. Environ. Dev. Sustain. 8, 119–137.

Pimental, D., Burgess, M., 2013. Soil erosion threatens food production. Agriculture 3, 443–463.

Pimental, D., Harvey, C., Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M., Crist, S., Shpitz, L., Fitton, L., Saffouri, R., Blair, R., 1995. Environmental and economic costs of soil erosion and conservation benefits. Science 267, 1117–1123.

Pimental, D., Kounang, N., 1998. Ecology of soil erosion in ecosystems. Ecosystems 1, 416–426.

Poesen, J.W., Torri, D., Bunte, K., 1994. Effects of rock fragments on soil erosion by water at different spatial scales: a review. Catena 23, 141–166.

R Core Team, 2019. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL: http://www.R-project.org/.

Rivas, T., 2006. Erosion Control Treatment Selection Guide. United States Department of Agriculture Forest Service.

Roberts, B.R., 1966. Conservation of the Vegetation of the Orange Free State. 6th A.E.T.A.T. Symposium, Uppsala.

Rousseau, L., Fonte, S.J., Téllez, O., van der Hoek, R., Lavelle, P., 2013. Soil macrofauna as indicators of soil quality and land use impacts in smallholder agroecosystems of western Nicaragua. Ecol. Indic. 27, 71–82.

Slingsby, P., 2017. Ants of Southern Africa: The Ant Book for All. Slingsby Maps, Muizenberg.

South African National Parks (SANParks), 2009. 2008-2009 SANParks annual report. SANParks.

South African National Parks (SANParks), 2013 Golden Gate Highlands National Park, Park Management Plan for the period 2013-2023. SANParks.

Theron, P.D., Ryke, P.A., 1969. The family Nanorchestidae Grandjean (Acari: Prostigmata) with descriptions of new species from South African soils. J. Entomol. Soc. South. Afr. 32, 31–60.

Triplehorn, C.A., Johnson, N.F., 2005. Borror and DeLong's Introduction to the Study of Insects 7th Edition. Thompson Brooks/Cole, Belmont.

Trombulak, S.C., Frissell, C.A., 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conserv. Biol. 14, 18–30.

Tullgren, A. 1918. Ein sehr einfacher Ausleseapparat fur terricole Tierformen. Z. Angew. Entomol. 4, 149–150.

van der Merwe, S.S., Swart, V.R., Bredenhand, E., Haddad, C.R., in press. Soil-dwelling arthropods as indicators of erosion in a South African grassland habitat. Pedobiologia. DOI: 10.1016/j.pedobi.2020.150647

van Maren, D.S., 2007. Grain size and sediment concentration effects on channel patterns of silt-laden rivers. Sediment. Geol. 202, 297–316.

van Oost, K., Govers, G., Desmet, P., 2000. Evaluating the effects of changes in landscape structure on soil erosion by water and tillage. Landsc. Ecol. 15, 577–589.

van Oudtshoorn, F., 2014. Guide to grasses of southern Africa. 3rd edition. Briza Publications, Pretoria.

Wall, D.H., Bardgett, R.D., Behan-Pelletier, V., Herrick, J.E., Jones, H., Rits, K., Six, J., Strong, D.R., van der Putten, W.H., 2012. Soil Ecology and Ecosystem Services. Oxford University Press, Oxford.

Wickham, H., 2016 ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag, New York.

Xu, X., Zhang, K., Kong, Y., Chen, J., Yu, B., 2006. Effectiveness of erosion control measures along the Qinghai-Tibet highway, Tibetan plateau, China. Transp. Res. D 11: 302–309.

Chapter 5:

The effect of renewal of previously implemented erosion rehabilitation methods on soil biota in a South African grassland habitat

Sylvia S. van der Merwe[a], Vaughn R. Swart[a], Emile Bredenhand[b], Charles R. Haddad[a]

[a] University of the Free State, Bloemfontein Campus, Bloemfontein, South Africa

[b] University of the Free State, Qwaqwa Campus, Phuthaditjhaba, South Africa.

Abstract

The use of rehabilitation methods is a useful tool against erosion in protected landscapes. These methods require monitoring and maintenance over time, being repaired and replaced as necessary. In certain cases, complete renewal of currently employed methods is needed, where entire sites may be reformed. This study looked at the effect of major restructuring of a single incidental site in the Golden Gate Highlands National Park, whereby previous rehabilitation structures were removed and the area flattened in preparation for renewed rehabilitation. Soil sampling was conducted for nine months before and nine months after renewal to observe changes in soil arthropod abundance and species richness in the affected site. Both species richness and diversity were notably higher after the renewal, but showed no significant difference in assemblage composition. Monte Carlo simulations showed no significant difference between species richness and diversity before and after the renewal. The results suggest that restructuring and renewal did not significantly affect soil arthropod assemblages in the affected site over the study period, but it could be argued that continued monitoring is necessary to fully gauge the effect of such disturbance on soil-dwelling arthropods.

Keywords: Arthropods; Diversity; Incidental; Restructuring; Species Richness

5.1 Introduction

Soils are an important component in monitoring the health of landscapes and have been used in assessing the health of environments (Doran and Zeiss, 2000; Arshad and Martin, 2002; Gulvik, 2007; Rosseau et al., 2013). The increased rate of soil erosion in terrestrial ecosystems is an important problem the environment is facing (Pimental et al., 1995; Pimental et al., 2004; Durán Zuazo and Rodríguez Pleguezuelo, 2008; Pimental and Burgess, 2013). Soil erosion reduces the quality and fertility of soils, subsequently decreasing the productivity of agricultural, natural, and forest ecosystems (Pimental, 2006). Landscapes in South Africa are no exception to this, as soil erosion is one of the most significant environmental disturbances causing various rates of ecosystem degradation (Meadows, 2003; Le Roux et al., 2007; Compton et al., 2010; Dlamini et al., 2011). Research suggests that over 70% of South African landscapes are at risk of the effects of soil erosion to varying degrees (Garland et al., 2000; Le Roux et al., 2008). The degree of erosion is dependent on ecosystem soil composition as well as the various sources of erosion (Meadows, 2003; Le Roux et al., 2007; Compton et al., 2010).

The Golden Gate Highlands National Park (GGHNP) in the eastern Free State Province of South Africa is well-known for its sandstone formations and cliffs (SANParks, 2013). Sandstone produces a shallow sandy soil with notably low fertility, which is highly susceptible to erosion losses (Robert, 1969). To combat the effects of soil erosion in a highly erodible landscape, yearly erosion control or rehabilitation projects are put in place by Park Management, concentrated specifically near the many streams running through the park. The initiatives are set in place by South African National Parks (SANParks) Management alongside SANParks Scientific Services as both an ecological conservation strategy as well as a community engagement and employment project, concentrated on the education and employment of the local population (SANParks, 2013). The Working for Water community group then implements the construction and maintenance of erosion control structures under the approval of management. In some cases, intervention on older rehabilitated structures is needed, and plans for renewal of site implements are put in place based on the level of deterioration. Many rehabilitation structures require maintenance on a regular basis, with structures being repaired and replaced as necessary. However, in certain cases, implements are too damaged to be maintained and renewal of implements is done to combat further deterioration.

The main aim of rehabilitation implementation in the GGHNP is to prevent siltation and excess sedimentation in rivers and streams in and around the park (SANParks, 2013). Many of the implemented methods, however, have little to no consideration given as to the effects of these implements, such as silt netting, rock-wall gabions and erosion control blankets, on soil-dwelling arthropods in the affected areas. Findings have shown that soil biota play a vital role in the health and functioning of ecosystems (Rousseau et al., 2010; Culliney, 2013), further emphasising the importance of considering soil assemblages specifically in landscapes at risk of deterioration (Orgiazzi and Panagos, 2018).

This study focuses on the effect of major restructuring of a rehabilitated site and the subsequent renewal of rehabilitation implements, after long-term use, on soil arthropod assemblages in an eroded site. The study site was part of a previous study investigating soil arthropods as bioindicators of erosion in the GGHNP, which was abruptly flattened and prepared for new rehabilitation implements in July 2018. It was hypothesised that restructuring of the site would result in a difference in soil arthropod assemblages before and after the renewed rehabilitation.

5.2 Materials and Methods

5.2.1 Study area and period

The study was conducted in the eastern parts of the GGHNP, located in the eastern Free State Province of South Africa. The national park borders the country of Lesotho, situated at the foothills of the Maluti-Drakensberg Mountains. It covers an area of 340 km^2 and has an abundance of several deeply eroded sandstone cliffs and outcroppings surrounded by large areas of natural grassland. Several marked river lines and small streams run through the landscape of the park, with wetlands forming during peak rainfall seasons (SANParks, 2013).

The affected site was located on the far eastern side of the park (28°25.284’S, 28°45.477’E, elevation: 1731 m), near part of a perennial river line. The site was initially selected as part of an already established erosion study alongside comparative sites, but was altered unexpectedly several months after the commencement of sampling. The site was previously rehabilitated using rock-wall gabions and silt catchment netting 6 years before the commencement of the study (SANParks personnel, personal communication, 2016[1]). During July 2018, previously implemented rehabilitation structures were removed and the area was flattened in preparation of renewed rehabilitation (Supplementary material, Chapter 5, Fig. S5.1a-b). New structures, including erosion control blankets and fibre rolls, were implemented by the start of August 2018 (Supplementary material, Chapter 5, Fig. S5.1c).

5.2.2 Soil arthropod sampling

Soil arthropods were initially sampled from this site to observe occurrence and overall abundance in order to establish soil faunal groups in erosion sites in the GGHNP. Soil samples were taken from the affected site nine months before the renewed rehabilitation and nine months after as a comparison before and after renewal. A single soil sample was defined as a soil quantity of between 400 and 500 g taken within the top 10 cm of soil and in a 10 cm radius of a chosen sampling spot. Five incorporated samples were taken in a linear design along the eroded site at the end of each month during the period of October 2017 to March 2019. Two sub-samples were taken at each sampling point and pooled together to form each incorporated sample. Sampling sequence was done from the highest point in the eroded furrow to the lowest point of soil deposition before reaching the water body.

[1] T. Nsibande, verbal communication, Working for Ecosystems Park Ranger - Golden Gate Highlands National Park, 21 September 2016.

All samples were transported to the Department of Zoology and Entomology, University of the Free State, South Africa, and placed into individual Berlese-Tullgren funnels with connected storage bottles containing 70% ethyl alcohol (Tullgren, 1918; Triplehorn and Johnson, 2005; Badenhorst, 2016) for a period of seven days, to allow for sufficient soil arthropod extraction. Arthropods were identified to morphospecies level, with special consideration given to groups of mites (Oribatida, Mesostigmata and Prostigmata), Collembola and other insects, with the assistance of specialists and associated literature (Theron and Ryke, 1969; Olivier and Theron, 1989; Triplehorn and Johnson, 2005; Dunger and Schlitt, 2011; Hugo-Coetzee, 2013; Badenhorst, 2016; Slingsby, 2017).

5.2.3 Statistical analysis

Observed species richness and Shannon-Wiener diversity for the affected study site was quantified using EstimateS, version 9.1.0 (Colwell, 2013), and plotted using boxplot analyses to investigate changes before and after rehabilitation renewal. Non-metric Two-dimensional Scaling ordination using Bray-Curtis similarities was done to identify differences between soil arthropod assemblages in each sample period. In addition, a cluster analysis was done using the Bray-Curtis similarity matrix, with paired group algorithms, to identify possible patterns of clustering of assemblage groups sampled before and after the rehabilitation renewal. In order to test for variance between the two periods, a Permutational Multivariate Analysis of Variance (PERMANOVA) was conducted. PERMANOVA, ordination and cluster analyses were done using Paleontological Statistics (PAST), version 3.16 (Hammer et al., 2001), while boxplot analyses were conducted using the GGPLOT2 package in R, version 3.5.3 (Wickham, 2016).

In order to quantify the effect of renewal on soil-dwelling species assemblages, a comparison of observed species richness and diversity before and after the implementation was conducted. Due to the incidental renewal process occurring in an already established site, data was analysed as a stand-alone occurrence, with samples taken over a time series. To overcome bias, due to non-independence in this time series, soil arthropod samples taken before and after renewal were compared using Monte Carlo simulations by bootstrapping, over 1000 permutations with replacement. Replacements and simulations were done using R, version 3.5.3 (R Core Team, 2019).

5.3 Results

Overall, 1125 individuals were sampled during the 18-month period, representing 13 orders and 54 families. Both diversity and species richness were observed to be slightly higher during the sampling period after the rehabilitation renewal (Fig. 5.1). The increase in observed species richness and diversity may be attributed to the displacement of soil-dwelling arthropods from the soil surrounding the original site into the furrows during the flattening of the site.

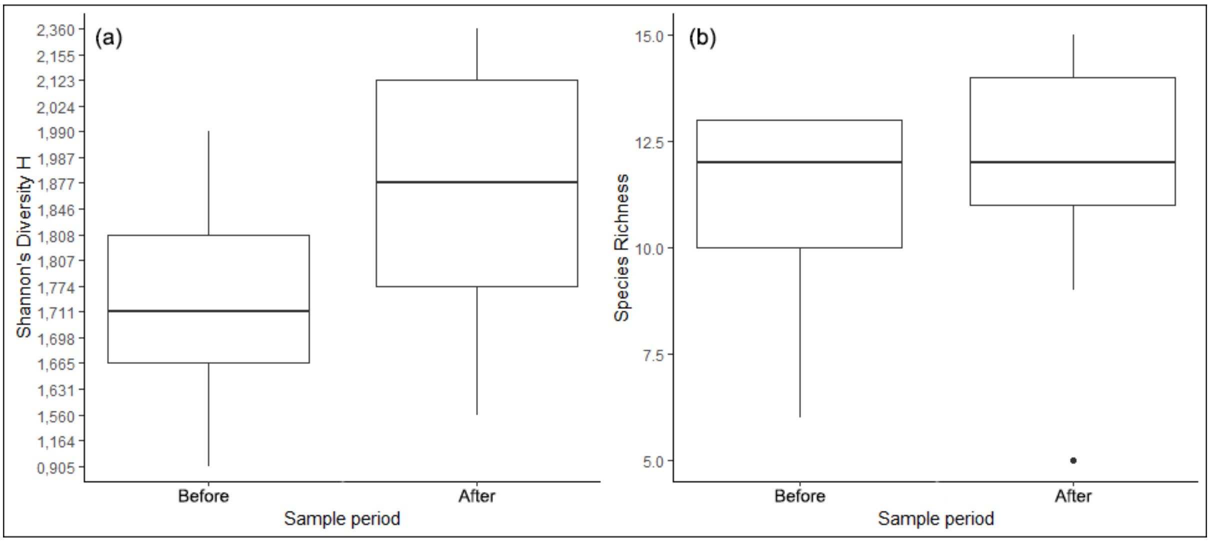

Fig. 5.1: Boxplot analyses of (a) soil arthropod species diversity, and (b) species richness before and after the implemented renewal, with dots representing outlier sample months.

Results of the ordination showed some overlap between assemblages of arthropods sampled before and after the renewal event, with clustering showing no distinct groupings separating assemblages sampled in each period (Fig. 5.2). Clustering between months showed variable levels of similarity.

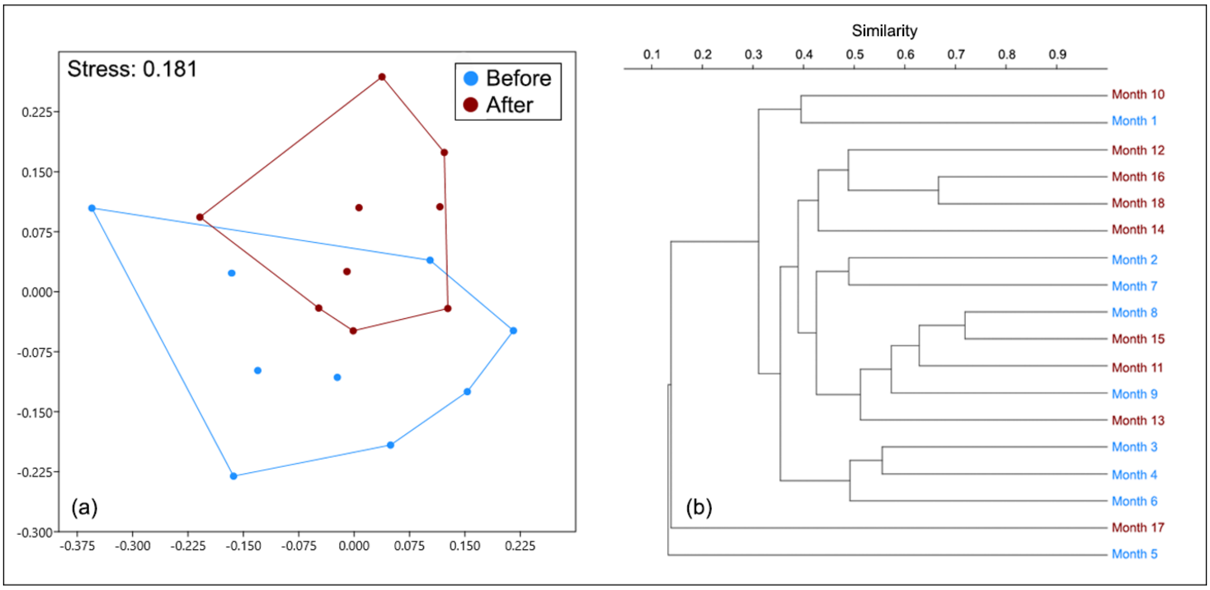

Fig. 5.2: (a) Non-metric Two-dimensional Scaling showing species overlap between assemblages before and after the renewal, and (b) Bray-Curtis similarity clustering, using paired group algorithms, showing pairings between monthly soil assemblages.

Permutational analysis showed no significant difference between assemblages before and after the renewal ($F_{1,16} = 1.450$; $p = 0.099$). The results of the Monte Carlo simulations supported this, with no significant differences found when comparing the diversity and species richness of assemblages before and after the renewal (Table 5.1).

Table 5.1: Results of Monte Carlo simulations with tested hypotheses. H - Shannon Wiener diversity

H_0 = before<after	
	p-value
Species richness	0.493
H	0.683
Therefore, Hypothesis rejected	
H_a = before>after	
	p-value
Species richness	0.507
H	0.317
Therefore, Hypothesis rejected	

5.4 Discussion

Soil arthropods showed variable responses to disturbances and environmental change (Neave and Fox, 1998; Frouz, 1999; Barbercheck et al., 2009; Coyle et al., 2017). The study provided a look into the effect of soil displacement into a previously rehabilitated furrow, and how newly implemented rehabilitation methods affect soil arthropod assemblage structure and abundance. The results showed a notable difference in species richness and diversity between the period before and after the renewal. It could be suggested that the increased species richness and diversity seen after the renewal was due to the mass displacement of soil from around the site, subsequently displacing soil arthropods into the furrow. Although the soil movement into the original furrow did alter the nature of assemblages over the course of the investigation, it is unsure as to whether this would be an indefinite change to the assemblage structure and whether this would promote soil health over time.

Results hinted strongly at the rehabilitation renewal having no significant effect on species assemblages over a nine-month period after the renewal compared to the nine months before. This can be explained by the similarity in species assemblages found before and after the rehabilitation, with species groups being similar between the original furrow site and its close surrounding soils. It is unlikely that individuals already in the furrow soils were able to migrate upwards by such a large distance due to their limited movement capabilities in soil (Bengtsson et al., 1994; Hubbell, 2001), suggesting that arthropods found after the rehabilitation were those most likely residing in the displaced soils.

Many methods of soil erosion rehabilitation are utilised primarily in the prevention of water siltation. Rock-wall gabions are used in a multitude of methods, with functions ranging from prevention of soil and land slide damage onto roads, near infrastructure and water bodies (Rivas, 2006). Similarly, silt netting structures are used in many forms, with their usage heavily dependent on size, height, build structure, and materials used to construct these soil barriers. Primarily, both these implements are utilised in the prevention of silt flow into water bodies and near infrastructure, while still allowing water flow through the structure (Rivas, 2006). Erosion control blankets and fibre roll implementation are regarded as a temporary method of mitigating soil erosion along sloping terrains as the materials used are typically biodegradable and tend to wash away over time if used alone (Mn/DOT, 2006; Rivas, 2006). These methods are more effective in conjunction with establishing vegetation and planting pioneering or cover seeds (Mn/DOT, 2006; Rivas, 2006).

In South Africa, the effect of erosion mitigation methods is not readily studied, and even less attention is usually given to erosion structures in respect to soil-dwelling arthropods. As the effects of all of the above methods on soil-dwelling arthropods were not quantified in comparison in this study, it is difficult to draw any conclusions on the direct effects of these implements on soil-dwelling arthropods and their function. However, this does emphasise a need to study soil erosion mitigation structures and its short- and long-term effects on soil arthropod assemblages.

It is often difficult to pinpoint the major drivers of soil arthropod population dynamics due to their varied and somewhat hidden lifestyles below-ground (Stork and Eggleton, 1992;

Behan-Pelletier, 1993; Wolters, 2001). Although studies have investigated soil arthropod responses to chemical change and pollutants (Eijsackers, 1983; Paoletti et al., 1991; Nahmani and Rossi, 2003; van Straalen, 2004), little is known as to their physiological and behavioural responses to terrain disturbances, and how disturbances, such as physical land reform, may alter assemblages over long-term. Certain soil arthropod groups may not show any response to environmental change whatsoever, and may respond to different drivers rather than changes in the soil (Madson, 2003). This study gives insight into the effect of renewed rehabilitation on soil arthropod assemblages in an area prone to sheet and gully erosion. However, the precise factors affecting soil arthropods in renewed rehabilitated areas in the GGHNP still needs to be investigated.

5.5 Conclusion

Although this study only includes the observations from one incidental site, the study results briefly addressed the effect of renewed rehabilitation on arthropod assemblages, and could be used as a starting point for studying the effect of rehabilitation structure maintenance methods. The results do not necessarily show that restructuring of the site would result in differences in soil arthropod assemblages before and after the renewal, but rather suggests that no significant change was caused by the rehabilitation over short-term. It is important to understand that this may have been an isolated case that may not be replicated in other terrains, suggesting that a closer look needs to be taken into rehabilitation practices across South Africa to accurately gauge the effect of these practices on soil assemblages.

5.6 References

Arshad, M.A., Martin, S., 2002. Identifying critical limits for soil quality indicators in agro-ecosystems. Agric. Ecosyst. Environ. 88, 153–160.

Badenhorst, H., 2016. Mesofaunal Assemblages in Soil of selected Crops under Diverse Cultivation Practices in Central South Africa, with Notes on Collembola Occurrence and Interactions. MSc dissertation, University of the Free State, Bloemfontein.

Barbercheck, M.E., Neher, D.A., Anas, O., El-Allaf, S.M., Weicht, T.R., 2009. Response of soil invertebrates to disturbance across three resource regions in North Carolina. Environ. Monit. Assess. 152, 283–298.

Behan-Pelletier, V.M., 1993. Diversity of soil arthropods in Canada: systematic and ecological problems. Mem. Entomol. Soc. Can. 125, 11–50.

Bengtsson, G., Hedlund, K., Rundgren, S., 1994. Food- and density-dependent dispersal: evidence from a soil collembolan. J. Anim. Ecol. 63, 513–520.

Buser, P., 2003 Soil macroinvertebrates as indicators of pollution by heavy metals. C. R. Biol. 326, 295–303.

Colwell, R.K., 2013. EstimateS: statistical estimation of species richness and shared species from samples. http://viceroy.eeb.uconn.edu/EstimateS.

Compton, J.S., Herbert, C.T., Hoffman, M.T., Schneider, R.R., Stuut, J-B., 2010 A tenfold increase in the Orange River mean Holocene mud flux: implications for soil erosion in South Africa. Holocene 20, 115–122.

Coyle, D.R., Nagendra, U.J., Taylor, M.K., Campbell, J.H., Cunard, C.E., Joslin, A.H., Mundepi, A., Phillips, C.A., Callaham Jr., M.A., 2017. Soil fauna responses to natural disturbances, invasive species, and global climate change: Current state of the science and a call to action. Soil Biol. Biochem. 110, 116–133.

Culliney, T.W., 2013. Role of arthropods in maintaining soil fertility. Agriculture 3, 629–659.

Dlamini, P., Orchard, C., Jewitt, G., Lorentz, S., Titshall, L., Chaplot, V., 2011. Controlling factors of sheet erosion under degraded grasslands in the sloping lands of KwaZulu-Natal, South Africa. Agric. Water Manag. 98, 1711–1718.

Doran, J.W., Zeiss, M.R., 2000 Soil health and sustainability: managing the biotic component of soil quality. Appl. Soil Ecol. 15, 3–11.

Dunger, W., Schlitt, B., 2011. Synopses on Paleartic Collembola. Museum of Natural History, Görlitz.

Durán Zuazo, V.H., Rodríguez Pleguezuelo, C.R., 2008. Soil-erosion and runoff prevention by plant covers. A review. Agron. Sustain. Dev. 28, 65–86.

Eijsackers, H., 1983. Soil fauna and soil microflora as possible indicators of soil pollution. Environ. Monit. Assess. 3, 307–316.

Frouz, J., 1999. Use of soil dwelling Diptera (Insecta, Diptera) as bioindicators: a review of ecological requirements and response to disturbance. Agric. Ecosyst. Environ. 74, 167–186.

Garland, G.G., Hoffman, M.T., Todd, S., 2000. Soil degradation. In: Hoffman, M.T., Todd, S., Ntshona, Z. and Turner, S. (Eds.), A National Review of Land Degradation in South Africa. South African National Biodiversity Institute, Pretoria, South Africa. pp 69–107.

Gulvik, M., 2007. Mites (Acari) as indicators of soil biodiversity and land use monitoring, a review. Pol. J. Ecol. 55, 415–440.

Hammer, Ø., Harper, D.A.T., Ryan, P.D., (2001) PAST: Palaeontological statistics software package for education and data analysis. Palaeontol. Electron. 4, 1–9.

Hubbell, S.P., 2001. The Unified Neutral Theory of Biodiversity and Biogeography. Princeton University Press, Princeton.

Hugo-Coetzee, E., 2013. New species of *Aleurodamaeus* Grandjean, 1954 (Oribatida: Aleurodamaeidae) from South Africa. Zootaxa 3670, 531–556.

Le Roux, J.J., Morgenthal, T.L., Malherbe, J., Pretorius, D.J., Sumner, P.D., 2008. Water erosion prediction at a national scale for South Africa. Water SA 34, 305–314.

Le Roux, J.J., Newby, T.S., Sumner, P.D., 2007. Monitoring soil erosion in South Africa at a regional scale: review and recommendations. S. Afr. J. Sci. 103, 329–335.

Madson, S.L., 2003. Response of Soil Microarthropods and Microclimatic Conditions to two-age Regeneration in Hardwood Forests in the Southern Appalachians. MSc dissertation, University of Georgia, Athens.

Meadows, M.E., 2003 Soil erosion in the Swartland, Western Cape Province, South Africa: implications of past and present policy and practice. Environ. Sci. Policy 6, 17–28.

Mn/DOT, 2006. Erosion Control Handbook II. Department of Transportation Minnesota.

Neave, P., Fox, C.A., 1998. Response of soil invertebrates to reduced tillage systems established on a clay loam soil. Appl. Soil Ecol. 9, 423–428.

Olivier, P.A.S., Theron, P.D., 1989. A new species of *Speleorchestes* (Nanorchestidae: Prostigmata) from a savanna biotope in South Africa. S. Afr. J. Zool. 24, 356–360.

Orgiazzi, A., Panagos, P., 2018. Soil biodiversity and soil erosion: it is time to get married. Glob. Ecol. Biogeogr. 27, 1155–1167.

Pimentel, D., 2006. Soil erosion: a food and environmental threat. Environ. Dev. Sustain. 8, 119–137.

Paoletti, M.G., Favretto, M.R., Stinner, B.R., Purrington, F.F., Bater, J.E., 1991. Invertebrates as bioindicators of soil use. Agric. Ecosyst. Environ. 34, 341–362.

Pimental, D., Burgess, M., 2013. Soil erosion threatens food production. Agriculture 3, 443–463.

Pimental, D., Harvey, C., Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M., Crist, S., Shpitz, L., Fitton, L., Saffouri, R., Blair, R., 1995. Environmental and economic costs of soil erosion and conservation benefits. Science 267, 1117–1123.

R Core Team, 2019. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL: http://www.R-project.org/.

Rivas, T., 2006. Erosion Control Treatment Selection Guide. United States Department of Agriculture Forest Service.

Rousseau, G.X., Silva, P.R.S., Carvalho, C.J.R., 2010. Earthworms, ants and other arthropods as soil health indicators in traditional and no-fire agro-ecosystems from eastern Brazillian Amazonia. Acta Zool. Mex. 2, 117–134.

Rousseau, L., Fonte, S.J., Téllez, O., van der Hoek, R., Lavelle, P., 2013. Soil macrofauna as indicators of soil quality and land use impacts in smallholder agroecosystems of western Nicaragua. Ecol. Indic. 27, 71–82.

Slingsby, P., 2017. Ants of Southern Africa: The Ant Book for All. Slingsby Maps, Muizenberg.

South African National Parks (SANParks), 2013 Golden Gate Highlands National Park, Park Management Plan for the period 2013-2023. SANParks.

Stork, N.E., Eggleton, P., 1992. Invertebrates as determinants and indicators of soil quality. Am. J. Altern. Agric. 7, 38–47.

Theron, P.D., Ryke, P.A., 1969. The family Nanorchestidae Grandjean (Acari: Prostigmata) with descriptions of new species from South African soils. J. Entomol. Soc. South. Afr. 32, 31–60.

Triplehorn, C.A., Johnson, N.F., 2005. Borror and DeLong's Introduction to the Study of Insects 7th Edition. Thompson Brooks/Cole, Belmont.

Tullgren, A. 1918. Ein sehr einfacher Ausleseapparat fur terricole Tierformen. Z. Angew. Entomol. 4, 149–150.

van Straalen, N., 2004. Chapter 6 The use of soil invertebrates in ecological surveys of contaminated soils. In: Doelman, P., Eijsackers, J.P. (Eds.), Vital Soil: Function, Value and Properties. Elsevier B.V., Amsterdam. pp. 159–195.

Wolters, V., 2001. Biodiversity of soil animals and its function. Eur. J. Soil Biol. 37, 221–227.

Chapter 6:

General Discussion

6. General Discussion

The studies reported are the first of their kind for the Golden Gate Highlands National Park (GGHNP), addressing a part of soil erosion and fire studies that is not readily investigated in South Africa. The findings in this thesis pave the way for soil arthropod research in the GGHNP, providing a look into the possible uses of soil arthropod assemblages in identifying the status of certain landscapes in the park.

6.1 Possible implications of findings and future investigation

In Chapter 2, soil arthropod diversity and richness was higher overall in the burnt site in comparison to the non-burnt site. The results hint at the possibility of fires promoting soil biodiversity in the park, with species being more adapted to burnt landscapes, as would be expected in a fire-prone ecosystem such as the GGHNP. This does not necessarily mean that regularly applied prescribed fires, or even more frequent sweeping wildfires, cannot affect these below-ground assemblages, as it is not clear whether fires applied more often than once a year in the GGHNP could drastically alter soil arthropod assemblages in and around affected areas. This definitely comes into question by the steep decrease in species richness and abundance during the month immediately after the implemented fire. In addition, the movement limitations of soil arthropods (Madson, 2003) could mean that, in the case of more frequent fires, the effect on assemblages could prove to be long-term, i.e. if fires affect these assemblages frequently, restoration of populations could slow down, affecting the rate of arthropod-driven ecosystem services. The lower diversity in the twelfth month post-fire compared to the one-month pre-fire hinted at changes in the assemblages over the 12-month period, possibly due to annual differences in assemblage structure. Although this may have not been due to the direct effects of the burning, this highlighted a need for investigation on fire effects in the GGHNP to identify the patterns and behaviour of soil arthropod assemblages when under fire disturbance, as well as the factors that affect these assemblages on a yearly cycle.

Although the findings of this section are based on results from a single burnt site compared to a non-burnt site, the findings suggest that it is worth further investigating the effects of prescribed fire on soil-dwelling arthropod assemblages in the GGHNP at a larger scale. Surprisingly, soil arthropods were found to be more species rich and abundant in the burnt site overall. As discussed in Chapter 2, it can be considered that the arthropod assemblages possess traits that help these populations survive frequent fires in the fire-prone area, suggesting that assemblages might in fact thrive in burnt sites. Fires are also a known factor of nutrient cycling, returning organic material back to the soil and, in turn, resulting in enrichment of soils that benefits both soil arthropods and their food sources. Ultimately, the results suggest that prescribed fires regulate areas that are already prone to burning, contributing to maintaining a differing niche for soil arthropod assemblages adapted to soils affected by fires and, in turn, maintain soil arthropod population heterogeneity among different sites. Future investigation to verify the benefits and effects of fires should include the study of burnt sites, both from areas burnt on a yearly prescribed basis and sites burnt by runaway fires, with monitoring of assemblages over long-term to assess the long-term effects of fire on soil arthropods and soil health. Determining the effect of fire on these assemblages could help in making informed schedules for burning for optimal conservation of soil assemblages to preserve arthropod-driven ecosystem services, while still maintaining the primary objectives of removing excess burnable biomass and mitigating possible future runaway fire damage.

The GGHNP is sectional in its structure, showing similar patterns of division in its soil arthropod structures and populations (Chapter 3). Given the history of land use in the park, the differences between assemblage diversity found in the eastern and western parts of the park start to make sense. Eastern sectors of the park were readily used as agricultural lands (Rademeyer and van Zyl, 2014), with some areas still being utilised as grazing and feeding lands for livestock being held in the area today (SANParks personnel, personal communication, 2019[1]). Studies have shown higher species richness and diversity of soil arthropods in agricultural soils of some landscapes, possibly due to agricultural practices, such as mixed crop rotation, cover crops and nutrient restoration, altering soil ecosystem function (Baker, 1998; e.g. Bardgett and Cook, 1998; Bedano et al., 2016). The western sector of the GGHNP, on the other hand, contains what can be considered as 'younger soils', i.e. majority of the western landscapes contain shallow, erodible, low fertility sandstone-derived soils (Roberts, 1969) with little exposure to grazing livestock and agricultural practices. The lower species richness and abundance in soil arthropods shown from the western sites support this, showing a clear difference in assemblage structure between eastern and western sites (Chapter 3). For many years, studies were done separately on the GGHNP and the Qwaqwa National Park before their amalgamation in 2008 (SANParks, 2013; e.g. checklists on mammals: Rautenbach, 1976; Avenant, 1997). Different land use practices between each sector have resulted in some defining ecosystem variation, some of which could explain why soil arthropod diversity was higher in the eastern sites containing higher fertility soils. Given that no known studies have been done on soil-dwelling arthropods in any ecosystems in the area, the findings emphasise the need to study the park in its respective landscape type, with each section of the park assessed independently.

More importantly, Chapter 3 showed that soil arthropods in eroded sites of the GGHNP have potential as bioindicators of landscape status. The results emphasise a rather unusual observation of soil arthropod groups being more species rich and abundant in non-rehabilitated

[1] D. Nariandas, verbal communication, Senior Section Range/Conservation Manager - Golden Gate Highlands National Park, 22 August 2019.

eroded sites, strongly suggesting that soil biota in these areas have adapted to erosion. This is not so out of the ordinary when considering that the mountainous area of the GGHNP experiences high levels of water erosion, due to its steeply sloped landscapes, during each yearly rainfall season. It can be assumed that soil biota in these areas have adapted to this regular phenomenon, as erosion can be regarded as a natural process which plays an integral role in the formation of mountainous slopes (Mhangara et al., 2012). It is possible that erosion sites in the GGHNP in fact provide a niche for soil arthropods adapted to erosion movements and changes. This, in turn, increases niche types in the park, which accommodate a higher diversity and richness of soil arthropod species similar to what was found in Chapter 2. Interestingly, indicator analyses only highlighted a single species of mite as a strong indicator of non-rehabilitated eroded sites (Chapter 3). The shared nature of species between landscapes may indicate that erosion and erosion rehabilitation does not necessarily result in drastic differences in species compositions of each site type. However, when examining species richness of soil arthropod groups against soil mineralogy, it can be deduced that it is possible to utilise overall soil arthropod richness of investigated eroded areas, in conjunction with changes in the soil, to gauge the effects of erosion on these assemblages (Chapter 3). This concretes the need to regard these organisms as important bioindicators of disturbance. Being one of the first of its kind in the GGHNP, the results of this study emphasises the importance of monitoring soil arthropods in eroded sites in the context of conservation strategies. Future studies should focus on the monitoring of soil arthropod groups in the various non-rehabilitated eroded sites of the GGHNP to grade their potential as unique niches in the park. An intensive look into the ecosystem processes these groups carry out would clarify their significance as role-players, and ultimately identify groups that should be considered during mitigation procedures for erosion.

Results of Chapter 4 and 5 fell under a similar category of investigating the effects of soil erosion mitigation on soil arthropod assemblages. The effects of current rehabilitation methodologies are sorely understudied despite their wide utilisation. Additionally, emphasis is commonly given to vegetation regrowth as an indication of ecosystem restoration, while responses of soil-faunal activity are often overlooked. The results of these two chapters provided a preliminary look at how soil arthropods respond to current erosion mitigation, with the most significant finding being the effect of sedimentation build-up on species richness. As discussed in Chapter 4, implementation of rehabilitation methods is often focussed on the prevention of siltation of water bodies, ensuring the conservation of one of South Africa's most limited resources.

Unlike other studies done on sediment caused by soil erosion, the study in Chapter 4 took a different approach in examining the effects of sediment build-up around rock-wall gabions. A significant interaction was found between sediment build-up and soil arthropod richness, implying that there may be a possible link between sediment accumulation around implemented rock-wall gabions and soil arthropod species richness in rehabilitated sites. This coincided with data obtained from a seven-year-old, similarly rehabilitated site in the GGHNP area, hinting at a possible pattern of long-term effects of silt build-up on soil assemblages in rehabilitated sites. Given the fact that rock-wall gabions and other erosion mitigation structures play a role specifically in the prevention of water sedimentation, it can be recommended that mitigation structure implementation not be used as a stand-alone solution to the problem of soil erosion. It should, instead, be included as a part of an integrated implementation plan including the planting of establishing pioneer plants, restoration of indigenous plant growth, and consistent monitoring of ecosystem effects brought about by mitigation structures.

The incidental restructuring of currently employed erosion mitigation methods from a single site provided a unique opportunity to investigate how large scale land reform may impact soil arthropod communities and their responses (Chapter 5). It was initially expected that the displacement of soil and incorporation of new structures would change the environmental conditions of a locality. However, this was not necessarily the case in this study, as results showed no significant effect on species assemblages over the months after removal of rock-wall gabions and silt netting, major restructuring of the landscape, and subsequent implementation of erosion control blankets and fibre rolls. The similarity between assemblages sampled before and after the renewal of the rehabilitated eroded site suggests that assemblages found within the eroded furrow were strikingly similar to assemblages in the surrounding soil of the affected area. While the results hinted at rehabilitation renewal having no significant impact on soil arthropod assemblages over the 18-month period, the question remains as to its long-term implications. Incidentally, the importance of constant monitoring of soil species groups was highlighted in this study. The observed increase in species richness and diversity could easily be mistaken for an increase because of the newly implemented erosion mitigation structures. However, as discussed in Chapter 5, the increase was most likely the result of mass displacement of soil from around the site, subsequently displacing soil arthropods into the furrow of the eroded site. It is unclear as to whether this would benefit the filled area over long-term, causing an indefinite change to the assemblage structure and promoting soil health to any extent. In light of this, it is vital that these site types are monitored on a regular basis to establish patterns of change in soil arthropod groups even long after the establishment of rehabilitation structures.

Despite the incidental nature of this study, the results have still provided an interesting insight into the use of rehabilitation methods and their implementation from a conservation standpoint. While the renewal of current rehabilitation structures may often be decided upon due to age and size of the eroded locality, the effect of certain structures on soil arthropod assemblages must be taken into consideration. Future emphasis is suggested to determine how different erosion rehabilitation methods impact soil arthropod assemblages, allowing for the cycling of certain structures to better benefit soil health.

6.2 Recommendations to SANParks Scientific Services and Final comments

Applying fire to an area that is already fire-prone has proven to be an effective regulatory practice that acts on various conservation levels. This rings particularly true for the preliminary study on prescribed fire in the GGHNP. The findings suggest that the prescribed fire did not have a detrimental effect on soil arthropod assemblages over a 12-month period. In fact, the burnt site exhibited a higher soil arthropod species abundance and richness in comparison to a non-burnt site. However, as this was an investigation on a single burnt site, it is recommended that this be further investigated and monitored in several sites. Something to keep in mind is that, although no significant changes were detected after a single burn, soil assemblages may react differently to burning due to their specific morphological and physiological traits. Monitoring of soil assemblages over short- and long-term periods in different localities should be given consideration to gauge the full effect of prescribed fire on soil arthropod assemblages in the park.

Studies in South Africa have implied that erosion can be regarded as wholly and indiscriminately 'bad', resulting in unsustainable landscapes and cascading deterioration of affected sites and the surrounding areas. It is true that the deterioration of soils could result in the ultimate collapse of an entire ecosystem, as soil is integrally linked to all terrestrial food webs (Coleman et al., 2004). However, results obtained from erosion areas in the GGHNP suggest that non-rehabilitated eroded sites, in fact, support more diverse and abundant soil arthropod populations in comparison to rehabilitated eroded and non-eroded undisturbed sites. Soil biota diversity is regularly sought after, as this usually means that populations carry out a larger number of interlinked functions in an ecosystem, subsequently contributing to soil health, and thus emphasising the importance of soil arthropod diversity in the protection of biodiversity (Coleman et al. 2004). The GGHNP is an area that naturally experiences high levels of water-caused erosion, such as sheet and gully erosion due to heavy rainfall, on a yearly basis due to the terrain's natural sloping structure. It is a safe assumption that, although new erosion areas have formed in the park, even during recent years, soil arthropod populations have adapted to surviving, and indeed thriving, in areas of erosion over many years in the GGHNP area. This means that erosion sites, in fact, result in a unique niche for these adapted soil assemblages, supporting a greater number of species in comparison to grassland soils.

A question remains: What does this mean for rehabilitation of erosion? Although the findings suggest that sediment build-up over long-term around rehabilitation structures could negatively affect soil arthropods over time, the benefits of erosion rehabilitation on the prevention of water siltation and sedimentation still need to remain a priority. Given that the park forms part of the most important water catchment systems in Southern Africa (SANParks, 2013), it is clear as to why biodiversity programmes in the park are focused on water in the landscape. However, all of this does not necessarily mean that there is no way to find methods to protect soil biodiversity while also achieving water protection goals. To date, no studies have been done on the effect of rehabilitation structures, such as rock-wall gabions and silt netting, on soil in South Africa. In some cases, these implements are used in isolation to block sediment only and slow down the flow of soil. However, this does not necessarily mitigate the loss of plant cover. A recommendation to consider here is to use static structures in conjunction with the plantation of pioneer plants. This plant growth is meant to promote further plant growth in rehabilitated areas, encouraging water retention to promote long-term plant establishment, and ultimately promote the establishment and thriving of beneficial biota in the affected area (Rivas, 2006). A similar approach could be used in the GGHNP, simultaneously meeting water protection goals and promoting soil biodiversity in rehabilitated areas. Ultimately, further study into integrated rehabilitation strategies will prove vital in mitigation of both water and biodiversity preservation.

Overall, the findings in this thesis emphasise a single overarching point: soil arthropods in the GGHNP deserve more study as useful tools in bioindication of disturbance in the park. These soil biota assemblages in the GGHNP show great potential as bioindication tools, specifically in prescribed fire-treated and naturally eroded areas. However, this only scratches the surface as to the potential of these organisms as indicators of other soil disturbances such as long-term effects of land reform and frequent, intense wildfires. I would strongly recommend that soil arthropods be taken into consideration, not only as a part of the biodiversity present in the park, but as a vital component of conservation ensuring the continued health of the GGHNP soils. Future investigation should focus on more intricate identification of soil arthropods in regards to their functional role in habitats of the GGHNP, mainly for the purpose of identifying drivers of ecosystem services in the various niches in the park. For example, this could include investigation into Oribatid mites (Acari, Oribatida), a group that is known to play a large role in a number of ecosystem functions due to their vastly diverse feeding and behavioural responses (Coleman et al., 2004; Caruso et al., 2012).

In conclusion, the study results expressed in this thesis, being the first of its kind for the park, emphasises how soil arthropods as a whole are vital components of its natural ecosystems. These soil arthropod groups, coupled with other soil attributes such as mineralogy and soil type, have potential as bioindicators of soil disturbance, and could possibly prove significant even in other niche types of the park. The GGHNP holds many opportunities for biodiversity studies, with the findings in this thesis highlighting only a fraction of these understudied factors. It is vital that these factors are taken into account as part of an holistic ecosystem conservation approach for the preservation of the protected environments of the GGHNP.

6.3 References

Avenant, N.L., 1997. Mammals recorded in the Qwaqwa National Park (1994-1995). Koedoe 40, 31–40.

Baker, G.H., 1998. Recognising and responding to the influences of agriculture and other land-use practices on soil fauna in Australia. Appl. Soil Ecol. 9, 303–310.

Bardgett, R.D., Cook, R., 1998. Functional aspects of soil animal diversity in agricultural grasslands. Appl. Soil Ecol. 10, 263–276.

Bedano, J.C., Domínguez, A., Arolfo, R., Wall, L.G., 2016. Effect of good agricultural practices under no-till on litter and soil invertebrates in areas with different soil types. Soil Tillage Res. 158, 100–109.

Caruso, T., Taormina, M., Migliorini, M., 2012. Relative role of deterministic and stochastic determinants of soil animal community: a spatially explicit analysis of oribatid mites. J. Anim. Ecol. 81, 214–221.

Coleman, D.C., Crossley Jr., D.A., Hendrix, P.F., 2004. Fundamentals of Soil Ecology 2nd Edition. Elsevier Academic Press, California.

Madson, S.L., 2003. Response of Soil Microarthropods and Microclimatic Conditions to two-age Regeneration in Hardwood Forests in the Southern Appalachians. MSc Dissertation, University of Georgia, Athens.

Mhangara, P., Kakembo, V., Lim, K.J., 2012. Soil erosion risk assessment of the Keiskamma catchment, South Africa using GIS and remote sensing. Environ. Earth Sci. 65, 2087–2102.

Rademeyer, C., van Zyl, W., 2014. Golden jubilee for Golden Gate - a concise history of Golden Gate Highlands National Park, 1963 to 2013. Mediterr. J. Soc. Sci. 5, 1169–1177.

Rautenbach, I.L., 1976. A survey of the mammals occurring in the Golden Gate Highlands National Park. Koedoe 19, 133–144.

Rivas, T., 2006. Erosion Control Treatment Selection Guide. United States Department of Agriculture Forest Service.

South African National Parks (SANParks), 2013. Golden Gate Highlands National Park, Park Management Plan for the period 2013-2023. SANParks.

Supplementary material

Chapter 2 – Chapter 5

Supplementary material: Chapter 2

Figures and tables

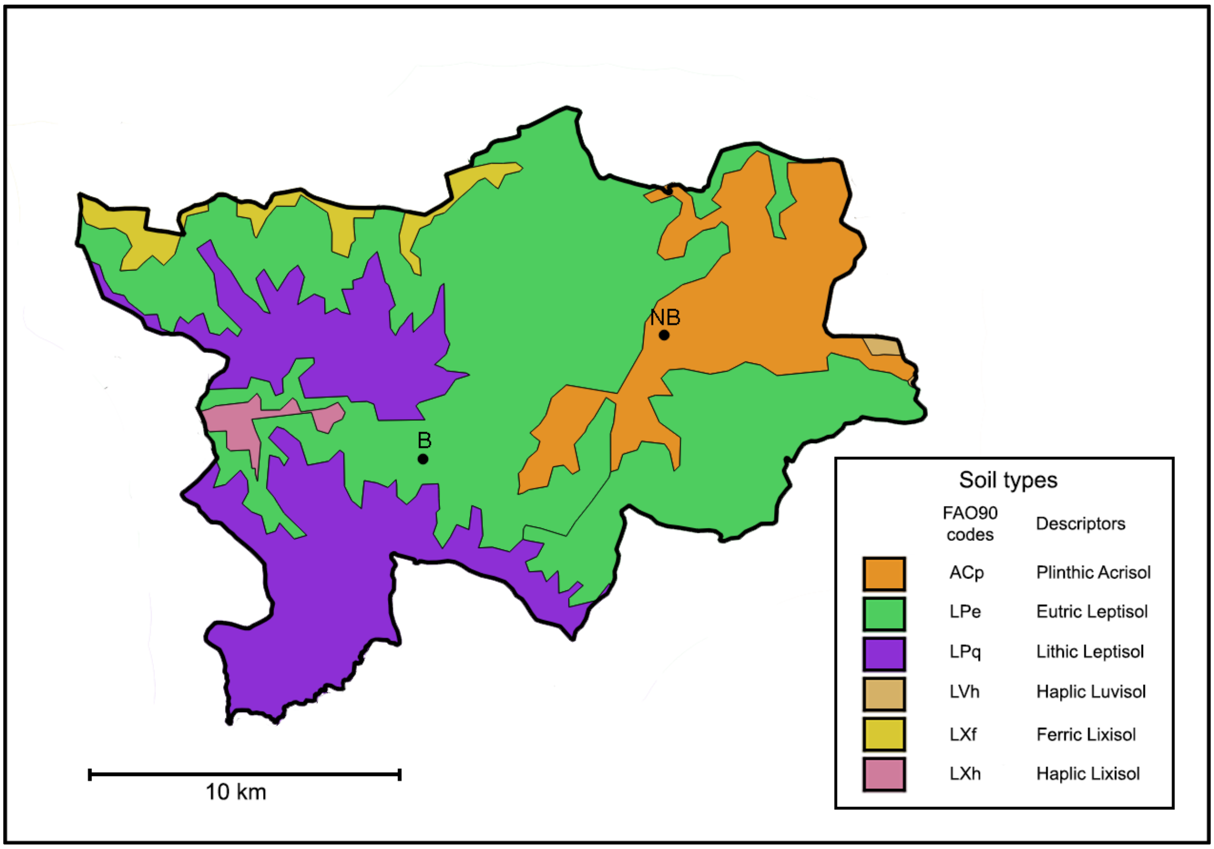

Fig. S2.1: Map of the Golden Gate Highlands National Park (GGHNP), showing described soil types, in accordance to FAO90 major groups and soil type codes and descriptors, within the park alongside site locality of the burnt (B) and non-burnt (NB) sites. Soil types, codes and soil mapping zones obtained, with full permission, from SANParks Scientific Services in South Africa. GGHNP map processed with QGIS, version 2.18.15.

Table S2.1: Soil sample analysis values for burnt and non-burnt sites investigated in the Golden Gate Highlands National Park, Free State Province, South Africa.

Groups	**Site**	**Mineralogy**										
		SiO2	**TiO2**	**Al2O3**	**Fe2O3**	**MgO**	**MnO**	**CaO**	**K2O**	**P2O5**	**Na2O**	**LOI**
Jul-17	NB	80.65	0.62	6.65	2.58	0.35	0.03	0.22	1.10	0.05	0.28	6.26
Jul-17	NB	86.97	0.43	5.56	1.95	0.06	0.01	0.11	0.68	0.03	0.2	4.58
Jul-17	NB	80.99	0.51	6.52	2.25	0.17	0.02	0.14	0.81	0.04	0.2	6.5
Nov-17	NB	77.52	0.63	8.27	3.20	0.33	0.02	0.19	1.00	0.05	0.23	8.27
Nov-17	NB	82.81	0.39	6.03	2.30	0.24	0.03	0.16	0.77	0.03	0.18	5.28
Nov-17	NB	80.69	0.54	7.75	2.93	0.14	0.02	0.10	0.89	0.04	0.21	6.91
Mar-18	NB	81.12	0.51	7.03	2.74	0.33	0.03	0.12	0.97	0.04	0.19	5.84
Mar-18	NB	83.56	0.45	5.25	2.28	0.49	0.03	0.17	0.78	0.05	0.22	5.49
Mar-18	NB	83.98	0.49	5.9	3.11	0.09	0.05	0.20	0.82	0.04	0.20	4.97
Jul-17	B	76.37	0.33	5.64	2.52	0.18	0.04	0.31	1.29	0.07	0.53	10.73
Jul-17	B	76.74	0.38	6.83	2.70	0.31	0.03	0.31	1.43	0.09	0.61	10.91
Jul-17	B	80.86	0.29	6.20	2.48	0.25	0.03	0.30	1.31	0.06	0.64	8.82
Nov-17	B	77.73	0.38	6.70	2.71	0.21	0.03	0.24	1.45	0.08	0.54	10.71
Nov-17	B	74.76	0.37	7.07	3.13	0.51	0.04	0.27	1.46	0.09	0.55	11.33
Nov-17	B	57.46	0.35	7.62	3.82	0.66	0.17	0.60	1.24	0.13	0.69	25.26
Mar-18	B	72.99	0.42	7.48	3.06	0.39	0.05	0.31	1.57	0.10	0.56	13.58
Mar-18	B	79.77	0.31	6.03	2.45	0.30	0.03	0.21	1.23	0.07	0.54	10.49
Mar-18	B	80.64	0.27	6.06	2.88	0.58	0.04	0.37	1.39	0.07	0.76	9.33

Legend/Symbol	**Descriptor**
Groups	Month and year sample groups
Site	Site symbol
NB	Non-burnt site
B	Burnt site
Mineralogy	Element results obtained by X-Ray Fluorescence (XRF) element analysis

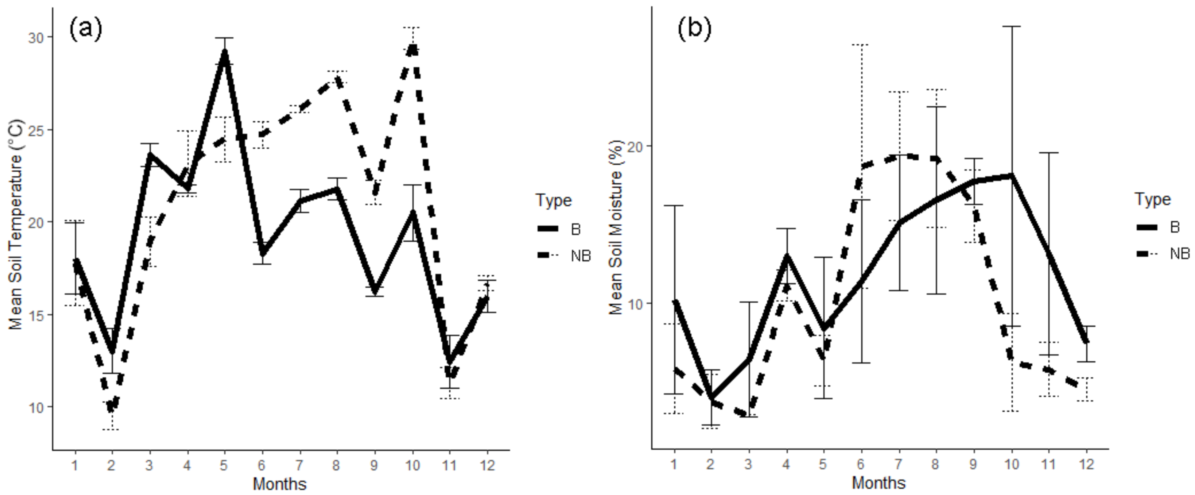

Fig. S2.2: General trends of (a) mean soil temperature and (b) mean soil moisture between the burnt and non-burnt sites at 0–10 cm soil depth. Site types: B - Burnt; NB - Non-burnt. Burning took place between months 1 and 2.

Supplementary material: Chapter 3

Figures and tables

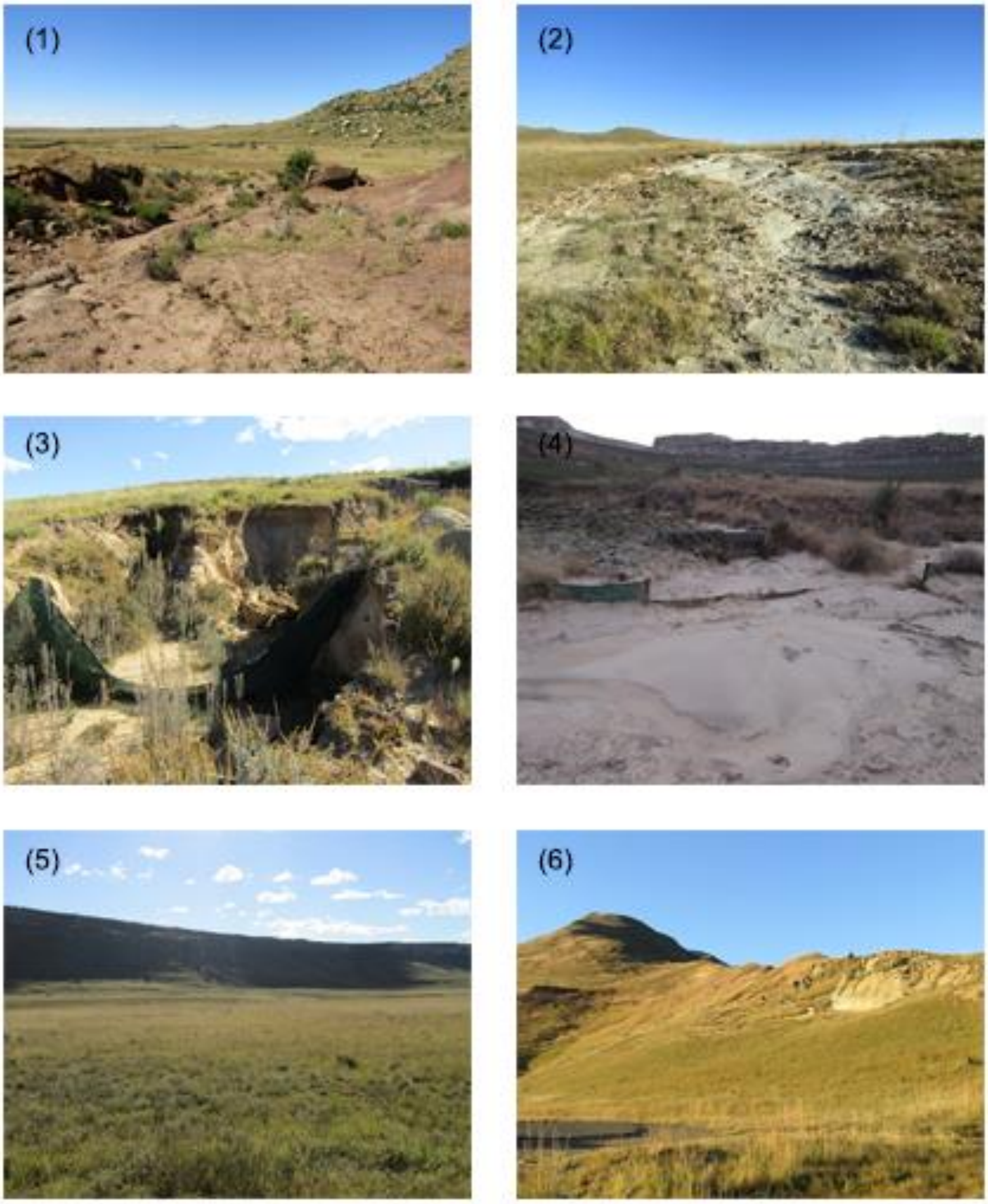

Fig. S3.1: Site photographs of site (1) to (6). (1) Site 1 - Non-rehabilitated eroded; (2) Site 2 - Non-rehabilitated eroded; (3) Site 3 - Rehabilitated eroded; (4) Site 4 - Rehabilitated eroded; (5) Site 5 - Non-eroded undisturbed; (6) Site 6 - Non-eroded undisturbed.

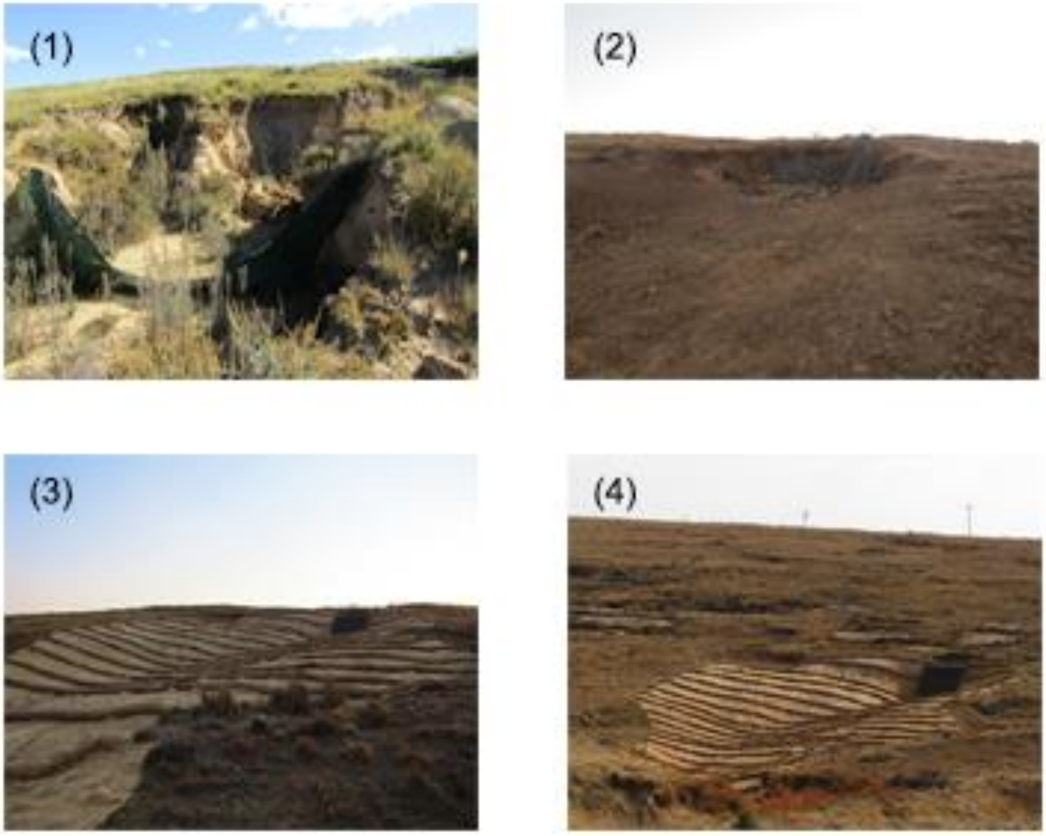

Fig. S3.2: Site 3 photographs, showing (1) Rehabilitated eroded site before demolition; (2) site after area was flattened for renewed rehabilitation; (3) and (4) showing new rehabilitation method implemented in August 2018.

Table S3.1: Soil sample analysis values for sites of the Golden Gate Highlands National Park, including soil temperature and soil moisture values.

Groups	Site	Mineralogy											STemp	Smoi
		SiO_2	TiO_2	Al_2O_3	Fe_2O_3	MgO	MnO	CaO	K_2O	Na_2O	P_2O_5	LOI		
Jul-17	Site 1 pt1	86.52	0.35	5.20	1.93	0.12	0.02	0.10	1.01	0.02	0.53	4.66	17.40	6.20
Jul-17	Site 1 pt2	79.31	0.61	9.29	4.93	0.72	0.18	0.46	1.50	0.07	0.48	3.50	15.60	2.80
Jul-17	Site 1 pt3	82.67	0.50	7.01	3.23	0.53	0.11	0.30	1.38	0.07	0.16	2.56	15.80	4.20
Nov-17	Site 1 pt1	76.99	0.67	12.09	4.75	1.03	0.13	0.35	2.16	0.42	0.08	3.17	28.20	23.20
Nov-17	Site 1 pt2	79.27	0.66	9.60	4.15	0.79	0.11	0.35	1.72	0.49	0.07	2.75	28.10	7.40
Nov-17	Site 1 pt3	94.21	0.31	3.27	1.12	0.00	0.02	0.08	0.91	0.34	0.02	1.47	28.70	9.00
Mar-18	Site 1 pt1	69.76	0.75	14.06	5.75	1.24	0.14	0.51	2.43	0.45	0.12	4.13	22.90	33.80
Mar-18	Site 1 pt2	79.90	0.65	8.37	3.62	0.78	0.10	0.39	1.54	0.46	0.07	2.78	21.10	20.10
Mar-18	Site 1 pt3	80.14	0.52	7.55	3.88	0.75	0.15	0.64	1.39	0.42	0.06	2.83	20.90	14.00
Jul-18	Site 1 pt1	73.96	0.65	12.02	4.74	1.01	0.10	0.35	0.38	2.10	0.08	3.55	19.70	10.40
Jul-18	Site 1 pt2	73.08	0.66	11.27	5.21	1.12	0.19	0.49	0.37	1.87	0.08	3.95	21.40	5.30
Jul-18	Site 1 pt3	77.28	0.62	10.14	4.55	0.78	0.12	0.32	0.34	1.78	0.07	3.09	19.80	3.50
Jul-17	Site 3 pt1	78.65	0.61	9.83	3.90	0.37	0.03	0.22	1.39	0.04	0.16	4.52	28.20	23.20
Jul-17	Site 3 pt2	69.48	0.73	14.42	5.80	1.32	0.12	0.45	2.48	0.11	0.17	4.33	28.10	7.40
Jul-17	Site 3 pt3	89.24	0.35	5.36	2.06	0.21	0.02	0.12	1.17	0.03	0.19	2.38	28.70	9.00
Nov-17	Site 3 pt1	81.64	0.56	7.31	3.50	0.57	0.14	0.45	1.36	0.60	0.06	1.99	26.20	24.80
Nov-17	Site 3 pt2	91.30	0.37	3.64	1.26	0.01	0.02	0.07	1.01	0.28	0.03	1.59	26.20	5.40
Nov-17	Site 3 pt3	83.67	0.48	7.88	3.19	0.73	0.03	1.04	1.21	0.11	0.04	3.07	25.80	18.80
Mar-18	Site 3 pt1	96.54	0.18	2.13	0.99	0.27	0.01	0.05	0.51	0.16	0.02	1.02	20.90	35.40
Mar-18	Site 3 pt2	88.31	0.39	5.26	1.72	0.11	0.02	0.10	1.01	0.23	0.03	3.03	20.20	30.40
Mar-18	Site 3 pt3	91.23	0.24	2.84	1.06	0.00	0.01	0.06	0.73	0.22	0.02	1.39	20.50	36.00
Jul-18	Site 3 pt1	90.07	0.29	3.79	1.31	0.28	0.02	0.09	0.23	0.82	0.03	2.39	21.90	8.30
Jul-18	Site 3 pt2	92.74	0.25	2.72	1.24	0.65	0.01	0.14	0.22	0.59	0.02	1.62	20.70	3.80
Jul-18	Site 3 pt3	89.17	0.27	3.52	1.27	0.04	0.02	0.08	0.26	0.82	0.03	1.98	22.60	4.40

Table S3.1 (cont.): Soil sample analysis values for sites of the Golden Gate Highlands National Park, including soil temperature and soil moisture values.

Groups	Site	Mineralogy											STemp	Smoi
		SiO_2	TiO_2	Al_2O_3	Fe_2O_3	MgO	MnO	CaO	K_2O	Na_2O	P_2O_5	LOI		
Jul-17	Site 5 pt1	80.65	0.62	6.65	2.58	0.35	0.03	0.22	1.10	0.05	0.28	6.26	13.50	3.40
Jul-17	Site 5 pt2	86.97	0.43	5.56	1.95	0.06	0.01	0.11	0.68	0.03	0.20	4.58	18.00	6.40
Jul-17	Site 5 pt3	80.99	0.51	6.52	2.25	0.17	0.02	0.14	0.81	0.04	0.20	6.50	20.20	5.50
Nov-17	Site 5 pt1	81.23	0.52	7.71	3.20	0.24	0.04	0.24	0.96	0.23	0.05	5.59	30.40	17.30
Nov-17	Site 5 pt2	82.92	0.48	6.09	2.21	0.10	0.02	0.14	0.90	0.21	0.05	5.01	28.80	6.50
Nov-17	Site 5 pt3	81.67	0.52	7.03	2.63	0.11	0.03	0.17	0.90	0.19	0.05	6.83	29.00	6.60
Mar-18	Site 5 pt1	84.93	0.53	6.29	2.34	0.26	0.02	0.18	0.94	0.21	0.04	5.38	22.50	15.20
Mar-18	Site 5 pt2	81.83	0.54	6.29	2.30	0.21	0.02	0.16	0.91	0.18	0.06	5.94	22.00	14.70
Mar-18	Site 5 pt3	79.35	0.50	8.47	2.89	0.18	0.04	0.12	1.01	0.15	0.04	5.61	21.40	17.70
Jul-18	Site 5 pt1	82.86	0.51	6.83	2.60	0.43	0.03	0.38	0.30	0.97	0.05	4.87	20.10	5.50
Jul-18	Site 5 pt2	83.44	0.50	5.86	2.22	0.18	0.03	0.15	0.20	0.86	0.06	6.43	21.20	6.30
Jul-18	Site 5 pt3	77.56	0.56	8.01	2.82	0.33	0.04	0.23	0.20	1.07	0.06	7.17	20.00	5.30

Legend	Descriptor
Group	Month and year sample groups
Site	Site and point number
Mineralogy	Element results obtained by X-Ray Fluorescence (XRF) element analysis
STemp	Soil Temperature (°C)
Smoi	Soil Moisture (%)

Table S3.2: Results of Linear Model analysis testing correlation between soil attributes and log-transformed species richness of sites in the eastern sectors of the GGHNP, with AIC values for best fit. n_{obs} = 36 for all models; *K*: site only = 4, all combination models = 7; p-values of site versus soil attributes: * = 0.05 ; ** = 0.01 ; *** = 0.001.

Model	**Multiple R^2**	**Adjusted R^2**	**p**	**p(site)**	**p(x)**	**p(slope diff)**			**AIC_c**	**ΔAIC_c**
						Site	**Estimate ± SE**	**p**		
Site	0.453	0.420	**p < 0.001**	**p < 0.001**	-		-	-	-18.577	12.753
Site/P_2O_5	-	0.631	**p < 0.001**	**p < 0.001**	**0.001**	1	-0.867 ± 0.248	**0.001*** **	-31.330	0.000
						3	-1.963 ± 0.638	**0.004** **		
						5	-0.177 ± 0.510	0.731		
Site/Soil temp	-	0.551	**p < 0.001**	**p < 0.001**	**0.013**	1	0.020 ± 0.010	0.058	-24.308	7.022
						3	-0.039 ± 0.014	**0.009** **		
						5	-0.009 ± 0.009	0.369		
Site/Organic matter	-	0.521	**p < 0.001**	**p < 0.001**	**0.033**	1	-0.088 ± 0.057	0.135	-21.944	9.386
						3	-0.115 ± 0.043	**0.012** **		
						5	-0.033 ± 0.059	0.577		
Site/CaO	-	0.509	**p < 0.001**	**p < 0.001**	**0.046**	1	0.735 ± 0.302	**0.021***	-21.076	10.254
						3	-0.276 ± 0.167	0.109		
						5	-0.377 ± 0.652	0.567		
Site/TiO_2	-	0.505	**p < 0.001**	**p < 0.001**	0.052	1	0.656 ± 0.361	0.079	-20.792	10.538
						3	-0.664 ± 0.287	**0.028***		
						5	-0.127 ± 1.064	0.906		
Site/Fe_2O_3	-	0.490	**p < 0.001**	**p < 0.001**	0.078	1	0.048 ± 0.036	0.198	-19.697	11.633
						3	-0.071 ± 0.033	**0.038***		
						5	-0.143 ± 0.139	0.314		
Site/SiO_2	-	0.489	**p < 0.001**	**p < 0.001**	0.079	1	-0.009 ± 0.008	0.233	-19.655	11.675
						3	0.013 ± 0.007	0.052		
						5	0.027 ± 0.020	0.177		
Site/Al_2O_3	-	0.486	**p < 0.001**	**p < 0.001**	0.086	1	0.018 ± 0.016	0.267	-19.447	11.883
						3	-0.031 ± 0.014	**0.032***		
						5	-0.053 ± 0.056	0.351		
Site/K_2O	-	0.465	**p < 0.001**	**p < 0.001**	0.148	1	0.087 ± 0.074	0.246	-17.970	13.360
						3	-0.160 ± 0.078	**0.049***		
						5	0.061 ± 0.155	0.695		
Site/Na_2O	-	0.459	**p < 0.001**	**p < 0.001**	0.171	1	0.025 ± 0.067	0.719	-17.562	13.768
						3	0.386 ± 0.173	**0.033***		
						5	-0.064 ± 0.133	0.634		
Site/MgO	-	0.441	**p < 0.001**	**p < 0.001**	0.261	1	0.250 ± 0.138	0.080	-16.379	14.950
						3	-0.125 ± 0.134	0.357		
						5	-0.116 ± 0.468	0.806		
Site/Soil moisture	-	0.436	**p < 0.001**	**p < 0.001**	0.290	1	0.010 ± 0.005	0.083	-16.075	15.255
						3	-0.001 ± 0.004	0.901		
						5	0.008 ± 0.010	0.414		
Site/MnO	-	0.435	**p < 0.001**	**p < 0.001**	0.298	1	1.651 ± 0.983	0.104	-15.999	15.331
						3	-0.595 ± 1.178	0.617		
						5	-4.727 ± 5.361	0.385		

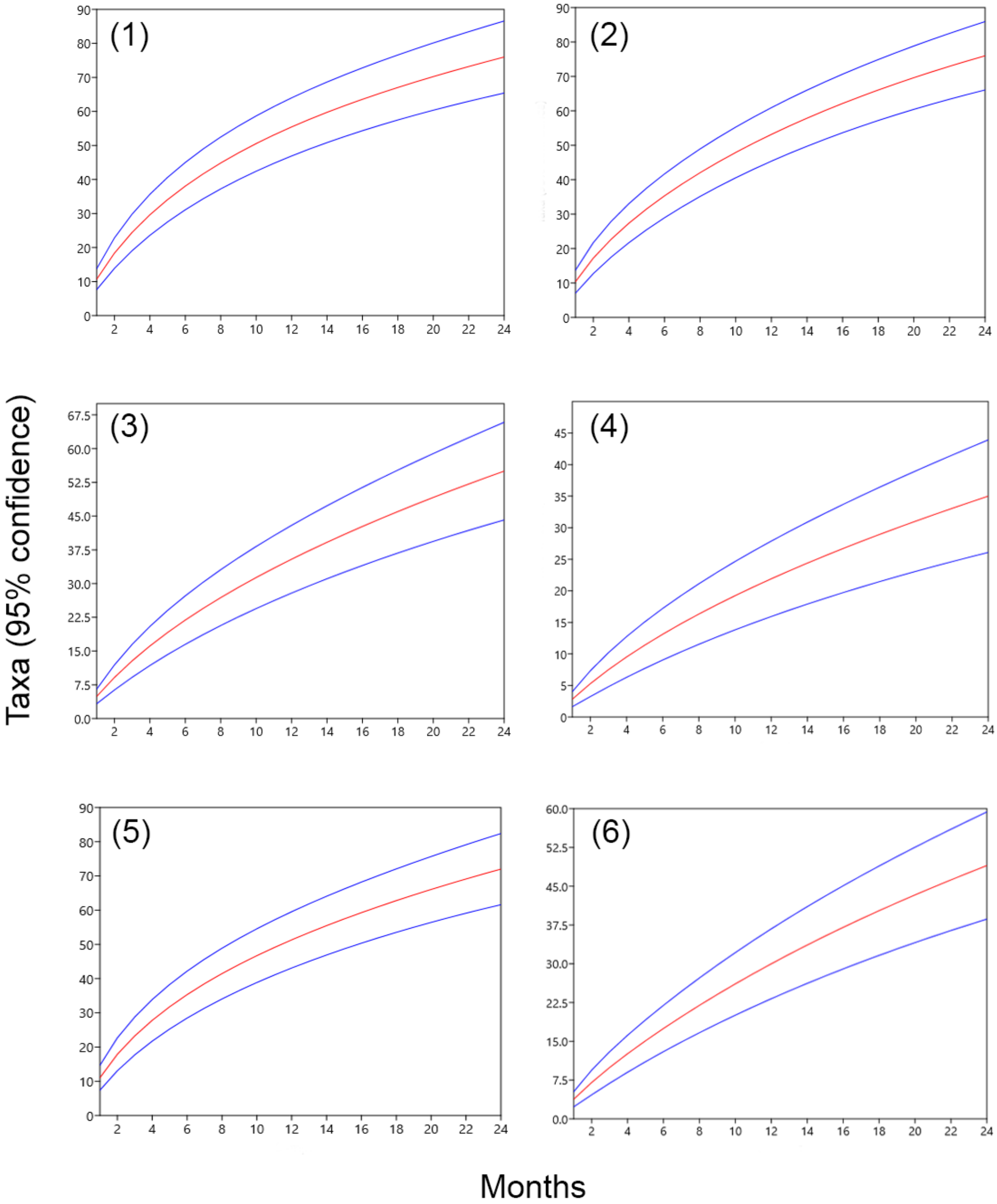

Fig. S3.3: Rarefaction curves (Mao Tau) of site (1) to (6). (1) Site 1 - Non-rehabilitated eroded; (2) Site 2 - Non-rehabilitated eroded; (3) Site 3 - Rehabilitated eroded; (4) Site 4 - Rehabilitated eroded; (5) Site 5 - Non-eroded undisturbed; (6) Site 6 - Non-eroded undisturbed.

Supplementary material: Chapter 4

Figures and tables

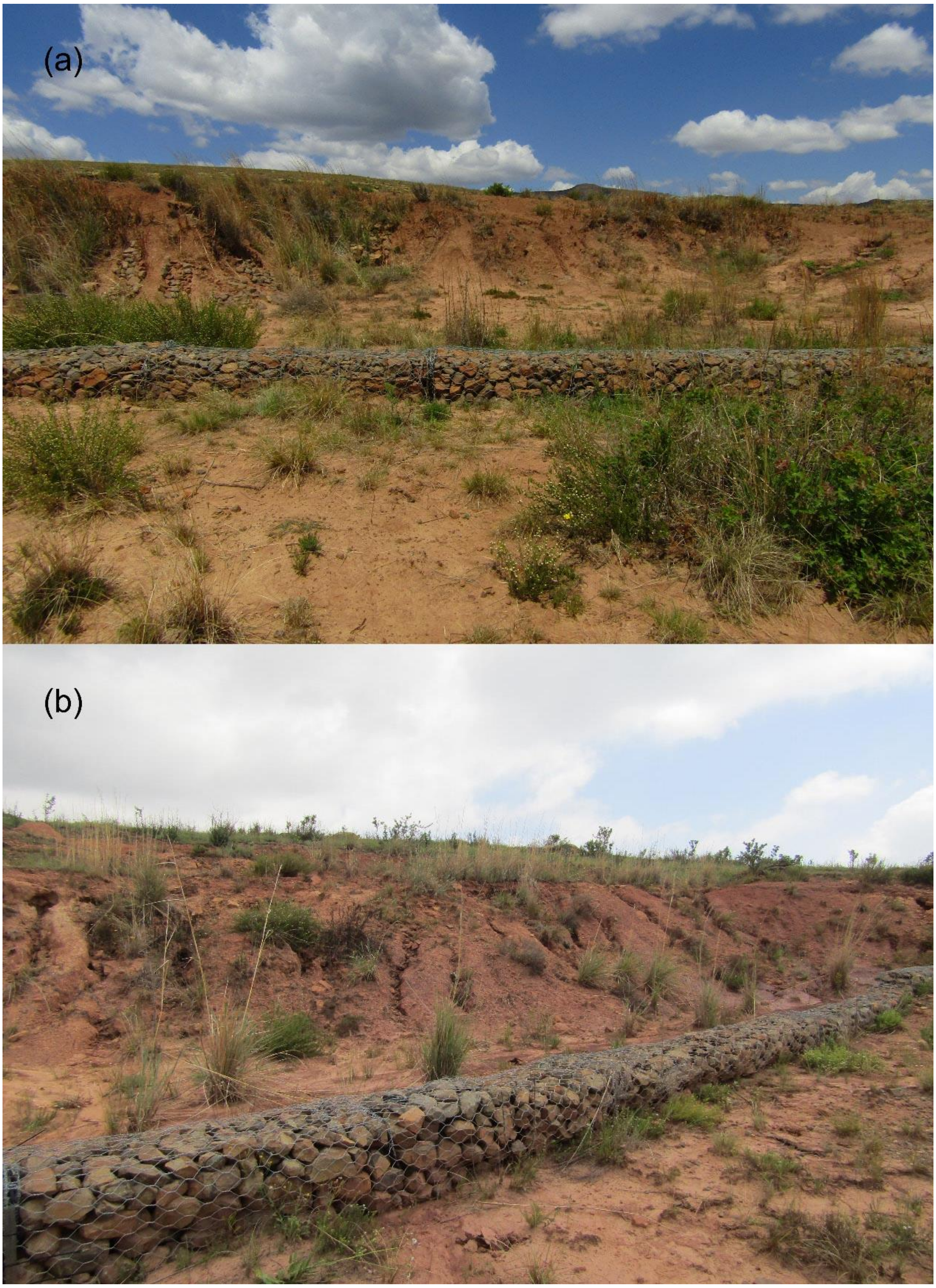

Fig. S4.1: Photographs of sites showing constructed and implemented rock-wall gabions of (a) Site A1 and (b) Site B1, taken in November 2017.

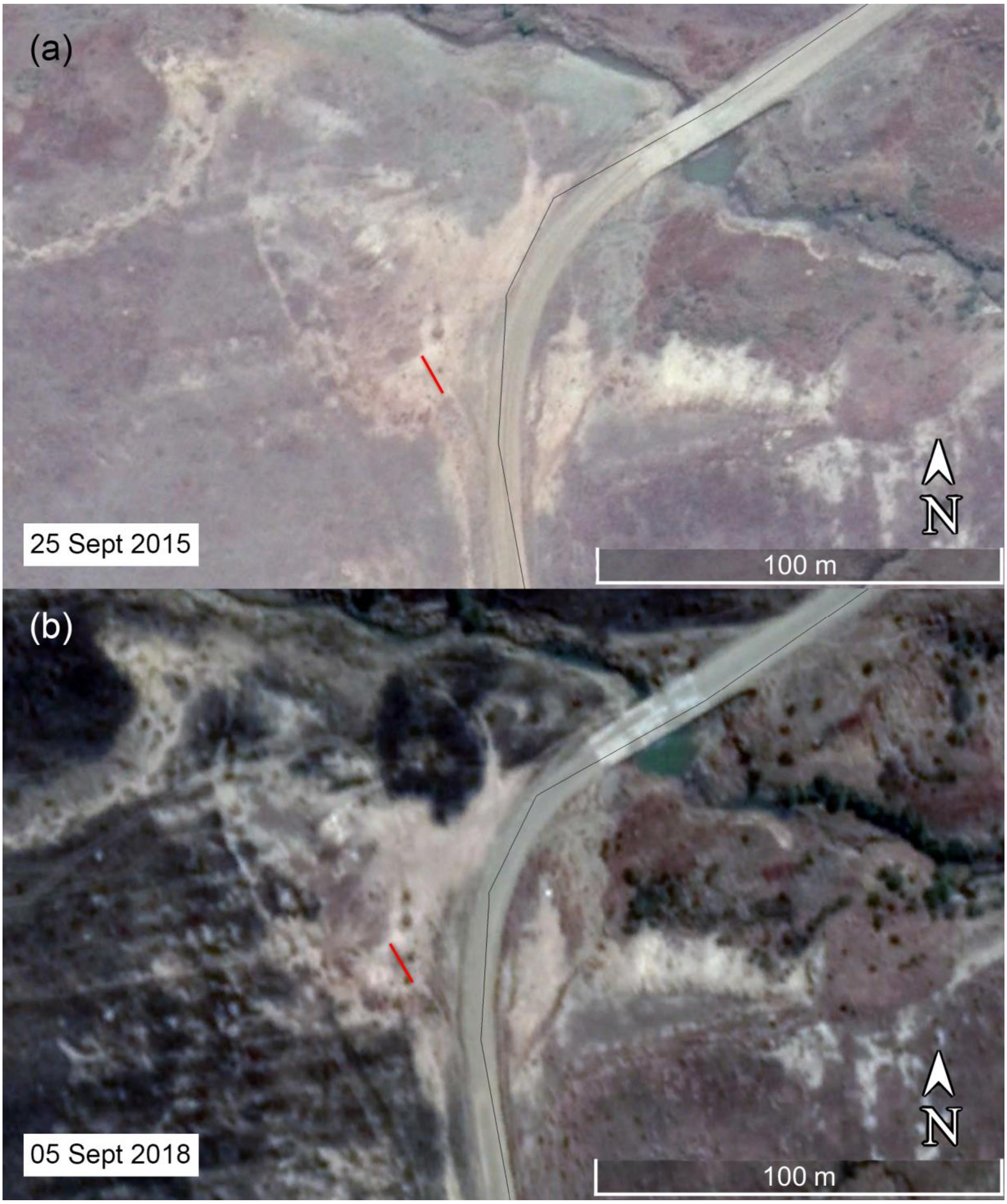

Fig. S4.2: Sattelite images of Site A on (a) 25 September 2015 and (b) 5 September 2018 showing further erosion over time. Images obtained from Google Earth 7.3.2.5776. Viewed 24 March 2020. Red line indicates position of the rock-wall gabion.

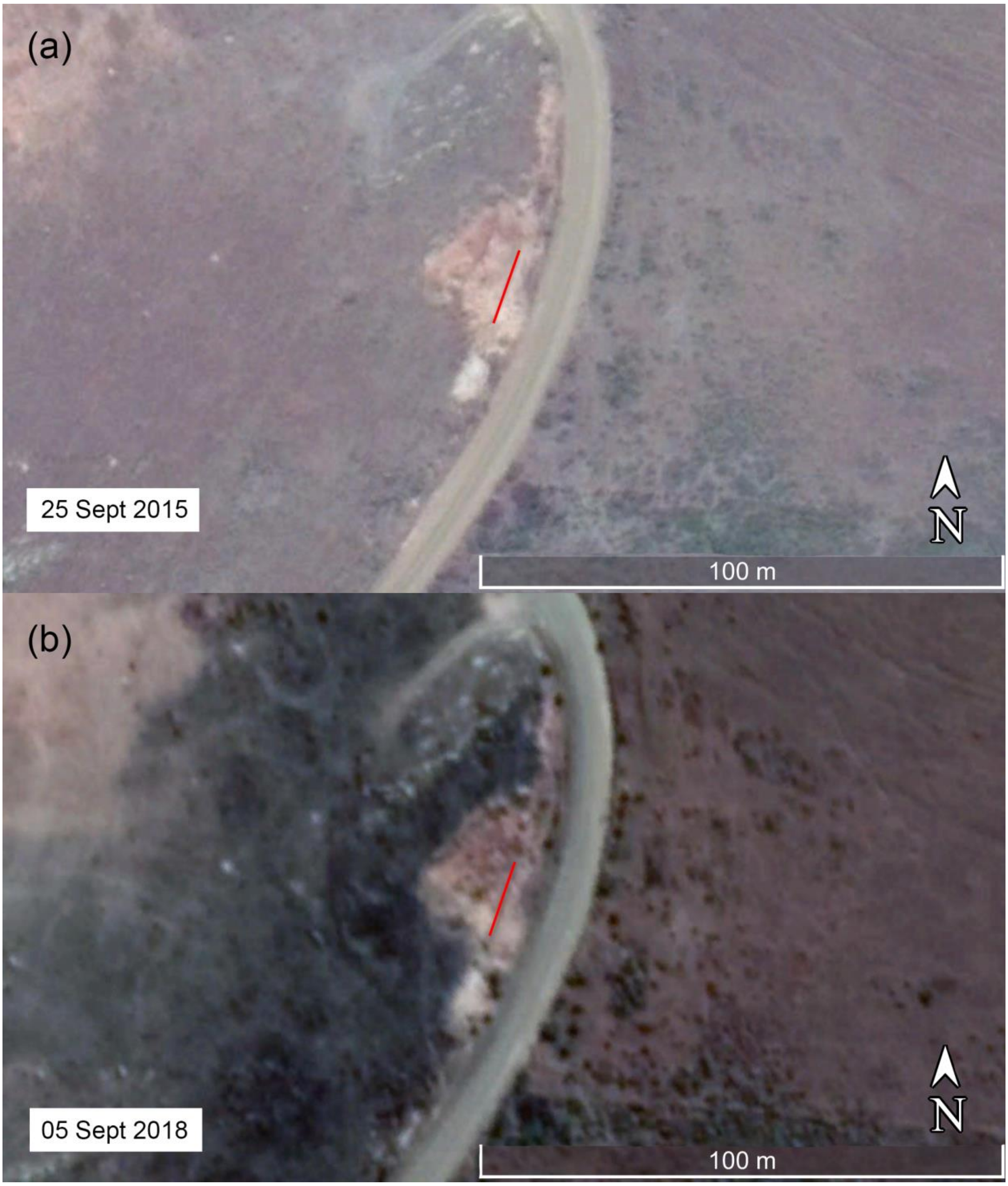

Fig. S4.3: Sattelite images of Site B on (a) 25 September 2015 and (b) 5 September 2018 showing further erosion over time. Images obtained from Google Earth 7.3.2.5776. Viewed 24 March 2020. Red line indicates position of the rock-wall gabion.

Table S4.1: Post-hoc PERMANOVA results for comparisons of communities across sub-sites, with adjusted p-values using 'fdr' correction; * = $p < 0.05$

	Site pairs	F-Model	R^2 value	p-value	p-adjusted
1	A1 behind vs A1 in front	1.204	0.039	0.264	0.512
2	A1 behind vs A2 source	1.100	0.035	0.308	0.512
3	A1 behind vs A2 base	1.571	0.050	0.092	0.286
4	A1 behind vs B1 behind	1.222	0.039	0.268	0.512
5	A1 behind vs B1 in front	1.112	0.036	0.333	0.518
6	A1 behind vs B2 source	1.293	0.041	0.219	0.512
7	A1 behind vs B2 base	0.667	0.022	0.714	0.909
8	A1 in front vs A2 source	1.823	0.057	0.031	0.145
9	A1 in front vs A2 base	2.601	0.080	0.002	**0.028***
10	A1 in front vs B1 behind	1.131	0.036	0.311	0.512
11	A1 in front vs B1 in front	1.210	0.039	0.276	0.512
12	A1 in front vs B2 source	0.516	0.017	0.902	0.955
13	A1 in front vs B2 base	1.159	0.037	0.309	0.512
14	A2 source vs A2 base	0.657	0.021	0.831	0.931
15	A2 source vs B1 behind	0.694	0.023	0.795	0.931
16	A2 source vs B1 in front	1.417	0.045	0.147	0.412
17	A2 source vs B2 source	1.841	0.058	0.029	0.145
18	A2 source vs B2 base	1.585	0.050	0.080	0.280
19	A2 base vs B1 behind	1.611	0.051	0.064	0.256
20	A2 base vs B1 in front	2.098	0.065	0.011	0.077
21	A2 base vs B2 source	2.547	0.078	0.001	**0.028***
22	A2 base vs B2 base	2.375	0.073	0.007	0.065
23	B1 before vs B1 in front	0.448	0.015	0.950	0.955
24	B1 before vs B2 source	1.002	0.032	0.460	0.644
25	B1 before vs B2 base	0.714	0.023	0.700	0.909
26	B1 in front vs B2 source	1.000	0.032	0.425	0.626
27	B1 in front vs B2 base	0.392	0.013	0.955	0.955
28	B2 source vs B2 base	0.592	0.019	0.807	0.931

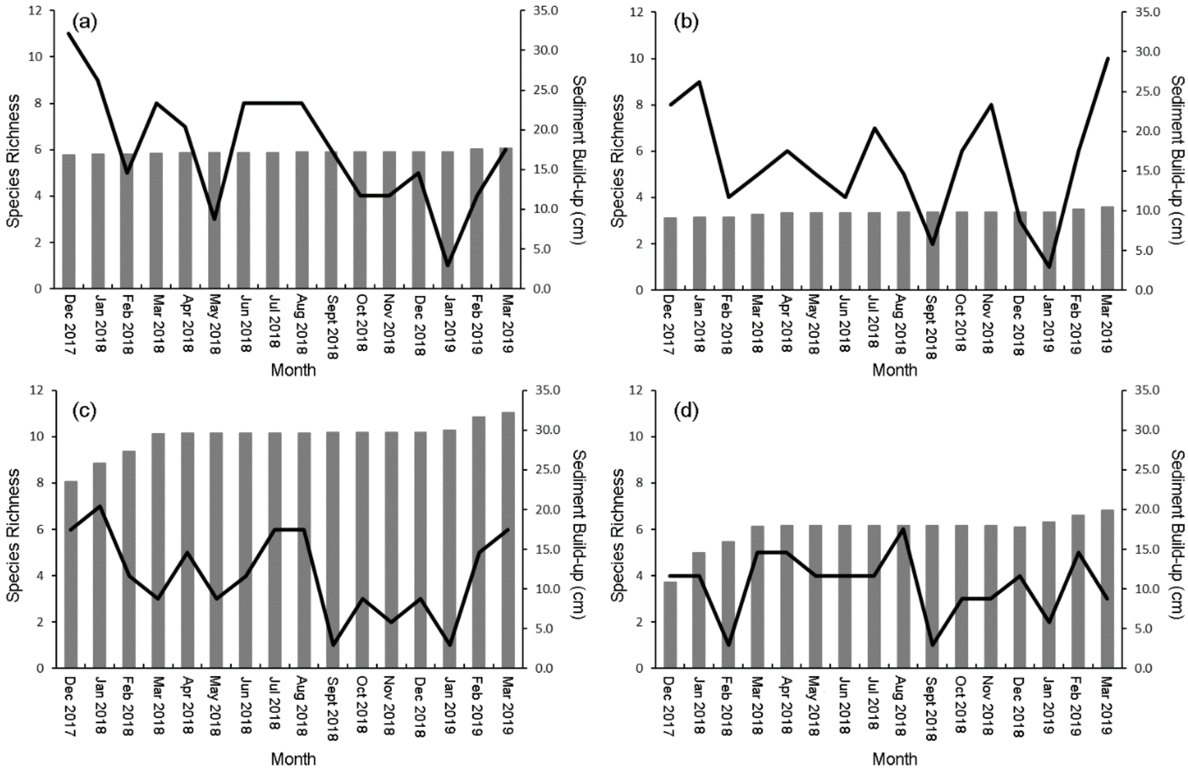

Fig. S4.4 Graphs indicating arthropod species richness compared to sediment depth over time. (a) Site A1 behind, (b) Site A1 in front, (c) Site B1 behind and (d) Site B1 in front.

Supplementary material: Chapter 5

Figures

Fig. S5.1a: Site with deteriorated rehabilitation implements at the start of the study, taken in October 2017.

Fig. S5.1b: Flattening of the site, with displacement of soils around the initial eroded site, taken during July 2018.

Fig. S5.1c: Newly rehabilitated site with new implements including erosion control blankets and fibre rolls placed over the displaced soils, taken in August 2018.

Appendices

Appendix 1: South African National Parks Research Agreement title page and signature authorisation page.

South African
NATIONAL PARKS

RESEARCH AGREEMENT

BETWEEN

SOUTH AFRICAN NATIONAL PARKS

herein represented by **Mr D. Pienaar**

in his/her capacity as **Senior GM: Scientific Services**

(hereinafter referred to as "SANParks")

AND

Miss SS van der Merwe

9103270212080

Id no. ______________________

(hereinafter referred to as "the Researcher")

WHEREAS the Researcher submitted a research application to SANParks to conduct a research on ""**Soil biota as bioindicators of levels of erosion and fire disturbances in Afromontane grassland areas within the Golden Gate Highlands** ("Research") and to obtain a sample of a biological resource (" Material") in the **"Golden Gate highlands National Park"** ("the Park");

SS

SIGNED at Skukuza on the 29th day of November 2016

WITNESSES

1. ____________________ ____________________
SANPARKS

2. ____________________

SIGNED at UFS, BLOEM on the 09 day of NOVEMBER 2016

WITNESSES:

1. ____________________ ____________________
RESEARCHER

2. ____________________

DIRECTOR RESEARCH DEVELOPMENT
21 FEB 2017
UNIVERSITY OF THE FREE STATE

Director: Contract

Appendix 2: University of the Free State Animal Research Ethics - Conditional approval and Full approval

UNIVERSITY OF THE FREE STATE
UNIVERSITEIT VAN DIE VRYSTAAT
YUNIVESITHI YA FREISTATA
UFS·UV
NATURAL AND AGRICULTURAL SCIENCES
NATUUR- EN LANDBOUWETENSKAPPE

Faculty of Natural and Agricultural Sciences

17-Feb-2017

Dear **Miss Sylvia Van Der Merwe**

Ethics Clearance: **Soil biota as bioindicators of levels of erosion and fire disturbances in Afromontane grassland areas within the Golden Gate Highlands**

Principal Investigator: **Miss Sylvia Van Der Merwe**

CONDITIONALLY APPROVED

This letter confirms that a research proposal with tracking number: **UFS-HSD2017/0074** and title: **Soil biota as bioindicators of levels of erosion and fire disturbances in Afromontane grassland areas within the Golden Gate Highlands** was given ethics clearance by the Ethical Committee pending clarification of the following:

Response from Chair:

Research contract needs to be signed by appropriate university authority. This should not block the application process, but before final approval can be given, a correctly signed research contract needs to be uploaded.

Reviewer 1

Approved

Reviewer 2

Approved

Please ensure that the ethical standards committee is notified should any substantive change(s) be made, for whatever reason, during the research process. This includes changes in investigators. Please also ensure that a brief report is submitted to the ethics committee on completion of the research. The purpose of this report is to indicate whether or not the research was conducted successfully, if any aspects could not be completed, or if any problems arose that the ethical standards committee should be aware of.

Note:

1. This clearance is valid from the date on this letter to the time of completion of data collection.
2. Progress reports should be submitted annually unless otherwise specified.

Yours Sincerely

Prof. RR (Robert) Bragg
Chairperson: Ethics Committee
Faculty of Natural and Agricultural Sciences

Natural and Agricultural Sciences Research Ethics Committee
Office of the Dean: Natural and Agricultural Sciences
T: +27 (0)51 401 2322 | F: +27 (0)51 401 3728 | E: Heidemannj@ufs.ac.za
Biology Building, Ground Floor, Room 9 | P.O. Box/Posbus 339 (Internal Post Box G44) | Bloemfontein 9300 | South Africa
www.ufs.ac.za

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

UFS·UV
NATURAL AND AGRICULTURAL SCIENCES
NATUUR- EN LANDBOUWETENSKAPPE

Faculty of Natural and Agricultural Sciences

21-Nov-2018

Dear **Miss Sylvia Van Der Merwe**

Ethics Clearance: **Soil biota as bioindicators of levels of erosion and fire disturbances in Afromontane grassland areas within the Golden Gate Highlands**

Principal Investigator: **Miss Sylvia Van Der Merwe**

Department: **Zoology and Entomology Department (Bloemfontein Campus)**

APPLICATION APPROVED

This letter confirms that a research proposal with tracking number: **UFS-HSD2017/0074** and title: **'Soil biota as bioindicators of levels of erosion and fire disturbances in Afromontane grassland areas within the Golden Gate Highlands'** was given ethical clearance by the Ethics Committee.

Your ethical clearance number, to be used in all correspondence is: **UFS-HSD2017/0074**

Please ensure that the Ethics Committee is notified should any substantive change(s) be made, for whatever reason, during the research process. This includes changes in investigators. Please also ensure that a brief report is submitted to the Ethics Committee on completion of the research.
The purpose of this report is to indicate whether or not the research was conducted successfully, if any aspects could not be completed, or if any problems arose that the Ethics Committee should be aware of.

Note:

1. This clearance is valid from the date on this letter to the time of completion of data collection.
2. Progress reports should be submitted annually unless otherwise specified.

Yours Sincerely

Dr. Karen Ehlers
Chairperson: Ethics Committee
Faculty of Natural and Agricultural Sciences

Natural and Agricultural Sciences Research Ethics Committee
Office of the Dean: Natural and Agricultural Sciences
T: +27 (0)51 401 2322 | +27 (0)82733 2696 | E: smitham@ufs.ac.za
Biology Building, Ground Floor, Room 9 | P.O. Box/Posbus 339 (Internal Post Box G44) | Bloemfontein 9300 | South Africa
www.ufs.ac.za

Appendix 3: Species lists of soil arthropods identified during all sections of the study, with associated literature and/or specialist(s) name(s).

Order	**Family/Species**	**Resources/Specialist(s)**
Collembola	*Priosotoma davidi*	Dunger and Shlitt, 2011; Janion-Scheepers, Potapov
Collembola	*Seira sp.1*	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	*Seira sp.2*	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Sminthurididae msp 1	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Isotomidae msp 1	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Sminthurididae msp 2	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	*Mesaphorura sp.1*	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Isotomidae msp 2	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Entomobryidae msp 1	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Isotomidae msp 3	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Entomobryidae msp 2	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	*Seira sp.3*	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Hypogasturidae msp 1	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Hypogasturidae msp 2	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Entomobryidae msp 3	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	*Seira sp.4*	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Entomobryidae msp 4	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	*Priosotoma davidi*	Janion-Scheepers; Potapov
Collembola	Odontellidae msp. 1	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Sminthurididae msp 3	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	*Seira sp.5*	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	*Tullbergia* sp. 1	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	*Capbrya* sp. 1	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Odontellidae msp. 2	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Sminthurididae msp 4	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	*Sminthurines* sp. 1	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	*Tullbergia sp. 2*	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Tullbergidae msp 1	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	*Entomobrya* sp. 1	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	*Seira sp. 6*	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Hypogastruridae msp 1	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Onychiuridae msp 1	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Hypograstruridae msp 2	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	*Capbrya sp. 2*	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Hypogastruridae msp 3	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	*Isotoma* sp. 1	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Sminthurididae msp 5	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	*Seira sp. 7*	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Sminthurididae msp 6	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	*Megalothorax sp.*	Dunger and Shlitt, 2011; Janion-Scheepers
Collembola	Entomobryidae msp 4	Dunger and Shlitt, 2011; Janion-Scheepers

Appendix 3 (cont.): Species lists of soil arthropods identified during all sections of the study, with associated literature and/or specialist(s) name(s).

Order	**Family/Species**	**Resources/Specialist(s)**
Acari	Trombidiidae msp 1	Badenhorst, 2016
Acari	Adamystidae msp 1	Badenhorst, 2016
Acari	*Speleorchestes meyeri*	Theron and Ryke, 1969 ;Badenhorst, 2016
Acari	Phytoseidae msp 1	Badenhorst, 2016
Acari	*Cunaxa sp.*	Badenhorst, 2016
Acari	Microtrombidiidae nymph	Badenhorst, 2016
Acari	Dermanyssidae msp 2	Badenhorst, 2016
Acari	*Bdellodes sp.*	Badenhorst, 2016
Acari	Paratydeidae msp 1	Badenhorst, 2016
Acari	Cheyletidae msp 1	Badenhorst, 2016
Acari	Eupodidae msp 1	Badenhorst, 2016
Acari	*Eupodes sp.*	Badenhorst, 2016
Acari	Rhodacaridae msp 1	Badenhorst, 2016
Acari	Phytoseidae msp 2	Badenhorst, 2016
Acari	Anystidae msp 2	Badenhorst, 2016
Acari	Phytoseidae nymph	Badenhorst, 2016
Acari	Erythraeidae msp 1	Badenhorst, 2016
Acari	Anystidae msp 3	Badenhorst, 2016
Acari	Trombidiidae msp 2	Badenhorst, 2016
Acari	Bdellidae msp 2	Badenhorst, 2016
Acari	Paratydeidae msp 2	Badenhorst, 2016
Acari	Adamystidae msp 2	Badenhorst, 2016
Acari	Microtrombidiidae msp 1	Badenhorst, 2016
Acari	Dermanyssidae msp 3	Badenhorst, 2016
Acari	Erythraeidae msp 2	Badenhorst, 2016
Acari	Erythraeidae msp 3	Badenhorst, 2016
Acari	Erythraeidae msp 4	Badenhorst, 2016
Acari	Caeculidae msp 3	Badenhorst, 2016
Acari	Erythraeidae msp 3	Badenhorst, 2016
Acari	Tetranychidae msp 2	Badenhorst, 2016
Acari	*Galumna sp.*	Badenhorst, 2016
Acari	Uropodidae msp 1	Badenhorst, 2016
Acari	Oribatulidae msp 1	Badenhorst, 2016
Acari	Dermanyssidae msp 1	Badenhorst, 2016
Acari	*Nanhermannia sp.*	Badenhorst, 2016
Acari	Chamobatidae msp 1	Badenhorst, 2016
Acari	Tydeidae msp 2	Badenhorst, 2016
Acari	Epilohmanniidae msp 1	Badenhorst, 2016
Acari	Caeculidae msp 1	Badenhorst, 2016
Acari	Uropodidae msp 2	Badenhorst, 2016
Acari	Uropodidae msp 3	Badenhorst, 2016

Appendix 3 (cont.): Species lists of soil arthropods identified during all sections of the study, with associated literature and/or specialist(s) name(s).

Order	Family/Species	Resources/Specialist(s)
Acari	Scutoverticidae msp 1	Badenhorst, 2016
Acari	Caeculidae msp 2	Badenhorst, 2016
Acari	Euphthiracaridae msp 1	Badenhorst, 2016
Acari	Tygaeidae nymph	Badenhorst, 2016
Acari	Uropodidae msp 4	Badenhorst, 2016
Acari	Oribatulidae msp 2	Badenhorst, 2016
Acari	Adamystidae msp 2	Badenhorst, 2016
Acari	Nanorchestidae msp 2	Badenhorst, 2016
Acari	Adamystidae msp 3	Badenhorst, 2016
Acari	*Pronematus sp.*	Badenhorst, 2016
Acari	Protoribatidae msp 1	Badenhorst, 2016
Acari	Tetranychidae msp 1	Badenhorst, 2016
Acari	Anystidae msp 1	Badenhorst, 2016
Acari	Dermanyssidae msp 4	Badenhorst, 2016
Acari	Bdellidae msp 3	Badenhorst, 2016
Acari	Bdellidae nymph	Badenhorst, 2016
Hymenoptera	Formicidae msp 1	Triplehorn and Johnson, 2005
Hymenoptera	*Solenopsis geminata*	Triplehorn and Johnson, 2005; Slingsby, 2017
Hymenoptera	Formicidae msp 3	Triplehorn and Johnson, 2005
Hymenoptera	Formicidae msp 4	Triplehorn and Johnson, 2005
Hymenoptera	Formicidae msp 5	Triplehorn and Johnson, 2005
Hymenoptera	Ceraphronidae msp 1	Triplehorn and Johnson, 2005
Hymenoptera	Eurytomidae msp 1	Triplehorn and Johnson, 2005
Hymenoptera	Braconidae msp 1	Triplehorn and Johnson, 2005
Hymenoptera	Platygastridae msp 1	Triplehorn and Johnson, 2005
Hymenoptera	Chalcididae msp 1	Triplehorn and Johnson, 2005
Hymenoptera	Eulopidae msp 1	Triplehorn and Johnson, 2005
Hymenoptera	Formicidae msp 6	Triplehorn and Johnson, 2005
Hymenoptera	*Solenopsis geminata*	Triplehorn and Johnson, 2005; Slingsby, 2017
Hymenoptera	Formicidae msp 8	Triplehorn and Johnson, 2005
Hymenoptera	Formicidae msp 9	Triplehorn and Johnson, 2005
Hymenoptera	Formicidae msp 10	Triplehorn and Johnson, 2005
Hymenoptera	Formicidae msp 11	Triplehorn and Johnson, 2005
Hymenoptera	Formicidae msp 12	Triplehorn and Johnson, 2005
Hymenoptera	Formicidae msp 13	Triplehorn and Johnson, 2005
Hymenoptera	Encyrtidae msp 1	Triplehorn and Johnson, 2005
Hymenoptera	Formicidae msp 14	Triplehorn and Johnson, 2005
Hymenoptera	Formicidae msp 15	Triplehorn and Johnson, 2005
Thysanoptera	Phlaeothripidae msp 1	Triplehorn and Johnson, 2005
Thysanoptera	Phlaeothripidae msp 2	Triplehorn and Johnson, 2005
Thysanoptera	Phlaeothripidae msp 3	Triplehorn and Johnson, 2005

Appendix 3 (cont.): Species lists of soil arthropods identified during all sections of the study, with associated literature and/or specialist(s) name(s).

Order	**Family/Species**	**Resources/Specialist(s)**
Thysanoptera	Phlaeothripidae msp 4	Triplehorn and Johnson, 2005
Thysanoptera	Phlaeothripidae msp 5	Triplehorn and Johnson, 2005
Coleoptera	Scarabaeidae larva	Triplehorn and Johnson, 2005
Coleoptera	Scarabaeidae msp 1	Triplehorn and Johnson, 2005
Coleoptera	Scarabaeidae msp 2	Triplehorn and Johnson, 2005
Coleoptera	Tenebrionidae msp 1	Triplehorn and Johnson, 2005
Coleoptera	Tenebrionidae msp 2	Triplehorn and Johnson, 2005
Coleoptera	Tenebrionidae msp 3	Triplehorn and Johnson, 2005
Coleoptera	Staphylinidae msp 1	Triplehorn and Johnson, 2005
Coleoptera	Staphylinidae msp 2	Triplehorn and Johnson, 2005
Coleoptera	Trogidae msp 1	Triplehorn and Johnson, 2005
Coleoptera	Tenebrionidae msp4	Triplehorn and Johnson, 2005
Coleoptera	Tenebrionidae msp 5	Triplehorn and Johnson, 2005
Coleoptera	Scarabaeidae msp 1	Triplehorn and Johnson, 2005
Coleoptera	Tenebrionidae msp 6	Triplehorn and Johnson, 2005
Coleoptera	Tenebrionidae msp 7	Triplehorn and Johnson, 2005
Diptera	Sciaridae msp 1	Triplehorn and Johnson, 2005
Diptera	Phoridae msp 1	Triplehorn and Johnson, 2005
Diptera	Phoridae msp 2	Triplehorn and Johnson, 2005
Diptera	Cecidomyidae msp 1	Triplehorn and Johnson, 2005
Diptera	Ceratopogonidae msp 1	Triplehorn and Johnson, 2005
Diptera	Muscidae msp 1	Triplehorn and Johnson, 2005
Diptera	Therevidae larvae	Triplehorn and Johnson, 2005
Diptera	Lauxanidae msp 1	Triplehorn and Johnson, 2005
Diptera	Psychodidae msp 1	Triplehorn and Johnson, 2005
Diptera	Sphaeroceridae msp 1	Triplehorn and Johnson, 2005
Diptera	Ceratopogonidae msp 2	Triplehorn and Johnson, 2005
Diptera	Phoridae msp 3	Triplehorn and Johnson, 2005
Diptera	Ceratopogonidae msp 3	Triplehorn and Johnson, 2005
Blattodea	Termitidae msp 1	Triplehorn and Johnson, 2005
Hemiptera	Cicadellidae msp 1	Triplehorn and Johnson, 2005
Hemiptera	Cicadellidae msp 2	Triplehorn and Johnson, 2005
Hemiptera	Cicadellidae msp 3	Triplehorn and Johnson, 2005
Hemiptera	Reduviidae msp 1	Triplehorn and Johnson, 2005
Psocoptera	Pachytroctidae msp 1	Triplehorn and Johnson, 2005
Diplura	Japygidae msp 1	Triplehorn and Johnson, 2005
Annelida	Annelida msp 1	Triplehorn and Johnson, 2005
Arachnida	Oonopidae sp. 1	Haddad
Arachnida	Trachelidae sp. 1	Haddad
Arachnida	Gnaphosidae sp. 1	Haddad