

**Guild structure and population dynamics of small mammal
assemblages in suburban patches of KwaZulu-Natal Sandstone
Sourveld grassland within eThekweni.**

by

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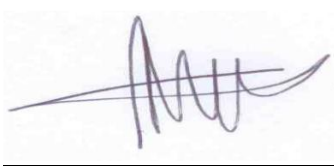
DECLARATION

I, Zamawelase Immaculate Mwelase, declare that this dissertation titled “Guild structure and population dynamics of small mammal assemblages in suburban patches of KwaZulu-Natal Sandstone Sourveld grassland within eThekweni” is my own original work. It has not been submitted, either in whole or in part, for any other degree at any other institution.

All sources of information, ideas, and data used in this study have been appropriately acknowledged and referenced. I also confirm that any assistance received during the course of this research, including guidance from my supervisors and support during fieldwork, has been properly recognised in the relevant sections of this dissertation.

I understand that submitting false information in this declaration may result in disciplinary action in accordance with the regulations of the University.

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Date: 29 July 2026

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ACRONYMS AND ABBREVIATIONS

AREC - Animal Research Ethics Committee

BMD - Biodiversity Management Division

CAES - College of Agricultural and Environmental Sciences

CBD - Convention on Biological Diversity

CMR – Capture–Mark–Recapture

DMOSS - Durban Metropolitan Open Space System

EMA - eThekweni Municipal Area

EMM - eThekweni Metropolitan Municipality

EPCPD – Environmental Planning and Climate Protection Division

FAO - Food and Agriculture Organization of the United Nations

GGEP - Giba Gorge Environmental Precinct

GGNR - Giba Gorge Nature Reserve

GNR – Giba Gorge Nature Reserve

ILC - International Land Coalition

ILRI - International Livestock Research Institute

IPBS - Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services

IPCC - Intergovernmental Panel on Climate Change

IUCN - International Union for Conservation of Nature

KNR - Krantzkloof Nature Reserve

KZN - KwaZulu-Natal

KZNSS – KwaZulu-Natal Sandstone Sourveld

LFA – Large Folding Aluminium

MEA – Millennium Ecosystem Assessment

MKA - Minimum Known Alive

MPA - Maputaland-Pondoland-Albany

OECMs - Other Effective Area-Based Conservation Measures

SAGraSS - Southern African Grassland Scoring System

SANBI - South African National Biodiversity Institute

SCA - Systematic Conservation Assessment

SFA - Small Folding Aluminium

SRA - Special Rating Area

SRA - Special Rating Area

UKZN – University of KwaZulu-Natal

UNEP - United Nations Environment Programme

UNISA - University of South Africa

WCMC - World Conservation Monitoring Centre

WRI - World Resources Institute

WWF - World Wide Fund for Nature

ABSTRACT

The KwaZulu-Natal Sandstone Sourveld (KZNSS) landscape is one of the Critically Endangered ecosystems that forms part of the Maputaland-Pondoland-Albany (MPA) biodiversity hotspot. It is a biodiversity hotspot which supports important assemblages of endemic fauna and flora and provides social and ecological services to the eThekweni assemblages. More than ~90% of this landscape has been completely altered and modified, and only ~0.1% is under statutory protection. To understand and improve conservation efforts for this threatened ecosystem, it's important to understand the effects of anthropogenic and environmental factors on biodiversity in this vegetation type. The Durban Metropolitan Open Space System (D'MOSS) initiative is one of the eThekweni efforts to conserve areas with high biodiversity value. Small mammals are key components in terrestrial ecosystems and their abundance and diversity are key bioindicators of ecosystem health, habitat quality, and the impacts of fragmentation. This study aimed to assess small mammal guild structure, species richness, diversity and population dynamics between D'MOSS and protected areas within this landscape. A Capture Mark Recapture (CMR) approach was used to monitor the seasonal dynamics of the native small mammal assemblages. Live trapping transects and H.B. Sherman™ live box traps were implemented inside and outside three protected areas (Krantzkloof Nature reserve, Giba Gorge Nature Reserve and Doornrug and Uitkomst Conservancy (reference site)) between July 2018 and June 2019. Over 16 586 trap nights, 890 unique individuals representing 11 small mammal species from 8 families and two Orders, were captured. The haul was dominated by rodent species. *Mastomys natalensis* (54%) and *Rhabdomys dilectus* (28%) suggesting that these species form the core of the small mammal assemblage in this threatened landscape. Local elevational gradients, regional variation and habitat management regimes (treatments) had a significant influence on total captures, trap success, minimum know alive (MKA), species richness and diversity. Slope aspect, one of the under-surveyed environmental variables in small mammal studies significantly influenced total captures, trap success, MKA, and species richness, while seasonal variation significantly influenced total captures and MKA. Grassland condition significantly influenced total captures, MKA, and species richness. The body size of several small mammal species varied in response to changes in elevational gradients, slope aspect, seasonal variation, grassland

condition, regional variation and habitat management regimes. Only the age and reproductive status of *M. natalensis* was significantly influenced by elevational gradients, slope aspect, seasonal variation, grassland condition and regional variation. The dominance of disturbance-tolerant generalist species and reduced representation of habitat specialists in more altered grassland sites indicate that shifts in small mammal assemblage composition reflect underlying habitat condition and fragmentation gradients. The results demonstrate the importance of small mammals as bio-indicators within this landscape which may provide guidance for conservation planning in the Critically Endangered KZNSS landscape particularly for the highly fragmented D'MOSS sites outside protected areas.

KEYWORDS

Small mammal assemblages, population dynamics, species diversity, species richness, functional guilds, assemblage structure, habitat heterogeneity, habitat disturbance, Maputaland-Pondoland-Albany Biodiversity Hotspot (MPA), Durban Metropolitan Open Space System (D'MOSS), KwaZulu-Natal Sandstone Sourveld (KZNSS), eThekweni

CHAPTER 1

INTRODUCTION

1.1 Background

Over the past century, anthropogenic activities have increasingly and profoundly altered natural environments worldwide. Key drivers of environmental degradation include pollution (Lelieveld et al., 2019), rapid population growth (Weber & Sciubba, 2018), deforestation (de-Miguel et al., 2025), the combustion of fossil fuels (Lelieveld et al., 2019; Li et al., 2024b), the emission of aerosols into the atmosphere (Amooli et al., 2025) and the spread of invasive alien species (Vilà et al., 2011). These pressures have resulted in widespread environmental change, manifested through increased soil erosion, declining air quality, altered climate patterns, and the deterioration of freshwater systems (Steffen et al., 2015). As a consequence, water resources have become less safe for human consumption and less capable of sustaining healthy aquatic and terrestrial ecosystems. Vörösmarty et al. (2010) demonstrated that increasing anthropogenic pressures have compromised freshwater security globally, while Kaushal (2016) and Kaushal et al. (2018) highlighted the effects of human activities on water quality and ecosystem functioning. In addition, climate change is increasingly recognised as a global amplifier of ecological stressors, including intensified human-wildlife conflict as species are forced to shift ranges and resources become more limited (Abrahms et al., 2023). Recent global scientific assessments, including the Sixth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC, 2022) and the United Nations Environment Programme's Global Environment Outlook 7 (UNEP, 2024), indicate that climate change and land degradation driven by human activities are placing unprecedented pressure on freshwater resources and terrestrial ecosystems at a global scale.

The Intergovernmental Panel on Climate Change (IPCC, 2022) reported that anthropogenic climate change has caused unprecedented shifts in seasonal temperature and precipitation patterns, intensifying water scarcity, degrading habitats, and undermining ecosystem stability. These shifts are more prevalent in ecosystems that are inherently fragile or highly specialised, such as grasslands, wetlands, mountain systems, coral reefs, and arid environments. Weiskopf et al. (2020) demonstrated that climate change is already altering biodiversity and ecosystem

functioning across multiple ecological systems, while Scheffers et al. (2016) showed that climate-driven disturbances are particularly severe in ecosystems with high environmental sensitivity and low adaptive capacity. Rapid climatic warming is occurring at rates that exceed the adaptive capacity of many species, disrupting life-history processes, altering phenology, shifting species distributions, and increasing extinction risk across terrestrial and marine ecosystems. Parmesan (2006) provided early evidence of widespread climate-driven shifts in phenology and species distributions. Pecl et al. (2017) showed that biodiversity redistribution is now occurring across both terrestrial and marine environments in response to global warming, and Urban (2015) demonstrated that these changes are contributing to elevated global extinction risk. These impacts are especially concerning for wetlands, which provide essential ecosystem services including water purification, flood regulation, carbon storage, and habitat provision, yet are highly sensitive to increases in temperature and changes in hydrological regimes. Bonebrake et al. (2018) highlighted that climate-driven phenological and hydrological shifts strongly affect ecosystem functioning in sensitive systems, while Kindlmann et al. (2025) emphasised that wetlands are among the ecosystems most vulnerable to altered temperature and water regimes under future climate scenarios.

More broadly, the cumulative effects of anthropogenic activities increasingly threaten the biological and ecological processes that underpin Earth system functioning. Steffen et al. (2005) demonstrated that human-driven global change is placing increasing pressure on Earth system stability, while Rockström et al. (2009) showed that these pressures are approaching critical planetary boundaries that regulate ecological processes. Díaz et al. (2018) and Costanza et al. (2017) reported that these processes are fundamental to the provision of ecosystem goods and services that support human survival and well-being, including food production, freshwater supply, soil formation, climate regulation, water purification, biological control, and recreational and cultural benefits. Cardinale et al. (2012) provided strong empirical evidence that biodiversity loss directly reduces the capacity of ecosystems to deliver these services, while McElwee et al. (2023) and Douglas (2024) further highlighted how ecosystem services underpin human well-being through multiple interacting ecological pathways. Hooper et al. (2005) demonstrated that biodiversity loss destabilises ecological systems by reducing functional diversity, disrupting energy flows and nutrient cycling,

and weakening trophic interactions. Building on this, Tilman et al. (2014) showed that reductions in biodiversity impair ecosystem functioning across multiple ecological processes, whereas Díaz et al. (2015) and Lefcheck et al. (2015) demonstrated that higher biodiversity enhances ecosystem productivity, resilience, stability, and multifunctionality across trophic levels and habitats. The scale and intensity of these changes are reflected in the designation of the current geological era as the Anthropocene, an era in which human activities have become a dominant force shaping the Earth's climate, ecosystems, and biogeochemical processes. Crutzen (2002) first introduced the concept of the Anthropocene, and Steffen et al. (2007) demonstrated that human activities are now comparable to major geological forces influencing Earth system processes. Lewis & Maslin (2015) further refined the concept by emphasising its global ecological and climatic signatures. Unlike earlier periods of environmental modification, the Anthropocene is distinguished by the unprecedented speed, magnitude, and global extent of human impacts. Ellis et al. (2021) showed that although human societies have influenced terrestrial ecosystems for millennia, recent changes are unprecedented in both rate and scale, raising significant concerns regarding ecosystem resilience, sustainability, and the continued provision of ecosystem services.

In response to these pressures, the establishment and effective management of protected areas have become increasingly critical for biodiversity conservation. Chape et al. (2005) highlighted their importance in achieving global conservation targets and documented the substantial expansion of protected area networks over recent decades. Geldmann et al. (2013) demonstrated that protected areas play an important role in reducing habitat loss and slowing population declines, while Watson et al. (2014) further emphasised their contribution to conserving biodiversity and maintaining ecological processes. Gray et al. (2016) subsequently showed that biodiversity is generally higher within protected areas than in surrounding unprotected landscapes. Despite these benefits, protected areas have been criticised for spatial biases, inadequate representation of biodiversity-rich regions, and limited ecological connectivity with surrounding human-modified landscapes, all of which can undermine their effectiveness. Greve et al. (2011) demonstrated that protected area effectiveness varies according to landscape context. Venter et al. (2018) showed that many protected areas are located in regions experiencing relatively low human pressure

rather than areas of highest conservation priority. More recently, Geldmann et al. (2019), DeFries et al. (2023), and Li et al. (2024a) highlighted that the capacity of protected areas to resist anthropogenic pressures and habitat loss remains highly variable among regions and depends strongly on management effectiveness, landscape configuration, and connectivity.

Landscape ecology provides an interdisciplinary framework for understanding how spatial patterns influence ecological processes across heterogeneous landscapes. Forman & Godron (1986) initially established landscape ecology as the study of interactions between spatial patterns and ecological processes. Wu & Hobbs (2007) further described it as an integrative discipline linking ecological patterns and processes across multiple spatial and temporal scales. McGarigal et al. (2012) highlighted the importance of quantifying landscape structure through spatial pattern analysis. More recently, Opdam et al. (2018) and Frazier (2024) emphasised its increasing relevance for sustainability science and contemporary environmental challenges. A central concept within landscape ecology is habitat fragmentation, whereby continuous habitats are divided into smaller and more isolated patches embedded within human-dominated matrices. Fahrig (2003) identified habitat fragmentation as a major driver of biodiversity loss, and Haddad et al. (2015) demonstrated its persistent and widespread impacts on ecosystems worldwide. Recent studies (Kuipers et al., 2021; Yuan et al., 2024) have shown that fragmentation amplifies the negative effects of habitat loss on biodiversity and continues to threaten ecological integrity across both natural and protected landscapes.

Habitat fragmentation alters habitat quality, increases edge effects, restricts dispersal, disrupts ecological interactions, and reduces species richness, population viability, and ecosystem functioning. Didham et al. (2012) and Laurance et al. (2018) highlighted the complex ecological consequences of fragmentation. Fletcher et al. (2023) emphasised the importance of spatial scale in understanding fragmentation effects. Recent studies (Gonçalves-Souza et al., 2025; Jian et al., 2025) further indicate that landscape edge heterogeneity and species turnover influence biodiversity responses in fragmented systems, although these processes do not fully compensate for biodiversity losses. These impacts are particularly severe in ecosystems that have already undergone extensive transformation, such as grasslands. Zhang et al. (2023) demonstrated that grazing and increasing aridity interact to reduce grassland

biodiversity and ecosystem functioning. Yan et al. (2024) showed that habitat loss weakens the positive relationship between plant species richness and above-ground biomass. Recent evidence from Jian et al., 2025 further indicates that landscape fragmentation continues to shape biodiversity outcomes and restoration potential in fragmented grassland ecosystems.

Petermann & Buzhdygan (2021) described grasslands as among the world's most biodiverse terrestrial ecosystems. Parr et al. (2014) and Bond (2019) further emphasised the ecological significance and increasing vulnerability of open ecosystems. Stevens et al. (2022) highlighted their importance for ecosystem functioning during the Anthropocene. Petermann & Buzhdygan (2021) noted that the high biodiversity of grasslands underpins numerous ecosystem services essential for human well-being, whereas Zhang et al. (2023) demonstrated that grassland biodiversity strongly influences ecosystem functioning under different grazing regimes. O'Mara (2012) highlighted the importance of grasslands for global food security and showed that these ecosystems support the livelihoods of more than one billion people worldwide, particularly through livestock production systems. Despite their ecological and socio-economic importance, Stevens et al. (2022) identified grasslands as among the most threatened biomes globally, primarily due to agricultural expansion, urbanisation, and infrastructure development. Similarly, Ma et al. (2024) demonstrated that increasing human expansion continues to intensify pressures on these ecosystems. Jian et al. (2025) further showed that landscape modification and fragmentation reduce habitat connectivity and negatively influence biodiversity patterns and ecological functioning in grassland ecosystems.

In South Africa, the South African National Biodiversity Institute (SANBI) (2013) and Carbutt & Kirkman (2022) reported that the Grassland Biome covers ~30% of the national land surface. It represents one of the country's most biodiverse and economically important biomes, although ~39-40% has already been irreversibly transformed. Li et al. (2024a) demonstrated that protected areas have had only limited success in preventing habitat loss, while Vukeya et al. (2025) highlighted the continued degradation of grassland ecosystems across southern Africa. These pressures have been exacerbated by urban expansion, which continues to fragment grassland remnants, reduce patch size and connectivity, and increase their susceptibility to degradation as reported by Haddad et al. (2015), Carbutt & Kirkman (2022) and

Vukeya et al. (2025). Consequently, the conservation of grassland remnants outside formally protected areas is becoming increasingly important for maintaining biodiversity and ecological functioning. Jian et al. (2025) demonstrated that landscape structure, habitat fragmentation, and edge heterogeneity strongly influence biodiversity outcomes in fragmented grasslands. Vukeya et al. (2025) emphasised the importance of maintaining ecological integrity to ensure long-term conservation success.

Within KwaZulu-Natal (KZN), the Durban Metropolitan Open Space System (D'MOSS) was established by the eThekweni Metropolitan Municipality (EMM) in the early 1980s in response to rapid urbanisation and biodiversity loss. Roberts (1994) first proposed an urban open-space network for eThekweni, laying the foundation for what has since evolved into a comprehensive biodiversity planning and conservation framework embedded within municipal decision-making processes. McLean et al. (2024) demonstrated that D'MOSS has become a globally recognised example of urban biodiversity planning and ecological infrastructure conservation. McLean et al. (2025a) further highlighted the increasing importance of local government initiatives in addressing contemporary urban biodiversity conservation challenges. The D'MOSS currently covers ~95 000 hectares (ha), representing nearly one-third of the eThekweni Municipal Area (EMA), and functions as both a spatial planning and biodiversity conservation framework. McLean et al. (2016) highlighted the role of systematic conservation assessments in guiding the identification and protection of priority conservation areas within the municipality. The system includes critically threatened ecosystems such as the KwaZulu-Natal Sandstone Sourveld (KZNSS) grassland landscape, which is a species-rich grassland endemic to KZN and classified as Critically Endangered in national and provincial biodiversity assessments. Mucina & Rutherford (2006) provided the formal classification of this vegetation type, while Boon et al. (2016) identified the KZNSS landscape as one of eThekweni's highest conservation priorities because of its exceptional biodiversity value and threatened conservation status.

Despite its importance, the KZNSS landscape has undergone extensive transformation due to multiple and interacting land-use pressures. Mucina & Rutherford (2006) identified commercial agriculture, particularly sugarcane cultivation and timber plantations, as major drivers of habitat conversion, while Nkambule et al.

(2016) further highlighted the contribution of high-density subsistence farming to habitat degradation and fragmentation. More recently, McLean et al. (2024) and McLean et al. (2025a) demonstrated that rapid urban expansion associated with population growth within the EMA continues to intensify pressures on the remaining KZNSS grassland fragments. Naicker et al. (2016) and Boon et al. (2016) further showed that these interacting land-use pressures have collectively driven substantial habitat loss and fragmentation across the KZNSS landscape. Current assessments estimate that ~90% of the original KZNSS landscape has been transformed, leaving only ~10-15% in a natural or semi-natural condition, of which >1% is formally protected. Jewitt et al. (2015) documented extensive land-cover change across KZN, demonstrating substantial habitat transformation within threatened ecosystems such as the KZNSS landscape. McLean et al. (2025a) emphasised the continued vulnerability of the remaining grassland remnants within eThekweni's urban landscape.

Within the EMA, only ~116 ha (0.74%) of the KZNSS landscape is afforded local protection, substantially below the systematic conservation planning target of ~3 920 ha required to ensure the adequate representation and long-term persistence of this vegetation type (eThekweni Municipality, 2013). Boon et al. (2016) subsequently highlighted the need to expand conservation coverage of the remaining KZNSS grassland remnants, while McLean et al. (2025b) identified opportunities for increasing terrestrial conservation areas within the municipality. Recent studies (Shih & Mabon, 2018; Andersson et al., 2025; Elmqvist et al., 2019; McPhearson et al., 2021; McLean et al., 2024) have emphasised that effective biodiversity conservation within urban landscapes requires proactive, science-based approaches that integrate ecological, social, and economic considerations into land-use planning. Shih & Mabon (2018) demonstrated the importance of incorporating biodiversity and ecosystem services into planning and decision-making processes. McLean et al. (2024) showed how these principles have been incorporated into eThekweni's open-space planning framework. To improve the persistence of KZNSS landscape and enhance ecological connectivity, several restoration and management initiatives, including ecosystem-based adaptation and community engagement projects, have been implemented within D'MOSS. Boon et al. (2016) highlighted the importance of active management interventions for maintaining KZNSS landscape, while Mugwedi et al. (2018) described restoration strategies aimed at improving ecological resilience. McLean et al. (2025a,

2025b) further highlighted ongoing conservation initiatives and opportunities to strengthen biodiversity persistence and connectivity within the municipal landscape. These initiatives emphasise the importance of collaborative governance and evidence-based planning in mitigating the ongoing impacts of urbanisation and land-use transformation. Roberts & Diederichs (2002) first advocated collaborative approaches to sustainable urban development within eThekweni. Cockburn et al. (2016) subsequently demonstrated how science-action partnerships can strengthen biodiversity conservation and improve its integration into municipal planning processes.

Identifying reliable biological indicators is essential for assessing ecosystem condition and informing conservation management in fragmented grassland landscapes. Terrestrial small mammals are particularly well suited for this role because of their close association with vegetation structure, limited dispersal ability, rapid responses to disturbance, and well-documented ecological requirements as reported by Avenant, (2000), (2003), Monadjem et al. (2015) Hope et al. (2017). Avenant (2000, 2003) and Avenant & Cavallini (2007) demonstrated that changes in small mammal assemblage composition closely reflect ecological disturbance and habitat integrity in South African ecosystems. Avenant (2011) further showed that rodent populations respond predictably to changes in grassland condition. Similarly, Hope et al. (2017), Green et al. (2020), and Ayodele (2025) highlighted the value of small mammals as indicators of climate, biodiversity, habitat health, and ecosystem change. Monadjem et al. (2015) demonstrated that many southern African small mammal species possess well-defined habitat requirements, making them particularly sensitive to environmental change. Consequently, small mammal assemblage composition, abundance, and functional guild structure have been shown to reflect gradients of habitat modification, disturbance history, patch size, vegetation structure, and landscape connectivity. Avenant et al. (2008) demonstrated strong relationships between assemblage characteristics and habitat integrity, while Haddad et al. (2015) showed that habitat fragmentation alters assemblage composition by reducing connectivity and increasing landscape isolation. More recently, Osman et al. (2022), Ramahlo et al. (2022), and Dondina et al. (2025) demonstrated that land-cover change, habitat degradation, seasonal variation, habitat quality, and landscape structure are important determinants of small mammal diversity and assemblage patterns. Importantly,

individual species respond differently to environmental change because of differences in nutritional requirements, physiological tolerances, anti-predator strategies, and social behaviour. Avenant (2011), Monadjem et al. (2015), de Klerk & Avenant (2017), Ancillotto et al. (2025), and Yang et al. (2025) demonstrated that these species-specific responses enable small mammal assemblages to reflect complex patterns of environmental change and distinguish between different levels of habitat disturbance and ecological integrity. Ancillotto et al. (2025) further showed that microhabitat characteristics shape urban small mammal assemblages, whereas Yang et al. (2025) demonstrated that long-term environmental change influences assemblage composition and species associations. Consequently, variation in species responses enables small mammal assemblages to serve as sensitive indicators of environmental change across fragmented landscapes.

Studies have further demonstrated that terrestrial small mammals are effective indicators across diverse landscapes and monitoring programmes. Avenant (2011) showed that rodents provide reliable indicators of ecosystem integrity in South African grasslands because their populations respond predictably to changes in habitat condition. Green et al. (2020) further demonstrated their usefulness for monitoring ecological restoration. Hope et al. (2017) highlighted their value for detecting climate-driven environmental change and associated biodiversity responses. More recently, Dondina et al. (2025) reported that small mammal assemblages reflect habitat quality and landscape characteristics within urban green spaces. These studies emphasise the utility of small mammals for assessing ecosystem integrity, restoration outcomes, and environmental change across multiple spatial scales.

Despite their ecological significance as bio-indicators, empirical information on small mammal assemblages within KZNSS landscape in the EMA remains limited, particularly regarding species-level responses to habitat fragmentation, degradation, and conservation interventions. Previous studies within the KZNSS landscape have primarily focused on vegetation composition and floristic patterns (Drury, 2016), habitat fragmentation and landscape connectivity (Naicker et al., 2016), alien plant invasion and vegetation mapping (Odindi et al., 2016), community-based conservation approaches (Nkambule et al., 2016), epigaeic ant diversity and distribution (Hlongwane et al., 2019), and the effects of fire and land-use practices on grassland structure and composition (Nelisi, 2025). Consequently, little attention has been given

to the diversity, composition, and assemblage structure of terrestrial small mammals within the fragmented KZNSS landscape. This knowledge gap limits the ability of conservation practitioners to adequately assess the ecological condition of KZNSS remnant grassland patches and to develop informed management strategies for their long-term conservation. Therefore, this study aims to investigate patterns of species composition, abundance, diversity, and assemblage structure of terrestrial small mammals within remnant KZNSS grassland patches, thereby contributing to evidence-based conservation planning and management within D'MOSS.

1.2 Motivation

The EMA lies within the Maputaland-Pondoland-Albany (MPA) biodiversity hotspot, which van Wyk & Smith (2001) recognised it for its exceptional levels of plant endemism and biodiversity. The EMA contains a diverse range of terrestrial and aquatic ecosystems that eThekweni Municipality (2020) attributed to the region's varied climate, geology, soils, and biogeography. Since the 1990s, however, rapid urbanisation and increasing human demands have placed considerable pressure on natural areas. Roberts & Diederichs (2002) identified balancing development and biodiversity conservation as a major challenge for municipal planning. Roberts et al. (2012), McLean et al. (2024), and McLean et al. (2025a) demonstrated the continuing difficulties associated with reconciling urban expansion and biodiversity conservation within eThekweni. In response to these pressures, D'MOSS was established to protect areas of high biodiversity value and integrate conservation into municipal planning processes. Roberts et al. (2012) described D'MOSS as an ecosystem-based adaptation initiative, and McLean et al. (2024) demonstrated its evolution into a science-based conservation planning framework. Through systematic conservation assessments, Jewitt et al. (2015) quantified extensive habitat transformation across KZN, whereas Boon et al. (2016) highlighted the conservation importance of the KZNSS landscape within the urban matrix. More recently, McLean et al. (2025b) identified opportunities to strengthen ecological connectivity and expand terrestrial conservation areas within the EMA. These studies demonstrate the importance of D'MOSS in guiding the conservation of fragmented, high-value ecosystems such as the KZNSS landscape within an increasingly urbanised environment.

1.3 Problem statement

The KZNSS landscape is a species-rich grassland ecosystem in terms of plant diversity endemic to South Africa and occurring largely within the EMA. Mucina & Rutherford (2006) recognised this vegetation type for its high biodiversity value and ecological importance. However, rapid urbanisation, agricultural expansion, commercial forestry, and other land-use changes have resulted in extensive habitat loss and fragmentation within the KZNSS landscape. Jewitt et al. (2015), Boon et al. (2016), Naicker et al. (2016), and Nkambule et al. (2016) demonstrated that these pressures have reduced the extent and connectivity of remaining grassland patches. Currently, only ~10-15% of the original KZNSS landscape remains in a natural or semi-natural condition, with >1% formally protected. McLean et al. (2025a) highlighted that existing conservation efforts remain insufficient to ensure the long-term persistence of this ecosystem, while McLean et al. (2025b) emphasised the need to improve ecological connectivity and strengthen conservation networks within fragmented urban landscapes. Consequently, the KZNSS landscape is recognised as one of the most threatened ecosystems in KZN. Despite its conservation importance, limited information exists regarding the biodiversity of the remaining KZNSS grassland remnants, particularly the responses of terrestrial small mammal assemblages to habitat fragmentation and degradation. Avenant (2000, 2003, 2011), Avenant & Cavallini (2007), Avenant et al. (2008), and de Klerk & Avenant (2017) demonstrated that small mammals are effective indicators of habitat condition and ecological disturbance. Therefore, understanding patterns of small mammal diversity and assemblage structure within remnant KZNSS landscape is essential for assessing ecosystem condition and informing evidence-based conservation management within D'MOSS.

1.4 Research questions

1. What is the current composition, diversity, and abundance of small mammal species in KZNSS patches within the EMA?
2. How does small mammal assemblages differ between KZNSS grassland patches that are inside protected areas and those that are unprotected outside protected areas (D'MOSS)?

3. How do environmental gradients, including elevational gradients, slope aspect, seasonal variation, grassland condition, regional variation, and habitat management regimes (treatments), and levels of human disturbance influence small mammal species richness, abundance, and functional guild composition across the KZNSS landscape?
4. Can small mammal assemblages serve as reliable bio-indicators to assess the ecological condition and conservation value of the KZNSS landscape?

1.5 Aim of the study

The aim of this study was to assess the composition, abundance, diversity, and functional guild structure of small mammal assemblages in the KwaZulu-Natal Sandstone Sourveld (KZNSS) landscape. The study focused on two highly urbanised protected areas, Krantzkloof Nature Reserve (Kloof) and Giba Gorge Nature Reserve (Hillcrest), and adjacent Durban Metropolitan Open Space System (D'MOSS) grassland patches within the Inner West of eThekweni. The Doornrug and Uitkomst Conservancy in Camperdown (Outer West of eThekweni) was used as a less urbanised reference site to assess the effectiveness of D'MOSS grassland patches in conserving small mammal diversity. Furthermore, the study aimed to examine the influence of environmental gradients, including local elevational gradients, slope aspect, seasonal variation, grassland condition, regional variation, and habitat management regimes (treatments), on small mammal species richness, abundance, and functional guild composition.

1.6 Objectives

1. To document the composition, abundance, diversity, and functional guild structure of small mammal assemblages occurring in KZNSS landscape within the EMA.
2. To assess how environmental gradients, including local elevational gradients, slope aspect, seasonal variation, grassland condition and regional variation influence small mammal species richness, abundance, assemblage composition, and functional guild structure.

3. To evaluate the effects of habitat management regimes(treatments), including protection status (inside and outside (D'MOSS patches) of protected areas and the reference site, urbanisation, and habitat fragmentation on small mammal assemblages within the KZNSS landscape. This was done to assess the effectiveness of D'MOSS grassland patches in conserving small mammal diversity and assemblage structure.

4. To determine whether small mammal assemblages can serve as reliable indicators of habitat condition, ecological integrity, and conservation value in remnant KZNSS grassland patches to support conservation planning and management of D'MOSS grassland patches.

1.7 Thesis Outline

This dissertation consists of seven chapters, a reference list and supplemental material in the form of appendices.

Chapter 1: Introduction

This chapter provides a general background to biodiversity loss caused by anthropogenic activities, with specific reference to grassland ecosystems. It introduces the study context, outlines the research problem, and presents the aims and objectives of the study, thereby justifying the research approach used to address the research questions.

Chapter 2: Literature Review

This chapter presents a review of relevant literature on the global biodiversity crisis and the effects of habitat loss and fragmentation on grassland biomes. It discusses grassland vegetation, conservation challenges, and current conservation strategies aimed at protecting grassland biodiversity. The chapter also reviews literature on small mammals as ecological bio-indicators of habitat condition and environmental integrity.

Chapter 3: Study Area

This chapter describes the study area and provides background information on the selected sites, including their location, vegetation types, climate, and general environmental characteristics.

Chapter 4: Materials and Methods

This chapter outlines the research methodology used in the study, including sampling design, data collection procedures, habitat assessments, and statistical analyses applied to address the study objectives.

Chapter 5: Results

This chapter presents the results of the data analyses, including patterns in small mammal species composition, abundance, and variation across environmental gradients (elevational gradients, slope aspect, seasonal variation, grassland condition, regional variation, and habitat managements regimes (treatments)).

Chapter 6: Discussion

This chapter interprets the results in relation to the study objectives and existing literature. The findings are discussed in the context of habitat condition, environmental gradients, and the effectiveness of D'MOSS in conserving biodiversity, with broader implications for urban grassland conservation.

Chapter 7: Conclusion, Study Limitations and Management Recommendations

This chapter summarises the key findings of the study and discusses their implications for conservation planning and biodiversity management. It also outlines study limitations and provides management recommendations to support the conservation of KZNSS within the EMA.

CHAPTER 2

LITERATURE REVIEW

2.1 Biodiversity crisis

Biodiversity conservation remains a central justification for establishing protected areas because biodiversity encompasses variation at genetic, species, and ecosystem levels and the interactions among organisms and their environments. McNeely et al. (1990) first highlighted this multidimensional nature of biodiversity. Scholes et al. (2008), Otegui et al. (2012), He & Wei (2023), and Silva et al. (2025) further emphasised the importance of conserving biodiversity across these interconnected levels. The importance of conserving biodiversity extends beyond the protection of individual species, as biodiversity is also fundamental for sustaining ecosystem services, including pollination, water purification, nutrient cycling, and climate regulation. McNeely et al. (1990) recognised these benefits as a major justification for biodiversity conservation, whereas Díaz et al. (2019), Kim et al. (2024), Qiu & Mitchell (2024), and Naderi & Fallahi-Akhlamad (2025) demonstrated that biodiversity underpins ecosystem functioning, resilience, and human well-being. Despite its importance, biodiversity continues to decline at multiple levels. Although estimates of global species richness remain uncertain, with May (1998) suggesting between ~5 and 50 million species and Mora et al. (2011) demonstrating that many species remain undescribed, contemporary extinction rates are substantially higher than natural background levels. The Millennium Ecosystem Assessment (MEA) (2005) reported extinction rates ~100-1 000 times above historical rates, while Díaz et al. (2019) estimated that ~1 million species are currently threatened with extinction. These declines are not limited to species loss, but also extend to genetic diversity, which is essential for maintaining the capacity of populations to respond to environmental change. Such losses raise concerns about the future persistence of biodiversity, particularly as multiple anthropogenic pressures increasingly act together. Shaw et al. (2025) further demonstrated significant declines in genetic diversity, potentially reducing species capacity to adapt to environmental change. Earlier projections by Pimm & Raven (2000) suggested that up to ~50% of extant species could be lost before 2050. Pimm et al. (2014), Urban (2015), and Urban et al. (2024) further

demonstrated that interactions between climate change, urbanisation, and other anthropogenic pressures are likely to intensify future biodiversity declines.

Barnosky et al. (2011) and Ceballos et al. (2015) provided evidence that current biodiversity losses may represent the onset of a sixth mass extinction event. Unlike previous mass extinctions which were primarily associated with natural disturbances, Keller et al. (2012) and Schulte et al. (2010) attributed these earlier events to natural disturbances, such as volcanism and asteroid impacts, whereas the current extinction crisis is primarily driven by human activities. Among these anthropogenic drivers, habitat transformation and land-use change remain among the most important drivers of biodiversity loss as described by Scholes et al. (2008). Steffen et al. (2015) demonstrated that human activities have pushed several Earth system processes beyond safe operating limits. Jaureguiberry et al. (2022) further highlighted agricultural expansion, deforestation, invasive species, and climate change as leading drivers of global biodiversity loss. More recently, Singini & Baso (2025) showed that these anthropogenic pressures continue to intensify biodiversity declines across different International Union for Conservation of Nature and Natural Resources (IUCN) conservation categories. Similarly, IPBES (2019) reported that ~75% of terrestrial and ~66% of marine environments have been significantly altered by human activities. Curtis et al. (2018) and Pendrill et al. (2022) further demonstrated extensive tropical forest loss associated with agricultural expansion. The World Resources Institute (WRI) (2025), Bourgoin et al. (2025), and Villegas (2025) further showed that recent deforestation rates remain incompatible with global conservation targets. Importantly, habitat transformation is not confined to areas outside protected areas. Despite global conservation commitments, habitat loss and fragmentation continue even within protected areas. Jones et al. (2018) and Geldmann et al. (2019) demonstrated that many protected areas remain subject to significant human pressures, thereby reducing their effectiveness. As multiple ecosystems are progressively transformed, one consequence is habitat fragmentation, where large, continuous areas of habitat are broken into smaller and more isolated patches. Habitat fragmentation reduces species population sizes, disrupts ecological processes, increases edge effects, and diminishes landscape connectivity. Leach & Givnish (1996) first highlighted these ecological consequences, whereas Haddad et al. (2015), Montes-Rojas et al. (2024), Yuan et al. (2024), and Gonçalves-Souza et al. (2025) demonstrated that

fragmentation reduces species richness, increases extinction risk, and compromises ecosystem integrity.

The human population explosion has contributed substantially to pressures on biodiversity. Cincotta et al. (2000) demonstrated that human populations within biodiversity hotspots were already growing faster than global averages during the late 20th century. Myers et al. (2000) highlighted that these biodiversity regions simultaneously contain exceptionally high levels of endemism and habitat loss. More recently, Cunningham & Beazley (2018) confirmed that biodiversity hotspots remain disproportionately populated relative to other regions, indicating that continued demographic growth is likely to place additional pressure on already threatened ecosystems. As human population grows, increasing demands for land and natural resources, together with expanding human footprints and landscape transformation, continue to amplify habitat conversion, resource demand, and environmental disturbance. Venter et al. (2016) documented substantial global increases in the human footprint since year 2000. These expanding human pressures are increasingly reflected in changes to natural habitats, particularly through conversion, habitat fragmentation, and disturbance. Yang et al. (2023, 2024) demonstrated that habitat conversion, fragmentation, and disturbance continue to increase within biodiversity-rich regions and key biodiversity areas. Similarly, Li et al. (2024a, 2024b) showed that many protected areas continue to experience habitat loss associated with increasing human pressures and ongoing land-use change. This indicates that human impacts are not confined to areas outside formal conservation boundaries but are increasingly widespread across protected and unprotected landscapes. Waldock et al. (2024) and Keck et al. (2025) further demonstrated that human impacts on biodiversity are geographically widespread and structurally embedded across multiple ecosystems. Looking ahead, these pressures are likely to intensify as human populations and activities increasingly overlap with wildlife habitats. Forward-looking projections by Ma et al. (2024) suggest that human-wildlife spatial overlap may increase across more than half of Earth's terrestrial surface by year 2070, potentially intensifying biodiversity risks and human-wildlife conflict.

These accelerating biodiversity declines raise fundamental ecological questions regarding how the loss of biological diversity influences ecosystem functioning and

whether critical thresholds exist beyond which ecosystems lose stability and resilience. Groffman et al. (2006) argued that identifying ecological thresholds is essential for effective environmental management. Mori et al. (2013), Isbell et al. (2015), and Oliver et al. (2015) demonstrated that biodiversity enhances ecosystem resilience and stability by maintaining ecological functions under environmental change. More recently, Fargeot et al. (2025) showed that genetic diversity can influence ecosystem functioning to an extent comparable to species diversity, while Shaw et al. (2025) demonstrated that continued losses of genetic diversity may compromise the adaptive potential and resilience of populations and ecosystems. These findings highlight the importance of understanding and conserving biodiversity at multiple levels, from genetic diversity to species and ecosystems. Conservation science seeks to address these challenges by generating robust evidence on the causes and consequences of biodiversity loss and by informing policies and management interventions aimed at promoting sustainable ecosystem management. Noss et al. (2012) advocated for more ambitious and ecologically representative conservation strategies, while Geldmann et al. (2013) demonstrated that protected areas can reduce habitat loss and biodiversity declines, although their effectiveness varies among regions and management contexts. Díaz et al. (2019) subsequently emphasised the need for transformative societal change to reverse global biodiversity decline, recognising that conventional conservation approaches alone may not be sufficient to address the scale of current pressures. More recent assessments by Senior et al. (2024) and Bell-James & Watson (2025) indicate that current conservation actions remain insufficient to achieve global biodiversity targets.

2.2 Protected areas and their effectiveness in conserving biodiversity.

The development of protected areas is a primary response to the biodiversity crisis, functioning as a core tool for in situ conservation of species, ecosystems, and ecological processes. Their effectiveness, however, depends not only on their establishment but also on their location, management, governance, and enforcement. Recent evidence demonstrates that protected areas can significantly reduce habitat loss, maintain ecological integrity, and generate positive conservation outcomes when strategically located and effectively managed. For example, Pulido-Chadid et al. (2023), Langhammer et al. (2024), and Li et al. (2024a) showed that protected areas

can provide measurable biodiversity and ecosystem benefits, while Cumming (2025) highlighted that their effectiveness varies considerably among regions and management contexts. These differences in conservation outcomes have reinforced the importance of systematic approaches to protected area planning and design. Early analyses by Margules & Pressey (2000) established the principles of systematic conservation planning. Chown et al. (2003) highlighted the importance of protected areas in conserving biodiversity within landscapes experiencing increasing human pressures. More recent studies by Geldmann et al. (2019) and Langhammer et al. (2024) confirm that well-managed protected areas continue to deliver measurable biodiversity benefits, particularly where governance, management effectiveness, and enforcement are strong.

Protected area coverage has expanded substantially over recent decades, with recent analyses reporting that protected and conserved areas now cover ~17.5% of terrestrial land and inland waters and ~8.4-8.5% of marine areas (United Nations Environment Programme World Conservation Monitoring Centre (UNEP-WCMC), & International Union for Conservation of Nature (IUCN), 2024). This represents considerable progress in global conservation efforts, however, current coverage remains below the 30x30 target established under the Kunming-Montreal Global Biodiversity Framework, which aims to conserve ~30% of the world's terrestrial, inland water, coastal, and marine environments by year 2030 (Convention on Biological Diversity (CBD), 2022). Importantly, achieving greater coverage does not necessarily translate into more effective biodiversity conservation. Geldmann et al. (2019) and Heywood (2019) emphasised that increasing protected area coverage alone does not guarantee effective biodiversity conservation, as many protected areas continue to experience biodiversity declines because of inadequate management, insufficient resources, and persistent external pressures. Similarly, Coad et al. (2015) demonstrated that management effectiveness is strongly influenced by governance, monitoring, and adequate resourcing. Allan et al. (2022) argued that achieving meaningful biodiversity conservation requires both expanding protected area networks and improving their management quality. In addition to challenges within protected areas, pressures from surrounding landscapes can further undermine their effectiveness. Increasing human population and associated land demands continue to compromise protected area effectiveness by reducing landscape connectivity, increasing edge effects, and driving

habitat conversion in surrounding unprotected areas (Watson et al., 2014). Laurance et al. (2018) and Maxwell et al. (2020) demonstrated that protected areas cannot function in isolation, as external land-use pressures frequently alter ecological processes and undermine conservation outcomes. These external pressures are particularly concerning where habitat is fragmented, as the loss and isolation of habitat patches can disrupt ecological networks and restrict species movement. Habitat fragmentation further threatens biodiversity persistence by disrupting ecological networks and restricting species movement, as shown by Haddad et al. (2015), Keeley et al. (2018), Scriven et al. (2019), and DeFries et al. (2023). Consequently, contemporary conservation approaches increasingly recognise that protected areas must be embedded within broader landscape-scale planning frameworks that maintain habitat connectivity and ecological processes. Earlier conservation planning studies similarly highlighted the importance of connected reserve networks. For example, Margules & Pressey (2000) argued that effective conservation planning requires reserve systems capable of maintaining ecological processes and viable populations. Reyers et al. (2002) demonstrated that habitat fragmentation and isolation can reduce biodiversity representation and conservation effectiveness within South African landscapes. Conservation outcomes also depend on governance arrangements and stakeholder participation. Studies by Alcorn (1993) and Hackel (1999) highlighted the limitations of exclusionary conservation approaches. Oldekop et al. (2016), Schleicher et al. (2019), and Dawson et al. (2024) demonstrated that involving local and indigenous communities in conservation can improve both biodiversity outcomes and social equity.

International commitments such as the Aichi Biodiversity Targets (CBD, 2010) and the Kunming-Montreal Global Biodiversity Framework (CBD, 2022) reflect growing recognition that expanding and strengthening protected area networks is essential for biodiversity conservation. However, Joppa & Pfaff (2009), Butchart et al. (2015), Allan et al. (2022), and Geldmann (2023) emphasised that conservation success depends not only on increasing protected area coverage but also on ensuring ecological representation, effective management, and long-term monitoring. Protected area effectiveness is strongly influenced by governance quality, political commitment, funding, and clearly defined conservation objectives. Leverington et al. (2010) and Geldmann et al. (2019) demonstrated that without adequate management and

monitoring frameworks, protected areas may fail to deliver meaningful biodiversity outcomes. Despite their central role in biodiversity conservation, protected areas continue to exhibit biases in their location and representation of threatened ecosystems. Margules & Pressey (2000) highlighted the importance of systematic conservation planning. Reyers et al. (2002) and Goodman (2003) demonstrated that conservation effectiveness in South Africa is constrained by uneven habitat representation and management challenges. More recently, Allan et al. (2022) and Dudley et al. (2024) argued that climate change and shifting species distributions require more adaptive and dynamic conservation planning approaches.

Protected areas are frequently too small and isolated to maintain viable populations of many species or sustain ecological processes over the long term. Margules & Pressey (2000) highlighted the importance of connected reserve networks while Haddad et al. (2015) demonstrated that habitat fragmentation continues to compromise biodiversity persistence and ecosystem functioning. More recently, Allan et al. (2022) emphasised that protected areas embedded within human-dominated landscapes are increasingly affected by land-use change and other external pressures, limiting their long-term effectiveness. There is also growing concern that many protected areas were established without considering the long-term effects of climate change on species distributions and ecosystem processes. Belote et al. (2021) and Allan et al. (2022) demonstrated that climate-driven range shifts may reduce the effectiveness of existing reserve networks unless adaptive and dynamic conservation planning approaches are implemented. Consequently, long-term biodiversity monitoring remains essential for assessing whether protected areas continue to conserve biodiversity and maintain ecological functions under changing environmental conditions.

Biodiversity hotspots remain important conservation priorities because they contain high levels of endemism and irreplaceable evolutionary history, as highlighted by Myers et al. (2000) and Mittermeier et al. (2004). However, there is increasing recognition that formal protected areas alone are insufficient to safeguard biodiversity. Tran et al. (2020) and Brodie et al. (2025) emphasised the importance of complementary approaches, including indigenous and community conserved areas, private reserves, and other effective area-based conservation measures, which can

enhance habitat connectivity and improve conservation resilience in human-modified landscapes.

2.3 Landscapes outside formally protected areas

Although protected areas are important for conserving biodiversity, Halpin (1997) and Hannah et al. (2002) recognised that the condition and context of the surrounding landscape are equally important for long-term conservation success because connectivity and landscape composition influence species movement, gene flow, and resilience to habitat fragmentation. Harvey et al. (2008) further emphasised that protected areas should not function as isolated "islands" of biodiversity within degraded landscapes but should instead be integrated with surrounding habitats to maintain connectivity and support broader ecosystem processes. Foundational studies by Forman & Godron (1986), later reinforced by Haddad et al. (2015), demonstrated that habitat fragmentation and the loss of connectivity negatively affect population viability by reducing habitat area, increasing isolation, and disrupting dispersal pathways. Subsequent work by Fahrig (2020) showed that edge effects and matrix quality strongly influence biodiversity persistence, with isolated reserves often failing to sustain long-term ecological processes in the absence of connected habitat networks. More recent studies by Travers et al. (2021), Mu et al. (2024), and Struebig et al. (2024) further demonstrate that conservation outcomes improve when protected areas are embedded within larger, connected landscapes.

Landscape ecology, as described by Wu & Hobbs (2007) and Wu (2013), seeks to understand ecological processes across multiple spatial scales by recognising that biodiversity patterns and ecosystem functions emerge from interactions among heterogeneous land units rather than from protected areas alone. Building on the foundations established by Forman & Godron (1986) and Turner (2005), the discipline emphasises the importance of landscape structure, spatial configuration, and habitat connectivity in shaping ecological processes. The principles of landscape composition, structure, function, and change outlined by Forman (1995) and Opdam et al. (2018) provide an important framework for integrating protected areas into broader socio-ecological landscapes and developing conservation strategies that are resilient to habitat fragmentation, land-use pressures, and climate change.

Habitat fragmentation has long been recognised as a major challenge to biodiversity conservation because it produces multiple, interacting effects on species and ecological processes. Early studies by Merriam (1988), Saunders & Hobbs (1991), and Wiens (1990) identified fragmentation as a process involving habitat loss, reductions in patch size, increased isolation, and the creation of habitat edges, all of which fundamentally alter ecological dynamics. Later syntheses by Fahrig (2003) and Haddad et al. (2015) demonstrated that these changes disrupt connectivity, alter assemblage structure, and ultimately threaten biodiversity persistence, findings that continue to be supported by recent studies such as Yuan et al. (2024) and Riva et al. (2025). Fragmentation also generates strong area effects, whereby smaller habitat patches support fewer individuals and species. Temple & Wilcox (1986) and Bierregaard et al. (1992) were among the first to demonstrate that reduced patch size increases local extinction risk and contributes to the loss of area-sensitive species. More recent studies (Prugh et al., 2008; Ramsay et al., 2023; Ong et al., 2024) have further shown that small and isolated fragments experience greater demographic stochasticity, reduced genetic diversity, and increased vulnerability to environmental fluctuations, resulting in persistent population declines and reduced ecosystem resilience. In addition, Hanski (1999b) and Ewers & Didham (2006) demonstrated that fragment isolation limits recolonisation following local extinctions, particularly for poor dispersers and habitat specialists, thereby reducing the functionality of metapopulation processes in fragmented landscapes.

Foundational empirical studies by Fahrig (2017) and Saura et al. (2017) demonstrated that reduced functional connectivity slows or prevents recolonisation, resulting in long-term declines in species richness and altered assemblage composition even where habitat remains. These effects are particularly pronounced for large predators, wide-ranging mammals, and habitat specialists, as shown by Ripple et al. (2014) and Tucker et al. (2018), because these species require extensive and connected habitats to maintain viable populations. Research has also highlighted the importance of edge effects in fragmented landscapes. Studies by Benítez-Malvido & Arroyo-Rodríguez (2008), Zurita et al. (2012), and Fahrig (2020) showed that although edge habitats may increase local species richness by favouring generalist and edge-tolerant species, this often occurs at the expense of interior-dependent specialist species. Ewers & Didham (2006) and Newbold et al. (2015) further suggested that such increases in diversity

may mask underlying biotic homogenisation and functional degradation. Some studies, including those of Haddad et al. (2015) and Pfeifer et al. (2017), demonstrated that edge effects can extend deep into habitat fragments, altering microclimatic conditions, vegetation structure, species interactions, and assemblage dynamics. These altered conditions may create ecological traps, a concept initially proposed by Murcia (1995) and later developed by Schlaepfer et al. (2002) and Battin (2004), whereby fragmented habitats appear suitable but ultimately reduce survival and reproductive success. These studies demonstrate that habitat fragmentation not only reduces habitat quantity but also fundamentally alters ecological functioning, highlighting the importance of maintaining habitat connectivity, improving matrix quality, and implementing landscape-scale conservation strategies.

The theory of island biogeography, developed by MacArthur & Wilson (1967), was one of the first frameworks used to explain how habitat fragmentation influences species diversity. It proposes that species richness in fragmented habitats is governed by the balance between colonisation and extinction, with both processes influenced by habitat patch size and isolation. Rybicki & Hanski (2013), together with Haddad et al. (2015), Saura et al. (2017), and Fahrig (2020), extended island biogeography concepts to terrestrial landscapes by demonstrating that fragmented habitat patches function as "habitat islands", where increasing isolation restricts dispersal among patches. Rosenzweig (2003) and Lomolino et al. (2010) reported that colonising species may alter assemblage composition through competition with resident species, while Gurevitch & Padilla (2004) and Colwell et al. (2012) highlighted the potential for novel predation and herbivory pressures. Haddad et al. (2015) and Newbold et al. (2015) further showed that colonisation can alter ecological processes, including nutrient cycling, disturbance regimes, and succession, thereby influencing long-term ecosystem dynamics. More recent studies by Saura et al. (2017) and Fahrig (2020) emphasised that biodiversity persistence in fragmented landscapes is strongly influenced by habitat connectivity and matrix quality, particularly under increasing land-use and climate pressures.

The metapopulation theory proposed by Opdam (1991) and further developed by Hanski (1999b) provides an important framework for understanding the consequences of habitat fragmentation. Hanski (1999a), together with Haddad et al. (2015) and Fahrig (2020), demonstrated that habitat fragmentation reduces patch size and

increases isolation, thereby restricting dispersal among populations and increasing the risk of local extinction. Metapopulations are therefore viewed as networks of spatially separated populations linked by dispersal, where long-term persistence depends on recolonisation following local extinction events, as described by Hanski (1999b) and Ovaskainen & Hanski (2004). More recent studies by Saura et al. (2017) and Fahrig (2020) further demonstrated that declining functional connectivity constrains recolonisation processes, leading to reductions in species richness and assemblage stability, particularly among habitat specialist species and poor dispersers, a pattern also highlighted by Haddad et al. (2015).

To address interactions between protected areas and their surrounding matrices, conservation science has increasingly promoted strategies such as buffer zones and ecological corridors, as highlighted by Bhagwat et al. (2008), Prevedello & Vieira (2010), and Haddad et al. (2015). These approaches can reduce the impacts of human activities around protected areas, provide supplementary habitat, enhance connectivity, and minimise edge effects. Heller & Zavaleta (2009), together with Schleicher et al. (2019), further demonstrated that such measures are most effective when implemented through landscape-scale planning and supported by local community participation. Furthermore, Hiley et al. (2013) showed that buffer zones can facilitate species range adjustments under changing climatic conditions by providing additional space for movement and colonisation.

2.4 The grassland biome

Grasslands are among the most widespread terrestrial vegetation types globally and are increasingly recognised for their ecological importance. Carlier et al. (2009) highlighted their multiple ecological functions, while Parr et al. (2014) described grassy biomes as globally extensive yet under recognised ecosystems. These systems are characterised by a dominant herbaceous layer composed primarily of grasses from the family Poaceae. Anderson (2006) identified grasses as the main structural component of grassland ecosystems, with Gibson (2009) and Bond (2019) further emphasising their role in maintaining ecosystem structure and function. Grasslands generally occur under intermediate environmental conditions where rainfall is insufficient to sustain continuous woody cover but adequate to prevent desert formation. Studies by Archibald et al. (2005) and Sankaran et al. (2005) demonstrated

that interactions among climate, fire, grazing, and rainfall are fundamental in maintaining open grassy ecosystems.

Occupying ~3.5 billion hectares (ha) worldwide, grasslands occur on every continent except Antarctica. Gibson (2009) and Blair et al. (2014) attributed this broad distribution to their adaptation to intermediate rainfall regimes and recurrent disturbances. Although regionally referred to as grasslands, pampas, prairies, or steppes, these ecosystems share similar ecological characteristics despite differing evolutionary histories (Gibson, 2009). Grasslands also support high levels of biodiversity and provide essential ecosystem services, including carbon storage, soil maintenance, livestock production, and the regulation of fire and herbivory, functions highlighted by Gibson (2009) and further emphasised by Bond (2019). Despite their ecological importance, grasslands have undergone extensive transformation through agricultural expansion, urbanisation, afforestation, industrial development, and altered disturbance regimes. Gibson (2009) identified these pressures as major drivers of grassland loss, while Veldman et al. (2015) and Sayre et al. (2020) demonstrated that many ancient grasslands remain poorly recognised and inadequately represented within protected-area networks. Research by Bender et al. (1998), Parr et al. (2014), and Veldman et al. (2015) further showed that substantial habitat loss has left many remaining grasslands highly fragmented and degraded. Such fragmentation disrupts ecological processes, reduces habitat quality, and contributes to population declines among grassland-dependent species, patterns demonstrated globally by Haddad et al. (2015) and within southern Africa by Carbutt & Kirkman (2022). Looking forward, Parr et al. (2014) identified continued land-use change as a major threat, while Petermann & Buzhdygan (2021) suggested that climate change will further alter grassland biodiversity, with particularly severe consequences for endemic and specialist species.

Increased anthropogenic activities have had detrimental effects on biodiversity and have caused widespread habitat loss for wildlife. Hunter & Gibbs (2007) identified habitat destruction as a major driver of global biodiversity decline, while Fuller & Perrin (2001) demonstrated that habitat loss negatively affects ecological processes and biodiversity conservation, particularly among indigenous plants and animal assemblages. These impacts are particularly relevant for small mammals, which can be sensitive to changes in habitat structure and connectivity. Small mammal species

are especially vulnerable to habitat fragmentation because of their limited dispersal ability, small body size, and specific habitat requirements. Fuller & Perrin (2001) showed that many small mammal species are closely associated with intact grassland habitats and respond negatively to disturbance. However, the effects of fragmentation are not necessarily uniform across species or landscapes. Fragmented landscapes associated with agriculture can sometimes create heterogeneous habitats that support increased diversity of rodents and shrews, particularly where edge habitats provide shelter and food resources. The importance of edge effects and habitat heterogeneity has been highlighted by Haddad et al. (2015), Pfeifer et al. (2017), and Bond (2019). In contrast, agricultural systems dominated by monocultures generally support lower small mammal diversity because they lack the structural complexity required by many species. Fuller & Perrin (2001) documented reduced habitat suitability in simplified agricultural landscapes, while Benedek & Sîrbu (2018) demonstrated that agricultural intensification reduces small mammal diversity and alters assemblage composition. Similar patterns were first observed in African rodent assemblages by Delany (1972) and later confirmed in South African landscapes by Lagesse & Thondhlana (2016).

Remnant grassland patches therefore act as important refugia for many species with narrow habitat requirements. Skinner & Chimimba (2005) and Monadjem et al. (2015) highlighted the dependence of numerous southern African small mammals on intact grassland ecosystems. Species such as the Southern African vlei rat (*Otomys irroratus*) and swamp musk shrew (*Crocidura mariquensis*) are closely associated with moist grassland and wetland habitats (Monadjem et al., 2015). Taylor (1998) documented the strong association of these species with mesic habitats, while Monadjem et al. (2015) highlighted the potential vulnerability of such habitat specialists to wetland degradation, drainage, and habitat loss. More recent studies (Umetsu & Pardini, 2007; Pardini et al., 2010; Fischer et al., 2011; Haddad et al., 2015; Fahrig, 2017) have shown that small mammal assemblages in fragmented landscapes are strongly influenced by landscape configuration, including habitat connectivity, patch size, and matrix quality. Fischer et al. (2011) demonstrated that both farming practices and landscape complexity influence small mammal assemblages in agricultural landscapes, with responses varying among species and landscape contexts. Fahrig (2017) and Haddad et al. (2015) showed that reduced connectivity and long-term

fragmentation contribute to declines in specialist species and changes in assemblage composition.

Grasslands are important centres of biodiversity and endemism, with White et al. (2000), Mucina & Rutherford (2006), and the South African National Biodiversity Institute (SANBI) (2013) highlighting their high species richness and ecological significance. In addition to supporting diverse floral and faunal assemblages, these ecosystems provide essential services such as carbon storage, water regulation, soil protection, grazing resources, and cultural benefits for many rural communities, as emphasised by SANBI (2013) and IPBES (2019). Within South Africa, Mucina & Rutherford (2006) and SANBI (2013) estimated that ~40-60% of the Grassland Biome has already been irreversibly transformed, largely through agricultural expansion, afforestation, urbanisation, and altered disturbance regimes. Formal protection of the biome remains limited, leaving many remaining grassland ecosystems under continued pressure. Nkambule et al. (2016) further demonstrated that the Ngongoni and KwaZulu-Natal Sandstone Sourveld (KZNSS) grassland landscapes are particularly vulnerable to land-use change and urban expansion, resulting in extensive habitat loss, fragmentation, and reduced connectivity among remnant patches.

Grasslands are classified into primary and secondary types according to their degree of modification. Mentis & Huntley (1982), O'Connor & Bredenkamp (1997), and Skowno et al. (2019) describe primary grasslands as relatively intact systems, whereas secondary grasslands develop following disturbance or previous land-use change. Primary grasslands generally retain greater biodiversity, ecological continuity, and ecosystem functioning because of their long-term persistence. Carbutt et al. (2011) and Veldman et al. (2015) showed that secondary grasslands often have altered species composition, vegetation structure, and soil characteristics. Although these grasslands may appear similar in vegetation cover, differences in ecological processes and resilience can occur, with degraded systems being more susceptible to invasive species, altered fire regimes, and nutrient changes as highlighted by D'Antonio & Vitousek (1992). Maintaining high-quality grasslands requires management approaches such as controlled burning, sustainable grazing, and preventing further conversion, which Fuhlendorf & Engle (2001) and Parr et al. (2014) identified as important for sustaining biodiversity and ecosystem functioning. The extensive transformation of grasslands also emphasises the need for effective

monitoring approaches using multiple biological indicators, as demonstrated by Avenant (2011), Parr et al. (2014), and IPBES (2019), to assess ecosystem condition and inform conservation management.

The identification of reliable indicator species is an important component of ecological monitoring because indicator species provide sensitive and cost-effective measures of habitat condition, ecological change, and responses to management interventions. Lindenmayer & Likens (2010) and McGeoch et al. (2011) highlighted the value of indicator species for detecting ecological responses across different environments. In grassland ecosystems, insect assemblages have been widely used as bio-indicators because of their high diversity, rapid generation times, and sensitivity to changes in vegetation structure, fire regimes, grazing intensity, and soil conditions, as demonstrated by McGeoch et al. (2011) and Landmann et al. (2023). However, Seaman & Louw (1999) identified limitations of early insect-based assessment approaches, including the Southern African Grassland Scoring System (SAGraSS), which required high sampling effort, specialist taxonomic knowledge, and regional calibration. More recent research by McGeoch et al. (2011) and Landmann et al. (2023) has supported integrated monitoring approaches that combine multiple species and functional traits to improve reliability across heterogeneous grassland systems. These approaches provide a broader understanding of ecological responses to land-use change and support adaptive conservation management, as emphasised by Lindenmayer & Likens (2010) and Parr et al. (2014).

2.5 Small mammals as biological indicators of ecological and environmental integrity.

Terrestrial small mammals are widely recognised as effective ecological indicators because their abundance, assemblage composition, and functional guild characteristics respond rapidly to changes in habitat condition, land-use intensity, and disturbance regimes as reported by Avenant (2011) and Osman et al. (2022). Avenant (2011) demonstrated the value of rodents and shrews for assessing ecosystem integrity in South African grasslands, showing that variation in assemblage structure reflects differences in habitat quality and ecological disturbance. Similar findings were reported by McCleery et al. (2018), who showed that land-use change alters small mammal assemblages, while Green et al. (2020) highlighted their usefulness in

monitoring ecological restoration and ecosystem recovery. Pearce & Venier (2005), Avenant (2011), Hope et al. (2017), Green et al. (2020), and Ayodele (2025) demonstrated that the short generation times, ecological versatility, and high sensitivity of small mammals to environmental change make them useful indicators of both immediate disturbance effects and longer-term ecosystem degradation and recovery.

A major strength of small mammals as ecological indicators lies in the ability to assess environmental responses through functional guilds and traits rather than species richness alone. Dietary guilds, including granivores, herbivores, insectivores, and omnivores, provide insight into how resource availability and habitat change influence assemblage structure as demonstrated by Fox (2011), Hurst et al. (2014), and Wilman et al. (2014). Wilman et al. (2014) highlighted the value of classifying species according to dietary and foraging traits, while Monadjem & Perrin (2003), Fox (2011), Kelt (2011), Hurst et al. (2014), and Ayodele (2025) demonstrated that variation in trophic guild composition and feeding strategies reflects environmental conditions and disturbance regimes. Functional guild ecology studies by Mouillot et al. (2011) and Gagic et al. (2015) further showed that changes in functional composition and diversity often provide greater insight into ecosystem processes than species richness alone. Consequently, trait-based approaches, including dietary guilds and body-size categories, provide an important framework for linking changes in small mammal assemblages with ecosystem functioning, resilience, and responses to environmental change as demonstrated by Wilman et al. (2014), Mouillot et al. (2011), Gagic et al. (2015), Dehling et al. (2016), and Paniccia et al. (2022).

Landscape structure and habitat modification strongly influence small mammal functional responses. Paniccia et al. (2022) showed that landscape composition and habitat configuration shape small mammal traits. Fuentes-Montemayor et al. (2019) demonstrated that both local habitat characteristics and landscape-scale factors influence small mammal responses to habitat creation and modification. In southern African grasslands, Avenant (2011) found that changes in guild structure correspond with differences in habitat integrity, while Monadjem & Perrin (2003) showed that variation in grassland resources and disturbance regimes affects assemblage composition. Disturbance responses are often context dependent; Avenant (2000), Gumbi et al. (2018), and Ramahlo et al. (2022) demonstrated that intermediate

disturbance can increase habitat heterogeneity and resource availability, whereas severe or prolonged disturbance reduces assemblage complexity and ecological functioning. Similarly, Loggins et al. (2019) showed that habitat modification can alter small mammal activity patterns and perceived predation risk.

Small mammals also contribute directly to ecosystem functioning through their roles as prey, seed consumers, seed dispersers, and ecosystem engineers. Avenant & Cavallini (2007) showed that small mammal assemblage characteristics can reflect ecological recovery following habitat restoration. Green et al. (2020) and Santos-Filho et al. (2024) linked small mammal diversity patterns with broader ecosystem processes. Changes in small mammal populations may therefore influence trophic interactions, vegetation dynamics, and soil processes. Recent studies continue to reinforce their value as indicators of ecological integrity, with Ancillotto et al. (2025) and Dondina et al. (2025) demonstrating that microhabitat conditions strongly influence small mammal assemblages, while Ayodele (2025) emphasised their application in assessing habitat health.

South Africa supports a diverse small mammal fauna, including 71 rodent species and 15 shrew species (SANBI, 2025). Research by Avenant (2000, 2003, 2011), Avenant & Cavallini (2007), Avenant et al. (2008), and de Klerk & Avenant (2017) has established small mammals as valuable indicators across South African grasslands, savannas, and other ecosystems. These studies demonstrated that assemblage composition, abundance, and functional structure respond to grazing pressure, habitat degradation, and restoration processes. Evidence from other regions supports these patterns, with Pearce & Venier (2005) reporting declines in forest-associated species following forest disturbance in the boreal forests of Ontario, Canada, and Umetsu & Pardini (2007) demonstrating that habitat fragmentation alters small mammal assemblages in the Atlantic Forest of São Paulo, Brazil. At broader scales, Clavel et al. (2011) and Newbold et al. (2015) showed that land-use intensification promotes functional homogenisation through declines in specialist species. These findings highlight the importance of small mammals as sensitive indicators for monitoring biodiversity change, evaluating ecological restoration, and assessing ecosystem resilience under increasing environmental pressures.

The municipality comprises several planning regions, notably the Inner West and Outer West regions. The Inner West includes Kloof and Hillcrest, where Krantzklloof Nature Reserve (KNR) and Giba Gorge Nature Reserve (GGNR) are situated, while the Outer West includes Cato Ridge and Camperdown Rural, where the reference site, Doornrug & Uitkomst Conservancy (DUC), is located (Figure 3.2B).

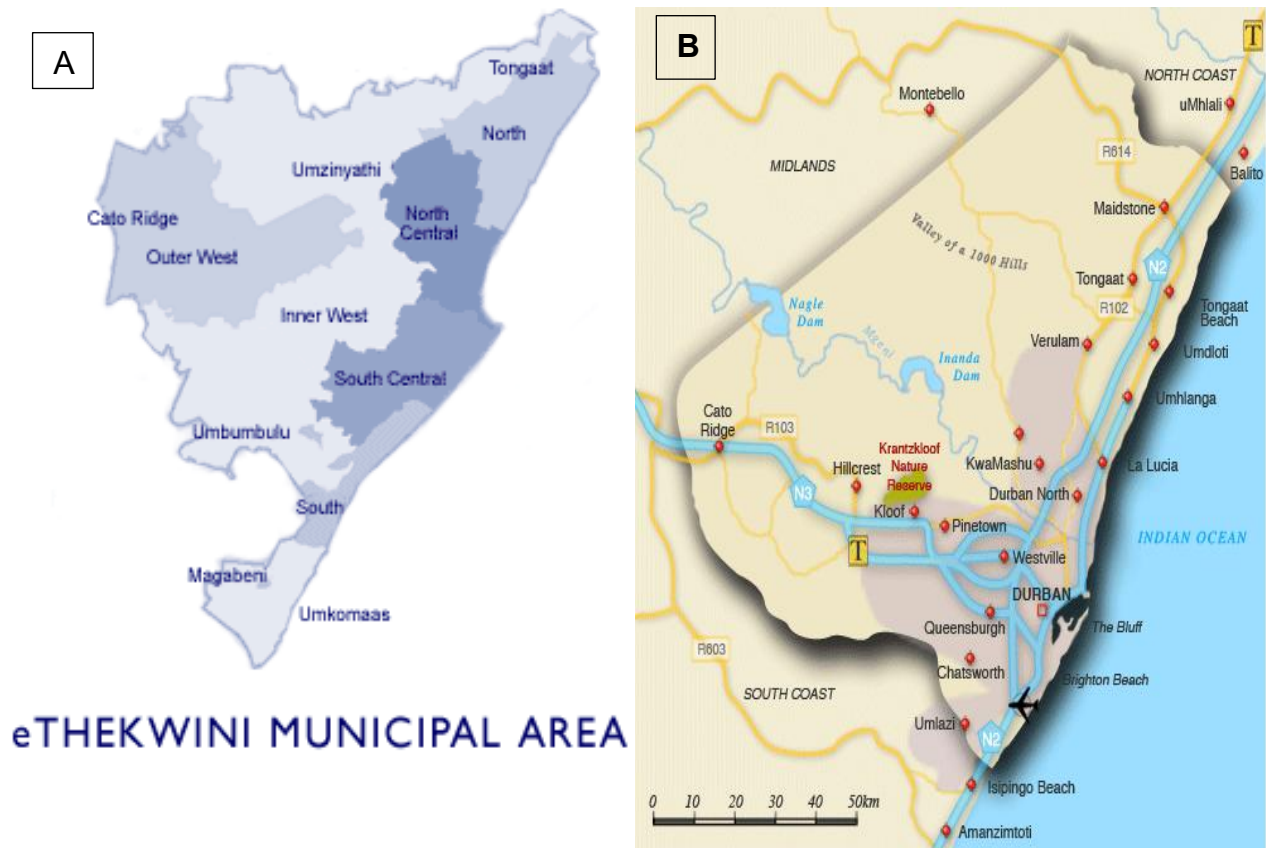


Figure 3.2. A. The eThekweni Municipal Area showing the Inner and Outer West regions included in the study (Google Maps, 2025). B. The study regions Kooof, Hillcrest and Camperdown, which are located within eThekweni, (Google Maps, 2025).

The EMA contains ~99 728 hectares (ha) of the original extent of the KZN Coastal Belt vegetation type, representing ~25% of its historical extent, as reported by McLean et al. (2016) based on the vegetation classification of Mucina & Rutherford (2006). The EMM has an estimated population of ~3.9-.2 million people (Statistics South Africa, 2022), with an annual growth rate of ~1.3% in 2023 (eThekweni Municipality, 2025). Despite continued economic growth, the municipality experiences substantial socio-economic challenges, with a Gini coefficient of ~0.60-0.61, reflecting persistent income inequality. Furthermore, an estimated of ~1.35 million residents live below the food

poverty line, highlighting widespread food insecurity and socio-economic vulnerability within the municipality (eThekweni Municipality, 2025).

Much of the EMA is characterised by rural and peri-urban land uses, including extensive areas of Zulu tribal land governed by traditional authorities (eThekweni Municipality, 2013). The incorporation of these areas into the municipal boundaries following the 2 000 local government restructuring resulted in a dual governance system in which traditional leadership structures coexist with municipal democratic governance (Sutherland et al., 2016). Mucina & Rutherford (2006) described the EMA as encompassing a broad environmental gradient, with vegetation patterns reflecting climatic transitions from humid subtropical conditions along the coast to increasingly temperate conditions at higher elevations. Similarly, Mushore et al. (2026), in their assessment of local climate zones and land surface temperature patterns within the municipality, demonstrated considerable spatial climatic heterogeneity across the EMA. In general, higher elevations are generally characterised by greater rainfall, stronger temperature seasonality, and lower mean annual temperatures. Lower elevations approaching sea level are characterised by warmer climatic conditions and reduced temperature seasonality, partly due to the moderating influence of the adjacent Indian Ocean. This pronounced climatic gradient contributes to the environmental heterogeneity that characterises the municipality. Together with variations in soils, lithology, physiography, and biogeographical position, these factors have produced a diverse landscape supporting a wide range of terrestrial and aquatic ecosystems and vegetation types. This includes tropical and subtropical assemblages, as well as temperate mist-belt ecosystems, as described by Mucina & Rutherford (2006) and further highlighted by Ground et al. (2016) and Rouget et al. (2016).

The EMA lies within the Maputaland-Pondoland-Albany (MPA) biodiversity hotspot, a globally recognised centre of biodiversity situated along the eastern seaboard of southern Africa. Van Wyk & Smith (2001) documented the exceptional floristic richness and endemism of the region, while McLean et al. (2016) highlighted the importance of the MPA hotspot within eThekweni, noting that it supports >~7 000 vascular plant species, ~25% of which are endemic. Despite this exceptional biodiversity value, rapid urbanisation, increasing service delivery demands, poverty, and unauthorised development have intensified pressures on the municipality's natural environment. Roberts (2008) highlighted that development priorities have frequently

superseded environmental concerns, while recent municipal biodiversity assessments continue to identify habitat transformation, fragmentation, and anthropogenic disturbance as major threats to biodiversity persistence within the EMA (eThekweni Municipality, 2025).

3.2. The Durban Metropolitan Open Space System (D'MOSS).

The EMM has instituted the Durban Metropolitan Open Space System (D'MOSS), a spatial planning and conservation framework established to conserve open spaces within the EMA, as described by Ground et al. (2016), McLean et al. (2016), Bux et al. (2021), and McLean et al. (2024). The concept of D'MOSS originated in the early 1980s in response to increasing urbanisation pressures and was first formally adopted by the former eThekweni City Council in 1989, as documented by Roberts (1994), and more recently by McLean et al. (2024). McLean et al. (2025b). The eThekweni Municipality (2023) reported that D'MOSS currently covers ~95 000 ha (Figure 3.3) of interconnected open space, representing about one-third of the metropolitan area and incorporating a range of high-value ecosystems, including grasslands, forests, wetlands, and estuarine environments.

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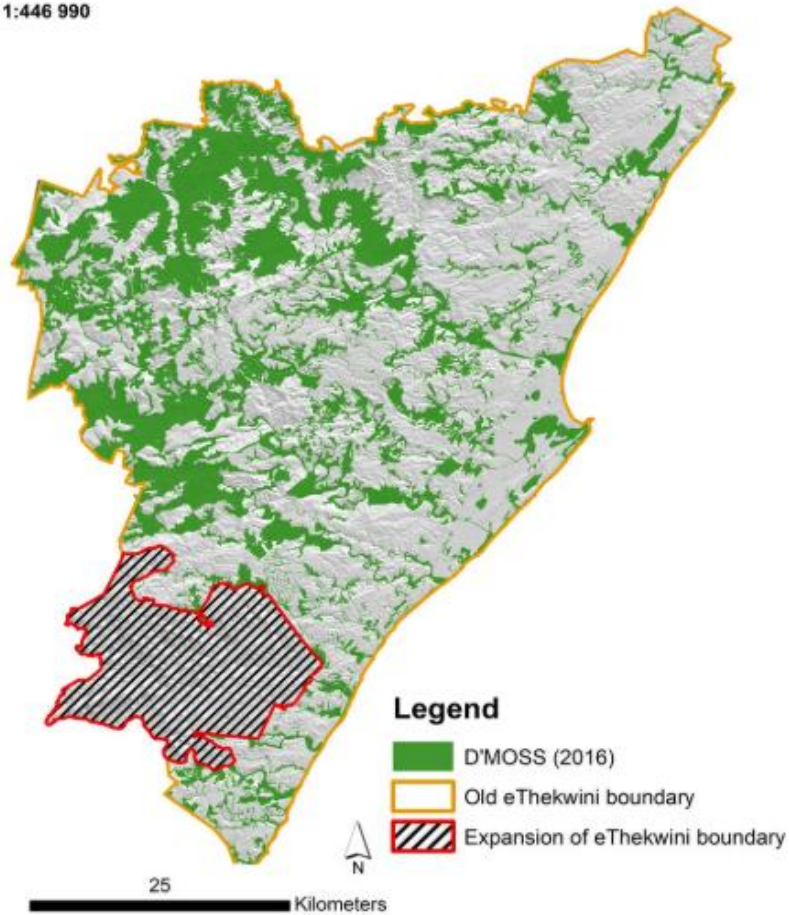


Figure 3.3. The current extent of Durban Metropolitan Open Space System (D'MOSS) (Bux et al., 2021).

McLean et al. (2024) noted that D'MOSS was formally integrated into municipal planning instruments in 2010 when it was adopted as a policy layer and incorporated into eThekweni's land-use and town planning schemes. Subsequent amendments during the 2017/2018 planning cycle updated its spatial extent and strategic role within municipal planning frameworks, as further reported by the eThekweni Municipality (2023) and McLean et al. (2025b). The eThekweni Municipality (2023), together with McLean et al. (2025a, 2025b), indicated that D'MOSS is now embedded within the City's Integrated Development Plan, Spatial Development Frameworks, and regional planning layers as a controlled biodiversity layer that guides development and environmental management to conserve critical natural habitats across the metropolitan area. Although D'MOSS covers ~95 000 ha of interconnected open space, only a small proportion is formally protected. The eThekweni Municipality (2025)

reported that ~7.25% of the D'MOSS area is subject to some level of legal protection, representing ~2.70% of the EMA. Ground et al. (2016), McLean et al. (2016), and McLean et al. (2025b) described these areas as comprising proclaimed and unproclaimed public and private protected areas, rural landscapes within upper river catchments, and riverine and coastal corridors linked through interconnected open spaces of high biodiversity value. Mucina & Rutherford (2006) highlighted that D'MOSS encompasses a diverse range of ecosystems, including dune, coastal and scarp forests; wetlands such as floodplains, mangroves, swamp forests, and reedbeds; grasslands including the threatened Ngongoni and KwaZulu-Natal Sandstone Sourveld (KZNSS) grasslands and associated woodland assemblages.

Roberts et al. (2012) and Boon et al. (2016) explained that although D'MOSS was initially established to prioritise the conservation of threatened species and habitats, its scope has progressively expanded to include the maintenance of ecosystem services within the urban landscape. Boon et al. (2016) and Dlamini (2021), further noted that increasing urbanisation, population growth, and changing land-use demands since the early 1990s have accelerated the transformation of open spaces and necessitated new approaches to environmental and land-use planning. In response, the EMM has increasingly recognised the important environmental benefits and ecosystem services provided by urban open spaces, which offer unique opportunities for conserving threatened ecosystems and species, as emphasised by Boon et al. (2016) and McLean et al. (2016). Ground et al. (2016), McLean et al. (2024), and Roberts et al. (2002) reported that improvements in conservation planning have been facilitated through the adoption of new tools and approaches, including systematic conservation assessment (SCA) and resource economics, which have enhanced conservation prioritisation and improved the scope and accuracy of the D'MOSS plan. More recently, Mugwedi et al. (2018) and McLean et al. (2025a) highlighted that growing concerns regarding climate change impacts, ecosystem resilience, and ecological restoration have further strengthened the implementation and strategic importance of the conservation network.

3.3 The KwaZulu-Natal Sandstone Sourveld (KZNSS) Grassland Vegetation.

Boon et al. (2016) reported that the KZNSS landscape is one of the nine vegetation types occurring within the EMM. Mucina & Rutherford (2006) first described and classified the KZNSS landscape as a component of the Savanna Biome. Boon et al. (2016) indicated that the KZNSS landscape is widely distributed from Maphumulo in the north to Port Shepstone in the south, including the municipal areas of Kloof, Hillcrest, Table Mountain, and Umbumbulu. Jewitt et al. (2015) estimated that the vegetation type covers ~179 671 ha, while Mucina & Rutherford (2006) reported that it occurs across an elevational range of ~500-1 100 m above sea level. The ecological importance of the KZNSS landscape has been highlighted by the eThekweni Municipality (2019), Drury (2016), and Boon et al. (2016), who recognised its indigenous nature, high levels of endemism, and provision of important ecosystem services, including soil formation, erosion control, carbon sequestration, and recreational opportunities. Mucina & Rutherford (2006) described this vegetation type as a short, species-rich grassland dominated by a matrix of grasses, with scattered shrubs, legumes, and occasional woody plants adapted to nutrient-poor sandstone-derived soils. Drury (2016), and Nkambule et al. (2016) further highlighted the high floristic diversity and ecological complexity of the vegetation.

Mucina & Rutherford (2006) identified dominant grasses within the KZNSS landscape as including red oat grass (*Themeda triandra*), three-awn grasses (*Aristida spp.*), and bog-rush grasses (*Schoenus spp.*). Hlanguza (2015) and Drury (2016) documented the occurrence of sandstone specialist forbs and legumes. Low shrubs and geoxyllic suffrutices, including members of the Proteaceae such as *Protea spp.* and *Leucospermum spp.* contribute to the structural diversity of the vegetation, whereas occasional trees such as *Faurea spp.* occur within remnant patches. Mucina & Rutherford (2006), Boon et al. (2016) and Naicker et al. (2016), indicated that the vegetation type has undergone extensive transformation and degradation and is consequently classified as Critically Endangered. Boon et al. (2016) further noted that the KZNSS landscape has received considerable conservation attention from the Biodiversity Management Division (BMD) of the EMM. Naicker et al. (2016) demonstrated that low levels of formal protection and increasing urban development continue to constrain conservation efforts. Hlanguza (2015), Drury (2016), Ground et al. (2016), Naicker et al. (2016), Nkambule et al. (2016), and Boon et al. (2016) all

emphasised that relatively little ecological research has been undertaken on the KZNSS landscape. Drury (2016), Boon et al. (2016), and Naicker et al. (2016) reported that ~90% of the original extent of the KZNSS landscape has been transformed or degraded through urbanisation, agricultural cultivation, invasion by bracken fern (*Pteridium aquilinum*), and the establishment of sugarcane (*Saccharum officinarum*) and timber plantations (*Pinus* and *Eucalyptus* spp.). Consequently, only a very small fraction of the KZNSS landscape remains under statutory protection. Mucina & Rutherford (2006), Drury (2016), and Boon et al. (2016) indicated that the Critically Endangered KZNSS landscape has undergone substantial transformation since the early 1990s owing to altered fire and grazing regimes, nutrient enrichment, and land-use change. As a result, only a small fraction of unaltered KZNSS landscape remains (Figure 3.4). These authors further noted that much of the original grassland has been replaced by woody vegetation and other transformed land covers in response to changes in fire dynamics and excessive grazing.

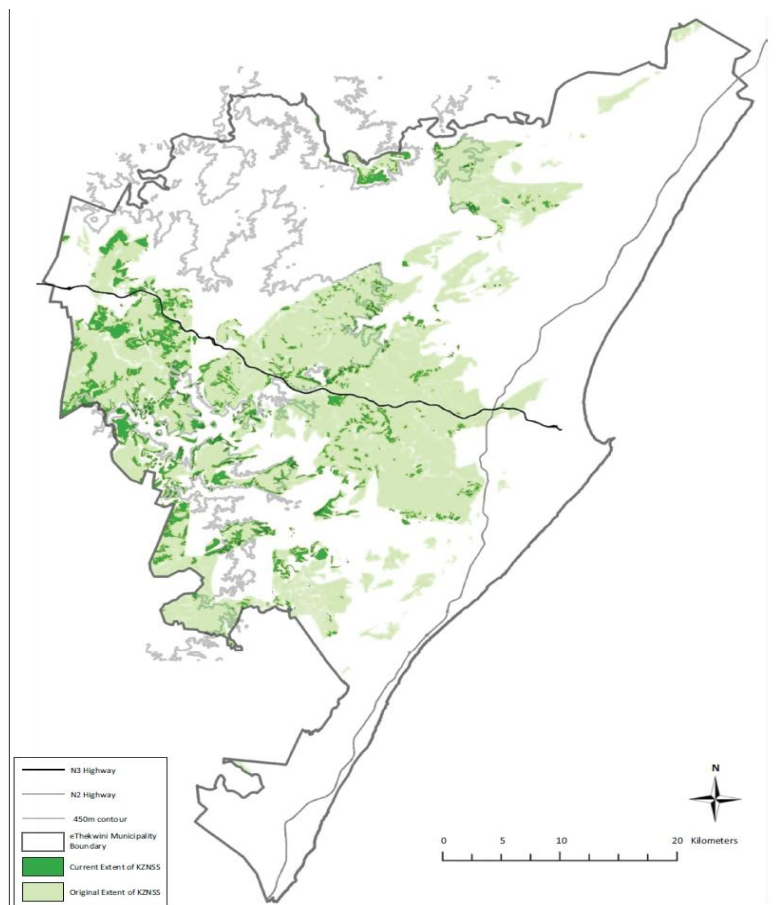


Figure 3.4. The original and current extent of KwaZulu-Natal Sandstone Sourveld in the eThekweni Municipal Area (Boon et al, 2016).

3.4 Krantzkloof Nature Reserve

Seaman (2006) and Ezemvelo KZN Wildlife (2023) described Krantzkloof Nature Reserve (KNR; -29.764303; 30.850883), which was formally proclaimed as a protected area in 1950, as comprising deeply incised gorges carved into the ancient Kloof Plateau by the Molweni and Nkutu Rivers, both tributaries of the uMngeni River, which drains into the Indian Ocean at Blue Lagoon. The protected area's geology is dominated by granite basement, Natal Sandstone, and intrusive dolerites. Granite is particularly evident along the Molweni Riverbed, where large boulders composed of quartz, feldspar, and biotite mica create striking river scenery. Seaman (2006) further described Natal Sandstone, deposited ~485-500 million years ago during the late Cambrian, as forming the steep gorge cliffs, while weathered dolerite intrusions have produced deep fissures, including the Nkonka Crack along the Forest Trail. Seaman

(2006) described the Molweni River as the protected area's principal watercourse, originating from Springside Nature Reserve ~4 km away and flowing through Everton Gorge before descending over Kloof Falls, the highest waterfall in the protected area, and eventually joining the uMngeni River near Marble Arches. Seaman (2006) and Ezemvelo KZN Wildlife (2023) further indicated that the river system comprises seven tributaries, including the Nkutu River and perennial streams such as the iNgungumbane, Mpithi, Ronald's Kloof, Uve, iNgolube, and Nkonka streams, collectively supporting nine major waterfalls within the protected area. Seaman (2006) and Boon & Croucamp (2002), described the reserve as encompassing a diverse mosaic of habitats ranging from coastal scarp and riverine forests to the highly threatened KZNSS landscape (Figure 3.5).

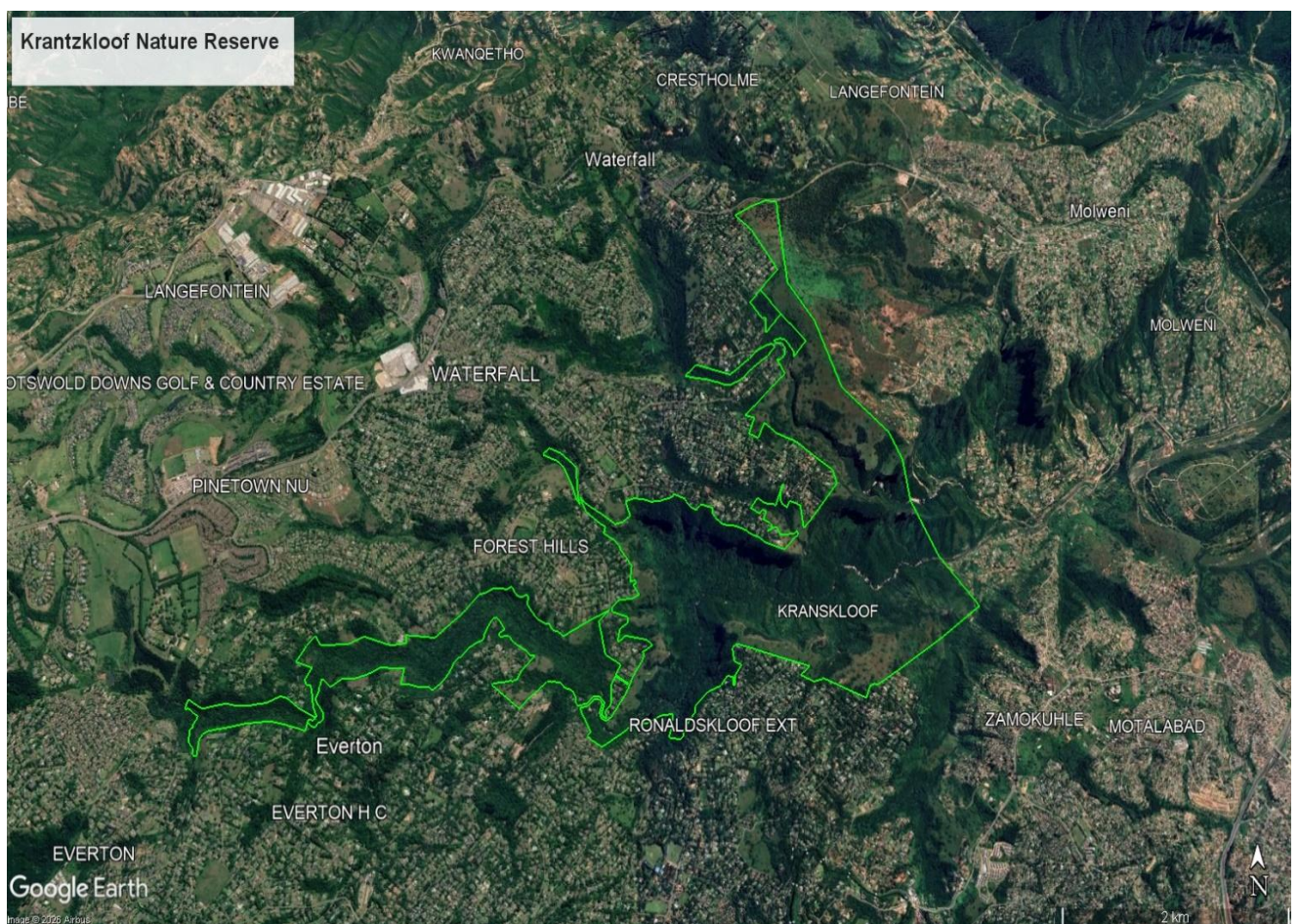


Figure 3.5. Satellite imagery showing the urbanised surroundings of Krantzklloof Nature Reserve, including the gorge system containing KwaZulu-Natal Sandstone Sourveld landscape (July 2018-June 2019; Google Earth, 2019).

Seaman (2006) further documented that the steep terrain and numerous inaccessible areas have contributed to the preservation of high biodiversity, with the protected area supporting more than ~50 mammal species, ~253 bird species, ~35 reptile species, ~150 butterfly species, and over ~1 500 plant species, including ~ 273 tree species. Seaman (2006) and van Niekerk (2010) documented several endemic and rare species within the protected area, including the Natal Sandstone-quince (*Dahlgrenodendron natalense*), *Brachystelma natalense*, *Streptocarpus molweniensis*, and the Ruby-footed Black Millipede (*Doratonagus rubipodus*). These records highlighted the importance of the protected area as a breeding habitat for several raptor species, including Crowned Eagles, Wahlberg's Eagles, Lanner Falcons, and Peregrine Falcons, further underscoring its conservation significance. Taylor (1991) noted that the small mammal fauna of KNR comprises species associated with both grassland and woodland habitats. This mixed fauna is consistent with the protected area's location within a transitional zone between the Coast Lowland and Mistbelt regions, as described by Mucina & Rutherford (2006). Historical records and previous surveys undertaken by the Durban Natural Science Museum further indicate that KNR supports a rich diversity of small terrestrial mammals, including rodents, shrews, and golden moles. Notable records include the southern vlei rat (*Otomys unisulcatus*) and the robust golden mole (*Amblysomus robustus*), both of which are of particular ecological interest owing to their specialised habitat requirements and relatively restricted distributions. Rautenbach et al. (2008) additionally documented species such as *Crocidura flavescens*, *Mastomys natalensis*, and *Lemniscomys rosalia* within the protected area. These records demonstrate that KNR supports both widespread generalist species, which can occupy a broad range of habitats and tolerate environmental disturbance, and habitat-specialist species, which possess more specific ecological requirements and are often restricted to particular forest, woodland, or grassland environments in KZN. Seaman (2006) highlighted that the protected is actively managed to conserve biodiversity and ecosystem services, with invasive species control, trail and infrastructure maintenance, and collaboration with local conservancies identified as key management priorities. Van Niekerk (2010) further recognised KNR as one of eThekweni's premier nature-based recreational destinations, offering extensive trail networks, environmental education programmes, and ecotourism opportunities. Ezemvelo KZN Wildlife (2023) additionally noted that visitor engagement is enhanced

through the installation of Bio-Guide interpretive plates along trails, enabling smartphone users to access detailed information on significant species and ecological features. The majority of KNR is covered by scarp forest with the lower lying areas supporting patches of grassland.

3.5 Giba Gorge Nature Reserve

Giba Gorge Nature Reserve (GGNR; 29.805065° S, 30.777396° E), formerly known as the Giba Gorge Environmental Precinct (GGEP), was established in 2009 through a collaborative initiative between private landowners and the EMM under a Special Rating Area (SRA) model. It was subsequently formally proclaimed as a protected area in 2018 under the National Environmental Management: Protected Areas Act (Act 57 of 2003). Chinzila (2015) described the initiative as aiming to conserve scarp forests, wetlands, cliff habitats, and endangered grasslands through coordinated biodiversity stewardship and structured management. The protected area was subsequently proclaimed under the National Environmental Management: Protected Areas Act (Act 57 of 2003) and has since been incorporated into the D'MOSS. The South African National Roads Agency SOC Ltd (SANRAL) and African Centre for Environmental Research (ACER) (2020) described GGNR as extending along a deeply incised gorge system forming part of the upper catchment of the uMhlatuzana River. The protected area lies within a transitional zone between the Indian Ocean Coastal Belt and Sub-Escarpment Savanna biomes, contributing to substantial environmental heterogeneity. The geology is dominated by Natal Group Sandstone overlying older granitic substrates, with prolonged erosional processes producing steep slopes, cliffs, and valley systems that create diverse microhabitats (Figure 3.6).

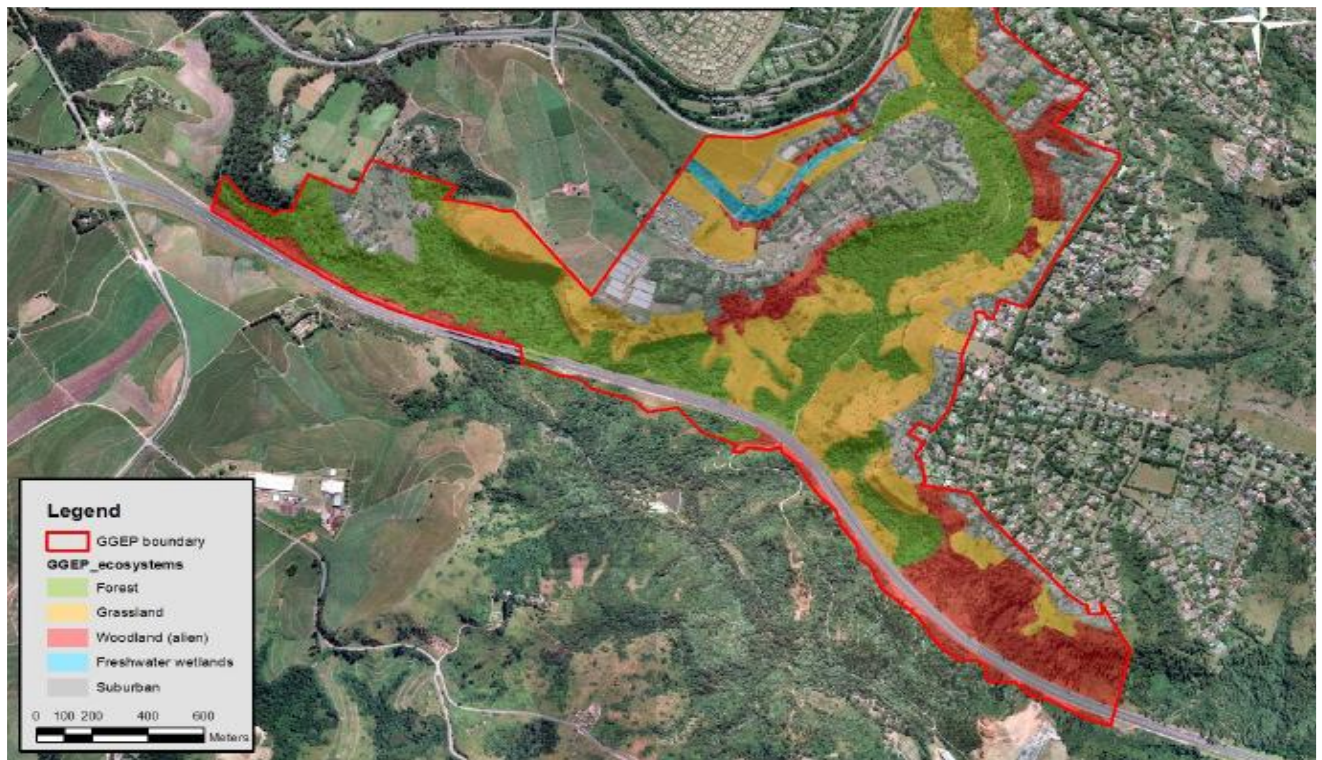


Figure 3.6. Satellite imagery of Giba Gorge Nature Reserve, formerly known as Giba Gorge Environmental Precinct, showing the reserve boundary, dominant habitat types, and the highly urbanised landscape surrounding the gorge system during the study period (July 2018-June 2019). Adapted from the Giba Gorge Nature Reserve Conservation Management Plan (2016/2017).

Vegetation within GGNR is highly heterogeneous and comprises endangered KZNSS landscape interspersed with scarp and riverine forests along valley floors and drainage lines. Drury (2016) highlighted that remnant KZNSS grassland patches within the protected area support high plant species richness and contribute significantly to regional biodiversity despite extensive historical transformation and fragmentation associated with urban expansion and agriculture. Boon & Croucamp (2002), Mucina & Rutherford (2006), and the Giba Gorge Environmental Precinct Conservation Management Plan (2016/2017) recognised five principal vegetation types within the protected area, namely KZNSS landscape, mixed woodland, wetlands, scarp forest, and sandstone cliffs, with scarp forest comprising the dominant vegetation type. Hydrologically, the Giba and uMhlatuzana rivers and their tributaries traverse the protected area and sustain extensive riparian habitats, thereby enhancing ecological complexity (SANRAL/ACER, 2020). The protected area experiences a subtropical climate characterised by warm, humid summers and mild, relatively dry winters, with rainfall concentrated during late spring and summer and a distinct dry period during

mid-winter. These climatic conditions influence vegetation dynamics, ecological processes, and recreational use patterns. Chinzila (2015) further highlighted the SRA model as an innovative biodiversity stewardship mechanism that facilitates cooperation between private landowners and local government through dedicated levies supporting invasive alien plant control, fire management, erosion mitigation, ranger patrols, and environmental education initiatives. Empirical assessments suggested that this management framework has contributed to improved ecological condition relative to unmanaged remnants. The combination of KZNSS landscape, forested gorges, and riparian habitats supports a rich diversity of fauna. Butterfly Gear (2025) reported that more than 100 butterfly species occur within the protected area, reflecting its high biodiversity value. Historical records housed at the Durban Natural Science Museum further document numerous small mammal species, including *Graphiurus murinus*, *Mastomys natalensis*, *Mus minutoides*, *Otomys auratus* (formerly known as *Otomys irroratus*), *Peomys melanotis*, *Rhabdomys dilectus*, and several shrew species. Kloof Conservancy (2025) additionally reported the occurrence of the endangered ruby-footed black millipede (*Doratonagus rubipodus*), further emphasising the conservation significance of the reserve.

3.6 Doornrug and Uitkomst Conservancy (Camperdown Rural)

Doornrug and Uitkomst Conservancy (DUC; -29.75294° S, 30.61193° E) is a privately owned rural conservancy situated near Camperdown Rural in KZN, north-west of the EMA and approximately midway between eThekweni and Pietermaritzburg. Unlike KNR and GGNR, DUC is not a formally proclaimed protected area but rather a voluntary conservation initiative comprising privately owned properties managed cooperatively for biodiversity conservation within a predominantly agricultural landscape. ShowMe South Africa (n.d.) and Mapcarta (n.d.) described the area as part of the Natal Midlands mistbelt, characterised by rolling agricultural landscapes and proximity to conservation areas such as Natal Lion Park and Tala Game Reserve. The region experiences a subtropical climate with warm, wet summers and cooler, drier winters. South African Explorer (n.d.) reported mean temperatures ranging from ~20.6 °C in winter to ~26.7 °C in summer and an average annual rainfall of ~580 mm, with most precipitation occurring during mid-summer. The conservancy occurs at elevations ranging from ~450 to 700 m above sea level, producing gently undulating

terrain and moderate relief relative to the higher elevations of the KZN Midlands. The DUC comprises a heterogeneous mosaic of habitats, including KZNSS landscape, eastern scarp forest, eastern valley bushveld, transitional forest, woodland, thicket, and wetlands. Compared with the more urbanised and fragmented protected areas of KNR and GGNR, the DUC study site is predominantly characterised by KZNSS vegetation and experiences lower levels of urban disturbance and habitat fragmentation. Consequently, DUC was selected as a reference site against which patterns of small mammal species richness and abundance in the more urbanised and fragmented protected areas could be evaluated (Figure 3.7).

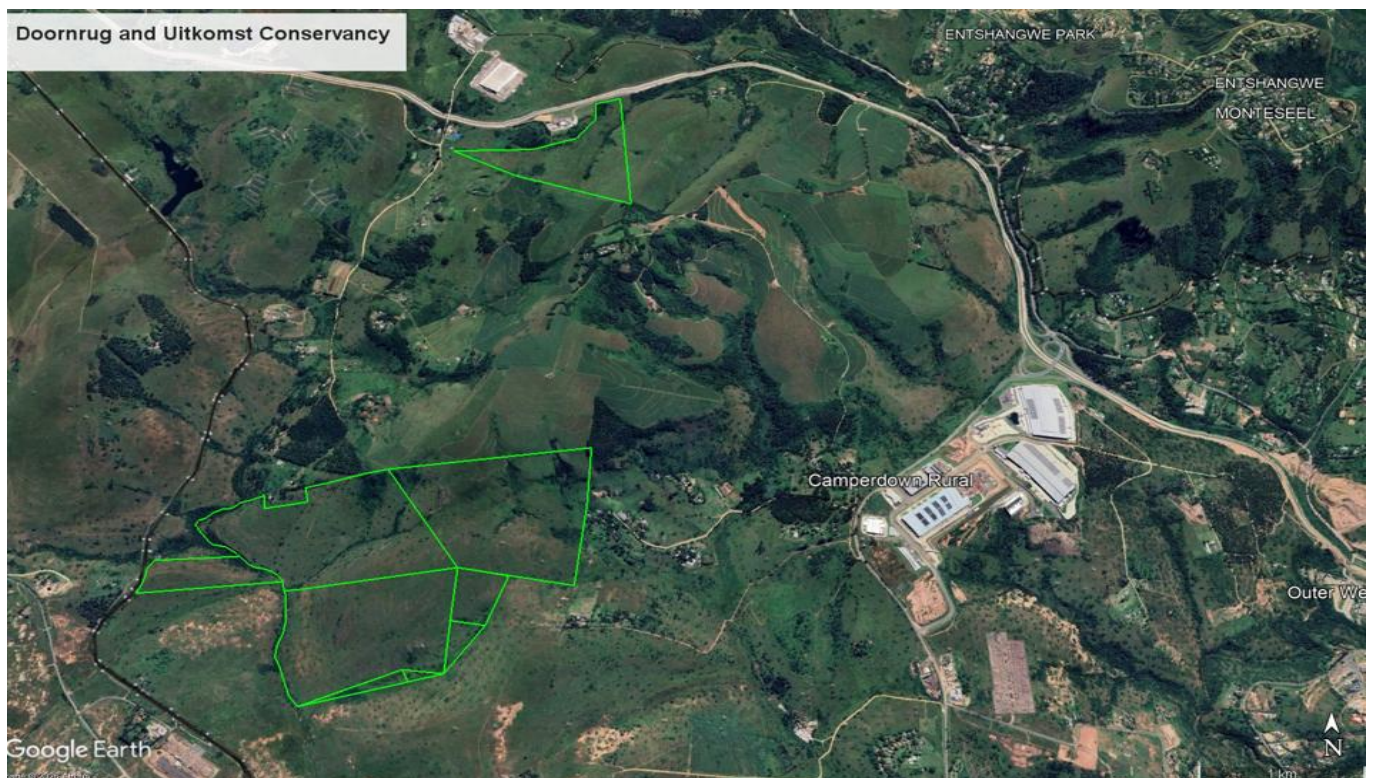


Figure 3.7. Satellite imagery of the KwaZulu-Natal Sandstone Sourveld grassland landscape at the Doornrug & Uitkomst Conservancy (Camperdown) study site, showing the relatively intact KwaZulu-Natal Sandstone Sourveld landscape structure and lower surrounding urban development observed during the study period (July 2018-June 2019; Google Earth, 2019).

The conservancy encompasses multiple land parcels on the historic Uitkomst and Doornrug farms (No. 852). eThekweni Municipality (2024) indicated that individual cadastral portions range from only a few hectares to > ~900 ha, collectively supporting extensive grassland, woodland, and riparian habitats that contribute to regional

biodiversity and landscape connectivity. As a recently proclaimed conservancy, little ecological information has been published for the site, and the present study represents the first formal ecological assessment undertaken within the area. Based on regional vegetation and faunal associations, DUC is likely to support characteristic mistbelt grassland and forest flora, including grasses such as *Themeda triandra* and a diverse assemblage of indigenous forbs, shrubs, and forest trees, including *Newtonia buchananii* and *Ficus* species. The habitat mosaic is also likely to support a range of small mammals typical of KZN Midlands grassland and forest systems, including *Myosorex varius*, *Crocidura flavescens*, *Lemniscomys rosalia*, *Mus minutoides*, *Aethomys ineptus* (the name *Aethomys ineptus* is used throughout this study, despite its treatment as a synonym of *A. chrysophilus* by Russo & da Silva (2025), *Rhabdomys dilectus*, and *Mastomys natalensis*. The occurrence of these species across open grassland, woodland, and riparian habitats highlights the ecological integrity and habitat heterogeneity of the conservancy.

CHAPTER 4

MATERIALS AND METHODS

4.1 Study area and site selection.

The study was conducted across three regions in KwaZulu-Natal (KZN), South Africa, including Kloof, Hillcrest, and Camperdown Rural, all of which harbour remnant patches of KwaZulu-Natal Sandstone Sourveld (KZNSS) grassland. Mucina & Rutherford (2006) described KZNSS landscape as a threatened grassland ecosystem that has undergone extensive transformation and fragmentation, while Jewitt et al. (2015) emphasised the conservation importance of its remaining fragments within a highly transformed landscape. The selected regions span a local elevational gradient of ~400-1100 m above sea level and differ in land-use intensity, habitat configuration, and conservation management practices, providing an appropriate framework for examining the influence of environmental and anthropogenic drivers on small mammal assemblages. The Kloof region is represented by Krantzkloof Nature Reserve (KNR), which occurs at lower elevations within an urban and peri-urban matrix. Hillcrest is represented by Giba Gorge Nature Reserve (GGNR), located at intermediate elevations and characterised by a mosaic of residential development and remnant natural vegetation. Camperdown Rural (reference site) is represented by the Doornrug & Uitkomst Conservancy (DUC), which occurs at higher elevations within a predominantly rural landscape experiencing comparatively lower levels of urban pressure. These sites were selected because they collectively represent the remaining distribution of KZNSS grassland across the eThekweni region and encompass variation in habitat condition, management regimes, and degrees of anthropogenic disturbance, providing an opportunity to assess small mammal responses to both natural environmental gradients and human-induced habitat modification.

4.2 Environmental variables influencing small mammal dynamics.

For this study, environmental variables were considered important determinants of small mammal assemblage structure because they influence microclimatic conditions, vegetation composition, habitat heterogeneity, and resource availability. Lomolino (2001) and McCain (2005) demonstrated that these factors shape species distributions along environmental gradients, while Tews et al. (2004) highlighted the importance of

habitat heterogeneity in maintaining biodiversity. Similarly, Monadjem et al. (2015) showed that environmental variation influences the abundance, distribution, and community composition of small mammals.

4.2.1 Elevational gradients

Sampling was conducted along a local elevational gradient ranging from ~400-1 100 m above sea level. Elevation for each transect was recorded using Google Earth Pro. Potere (2008) demonstrated that Google Earth's high-resolution imagery possesses high positional accuracy. Benker et al. (2011) showed that Google Earth-derived terrain models provide reliable topographic information for scientific applications. The utility of Google Earth for environmental and ecological assessments has also been highlighted by Large & Gilvear (2015). Elevation was included as a key environmental variable because elevational gradients influence microclimatic conditions, vegetation structure, species distributions, and assemblage composition, as demonstrated by Lomolino (2001), and McCain (2007). In southern Africa, Rowe-Rowe & Meester (1981) further demonstrated the importance of elevation in structuring small mammal distributions, while Bateman et al. (2010) reported similar relationships between elevation and small mammal assemblages. For analytical purposes, study sites were grouped into lower (401-500 m), intermediate (501-600 m), and higher (>600-700 m) elevation categories. Although the elevational range examined in this study was relatively modest, lower elevations are generally associated with greater anthropogenic influence owing to their accessibility and proximity to urban and peri-urban areas, whereas higher elevations may experience slightly cooler and more mesic conditions. As noted by Körner (2007), such elevational differences can contribute to environmental heterogeneity and influence species distributions. The inclusion of elevation therefore enabled an assessment of how local environmental variation influenced small mammal assemblages across remnant KZNSS grasslands.

4.2.2 Slope aspect

Slope aspect was determined using the terrain visualisation functionality in Google Earth Pro and classified into eight directional categories (north, north-east, east, south-east, south, south-west, west, and north-west). Benker et al. (2011) demonstrated that Google Earth-derived terrain data provide sufficiently accurate topographic

information for landscape analyses. Large & Gilvear (2015) highlighted the utility of Google Earth as a platform for environmental and ecological assessments. The terrain functionality provides three-dimensional representations of the landscape, facilitating the interpretation of slope orientation and local topographic patterns. Bennie et al. (2008) demonstrated that variation in slope orientation influences solar radiation exposure, thereby creating differences in local temperature, soil moisture, and vegetation structure. In the Southern Hemisphere, north-facing slopes generally receive greater solar radiation and experience warmer and drier conditions, whereas south-facing slopes receive less direct solar radiation and are typically cooler and moister. Similarly, Måren et al. (2015) showed that slope aspect influences environmental conditions, including soil properties and vegetation characteristics, while Elnaker & Zaleski (2021) reported that slope orientation affects soil temperature and water availability. Qin et al. (2019) and Yin et al. (2023) further demonstrated that aspect-driven microclimatic variation can influence plant assemblage composition and vegetation productivity. These topographically mediated differences in microclimate and vegetation structure may subsequently influence small mammal assemblages by affecting habitat suitability, resource availability, and shelter conditions. Chirichella et al. (2022) demonstrated that topographic and microclimatic factors contribute to variation in small mammal assemblage structure, while Chidodo et al. (2020) identified environmental variables as important drivers of rodent assemblage composition. Despite its potential ecological importance, slope aspect remains relatively understudied as a driver of small mammal assemblage patterns in South African grasslands, particularly within the KZNSS landscape. Consequently, slope aspect was incorporated into this study to assess its potential influence on small mammal assemblages within this threatened grassland ecosystem.

4.2.3. Seasonal variation

Seasonal variation was assessed by classifying sampling periods according to the climatic seasons of KZN, categorised into summer (December-February), autumn (March-May), winter (June-August), and spring (September-November), reflecting the seasonal climatic patterns of the summer-rainfall region of South Africa. The distinction between wet (spring and summer) and dry (autumn and winter) periods reflects the summer-rainfall climatic regime characteristic of eastern South Africa, as described by

Tyson & Preston-Whyte (2000) and illustrated in the agroclimatic analyses of Schulze (1997b). Seasonal fluctuations in rainfall and temperature influence hydrological processes and vegetation productivity across South African landscapes (Schulze, 1997a). These seasonal changes can alter habitat conditions and resource availability, which in turn influence small mammal abundance, activity patterns, and assemblage composition (Monadjem & Perrin, 2003). Studies in southern Africa have demonstrated that seasonal variation influences small mammal capture frequencies, species richness, diversity, and assemblage structure (Ramahlo et al., 2022; Shilereyo et al., 2023). Ferrari et al. (2022) further highlighted the importance of seasonal and environmental factors in shaping small mammal distributions across ecological gradients, while Jolly et al. (2024) demonstrated interactions between climatic conditions, resource availability, and population dynamics. Because KZNSS landscapes occur within the summer-rainfall region of eastern South Africa, they experience distinct wet and dry periods that influence vegetation dynamics and ecosystem processes. Mucina & Rutherford (2006) describe KZNSS landscapes as grassland ecosystem shaped by regional climatic conditions, while Tyson & Preston-Whyte (2000) highlight the strong seasonality of rainfall and temperature across southern Africa. For this study, sampling was conducted during winter (July-August 2018), spring (October-November 2018), late summer (March 2019), and late autumn (June 2019) to assess temporal variation in small mammal abundance, capture rates, species richness, and assemblage patterns.

4.2.4. Grassland condition

Grassland condition was classified into four categories: primary-good, primary-intermediate, secondary-good, and secondary-intermediate. Grassland condition categories were obtained from the Biodiversity Management Department (BMD) of the eThekweni Metropolitan Municipality (EMM) and were derived from in situ veld condition assessments and habitat history. Spatial shapefiles delineating grassland condition classes were overlaid with GPS coordinates of sampled transects in Google Earth Pro to assign the corresponding grassland condition category to each study site. These categories reflect both current habitat condition and historical disturbance and recovery status, recognising that some currently protected areas may have experienced previous anthropogenic disturbance. This variable was included because

grassland condition influences vegetation structure, habitat heterogeneity, and resource availability, which are important determinants of small mammal assemblage composition and abundance. Churchfield et al. (1997) demonstrated that small mammal assemblages vary among grasslands of different successional stages, while Mdluli et al. (2022) showed that reductions in vegetation cover can negatively affect vertebrate assemblages in grassland ecosystems. More recent studies by Misher et al. (2022) and Minor & Eichholz (2024) further highlighted the importance of vegetation structure and habitat complexity in shaping small mammal occupancy and assemblage composition. Similarly, Shilereyo et al. (2021) and Hempel et al. (2025) showed that land-use intensity and grassland degradation can alter habitat quality and favour disturbance-tolerant species. Grassland condition was therefore incorporated into this study to account for variation in habitat quality and its potential influence on capture rates and small mammal assemblage patterns across remnant KZNSS grasslands.

4.2.5 Regional variations

Region was incorporated as a spatial environmental variable based on the geographic location of study sites within the EMM and surrounding rural landscape. The three regions sampled in this study were Kloof (KNR; lower elevation, ~584 ha), Hillcrest (GGNR; mid elevation, ~400 ha), and Camperdown Rural (DUC; higher elevation, comprising multiple privately owned land parcels ranging from ~4-900 ha). Regions were assigned according to the location of sampling sites and represented contrasting landscape contexts, including differences in elevation, habitat continuity, land-use history, and surrounding anthropogenic pressures. Spatial locations of sampled transects were assigned to their respective regions based on GPS coordinates and site boundaries. Regional variation among the study areas was included to account for differences in landscape context, including reserve size, surrounding land use, and the extent and isolation of remnant KZNSS grassland habitats. Both Kloof and Hillcrest (Figure 3.5 & 3.6) are situated within highly urbanised landscapes characterised by greater habitat fragmentation and smaller, more isolated remnant grassland patches, whereas Camperdown Rural (Figure 3.7) consists of larger and more contiguous grassland remnants embedded within a less urbanised matrix and subjected to comparatively lower anthropogenic pressure. The KZNSS landscape is highly

fragmented due to agricultural expansion, urbanisation, and land transformation, resulting in spatial variation in habitat condition and ecological processes across remaining grassland patches (Jewitt et al., 2015). Landscape composition and connectivity are recognised as important determinants of mammal assemblages because they influence habitat quality, movement, colonisation processes, and access to regional species pools. Taylor et al. (1993) identified connectivity as a fundamental component of landscape structure, while Bedoya-Durán et al. (2021) and Paniccia et al. (2022) demonstrated that landscape context strongly influences mammal assemblages and functional composition. More recent studies by Brodie et al. (2023), Magoulick et al. (2024), and Westaway et al. (2024) further highlighted the importance of connected landscapes and habitat availability in maintaining mammal diversity and assemblage structure. Regional variation was therefore incorporated into this study to examine whether differences in landscape context and local habitat characteristics among Kloof, Hillcrest, and Camperdown Rural were associated with variation in small mammal assemblages across remnant KZNSS grasslands.

4.2.6 Habitat management regimes (hereafter referred to as treatments)

To assess the influence of conservation status and human disturbance, sites were categorised into three habitat management regimes: inside protected areas, D'MOSS grasslands outside protected areas, and control sites. Protected area status was assigned based on the formal proclamation and legal designation of conservation areas under relevant biodiversity conservation legislation, including the National Environmental Management: Protected Areas Act (Act 57 of 2003), which provides the framework for the declaration and management of protected areas in South Africa (Republic of South Africa, 2003). Krantzkloof Nature Reserve was classified as a protected area based on its formal proclamation and management under an approved protected area management framework (Ezemvelo KZN Wildlife, 2010). Giba Gorge Nature Reserve was classified as a protected conservation area based on its recognised biodiversity conservation role within the eThekweni landscape (Giba Gorge Environmental Precinct Conservation Management Plan, 2016/2017). In contrast, D'MOSS grasslands outside protected areas were classified as areas identified within the eThekweni Municipality's D'MOSS, a municipal biodiversity planning initiative aimed at conserving areas of ecological importance and maintaining landscape

connectivity, but lacking formal protected area status (eThekweni Municipality, 2016). Control sites represented grasslands outside formally protected areas and D'MOSS conservation areas. Despite lacking formal conservation status, these sites exhibited relatively low levels of disturbance and well-preserved grassland habitat, serving as reference sites for comparison with protected and D'MOSS grasslands. It is recognised that current protection status does not necessarily reflect historical land-use or disturbance history, and some protected areas contain grassland patches that were previously degraded or transformed before being incorporated into the protected area network. This was evident in the present study, where several secondary-intermediate grassland patches occurred within GGNR, indicating that habitat condition and conservation status were not always directly aligned. Consequently, habitat management regime was considered independently of grassland condition, allowing the effects of legal protection and vegetation condition to be evaluated separately. Protected areas generally maintain more intact vegetation structure and experience lower levels of disturbance due to conservation-focused management. Geldmann et al. (2019) and Gray et al. (2016) demonstrated that effective protected areas can support biodiversity persistence by reducing anthropogenic pressures and maintaining suitable habitat conditions for disturbance-sensitive species. Conversely, habitats outside protected areas may experience greater edge effects, habitat modification, and altered ecological processes. Hurst et al. (2013), Umetsu & Pardini (2007), and Torre et al. (2024) showed that changes in land use and habitat disturbance can influence habitat quality and favour disturbance-tolerant species. Fahrig (2003) highlighted that landscape fragmentation can alter habitat availability and connectivity, affecting movement and colonisation processes, while Bedoya-Durán et al. (2021) demonstrated the importance of landscape context in shaping mammal assemblages. Laurance & Bierregaard (1997) highlighted that protected areas are not immune to ecological pressures associated with habitat fragmentation, including edge effects, while Laurance et al. (2002) demonstrated that such effects can persist within protected habitat fragments. Similarly, invasive alien species may continue to affect protected areas where they occur as small or fragmented patches embedded within human-modified landscapes. Foxcroft et al. (2013) highlighted that protected areas are not immune to biological invasions, as invasive alien species can establish and spread within conservation areas despite formal protection. In South Africa, Richardson & van Wilgen (2004) demonstrated that invasive alien plants

represent a major ecological threat to native ecosystems, including conservation landscapes. Recent studies (Paniccia et al., 2022; Shilereyo et al., 2023; Magoullick et al., 2024; Westaway et al., 2024) have further shown that protection status, landscape composition, and land-use intensity influence mammal assemblage structure and functional composition. Habitat management regimes were therefore incorporated into this study to evaluate how varying levels of protection and human disturbance influence small mammal assemblages across remnant KZNSS grasslands.

4.2.7. Focal species (rodents and shrews)

The focal species for this study included rodents and shrews, which represent important components of small mammal assemblages in southern African grassland ecosystems. Despite their ecological importance, small mammal assemblages within KZNSS landscape remain relatively understudied. These species represent diverse ecological roles, encompassing functional guilds such as granivores, herbivores, insectivores, and omnivores. Small mammals were selected as focal organisms because their abundance, diversity, and assemblage composition are responsive to environmental variation, including changes in vegetation structure, habitat disturbance, and land-use intensity. Studies by Lagesse & Thondhlana (2016), Loggins et al. (2019), and Ramahlo et al. (2022) have demonstrated the value of small mammals as indicators of habitat condition and ecological change across modified landscapes. Species with very low capture frequencies (less than 30 captures) were excluded from species-specific analyses to minimise the influence of rare observations on statistical outcomes and to ensure reliable comparisons among study sites. Captured individuals were assigned to sex categories (male or female) and age classes (juvenile, subadult, or adult) using a combination of body mass, external morphological measurements, pelage characteristics, and reproductive condition. Reproductive condition was assessed based on external characteristics, including testicular position in males (abdominal, semi-scrotal, or scrotal) and reproductive signs in females (imperforate, perforate, pregnant, or lactating). These classifications followed established small mammal handling and demographic assessment protocols described by Hoffmann et al. (2010), Monadjem et al. (2015), and Balčiauskas & Balčiauskienė (2022). Functional guild classifications were based on published ecological trait information, with species assigned according to dietary preferences,

trophic position, foraging behaviour, body size, and ecological niche characteristics. This approach followed the trait-based framework described by Wilman et al. (2014), which integrates multiple ecological attributes to characterise species functional roles.

4.3 Small mammal trapping

A combination of large folding aluminium ($7.6 \times 8.9 \times 22.9$ cm; LFA) and small folding aluminium ($5.1 \times 6.4 \times 16.5$ cm; SFA) H.B. Sherman™ live box traps were used to capture small mammal species, as these traps are suitable for animals weighing between 1.8 and 150 g (H. B. Sherman Traps, n.d.). Traps were set along 125 m long transects comprising 25 traps spaced 5 m apart, following standard small mammal trapping methodology described by Sikes et al. (2011). The start, middle, and end points of each trap line were marked with grass knots or danger tape, and GPS coordinates were recorded. Transects were established as close to the centre of each grassland patch as possible to reduce edge effects. Transects were preferred to grids due to their effectiveness in avoiding patch boundaries in fragmented landscapes, as discussed by Murcia (1995) and Debinski & Holt (2000). The traps were alternated along each transect (one SFA followed by one LFA), with 13 SFA and 12 LFA traps used per trap line. A total of 30 grassland patches were sampled, with one transect established per patch. In both KNR and GGNR, five transects were deployed inside the protected areas (transects 1-5) and five outside (D'MOSS) the protected area boundaries (transects 6-10; not less than 500 m away), resulting in 10 replicates inside and 10 outside protected areas. The control site (D'MOSS reference site) was sampled across 10 transects.

To ensure spatial independence and minimise spatial autocorrelation and reduce the likelihood of repeatedly sampling the same individuals, transects were spaced more than 500 m apart. This separation distance exceeded the reported home-range sizes of the southern African vlei rat (*Otomys auratus*), the largest small mammal recorded during the study, for which Davis (1973) reported relatively small home ranges. The spacing of transects was therefore considered adequate to reduce potential overlap among sampled populations and promote relative spatial independence among sampling units, with Krebs (1966), Legendre (1993), and Dormann et al. (2007) providing recommendations regarding spatial independence and ecological sampling design. Sampling was scheduled to avoid full moon periods and coincide with new

moon phases because lunar illumination can influence small mammal activity patterns and detectability, as demonstrated by Soulsbury (2021) and Oosthuizen et al. (2025). Transects were positioned ≥ 150 m from patch edges where feasible to minimise edge effects, following recommendations by Murcia (1995) and Harper et al. (2005). However, within D'MOSS, grassland patches were highly fragmented and small, making it impossible to completely avoid edge effects. In such cases, larger and more contiguous patches were preferentially selected. Traps were placed systematically along transects to standardise sampling effort and reduce spatial bias in capture probability, following the recommendations of Cunningham et al. (2005).

Traps were baited with a mixture of peanut butter and rolled oats, a commonly used and effective attractant in small mammal live-trapping protocols (Sikes et al., 2011). Carrot (*Daucus carota*) discs were placed inside traps to provide a source of moisture and reduce dehydration in captured individuals (Sikes et al., 2011). A small amount of fluffed cotton wool was inserted into each trap as insulation and bedding to improve thermoregulation and animal welfare, ensuring that it did not obstruct the trigger mechanism, as recommended by Mills et al. (1995) and Sikes et al. (2011). Traps were covered with grass and leaf litter to reduce exposure to weather conditions and provide concealment, consistent with standard small mammal trapping protocols described by Sikes et al. (2011). During each seasonal sampling session, traps were deployed for five consecutive days and four consecutive nights. Traps were checked twice daily, in the early morning (~10:00) and late afternoon (~14:30), and during periods of extreme heat or when diurnal species were recorded, traps were checked at midday as well. Captured individuals were collected, processed, and released, after which traps were rebaited each morning.

4.4 Capture-Mark-Recapture (CMR).

The Capture-Mark-Recapture (CMR) method described by Otis et al. (1978) was used to estimate small mammal population size. Captured individuals were marked using non-invasive, non-permanent techniques, including non-toxic marker pens and fur clipping, to facilitate individual identification while minimising handling time and stress, consistent with the ethical recommendations for small mammal research outlined by Sikes et al. (2011). Fur clipping was primarily used to distinguish individuals between seasonal sampling sessions, whereas identification within each sampling session

relied primarily on temporary marker patterns, supplemented by information on sex, morphometric measurements, and distinctive morphological characteristics such as truncated tails, ear notches, scars, and other unique morphological features. This approach was preferred because it minimised handling stress and reduced the potential welfare concerns associated with permanent and more invasive marking techniques, such as toe clipping, as discussed by Powell & Proulx (2003) and Sikes et al. (2011) (Figure 4.1).

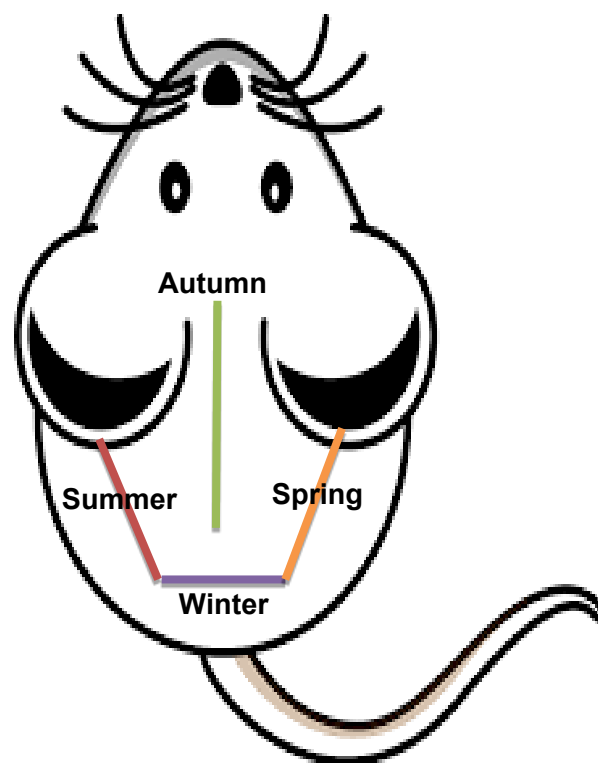


Figure 4.1: Fur clipping was used primarily for between-season identification, whereas identification within sampling sessions relied on temporary marker patterns supplemented by sex, morphometric measurements, and distinctive morphological characteristics.

Morphometric data, including body weight, total length, and tail length, were also used to support individual recognition, and each individual was assigned a unique number for identification during subsequent captures. Although these non-permanent marking methods may result in occasional misidentification, particularly if marks fade or animals are recaptured after long intervals, every effort was made to reduce errors by using a combination of markings, unique physical characteristics, and morphometric measurements to reliably identify individuals during each trapping session.

Measurements from each captured animal were recorded on standardised data sheets as either a new individual or a recapture based on the presence of marks and the correspondence of recorded characteristics.

Animal capture and handling procedures followed the animal welfare and handling guidelines outlined by Sikes et al. (2011) and the American Veterinary Medical Association (2020) and were conducted under permits OP 1104/2017 and OP 143/2018 issued by Ezemvelo KZN Wildlife. Ethical clearance for the study was granted by the College of Agricultural and Environmental Sciences (CAES) Animal Research Ethics Committee (AREC) of the University of South Africa (UNISA), reference number 2018/CAES/044. Personal protective equipment, including facemasks and durable plastic garden gloves and/or latex gloves, were worn during all animal handling and processing. Captured animals were carefully removed from live traps by gently encouraging individuals into durable plastic zip-seal bags to minimise handling stress and avoid injury.

Animals were manoeuvred into a corner of the bag to facilitate safe handling, and a small air hole (~5 mm) was made in the corner of each bag to allow ventilation during processing. Morphometric measurements were recorded for each captured individual, including head and body length (HB, mm), total length (TL, mm), and tail length (TAL, mm), measured using a steel ruler. Ear length (EL, mm) and hind foot length (FL, including the claw, mm) were measured using Mitutoyo digital callipers (Mitutoyo American Corporation, 2019), while body weight (BW, g) was recorded using calibrated Pesola spring balances (10 g, 30 g, and 100 g), selected according to the size of the individual. The use of these measurements follows standard southern African mammal measurement protocols described by De Graaff (1981), Taylor (1998), and Sikes et al. (2011).

Sex (male or female), age class (juvenile, sub-adult, or adult), and reproductive condition for males (abdominal, semi-scrotal and scrotal) and females (imperforate, perforate, pregnant and lactating) were recorded for each individual following standard small mammal field protocols described by Hoffmann et al. (2010), Sikes et al. (2011), and Monadjem et al. (2015), as explained above. Female reproductive condition was classified as imperforate hymen (sexually immature or non-reproductive), perforate hymen (sexually mature or reproductively active), pregnant, or lactating, based on

external reproductive characteristics, including the presence of enlarged teats and/or milk, following classifications described by Hoffmann et al. (2010) and Monadjem et al. (2015). Male reproductive status was assessed based on testicular position (abdominal, semi-scrotal, or scrotal), which is widely used as an indicator of reproductive activity in small mammals and has been described by Monadjem et al. (2015) and Jung et al. (2020).

The reproductive status of shrew species was not recorded during this study due to difficulties in accurately determining reproductive condition under field conditions. Churchfield (1990), Taylor et al. (1998), and Monadjem et al. (2015) note that, in shrews, both reproductive and excretory openings are enclosed within a cloaca, limiting reliable external identification of reproductive state without invasive examination. To minimise handling stress and avoid procedures that could compromise animal welfare, invasive methods required for definitive reproductive assessment were not performed, consistent with the recommendations of Sikes et al. (2011).

Captured rodents were classified into age classes using relative body size and body mass in combination with external morphological and reproductive indicators. Individual body mass and linear measurements were compared with species-specific adult reference values derived from this study and published sources, including Hoffmann et al. (2010), Monadjem et al. (2015), and Balčiauskas & Balčiauskienė (2022). Animals were assigned to one of three age classes: juvenile, subadult, or adult as described above. When body mass alone did not allow clear classification, additional indicators such as reproductive status (e.g. lactation, perforate vagina, or scrotal testes) were used to support age assignment, following standard small mammal field protocols described by Sikes et al. (2011) and Jung et al. (2020). The minimum body mass of reproductively active individuals observed during this study was used as the threshold separating juveniles from adults for each sex.

Functional guilds were assigned based on species-specific ecological traits, particularly dominant diet, body size, foraging behaviour, and habitat associations (Wilman et al., 2014). Species were classified following a trait-based approach, whereby species were grouped according to dominant dietary preferences and subsequently differentiated by body size where appropriate (de Bello et al., 2021).

Mean body mass values and ecological trait information were obtained from published literature, regional mammal references, and curated ecological trait databases (Wilman et al., 2014). Any small mammals found deceased in traps were carefully assessed for the presence of marks, identified to species level and, where possible, to individual based on applied markings and recorded characteristics, and documented accordingly on standardized data sheets. These individuals were retained as voucher specimens and deposited in the Durban Natural Science Museum mammal collection following appropriate processing. The vouchered specimens serve both as a permanent record of species occurrence and as reference material for verifying and confirming species identifications made in the field. All live-captured animals were released at their site of capture immediately after processing.

4.5 Data analysis

Population size and capture patterns of small mammals were assessed by recording total captures, the number of new captures, and recaptures for each trapline during each sampling session, as described by Delany (1974), Krebs (1999), and Eymann et al. (2010). This approach enabled the quantification of species richness and relative abundance across the study sites, as reported by Harkins et al. (2019) and Kayala (2024). Minimum Known Alive (MKA) values were calculated to provide a conservative estimate of population size for each species based on capture histories across sampling sessions. Delany (1974), Shanker (2000), and Hammond & Anthony (2006) demonstrated that the MKA value represents the minimum number of individuals of a given species known to be alive on a single transect during each season, based on capture histories of individuals recorded before, during, and after each sampling session. Specifically, Krebs (1966) defined MKA as the number of individuals captured during a given trapping session, plus individuals not captured during that session but known to be alive based on captures recorded both before and after the session. Sullivan & Sullivan (2013) and Barros et al. (2015) demonstrated that this approach provides a conservative estimate of population size by accounting for individuals that may have temporarily avoided capture. Harkins et al. (2019) and Gračanin et al. (2022) further reported that the use of MKA reduces bias associated with variable capture probabilities and is commonly applied in small mammal population studies where detection may vary among individuals and sampling occasions. By incorporating

information from recaptures across sessions, this method reduces the likelihood of underestimating population size. In addition to MKA, the Lincoln Index, also known as the Lincoln-Petersen estimator, was used to estimate total population size for each species on each transect at each site. Petersen (1896) first described this estimator, which was subsequently refined by Lincoln (1930). Following the recommendations of Chao et al. (1992), Shanker (2000), and Hammond & Anthony (2006), Lincoln Index analyses were restricted to species with >30 recorded captures to ensure adequate sample size and statistical reliability. The Lincoln Index was calculated using the following formula:

$$N = \frac{n_1 \times n_2}{m_2}$$

In this formula, N is the estimated population size, n_1 is the number of individuals captured and marked in the first session, n_2 is the total number of individuals captured in the second session, and m_2 is the number of marked individuals recaptured in the second session, as described by Lincoln (1930) and Krebs (1999). Krebs (1999) and Williams et al. (2002) stated that this estimator assumes a closed population with no births, deaths, immigration, or emigration between capture sessions. By combining total captures, MKA, and Lincoln-Petersen population estimates originally proposed by Petersen (1896) and later refined by Lincoln (1930), a robust assessment of population structure, capture efficiency, and spatial distribution patterns of small mammals was obtained, allowing comparisons among sampled sites with varying levels of habitat fragmentation.

Krebs (1999) and Sullivan & Sullivan (2013) reported that trap success is directly related to population size and can be used to evaluate population trends. Accordingly, trap success was used as an indicator of species abundance within the study area and was calculated as the number of animals captured per 100 trap nights using the following formula:

$$\text{trap success} = (\text{total catch} / \text{trap-nights}) \times 100,$$

where total catch represents the total number of individuals captured, irrespective of species, as described by Krebs (1999) and Sullivan & Sullivan (2013). Trap nights were calculated as the number of traps deployed multiplied by the number of nights sampled, minus misfires and captures of non-target species, following Krebs (1999).

Misfires were defined by Nelson & Clark (1973) as traps that were triggered without capturing an animal, commonly caused by wind, vegetation interference, or trap sensitivity issues.

Krebs (1999) and Magurran (2004) defined species richness as the total number of species captured per trapline during each sampling period. This measure was used to describe the variety of small mammal species present within each trapline and to enable comparisons among traplines and sampling sites. Species diversity within each trapline was quantified using the Shannon-Wiener diversity index, originally developed by Shannon (1948). This index incorporates both species richness and the relative abundance of individuals within each species, thereby providing a comprehensive measure of assemblage diversity. Relative abundance was calculated as the proportion of individuals of a given species relative to the total number of individuals captured within each trapline. The Shannon-Wiener diversity index, as originally developed by Shannon (1948) and described by Magurran (2004), was calculated using the following equation:

$$H = -\sum[(p_i) * \log(p_i)]$$

In this equation H is the Shannon Diversity Index, p_i is the proportion of the total sample represented by species i in a whole community: $p_i = n / N$, where: n is the number of individuals of a given type/species and N is the total number of individuals in an assemblage, $\sum$ is a sum symbol and $\log$ is the natural logarithm.

Data were analysed to estimate population size and assess differences among groups and sampling treatments. Prior to conducting statistical analyses, the data were tested for normality using the Shapiro-Wilk test in IBM SPSS Statistics Version 29.0, developed by International Business Machines Corporation (IBM Corp., 2023). Differences among multiple groups were evaluated for a range of response variables, including total small mammal captures, trap success, minimum known alive (MKA), species richness, species diversity, and functional guild composition. These parameters were compared across environmental gradients and sampling factors such as elevation, slope aspect, seasonal variation, grassland condition, regional variation, and habitat management regimes (treatments). Kruskal & Wallis (1952) developed the Kruskal-Wallis test, which was used to identify significant differences among groups. Where significant effects were detected, post-hoc pairwise

comparisons were conducted using Dunn's test, developed by Dunn (1964), with a Bonferroni correction, as described by Armstrong (2014), to adjust for multiple comparisons. Due to the large volume of statistical outputs generated, only the most pertinent post hoc results are presented in the main text, while the full set of outputs is provided in the appendices and referenced accordingly (Appendices 1-51). Due to low capture frequencies for some species, species-specific demographic analyses were restricted to species with >30 recorded captures to ensure adequate sample size and statistical reliability. This threshold was adopted in accordance with the general recommendations of Chao et al. (1992), Shanker (2000), and Hammond & Anthony (2006), who emphasised the importance of adequate sample sizes for reliable capture-recapture and population analyses. To assess relationships between small mammal demographics (age classes, reproductive status, and sex categories) and environmental predictors, multinomial logistic regression was used to analyse data for age classes and reproductive status, while binomial logistic regression was used to analyse sex categories. These analyses were conducted using IBM SPSS Statistics Version 29.0 (IBM Corp., 2023). This analysis evaluated the influence of elevation, slope aspect, seasonal variation, grassland condition, regional variation, and habitat management regimes (treatments) on the distribution of demographic categories. Statistical significance was assessed at $\alpha = 0.05$. Results were summarised using descriptive statistics, including means and standard deviations, where appropriate, to highlight general trends and provide context for observed patterns.

CHAPTER 5

RESULTS

5.1 eThekwini KwaZulu-Natal Sandstone Sourveld Grasslands

This study included 16 586 trap nights, a total of 1 199 captures including 890 unique captures and 309 recaptures of small mammals, from two orders (Eulipotyphla and Rodentia) and three families (Muridae, Nesomyidae and Soricidae) were made (Table 5.1). The mean number of total captures per trap line ($\bar{X} = 9.98 \pm 9.91$) ($n = 30$ traplines) was higher than Minimum Known Alive (MKA) ($\bar{X} = 7.18 \pm 6.83$) values.

Table 5.1. Metrics associated with total small mammal captures in eThekwini KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Metrics	Total	Mean	Std. deviation
Total captures	1199	9.98	9.91
Trap success	-	12.55	12.77
MKA ^a	889	7.18	6.83
Species richness	11	1.92	1.27
Shannon Diversity index	-	0.43	0.44

^aMKA – minimum known alive

The haul of small mammals comprised individuals from 11 species across three study locales in the eThekwini KZNSS grasslands (Appendix 1, 2, and 3). From 30 traplines equally distributed among the three component study sites (Krantzkloof Nature Reserve - KNR, Giba Gorge Nature Reserve - GGNR, and the Doornrug and Uitkomst Conservancy - DUC) a total of 890 small mammals (KNR: $n=186$; GGNR: $n=318$, DUC: $n=386$) were captured. The haul comprised nine species of rodents (Muridae: Tete veld rat, *Aethomys ineptus* (the name *Aethomys ineptus* is used throughout this study, despite its treatment as a synonym of *Aethomys chrysophilus* by Russo & da Silva (2025); single-striped grass mouse, *Lemniscomys rosalia*; Natal multimammate mouse, *Mastomys natalensis*; Namaqua rock rat, *Micaelamys namaquensis*; African pygmy mouse, *Mus minutoides*; southern African vlei rat, *Otomys auratus*; mesic four-striped grass rat, *Rhabdomys dilectus* (formerly *Rhabdomys pumilio*); Nesomyidae: grey climbing mouse, *Peomys melanotis* (formerly *Dendromus melanotis*), fat mouse, *Steatomys* spp.) and two species of shrews (Soricidae: greater red musk shrew, *Crocidura flavescens*; forest shrew, *Myosorex varius*) (Figure 5.1). Five functional

guilds were represented in the species haul, granivores (*R. dilectus*, *Steatomys* spp.), herbivores (*L. rosalia* and *O. auratus*), small omnivores (<30 g) (*P. melanotis* and *M. minutoides*), large omnivores (≥30 g) (*A. ineptus*, *M. namaquensis*, *M. natalensis*) and insectivores (*C. flavescens*, *M. varius*) (Appendix 5).

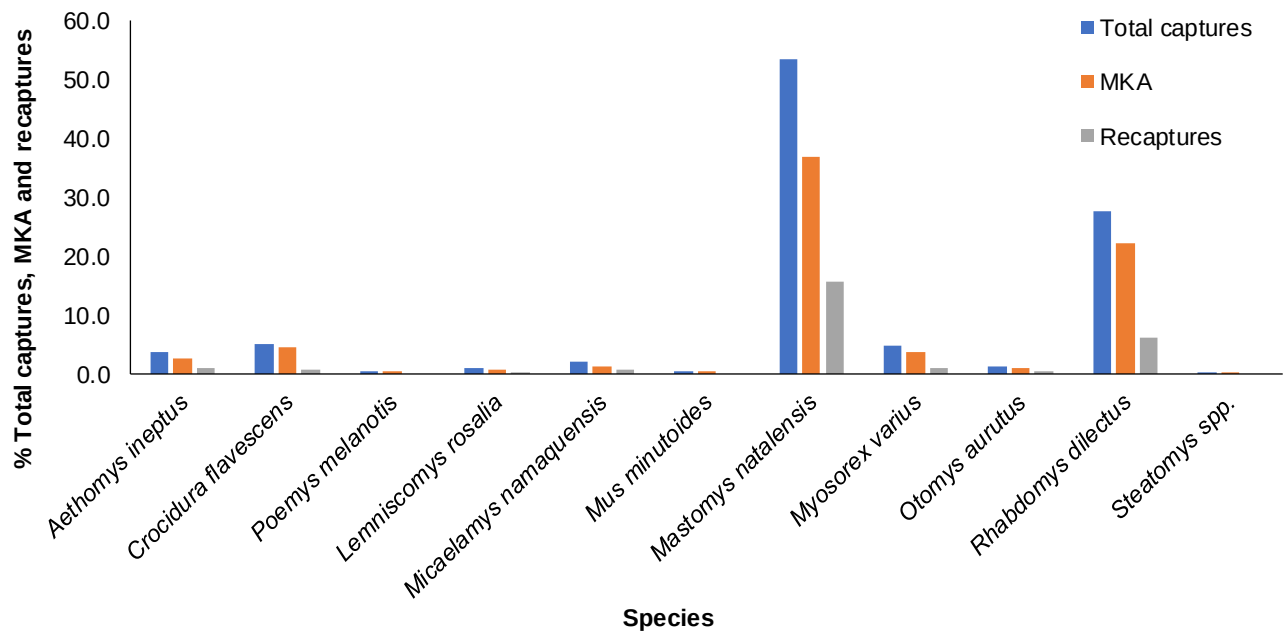


Figure 5.1. Percentage contribution of individual small mammal species to total captures, minimum known alive (MKA) and recaptures per species within eThekwin i KwaZulu-Natal Sandstone Sourveld grasslands (from July 2018 to June 2019).

The total number of captures, recaptures and MKA values differed among species (Figure 5.1). The two most frequently captured small mammal species were *M. natalensis* and *R. dilectus*. *Mastomys natalensis* was the most abundant species ($n_{\text{total captures}}=641$; $n_{\text{recaptures}}=188$; $n_{\text{MKA}}=443$) and contributed 54% of the total small mammal captures during this study. *Rhabdomys dilectus* contributed 28% of total captures ($n_{\text{total captures}}=331$; $n_{\text{recaptures}}=74$; $n_{\text{MKA}}=267$). The specialist species *P. melanotis* ($n_{\text{total captures}}=7$; $n_{\text{recaptures}}=0$; $n_{\text{MKA}}=7$) and *Steatomys* spp. ($n_{\text{total captures}}=2$; $n_{\text{recaptures}}=0$; $n_{\text{MKA}}=2$) had the least total captures, MKA and recaptures and comprised only 0.7% of the combined total small mammal captures. Shrews contributed 10% of total captures combined - *C. flavescens*: ($n_{\text{total captures}}=69$; $n_{\text{recaptures}}=18$; $n_{\text{MKA}}=45$); *M. varius*: ($n_{\text{total captures}}=56$; $n_{\text{recaptures}}=17$; $n_{\text{MKA}}=39$) (Figure 5.1).

As anticipated, representatives from five small mammal functional guilds were identified during the study (Figure 5.2). The functional guild associated with the highest number of captures and recaptures were large omnivores ($n_{\text{total captures}}=711$; $n_{\text{MKA}}=490$; $n_{\text{recaptures}}=210$). The functional guild with the lowest number of captures and recaptures was the small omnivores ($n_{\text{total captures}}=10$; $n_{\text{MKA}}=10$; $n_{\text{recaptures}}=0$).

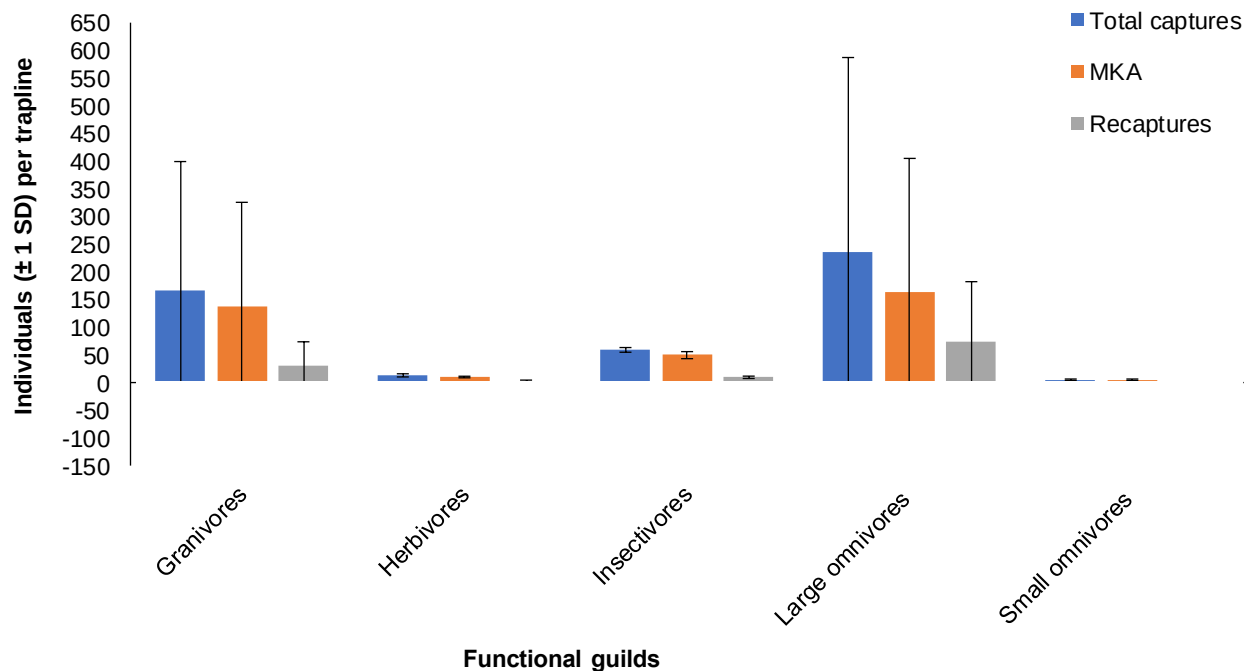


Figure 5.2. Total captures, minimum known alive (MKA) values and recaptures per functional guild recorded from eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (from July 2018 to June 2019).

5.1.1 Elevation

The total captures, trap success, MKA, species richness and species diversity, were significantly influenced by elevation, with values generally increasing with increased elevation (Table 5.2; Appendix 4).

Table 5.2. The outcomes from Kruskal-Wallis tests on metrics associated with the effect of elevational gradients on small mammal captures within eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Metrics	Elevation ($\bar{X} \pm SD$)			Z-test	P-value
	Low	Mid	High	DF	
Total captures	5.95 $\pm$ 6.30	11.30 $\pm$ 10.55	10.63 $\pm$ 10.01	9.86 (2)	0.007*
Trap success	7.43 $\pm$ 7.80	9.85 $\pm$ 9.11	14.44 $\pm$ 14.08	12.33 (2)	0.002*
MKA	4.40 $\pm$ 4.14	7.85 $\pm$ 7.08	7.63 $\pm$ 7.28	9.29 (2)	0.010*
Species richness	1.28 $\pm$ 0.99	1.75 $\pm$ 0.97	2.00 $\pm$ 1.90	19.16 (2)	<0.001*
Species diversity	0.20 $\pm$ 0.41	0.70 $\pm$ 0.57	0.25 $\pm$ 0.45	20.91 (2)	<0.001*

* Significant differences when evaluated at $\alpha = 0.05$

Total captures, MKA, and species diversity were highest at mid elevations, whereas trap success and species richness were highest at high elevations (Table 5.2). Total captures were significantly greater at both mid and high elevations than at low elevations, with no significant difference between mid and high elevations (Table 5.2; Appendix 4). Trap success, MKA, and species richness were each significantly higher at high than at low elevations but did not differ significantly between low and mid elevations or between mid and high elevations (Table 5.2; Appendix 4). Species diversity was significantly higher at both mid and high elevations than at low elevations, with no significant difference between mid and high elevations (Table 5.2; Appendix 4).

The small mammal functional guilds were influenced by elevation (Table 5.3). The total captures and MKA values for granivores, herbivores, insectivores and large omnivores were significantly influenced by elevation (Table 5.3). The total captures and MKA values for granivores and insectivores were greatest at high elevations while that of herbivores, large omnivores and small omnivores were greatest at mid elevations (Table 5.3).

Table 5.3. The outcomes from Kruskal-Wallis tests on functional guild metrics for total captures and minimum known alive (MKA) values associated with the effect of elevational gradients on small mammal captures within eThekwinini KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Metrics	Functional guilds	Elevation ($\bar{X} \pm SD$)			Z-test	P-value
		Low	Mid	High	DF	
Total captures	Granivores	0.12 $\pm$ 0.44	0.22 $\pm$ 0.69	5.91 $\pm$ 6.50	59.78 (2)	<0.001*
	Herbivores	0.12 $\pm$ 0.33	0.41 $\pm$ 1.16	0.11 $\pm$ 0.42	11.72 (2)	0.003*
	Insectivores	0.48 $\pm$ 1.12	0.51 $\pm$ 0.98	1.57 $\pm$ 2.17	12.72 (2)	0.002*
	Large Omnivores	5.56 $\pm$ 6.10	9.41 $\pm$ 8.37	3.44 $\pm$ 6.06	5.58 (2)	0.061*
	Small Omnivores	0.13 $\pm$ 0.44	0.17 $\pm$ 0.59	-	1.40 (2)	0.497
MKA	Granivores	0.12 $\pm$ 0.44	0.22 $\pm$ 0.69	4.76 $\pm$ 5.23	58.53 (2)	<0.001*
	Herbivores	0.12 $\pm$ 0.33	0.30 $\pm$ 0.72	0.07 $\pm$ 0.33	14.49 (2)	<0.001*
	Insectivores	0.48 $\pm$ 1.12	0.51 $\pm$ 0.98	1.30 $\pm$ 1.74	13.13 (2)	0.001*
	Large Omnivores	5.56 $\pm$ 6.10	6.41 $\pm$ 5.47	2.41 $\pm$ 3.81	5.61 (2)	0.061
	Small Omnivores	0.13 $\pm$ 0.44	0.17 $\pm$ 0.59	-	1.40 (2)	0.497

* Significant differences when evaluated at $\alpha = 0.05$

The total captures and MKA of granivores were significantly higher at high than at both mid and low elevations, with no significant difference between low and mid elevations (Table 5.3; Appendix 5). For herbivores, total captures and MKA were significantly higher at mid than at high elevations. Total captures were also significantly higher at mid than at low elevations, whereas MKA did not differ significantly between these elevations. There was no significant difference in herbivore total captures between low and high elevations, although MKA was significantly higher at low than at high elevations (Table 5.3; Appendix 5). Insectivore total captures and MKA were significantly higher at high than at mid elevations. Total captures were also significantly higher at high than at low elevations, whereas MKA did not differ significantly between these elevations. Neither total captures nor MKA differed significantly between low and mid elevations (Table 5.3; Appendix 5). Large omnivore total captures and MKA were significantly higher at mid than at high elevations but did not differ significantly between mid and low elevations or between high and low elevations (Table 5.3; Appendix 5). Elevation had no significant effect on the total captures or MKA of small insectivores (Table 5.3; Appendix 5).

5.1.2 Slope aspect

Total captures, trap success, MKA, and species richness were significantly influenced by slope aspect, whereas species diversity was not (Table 5.4). Total captures and MKA were highest on west-facing slopes, while trap success was highest on south-

east facing slopes. Species richness was highest on both south-east and south-west facing slopes (Table 5.4). Total captures on south facing slopes were significantly lower than those on south-east, south-west, and west facing slopes (Table 5.4; Appendix 6). Similarly, total captures on north facing slopes were significantly lower than those on south-east, south-west, and west facing slopes, whereas east facing slopes had significantly lower total captures than north-west, north-east, south-east, south-west, and west facing slopes. Conversely, total captures on west facing slopes were significantly higher than those on north-west and north-east facing slopes (Table 5.4; Appendix 6). Trap success was significantly lower on north facing slopes than on west, south-west, north-east, and south-east facing slopes, while north-west facing slopes had significantly lower trap success than west, north-east, and south-east facing slopes. In addition, trap success on east facing slopes was significantly lower than on south-east facing slopes, and west facing slopes also had significantly lower trap success than south-east facing slopes (Table 5.4; Appendix 6).

Minimum known alive (MKA) values differed significantly among slope aspects (Table 5.4). MKA on south facing slopes was significantly lower than on south-east, south-west, and west facing slopes. Similarly, MKA on north facing slopes was significantly lower than on south-east, south-west, and west facing slopes. East-facing slopes had significantly lower MKA values than north-west, north-east, south-east, south-west, and west facing slopes. In contrast, MKA on west facing slopes was significantly higher than on north-west and north-east facing slopes (Table 5.4; Appendix 6). Species richness also differed significantly among slope aspects. Species richness on south facing slopes was significantly lower than on west- and south-east facing slopes, while north-facing slopes had significantly lower species richness than west and south-east facing slopes. East facing slopes had significantly lower species richness than west and south-east facing slopes. Similarly, species richness on north-east and north-west facing slopes was significantly lower than on west and south-east facing slopes (Table 5.4; Appendix 6). Slope aspect did not significantly influence species diversity (Table 5.4).

Table 5.4. The outcomes from Kruskal-Wallis tests on metrics associated with small mammals captures relative to slope aspect across eThekwini KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Metric	Slope aspect ($\bar{X} \pm \text{SD}$)								Z-test	P-value
	North-West	North-East	North	South-East	South-West	South	East	West	DF	
Total captures	8.08 $\pm$ 7.44	11.92 $\pm$ 13.04	2.63 $\pm$ 1.30	14.81 $\pm$ 13.73	12.50 $\pm$ 9.75	2.00 $\pm$ 0.82	3.17 $\pm$ 3.43	16.40 $\pm$ 8.41	31.83 (7)	<0.001*
Trap success	7.59 $\pm$ 7.98	19.32 $\pm$ 16.18	4.51 $\pm$ 4.85	25.79 $\pm$ 18.31	6.59 $\pm$ 7.90	11.43 $\pm$ 10.43	8.67 $\pm$ 7.61	13.45 $\pm$ 11.18	24.79 (7)	<0.001*
MKA	6.00 $\pm$ 4.89	8.50 $\pm$ 9.23	2.50 $\pm$ 1.31	10.31 $\pm$ 9.39	9.63 $\pm$ 6.78	2.00 $\pm$ 0.58	2.17 $\pm$ 2.08	11.70 $\pm$ 5.91	31.58 (7)	<0.001*
Species richness	1.70 $\pm$ 1.09	1.58 $\pm$ 1.31	1.25 $\pm$ 0.89	2.69 $\pm$ 4.49	2.00 $\pm$ 1.07	1.25 $\pm$ 0.50	1.33 $\pm$ 1.23	2.65 $\pm$ 1.27	18.27 (7)	0.011*
Species diversity	0.46 $\pm$ 0.47	0.54 $\pm$ 0.46	0.20 $\pm$ 0.36	0.48 $\pm$ 0.43	0.67 $\pm$ 0.24	0.17 $\pm$ 0.35	0.32 $\pm$ 0.49	0.38 $\pm$ 0.45	8.84 (7)	0.266

* Significant differences when evaluated at $\alpha = 0.05$

Slope aspect only had a significant influence on total captures and MKA for granivores, large omnivores and small omnivores (Table 5.5). The mean value for total captures for granivores was greatest on the north-east facing slope, while MKA mean value was greatest on the south-west facing slopes (Table 5.5). The mean values for total captures and MKA for herbivores were highest on the north-west facing slopes while that of insectivores was highest on the south-east and south-west facing slopes (Table 5.5). The mean value for large and small omnivores was highest on the west facing slope (Table 5.5). The mean values for total captures and MKA for granivores were significantly lower on the north facing slopes than north-west, north-east, south-west and south-east facing slopes (Table 5.5; Appendix 7). The mean value for total captures and MKA for granivores on the south facing slope was significantly lower than those on the south-east facing slopes, while that on the east facing slopes was significantly lower than those on the south-east facing slopes (Table 5.5; Appendix 7). The mean value for granivores on west facing slopes was significantly lower than those on the south-east facing slopes, while that on the north-west facing slopes was significantly lower than those on the south-east facing slopes (Table 5.5; Appendix 7). The total captures mean value for large omnivores was significantly lower on the east facing slopes than on north-west, south-west, north, or south-east facing slopes (Table 5.5; Appendix 7). The mean total captures and MKA for large omnivores on the south facing slopes were significantly lower than those on the south-east and west facing slopes (Table 5.5; Appendix 7). The total capture mean value for large omnivores on the west facing slopes was significantly higher than those on the north-east, north-west, south-west, north and south-east facing slopes (Table 5.5; Appendix 7).

Table 5.5. The outcomes from Kruskal-Wallis tests on functional guild metrics for total captures and minimum known alive (MKA) values associated with the effect of slope aspect on small mammal captures within eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Metrics	Functional guilds	Slope aspect ($\bar{X} \pm SD$)								Z-test	P-value
		North-West	North-East	North	South-East	South-West	South	East	West	DF	
Total captures **	Granivores	1.80 ± 2.67	6.50 ± 10.18	-	6.31 ± 7.27	3.75 ± 5.28	-	1.50 ± 2.39	1.60 ± 3.35	18.20 (7)	0.011*
	Herbivores	0.38 ± 1.17	0.17 ± 0.39	-	0.25 ± 0.68	0.13 ± 0.35	-	-	0.20 ± 0.41	4.63 (7)	0.705
	Insectivores	0.68 ± 1.33	1.42 ± 2.57	0.25 ± 0.46	1.88 ± 2.36	1.88 ± 2.42	1.00 ± 0.82	0.42 ± 1.16	0.90 ± 1.33	11.38 (7)	0.123
	Large Omnivores	4.65 ± 5.12	4.00 ± 6.27	4.03 ± 3.52	5.81 ± 5.38	5.00 ± 5.76	0.50 ± 0.58	0.50 ± 0.67	15.15 ± 10.14	35.27 (7)	<0.001*
	Small Omnivores	-	-	-	-	-	-	0.25 ± 0.62	0.35 ± 0.81	17.40 (7)	0.015*
MKA	Granivores	1.48 ± 2.30	4.83 ± 7.93	-	5.00 ± 5.68	3.50 ± 5.01	-	1.17 ± 1.85	1.50 ± 3.07	17.87 (7)	0.013*
	Herbivores	0.26 ± 0.72	0.17 ± 0.39	-	0.13 ± 0.50	0.13 ± 0.35	-	-	0.20 ± 0.41	5.48 (7)	0.601
	Insectivores	0.50 ± 0.99	1.17 ± 1.99	0.25 ± 0.46	1.44 ± 1.71	1.75 ± 2.31	1.00 ± 0.82	0.42 ± 1.16	0.85 ± 1.27	12.00 (7)	0.101
	Large Omnivores	3.58 ± 4.00	2.92 ± 4.98	3.00 ± 2.20	4.38 ± 5.33	4.50 ± 6.30	0.50 ± 0.58	0.42 ± 0.51	8.75 ± 5.84	29.91 (7)	<0.001*
	Small Omnivores	-	-	-	-	-	-	0.25 ± 0.62	0.35 ± 0.81	17.40 (7)	0.015*

* Significant differences when evaluated at $\alpha = 0.05$

5.1.3. Seasons

Seasons had a significant influence on total captures and MKA, and had no significant effect on trap success, species richness and species diversity (Table 5.6). The general trends show that the mean values for total captures, MKA, species richness and species diversity were highest in the autumn while that of trap success was highest in the winter season (Table 5.6).

Table 5.6. The outcomes from Kruskal-Wallis tests on metrics associated with seasonal small mammal captures across eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Metrics	Seasons ($\bar{X} \pm SD$)				Z-test	P-value
	Winter	Spring	Summer	Autumn	DF	
Total Captures	8.47 $\pm$ 11.02	7.97 $\pm$ 8.48	9.80 $\pm$ 7.67	13.70 $\pm$ 11.41	8.63 (3)	0.035*
Trap success	17.70 $\pm$ 13.89	9.43 $\pm$ 9.04	12.07 $\pm$ 9.99	15.93 $\pm$ 16.43	1.88 (3)	0.598
MKA	5.97 $\pm$ 7.65	6.13 $\pm$ 5.98	6.77 $\pm$ 4.65	9.87 $\pm$ 8.09	8.01 (3)	0.046*
Species Richness	1.73 $\pm$ 1.51	1.80 $\pm$ 1.19	1.80 $\pm$ 1.19	2.33 $\pm$ 1.15	5.18 (3)	0.159
Species diversity	0.53 $\pm$ 0.51	0.47 $\pm$ 0.51	0.33 $\pm$ 0.55	0.60 $\pm$ 0.50	3.65 (3)	0.302

* Significant differences when evaluated at $\alpha = 0.05$

The total captures and MKA were significantly higher in the autumn than winter season and higher still in the autumn than the spring season (Table 5.6; Appendix 8). No significant differences were observed between the other seasons for either captures or MKA (Table 5.6; Appendix 8). When evaluated on a feeding guild basis, the total captures and MKA for granivores, herbivores, insectivores and small omnivores were not significantly influenced by seasons (Table 5.7). Total captures for large omnivores were not significantly influenced by seasons, however MKA values for this group were significantly different and higher in summer and autumn than in winter (Table 5.7; Appendix 9).

Table 5.7. The outcomes from Kruskal-Wallis tests on functional guild metrics for total captures and minimum known alive (MKA) values associated with the effect of seasons on small mammal captures within eThekwin KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Metric	Functional guilds	Seasons ($\bar{X} \pm SD$)				Z-test	P-value
		Winter	Spring	Summer	Autumn	DF	
Total captures	Granivores	2.93 $\pm$ 5.83	2.20 $\pm$ 4.00	1.77 $\pm$ 2.67	4.13 $\pm$ 7.14	1.09 (3)	0.779
	Herbivores	0.23 $\pm$ 1.10	0.13 $\pm$ 0.43	0.17 $\pm$ 0.46	0.33 $\pm$ 0.84	2.56 (3)	0.465
	Insectivores	0.90 $\pm$ 1.86	0.60 $\pm$ 1.10	0.63 $\pm$ 1.19	1.80 $\pm$ 2.25	6.86 (3)	0.076
	Large Omnivores	4.87 $\pm$ 8.69	5.47 $\pm$ 6.83	5.97 $\pm$ 6.08	7.40 $\pm$ 7.79	4.95 (3)	0.175
	Small Omnivores	0.07 $\pm$ 0.37	0.20 $\pm$ 0.66	-	0.07 $\pm$ 0.25	3.50 (3)	0.320
MKA	Granivores	2.60 $\pm$ 5.12	1.70 $\pm$ 3.36	1.27 $\pm$ 1.78	3.40 $\pm$ 5.37	1.66 (3)	0.646
	Herbivores	0.13 $\pm$ 0.57	0.10 $\pm$ 0.31	0.27 $\pm$ 0.69	0.14 $\pm$ 0.35	1.65 (3)	0.648
	Insectivores	0.73 $\pm$ 1.48	0.57 $\pm$ 1.01	0.60 $\pm$ 1.16	1.40 $\pm$ 1.75	6.16 (3)	0.104
	Large Omnivores	2.43 $\pm$ 4.01	3.90 $\pm$ 4.58	4.60 $\pm$ 4.83	5.40 $\pm$ 6.10	7.84 (3)	0.049 *
	Small Omnivores	0.07 $\pm$ 0.37	0.20 $\pm$ 0.66	-	0.07 $\pm$ 0.25	3.50 (3)	0.320

* Significant differences when evaluated at $\alpha = 0.05$

Despite the non-significant Kruskal-Wallis test outcomes, the general trends show that the total captures and MKA values for granivores, insectivores and large omnivores were highest in autumn. The total capture value for herbivores was highest in autumn while the MKA value was highest in summer. The total captures and MKA values for small omnivores were highest in spring (Table 5.7).

5.1.4 Grassland condition

Although small mammal capture metrics varied in relation to grassland condition, the grassland in the best condition was not necessarily correlated to the highest number of captures or the most diverse small mammal assemblage. Grassland condition was associated with a significant effect on total captures, MKA and species richness (Table 5.8). However, mean values for trap success and species diversity did not show any significant variation with grassland conditions (Table 5.8).

Table 5.8. The outcomes from Kruskal-Wallis tests on metrics associated with small mammal captures relative to grassland conditions across eThekwin KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Metrics	Grassland Condition ($\bar{X} \pm SD$)				Z-test	P-value
	Primary-Good	Primary-Intermediate	Secondary-Good	Secondary-Intermediate	DF	
Total captures	10.47 $\pm$ 10.93	6.75 $\pm$ 3.20	6.86 $\pm$ 8.67	12.88 $\pm$ 8.30	9.50 (3)	0.023 *
Trap success	13.41 $\pm$ 14.37	6.50 $\pm$ 2.08	10.54 $\pm$ 11.23	13.54 $\pm$ 10.58	2.14 (3)	0.544
MKA	7.47 $\pm$ 743	4.25 $\pm$ 1.50	5.14 $\pm$ 6.13	9.29 $\pm$ 5.87	9.11 (3)	0.028 *
Species richness	1.86 $\pm$ 1.22	2.00 $\pm$ 0.00	1.36 $\pm$ 1.19	2.71 $\pm$ 1.27	14.06 (3)	0.003 *
Species diversity	0.44 $\pm$ 0.53	0.75 $\pm$ 0.50	0.50 $\pm$ 0.51	0.54 $\pm$ 0.51	4.47 (3)	0.215

* Significant differences when evaluated at $\alpha = 0.05$

The data suggest that total captures, trap success, MKA and species richness were higher in secondary-intermediate grasslands than in other grasslands while species diversity values were highest in primary-intermediate grasslands (Table 5.8). However, only total captures, MKA and species richness were significantly influenced by grassland condition (Table 5.8). The total captures were only significantly higher in the primary-good than in the secondary-good grasslands (Table 5.8; Appendix 10). Both total captures and MKA were significantly higher in the secondary-intermediate than in the secondary-good grasslands (Table 5.8; Appendix 10). No significant differences were observed between the other grassland types for either total captures or MKA (Table 5.8; Appendix 10). Species richness was only significantly higher in secondary-intermediate than in secondary-good grasslands and in the secondary-intermediate than in primary-good grasslands (Table 5.8; Appendix 10). No significant differences were observed between the other grasslands for species richness (Table 5.8; Appendix 10).

When evaluated on a feeding guild basis, grassland condition had a significant effect on total captures and MKA for granivores, large omnivores and small omnivores (Table 5.9). The general trends that were observed included total captures and MKA for granivores were highest in the primary intermediate grasslands, herbivore captures and MKA values were highest in secondary-good grasslands, while captures and MKA values for insectivores and small omnivores were highest in the secondary-intermediate grasslands (Table 5.9). For large omnivores, total captures were highest in the primary-good grasslands while the MKA value was highest in the secondary intermediate grasslands (Table 5.9).

Table 5.9. The outcomes from Kruskal-Wallis tests on functional guild metrics for total captures and minimum known alive (MKA) associated with the effect of grassland conditions on small mammal captures within eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Metrics	Functional guilds	Grassland Condition ($\bar{X} \pm SD$)				Z-test	P-value
		Primary-Good	Primary-Intermediate	Secondary-Good	Secondary-Intermediate	DF	
Total captures	Granivores	2.83 $\pm$ 6.39	4.50 $\pm$ 1.73	1.79 $\pm$ 3.25	3.42 $\pm$ 3.74	15.84 (3)	0.001 *
	Herbivores	0.23 $\pm$ 0.53	-	0.36 $\pm$ 1.34	0.04 $\pm$ 0.20	4.73 (3)	0.193
	Insectivores	1.05 $\pm$ 1.86	-	0.71 $\pm$ 1.58	1.29 $\pm$ 1.57	7.00 (3)	0.072
	Large Omnivores	6.47 $\pm$ 6.35	1.25 $\pm$ 0.50	2.57 $\pm$ 4.53	5.04 $\pm$ 5.08	12.42 (3)	0.006 *
	Small Omnivores	-	-	0.11 $\pm$ 0.42	0.29 $\pm$ 0.75	10.66 (3)	0.014 *
MKA	Granivores	2.19 $\pm$ 64.97	3.53 $\pm$ 1.29	1.64 $\pm$ 3.06	2.88 $\pm$ 3.37	15.28 (3)	0.002 *
	Herbivores	0.19 $\pm$ 0.43	-	0.22 $\pm$ 0.80	0.04 $\pm$ 0.20	3.84 (3)	0.280
	Insectivores	0.84 $\pm$ 1.43	-	0.64 $\pm$ 1.47	1.13 $\pm$ 1.36	7.00 (3)	0.072
	Large Omnivores	4.66 $\pm$ 5.32	1.00 $\pm$ 0.00	2.39 $\pm$ 4.01	5.04 $\pm$ 5.08	8.74 (3)	0.033 *
	Small Omnivores	-	-	0.11 $\pm$ 0.42	0.29 $\pm$ 0.75	10.66 (3)	0.014 *

* Significant differences when evaluated at $\alpha = 0.05$

The total captures and MKA for granivores were significantly higher in the secondary-intermediate than the primary-good grasslands, higher in the primary-intermediate than in the primary-good grasslands, higher in the secondary-intermediate than secondary-good grasslands and, primary-intermediate than secondary-good grasslands (Table 5.9; Appendix 11). No significant differences were observed between total captures and MKA for primary-good and secondary-good grasslands and secondary-intermediate and primary-intermediate grasslands (Table 5.9; Appendix 11). For large omnivores, the total captures and MKA were only significantly higher in the primary-good than in the secondary-good grasslands and higher in the secondary-intermediate than in the secondary-good grasslands (Table 5.9; Appendix 11). No significant differences were observed between the other grasslands for large omnivore total captures and MKA (Table 5.9; Appendix 11). The total captures and MKA for small omnivores were only significantly higher in the secondary-intermediate than in primary-good grasslands (Table 5.9; Appendix 11). No significant differences were observed between the other grassland patches for total captures and MKA for small omnivores (Table 5.9; Appendix 11).

5.2 The Influence of regional variation and habitat management regimes (treatments) on small mammal populations.

5.2.1 Regional variation

The total captures, trap success, MKA, species richness and species diversity were significantly influenced by region Kloof (KNR), Hillcrest (GGNR) and Camperdown Rural (DUC) in which sampling was conducted (Table 5.10).

Table 5.10. The outcomes from Kruskal-Wallis tests on metrics associated with small mammals captures relative to regional variation across the sampled eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Metrics	Regions ($\bar{X} \pm SD$)			Z-test	P-value
	Kloof	Hillcrest	Camperdown Rural	DF	
Total captures	5.95 $\pm$ 6.30	11.50 $\pm$ 10.11	12.50 $\pm$ 11.50	10.17 (2)	0.006 *
Trap success	7.53 $\pm$ 7.82	13.20 $\pm$ 11.94	16.88 $\pm$ 15.70	9.66 (2)	0.008 *
MKA	4.40 $\pm$ 4.14	8.10 $\pm$ 7.04	9.05 $\pm$ 7.95	20.81 (2)	<0.001 *
Species richness	1.28 $\pm$ 0.99	1.95 $\pm$ 1.41	2.53 $\pm$ 1.09	24.84 (2)	<0.001 *
Species diversity	0.23 $\pm$ 0.42	0.50 $\pm$ 0.55	0.73 $\pm$ 0.45	10.58 (2)	0.005 *

* Significant differences when evaluated at $\alpha = 0.05$

The Camperdown Rural region yielded higher mean total captures, trap success, MKA, species richness and species diversity (Table 5.10). The total captures were significantly higher in the Camperdown Rural and Hillcrest than in the Kloof region and there were no significant differences between Hillcrest and Camperdown Rural region (Table 5.10; Appendix 12). Significantly greater trap success was achieved in the Camperdown Rural and Hillcrest than in the Kloof region. There was no significant difference in trap success between the Hillcrest and Camperdown Rural regions (Table 5.10; Appendix 12). Significantly greater MKA values were recorded in the Camperdown Rural than in the Kloof or Hillcrest regions. There was no significant difference in MKA values between Hillcrest and Camperdown Rural regions (Table 5.10; Appendix 12). Species richness was significantly higher in the Camperdown Rural than in the Kloof region, it was higher in the Hillcrest than Kloof region, and higher in the Camperdown Rural than in the Hillcrest region (Table 5.10; Appendix 12). The species diversity was significantly higher in the Camperdown Rural than Kloof region, higher in the Hillcrest than Kloof region and higher in the Camperdown Rural than Hillcrest region (Table 5.10; Appendix 12).

Regions had a significant effect on total captures and MKA for granivores, insectivores and large omnivores (Table 5.11). The general trends showed that the total captures and MKA for granivores and insectivores were higher in the Camperdown Rural than Hillcrest and Kloof regions, herbivores lowest in the Kloof and Camperdown Rural regions and highest in the Hillcrest region, whilst such values for large and small omnivores were higher in the Hillcrest than Kloof and Camperdown Rural regions (Table 5.11).

Table 5.11. The outcomes from Kruskal-Wallis tests on functional guild metrics for total captures and minimum known alive (MKA) values associated with the effect of regional variation on small mammal captures within eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Metric	Functional guilds	Regions ($\bar{X} \pm SD$)			Z-test	P-value
		Kloof	Hillcrest	Camperdown Rural	DF	
Total captures	Granivores	-	1.05 $\pm$ 2.50	7.23 $\pm$ 6.74	81.03 (2)	<0.001 *
	Herbivores	0.10 $\pm$ 0.30	0.43 $\pm$ 1.17	0.13 $\pm$ 0.46	3.26 (2)	0.196
	Insectivores	0.35 $\pm$ 0.77	0.78 $\pm$ 1.33	1.83 $\pm$ 2.34	11.86 (2)	0.003 *
	Large Omnivores	5.90 $\pm$ 6.61	8.95 $\pm$ 9.27	2.93 $\pm$ 4.22	10.14 (2)	0.006 *
	Small Omnivores	0.08 $\pm$ 0.35	0.18 $\pm$ 0.59	-	4.19 (2)	0.123
MKA	Granivores	-	1.00 $\pm$ 2.31	5.73 $\pm$ 5.41	77.87 (2)	<0.001 *
	Herbivores	0.10 $\pm$ 0.30	0.31 $\pm$ 0.73	0.08 $\pm$ 0.35	4.76 (2)	0.092
	Insectivores	0.35 $\pm$ 0.77	0.63 $\pm$ 1.10	1.50 $\pm$ 1.87	11.67 (2)	0.003 *
	Large Omnivores	4.13 $\pm$ 4.48	5.85 $\pm$ 5.92	2.28 $\pm$ 3.82	9.18 (2)	0.010 *
	Small Omnivores	0.08 $\pm$ 0.35	0.18 $\pm$ 0.59	-	4.19 (2)	0.123

* Significant differences when evaluated at $\alpha = 0.05$

The total captures and MKA for granivores were significantly higher in the Camperdown Rural than in the Hillcrest region, higher in the Camperdown Rural than in the Kloof region and higher in the Hillcrest than Kloof region (Table 5.11; Appendix 13). Total captures and MKA for insectivores were significantly higher in the Camperdown Rural than Hillcrest region and higher in the Camperdown Rural than Kloof region and did not show any significant differences between the Kloof and Hillcrest regions (Table 5.11; Appendix 13). The total captures and MKA for large omnivores were significantly lower in the Hillcrest than Camperdown Rural region and the comparison was non-significant between Camperdown Rural and Kloof region and between Hillcrest and Kloof regions (Table 5.11; Appendix 13). The total captures and MKA for herbivores and small omnivores were not significantly influenced by regions (Table 5.11; Appendix 13).

5.2.2 Habitat management regimes (hereafter referred to as treatments).

The treatments (control site, inside protected areas and outside (D'MOSS) protected areas) influenced total captures, trap success, MKA, species richness and species diversity (Table 5.12).

Table 5.12. The outcomes from Kruskal-Wallis tests on metrics associated with small mammal captures relative to treatments across eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Metrics	Treatments ($\bar{X} \pm SD$)			Z-test	P-value
	Inside Protected Areas	Outside Protected Areas	Control	DF	
Total captures	11.03 $\pm$ 8.59	6.43 $\pm$ 8.55	12.50 $\pm$ 11.50	14.15 (2)	<0.001 *
Trap success	8.73 $\pm$ 9.75	12.00 $\pm$ 10.93	16.88 $\pm$ 15.70	8.83 (2)	0.012 *
MKA	7.90 $\pm$ 5.83	4.60 $\pm$ 5.86	9.05 $\pm$ 7.95	14.54 (2)	<0.001 *
Species richness	2.08 $\pm$ 1.33	1.15 $\pm$ 1.00	2.53 $\pm$ 1.09	25.92 (2)	<0.001 *
Species diversity	0.40 $\pm$ 0.55	0.33 $\pm$ 0.47	0.73 $\pm$ 0.45	16.54 (2)	<0.001 *

* Significant differences when evaluated at $\alpha = 0.05$

The total captures, trap success, MKA, species richness and species diversity were higher in the control sites than either inside or outside protected areas (Table 5.12). The total captures were significantly higher in the control site than outside protected areas, furthermore, they were significantly higher inside than outside protected areas. No significant differences were observed between control site and inside protected areas (Table 5.12; Appendix 14). The MKA was significantly higher in the control site than outside protected areas and higher inside than outside protected areas (Table 5.12; Appendix 14). The trap success was significantly higher in the control sites than inside protected areas and comparisons were non-significant between inside and outside protected areas and between outside and control sites (Table 5.12; Appendix 14). Species richness was significantly higher in the control site than outside protected areas and was significantly higher inside than outside protected areas. There was no significant difference between the species richness in the control site and inside protected areas (Table 5.12; Appendix 14). The species diversity was significantly higher in the control site than inside protected areas and higher in the control site than outside protected areas. There was no significant difference in species diversity between inside and outside protected areas (Table 5.12; Appendix 14).

The total captures and MKA values for granivores, insectivores and large omnivores were influenced by treatments (Table 5.13).

Table 5.13. The outcomes from Kruskal-Wallis tests on functional guild metrics for total captures and minimum known alive values associated with the effect of habitat management regimes (treatments) on small mammal captures within eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Metric	Functional guilds	Treatments ($\bar{X} \pm SD$)			Z-test	P-value
Metric		Inside Protected Areas	Outside Protected Areas	Control	DF	
Total captures	Granivores	0.35 $\pm$ 0.86	0.70 $\pm$ 2.45	7.23 $\pm$ 6.74	75.92 (2)	<0.001 *
	Herbivores	0.40 $\pm$ 1.17	0.13 $\pm$ 0.33	0.13 $\pm$ 0.46	1.84 (2)	0.398
	Insectivores	0.75 $\pm$ 1.21	0.38 $\pm$ 0.95	1.83 $\pm$ 2.34	13.97 (2)	<0.001 *
	Large Omnivores	11.38 $\pm$ 8.76	3.48 $\pm$ 5.08	2.93 $\pm$ 4.22	27.51 (2)	<0.001 *
	Small Omnivores	0.18 $\pm$ 0.59	0.08 $\pm$ 0.35	-	4.19 (2)	0.123
MKA	Granivores	0.35 $\pm$ 0.86	0.65 $\pm$ 2.25	5.73 $\pm$ 5.41	72.17 (2)	<0.001 *
	Herbivores	0.28 $\pm$ 0.72	0.13 $\pm$ 0.33	0.08 $\pm$ 0.35	3.20 (2)	0.202
	Insectivores	0.65 $\pm$ 1.05	0.33 $\pm$ 0.83	1.50 $\pm$ 1.87	13.89 (2)	<0.001 *
	Large Omnivores	6.73 $\pm$ 5.02	3.25 $\pm$ 5.02	2.28 $\pm$ 3.82	20.00 (2)	<0.001 *
	Small Omnivores	0.18 $\pm$ 0.59	0.08 $\pm$ 0.35	-	4.19 (2)	0.123

* Significant differences when evaluated at $\alpha = 0.05$

The total captures and MKA for granivores were significantly higher in the control site than either inside or outside protected areas (Table 5.13; Appendix 15). The total captures and MKA for insectivores were significantly higher in the control sites than inside or outside protected areas (Table 5.13; Appendix 15). The total captures and MKA for large omnivores were significantly higher inside protected areas than outside protected area or in control site (Table 5.13; Appendix 15). The total captures and MKA for herbivores and small omnivores were not significantly influenced by treatments (Table 5.13; Appendix 15).

5.3 Small Mammal Population Size Estimation per Trapline.

The highest mean values for total captures, MKA and recaptures were associated with *M. natalensis* and *R. dilectus* (Figure 5.3). Of the insectivorous species *C. flavescens* had higher mean total captures and MKA values than *M. varius* (Figure 5.3, Table 5.14).

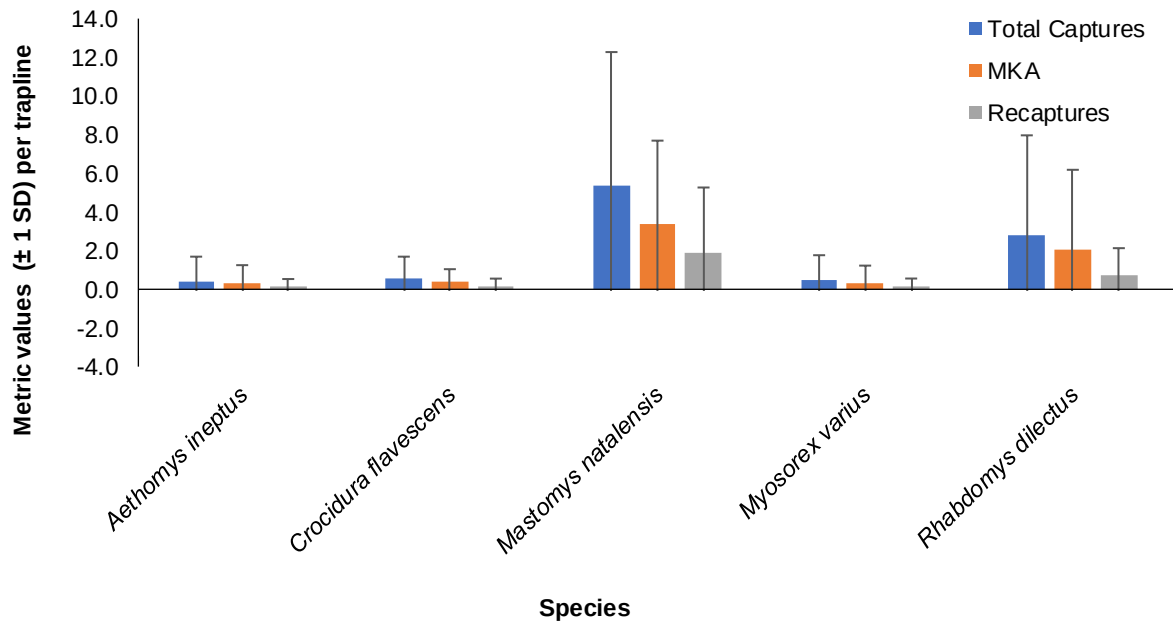


Figure 5.3: Species specific metrics associated with captures of small mammals from eThekwin KwaZulu-Natal Sandstone Sourveld Grasslands (from July 2018 to June 2019).

Some general trends were observed for the mean values relating to total captures, MKA and recaptures including the population estimates and trap success for small mammals (Table 5.14).

Table 5.14: Species-specific metrics associated with the small mammals captured from eThekwin KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Species	Metrics ($\bar{X} \pm SD$)			Population estimates	Trap success %
	Total captures	MKA	Recaptures		
<i>Aethomys ineptus</i>	4.60 ± 1.28	3.30 ± 0.96	1.33 ± 0.41	138	0.30
<i>Mastomys natalensis</i>	5.40 ± 6.90	3.40 ± 4.30	2.00 ± 3.50	1285	3.89
<i>Rhabdomys dilectus</i>	2.80 ± 5.20	2.10 ± 4.10	0.70 ± 1.40	1163	2.01
<i>Crocidura flavescens</i>	0.51 ± 1.00	0.37 ± 0.67	0.15 ± 0.40	74	0.42
<i>Myosorex varius</i>	0.50 ± 1.31	0.30 ± 0.90	0.10 ± 0.40	115	0.34

The population estimates for the rodent species, *M. natalensis* and *R. dilectus* were higher than those for other species (Table 5.14). The trap success for *M. natalensis* and *R. dilectus* was higher than that of other species (Table 5.14). For the insectivorous species, *M. varius* was consistently associated with higher population estimates than *C. flavescens*. However, traps success for *C. flavescens* was higher than that for *M. varius* (Table 5.14)

5.3.1 Elevation

Despite elevation not having a significant influence on small mammal population estimates, except for *R. dilectus*, some general trends were apparent (Table 5.15). The population estimate for *A. ineptus* was highest at low elevations while that of *M. natalensis* was highest at mid elevation and that of *R. dilectus* highest at high elevations. The shrew species, *C. flavescens*, had its highest population estimate at high elevations (Table 5.15). The population estimate for *R. dilectus* was significantly higher at high elevations than mid and low elevations (Table 5.15, Appendix 16).

Table 5.15: The outcomes from Kruskal-Wallis tests of species-specific effects of elevational gradients on small mammal populations in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Species	Population estimates ($\bar{X} \pm SD$)			Z-test	P-value
	Low	Mid	High	DF	
<i>Aethomys ineptus</i>	2.55 ± 9.47	-	0.59 ± 2.78	2.00 (2)	0.368
<i>Mastomys natalensis</i>	10.04 ± 16.90	22.73 ± 38.33	7.14 ± 15.14	1.99 (2)	0.370
<i>Rhabdomys dilectus</i>	-	0.78 ± 1.87	19.12 ± 34.85	43.10 (2)	<0.001 *
<i>Crocidura flavescens</i>	0.35 ± 1.92	0.58 ± 1.58	0.83 ± 2.42	5.52 (2)	0.063

* Significant differences when evaluated at $\alpha = 0.05$

Elevation only had a significant influence on the trap success for *R. dilectus* (Table 5.16; Appendix 16). The general trends for trap success shows that trap success for *A. ineptus* was highest at low elevations. *Mastomys natalensis* and *R. dilectus* had highest trap success at high elevations. For the shrew species, *C. flavescens*, trap success was highest at high elevations (Table 5.16).

Table 5.16: The outcomes from Kruskal-Wallis tests of species-specific effects of elevational gradients on small mammal populations in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Species	Trap success %			Z-test	P-value
	Low	Mid	High	DF	
<i>Aethomys ineptus</i>	0.13	0.01	0.08	2.07 (2)	0.355
<i>Mastomys natalensis</i>	0.80	0.66	1.01	5.40 (2)	0.067
<i>Rhabdomys dilectus</i>	0.00	0.04	1.47	59.77 (2)	<0.001 *
<i>Crocidura flavescens</i>	0.07	0.04	0.16	1.48 (2)	0.478

* Significant differences when evaluated at $\alpha = 0.05$

The trap success for *R. dilectus* was significantly higher at high elevations than mid and low elevations (Table 5.16; Appendix 16).

5.3.2 Slope aspect

Slope aspect had a significant influence on the population estimate for *R. dilectus* and *M. varius* (Table 5.17). The general trends show that the population estimates for *A. ineptus* was highest in the north facing slopes while that of *M. natalensis* was highest in the west facing slopes and that of *R. dilectus* was highest in the south-west facing slopes (Table 5.17). The population estimates for shrew species, *C. flavescens*, was highest in the west facing slopes while that of *M. varius* was highest in the south-west facing slopes (Table 5.17).

The population estimate for *R. dilectus* was significantly lower on the north than on the north-west, north-east and south-east facing slopes, and was also significantly lower in the south than south-east facing slopes and lower in the west than north-west and south-east facing slopes (Table 5.17; Appendix 17). For the shrew species, *M. varius*, the population estimates were significantly lower for the west than for the east, north-east and south-west facing slopes (Table 5.17; Appendix 17). The population estimates for the north facing slopes were significantly lower than those for the south-west, while those for the south-west were significantly higher in the south-west than south-east and north-west facing slopes (Table 5.17, Appendix 17).

Table 5.17: The outcomes from Kruskal -Wallis tests of species-specific effects of slope aspect on small mammal populations in eThekwini KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Species	Population estimates ($\bar{X} \pm SD$)								Z-test	P-value
	East	North	North-East	North-West	South	South-East	South-West	West	DF	
<i>Aethomys ineptus</i>	-	10.25 $\pm$ 19.75	0.67 $\pm$ 1.78	0.15 $\pm$ 0.94	-	1.84 $\pm$ 5.09	-	0.60 $\pm$ 2.68	11.34 (7)	0.124
<i>Mastomys natalensis</i>	0.17 $\pm$ 0.58	0.34 $\pm$ 0.64	6.86 $\pm$ 12.55	11.92 $\pm$ 28.07	-	13.91 $\pm$ 21.74	13.03 $\pm$ 29.08	19.72 $\pm$ 17.85	4.67 (7)	0.700
<i>Rhabdomys dilectus</i>	3.29 $\pm$ 6.54	-	18.85 $\pm$ 34.38	4.84 $\pm$ 12.02	-	21.21 $\pm$ 27.90	31.69 $\pm$ 72.52	5.53 $\pm$ 16.06	16.38 (7)	0.022 *
<i>Crocidura flavescens</i>	1.00 $\pm$ 3.46	-	0.17 $\pm$ 0.58	0.38 $\pm$ 1.20	0.50 $\pm$ 1.00	0.63 $\pm$ 1.59	0.25 $\pm$ 0.71	1.60 $\pm$ 3.82	5.77 (7)	0.567
<i>Myosorex varius</i>	-	-	2.13 $\pm$ 5.15	0.50 $\pm$ 1.62	-	2.09 $\pm$ 4.31	4.50 $\pm$ 6.21	-	13.57 (7)	0.059 *

* Significant differences when evaluated at $\alpha = 0.05$

The trap success for *M. natalensis* and *R. dilectus* were significantly influenced by slope aspect (Table 5.18). The trap success for *A. ineptus* was highest on the north facing slopes, that of *M. natalensis* was highest on the west facing slopes and that of *R. dilectus* was highest in the south-east facing slope (Table 5.18). The trap success for *C. flavescens* was highest on the west facing slopes while that of *M. varius* was highest on the south-east facing slopes (Table 5.18).

The trap success for *M. natalensis* was significantly lower on the east than on the north-west, south-east and west facing slopes (Table 5.18; Appendix 17). The trap success for west facing slopes was significantly higher than for south, north-east, north, south-west, north-west and south-east facing slopes (Table 5.18; Appendix 17). For *R. dilectus*, the trap success was significantly lower on the north than north-west, south-west, south-east facing slopes (Table 5.18; Appendix 17). The trap success on the south-east facing slopes was significantly higher than on the south, east, west and north-west facing slopes (Table 5.18, Appendix 17).

Table 5.18: The outcomes from Kruskal-Wallis tests of species-specific effects of slope aspect on small mammal populations in eThekwini KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Species	Trap success %								Z-test	P-value
	East	North	North-East	North-West	South	South-East	South-West	West	DF	
<i>Aethomys ineptus</i>	0.01	0.08	0.02	0.02	0.00	0.05	0.00	0.02	13.12 (7)	0.069
<i>Mastomys natalensis</i>	0.02	0.05	0.16	0.78	0.01	0.34	0.18	0.91	30.89 (7)	<0.001 *
<i>Rhabdomys dilectus</i>	0.08	0.00	0.35	0.30	0.00	0.46	0.15	0.17	18.03 (7)	0.010 *
<i>Crocidura flavescens</i>	0.02	0.01	0.02	0.05	0.02	0.05	0.01	0.08	12.31 (7)	0.091
<i>Myosorex varius</i>	0.00	0.00	0.05	0.04	0.00	0.08	0.06	0.00	0.00 (7)	1.000

*Significant differences when evaluated at $\alpha = 0.05$

5.3.3 Seasonal variation

The seasons did not have a significant influence on small mammal population estimates (Table 5.19).

Table 5.19: The outcomes from Kruskal -Wallis tests of species-specific effects of seasons on small mammal populations in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Species	Population estimates ($\bar{X} \pm SD$)				Z-test	P-value
	Winter	Spring	Summer	Autumn	DF	
<i>Aethomys ineptus</i>	0.72 $\pm$ 2.43	2.33 $\pm$ 10.25	0.87 $\pm$ 3.78	0.67 $\pm$ 3.65	0.58 (3)	0.901
<i>Mastomys natalensis</i>	4.13 $\pm$ 10.65	10.02 $\pm$ 15.42	20.05 $\pm$ 34.47	8.64 $\pm$ 15.55	4.16 (3)	0.245
<i>Rhabdomys dilectus</i>	10.85 $\pm$ 25.82	6.79 $\pm$ 15.78	3.20 $\pm$ 5.43	17.91 $\pm$ 42.10	0.54 (3)	0.909
<i>Crocidura flavescens</i>	0.58 $\pm$ 2.23	0.40 $\pm$ 1.22	0.40 $\pm$ 2.19	1.13 $\pm$ 2.66	0.32 (3)	0.957
<i>Myosorex varius</i>	0.65 $\pm$ 3.20	0.40 $\pm$ 2.19	0.73 $\pm$ 2.43	2.05 $\pm$ 4.03	2.88 (3)	0.411

* Significant differences when evaluated at $\alpha = 0.05$

However, some general trends show that the population estimate for *A. ineptus* was highest in the spring season while that of *M. natalensis* was highest in the summer season and that of *R. dilectus* highest in the autumn season (Table 5.19). The shrew species, *C. flavescens* and *M. varius* had highest population estimates in the autumn season (Table 5.19)

Table 5.20: The outcomes from Kruskal-Wallis tests of species-specific effects of seasons on small mammal populations in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Species	Trap success %				Z-test	P-value
	Winter	Spring	Summer	Autumn	DF	
<i>Aethomys ineptus</i>	0.05	0.08	0.05	0.02	3.65 (3)	0.302
<i>Mastomys natalensis</i>	0.35	0.59	0.72	0.8	8.87 (3)	0.031 *
<i>Rhabdomys dilectus</i>	0.43	0.28	0.21	0.58	1.47 (3)	0.690
<i>Crocidura flavescens</i>	0.07	0.05	0.04	0.11	7.53 (3)	0.057
<i>Myosorex varius</i>	0.04	0.03	0.05	0.11	0.00 (3)	1.000

* Significant differences when evaluated at $\alpha = 0.05$

The seasons only had a significant influence on the trap success for *M. natalensis* (Table 5.20). The general trends observed show that the trap success for *A. ineptus* was highest in the spring season while that of *M. natalensis* and *R. dilectus* was highest in the autumn season (Table 5.20). The trap success for the shrew species, *C. flavescens* and *M. varius* were highest in the autumn season (Table 5.20).

The trap success for *M. natalensis* was significantly lower in the winter than in either the autumn or summer seasons (Table 5.20; Appendix 18). The trap success for *C. flavescens*, was significantly higher in the autumn than summer and spring seasons (Table 5.20; Appendix 18).

5.3.4 Grassland condition

Grassland condition only appeared to have a significant influence on the population estimates for *M. varius* and *R. dilectus* (Table 5.21).

Table 5.21: The outcomes from Kruskal-Wallis tests of species-specific effects of grassland conditions on small mammal populations in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Species	Population estimates ($\bar{X} \pm SD$)				Z-test	P-value
	Primary-Good	Primary-Intermediate	Secondary-Good	Secondary-Intermediate	DF	
<i>Aethomys ineptus</i>	2.05 $\pm$ 7.90	-	-	0.25 $\pm$ 1.22	5.15 (3)	0.161
<i>Mastomys natalensis</i>	12.80 $\pm$ 19.93	0.50 $\pm$ 1.00	9.35 $\pm$ 30.82	8.42 $\pm$ 14.29	2.07 (3)	0.557
<i>Rhabdomys dilectus</i>	8.87 $\pm$ 22.22	9.88 $\pm$ 8.37	10.46 $\pm$ 39.51	10.94 $\pm$ 19.60	14.37 (3)	0.002 *
<i>Crocidura flavescens</i>	0.49 $\pm$ 1.78	-	0.36 $\pm$ 1.22	1.42 $\pm$ 3.51	4.05 (3)	0.257
<i>Myosorex varius</i>	0.92 $\pm$ 3.19	-	1.29 $\pm$ 3.78	0.83 $\pm$ 2.04	18.11 (3)	<0.001 *

* Significant differences when evaluated at $\alpha = 0.05$

The population estimates for *A. ineptus* and *M. natalensis* were highest in the primary-good grasslands while that of *R. dilectus* was highest in the secondary-intermediate grasslands (Table 5.17). The shrew species, *C. flavescens* had its highest population estimate in the secondary-intermediate grasslands while that of *M. varius* was highest in the secondary-good grasslands (Table 5.17). The population estimate for *R. dilectus* was significantly lower in the primary-good than in the secondary-intermediate and primary-intermediate grasslands, and significantly lower in the primary-intermediate than in the secondary-good grasslands and significantly higher in the secondary-intermediate than primary-intermediate grasslands (Table 5.17; Appendix 19). The population estimate for *M. varius* was significantly higher in the primary-good than in the primary-intermediate grasslands, higher in the secondary-intermediate than in the primary-intermediate grasslands, and higher in the secondary-good than in the primary-intermediate grasslands (Table 5.17; Appendix 19).

The grassland conditions had a significant influence on the trap success for *A. ineptus*, *R. dilectus* and *C. flavescens* (Table 5.22). The trap success for *A. ineptus*, *M. natalensis* and *R. dilectus* was highest in the primary-good grasslands while that of

shrew species, *C. flavescens* and *M. varius*, were also highest in the primary-good grasslands (Table 5.22).

Table 5.22: The outcomes from Kruskal-Wallis test for species-specific effects of grassland conditions on small mammal populations in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Species	Trap success %				Z-test	P-value
	Primary-Good	Primary-Intermediate	Secondary-Good	Secondary-Intermediate	DF	
<i>Aethomys ineptus</i>	0.19	0.01	-	0.02	7.93 (3)	0.048 *
<i>Mastomys natalensis</i>	1.47	0.02	0.35	0.63	7.23 (3)	0.065
<i>Rhabdomys dilectus</i>	0.81	0.08	0.24	0.38	16.17 (3)	0.001 *
<i>Crocidura flavescens</i>	0.15	-	0.02	0.09	8.49 (3)	0.037 *
<i>Myosorex varius</i>	0.13	-	0.06	0.04	0.00 (3)	1.000

* Significant differences when evaluated at $\alpha = 0.05$

The trap success for *A. ineptus*, was significantly lower in the secondary-good than in primary-good grasslands (Table 5.22; Appendix 19). The trap success for *R. dilectus* was significantly higher in the primary-good than in the secondary-intermediate and primary-intermediate grasslands and higher in the secondary-good than in the secondary-intermediate and primary-intermediate grasslands (Table 5.22; Appendix 19). The trap success for *C. flavescens*, was significantly lower in the secondary-good than in the primary-good and secondary-intermediate grasslands (Table 5.22; Appendix 19).

5.3.5 Regional variation

All species, except for *A. ineptus* and *C. flavescens* showed a significant variation in population estimates in relation to region (Table 5.23). The general trends showed that *A. ineptus* had highest population estimate in the Kloof region while that of *M. natalensis* was highest in the Hillcrest region and that of *R. dilectus* higher in the Camperdown Rural region (Table 5.32). The insectivorous species, *C. flavescens* had its highest population estimates in the Hillcrest region (Table 5.23). *Myosorex varius* was not included in the analysis, as it was recorded only at the Camperdown Rural region which would have biased group comparisons and violated assumptions of the statistical analyses.

Table 5.23: The outcomes from Kruskal-Wallis test of species-specific effects of regional variation on small mammal populations in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Species	Population estimates ($\bar{X} \pm SD$)			Z-test	P-value
	Kloof	Hillcrest	Camperdown	DF	
<i>Aethomys ineptus</i>	2.55 ± 9.47	-	0.89 ± 3.38	5.26 (2)	0.072
<i>Mastomys natalensis</i>	10.04 ± 16.91	16.52 ± 29.41	5.56 ± 15.14	6.37 (2)	0.041 *
<i>Rhabdomys dilectus</i>	-	3.15 ± 11.54	25.92 ± 39.62	74.48 (2)	<0.001 *
<i>Crocidura flavescens</i>	0.35 ± 1.92	1.24 ± 3.03	0.30 ± 0.72	5.06 (2)	0.080

* Significant differences when evaluated at $\alpha = 0.05$

The population estimate for *M. natalensis* was significantly lower in the Camperdown Rural region than either the Hillcrest or Kloof regions, while that of *R. dilectus* was significantly higher in the Camperdown Rural region than in either the Kloof or Hillcrest regions (Table 5.23; Appendix 20).

Table 5.24: The outcomes from Kruskal-Wallis test of species-specific effects of regional variation on small mammal populations in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Species	Trap success %			Z-test	P-value
	Kloof	Hillcrest	Camperdown Rural	DF	
<i>Aethomys ineptus</i>	0.13	0.01	0.08	6.23 (2)	0.044 *
<i>Mastomys natalensis</i>	0.80	1.20	0.46	8.57 (2)	0.014 *
<i>Rhabdomys dilectus</i>	0.00	0.21	1.30	79.26 (2)	<0.001 *
<i>Crocidura flavescens</i>	0.01	0.13	0.00	1.22 (2)	0.545

* Significant differences when evaluated at $\alpha = 0.05$

The trap success for *A. ineptus*, *M. natalensis* and *R. dilectus* were significantly influenced by regions (Table 5.24). Trap success for *A. ineptus* was highest in the Kloof region, *M. natalensis* in the Hillcrest region while *R. dilectus* was highest in the Camperdown Rural region (Table 5.24). *Crocidura flavescens* had highest trap success in the Hillcrest region (Table 5.24). The trap success for *A. ineptus* was significantly lower in the Hillcrest region than in the Kloof and Camperdown Rural regions, while that of *M. natalensis* was significantly higher in the Hillcrest than Camperdown Rural region (Table 5.24; Appendix 20). The trap success for *R. dilectus* was significantly higher in the Camperdown Rural than Kloof and Hillcrest regions (Table 5.24; Appendix 20).

5.3.6 Habitat management regimes (treatments)

Treatment only had a significant influence on the populations of two of the focal species, *M. natalensis* and *R. dilectus* (Table 5.25). The population estimates for *A. ineptus* and *M. natalensis* were highest inside protected areas while that of *R. dilectus* were highest in the control site. The population estimates for the shrew species, *C. flavescens*, was highest inside protected areas (Table 5.25). *Myosorex varius* was not included in the analysis, as it was recorded exclusively at the control site, which would have biased group comparisons and violated assumptions of the statistical analyses.

Table 5.25: The outcomes from Kruskal-Wallis test of species-specific effects of habitat management regimes (treatments) on small mammal populations in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Metrics	Population Estimates ($\bar{X} \pm SD$)			Z-test	P-value
	Inside Protected Area	Outside Protected Area	Control	DF	
<i>Aethomys ineptus</i>	2.55 ± 9.47	-	0.89 ± 3.38	5.26 (2)	0.072
<i>Mastomys natalensis</i>	17.79 ± 26.77	8.77 ± 20.35	5.56 ± 15.14	6.21 (2)	0.045 *
<i>Rhabdomys dilectus</i>	0.39 ± 1.36	2.76 ± 11.55	25.92 ± 39.62	73.45 (2)	<0.001 *
<i>Crocidura flavescens</i>	0.94 ± 2.82	0.65 ± 2.28	0.30 ± 0.72	2.22 (2)	0.329

* Significant differences when evaluated at $\alpha = 0.05$

The population estimates for *M. natalensis* were significantly lower in the control site than either inside or outside protected areas while that of *R. dilectus* were significantly higher in the control site than either inside or outside protected areas (Table 5.25; Appendix 21).

The trap success for *A. ineptus*, *M. natalensis* and *R. dilectus* were significantly influenced by treatments (Table 5.26). The general trends show that trap success for *A. ineptus* and *M. natalensis* were highest inside protected areas, while that of *R. dilectus* was highest in the control site (Table 5.26). The trap success for shrew species, *C. flavescens* was highest inside protected areas (Table 5.26).

Table 5.26: The outcomes from Kruskal-Wallis test of species-specific effects of habitat management regimes (treatments) on small mammal populations in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Species	Trap Success %			Z-test	P-value
	Inside Protected Area	Outside Protected Area	Control	DF	
<i>Aethomys ineptus</i>	0.13	0.01	0.00	6.23 (2)	0.044 *
<i>Mastomys natalensis</i>	1.35	0.65	0.46	15.35 (2)	<0.001 *
<i>Rhabdomys dilectus</i>	0.00	0.15	1.30	73.81 (2)	<0.001 *
<i>Crocidura flavescens</i>	0.13	0.07	0.07	4.80 (2)	0.091

* Significant differences when evaluated at $\alpha = 0.05$

The trap success for *A. ineptus* was significantly higher inside than outside protected areas and higher outside protected areas than in control sites (Table 5.26; Appendix 21). The trap success for *M. natalensis* was significantly higher inside than outside protected areas or in control sites and that for *R. dilectus* was significantly higher in the control site than inside or outside protected areas (Table 5.26; Appendix 21).

5.4. Small mammal demographic responses to environmental gradients and habitat management regimes (treatments).

The small mammal demographic traits and morphometrics were examined in relation to environmental gradients and habitat characteristics, with patterns in age, reproductive status, sex and head-body size interpreted across elevation, slope aspect, seasonality, grassland condition, regional effects, and habitat treatments.

5.4.1 Elevation

General trends were observed in small mammal age categories, sex ratios, and reproductive status along the elevational gradient (Table 5.27).

Table 5.27. The total captures of focal small mammal species in relation to elevational gradients in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Elevation	Relative Age				Sex							
	Juv	Subad	Ad	Imp	Perf	Preg	Lact	Total	Ab	Males S-scrot	Scrot	Total
<i>Aethomys ineptus</i>												
Low	-	1	18	4	6	-	1	11	2	5	1	8
Mid	-	-	1	-	-	-	-	-	1	-	-	1
High	-	1	9	1	1	1	3	6	3	1	-	4
Total	-	2	28	5	7	1	4	17	6	6	1	13
<i>Mastomys natalensis</i>												
Low	25	36	82	36	15	10	13	74	34	16	19	69
Mid	13	16	99	35	19	5	4	63	45	15	5	65
High	10	49	113	33	34	1	18	86	52	25	9	86
Total	48	101	294	104	68	16	35	223	131	56	33	220
<i>Rhabdomys dilectus</i>												
Low	-	-	-	-	-	-	-	-	-	-	-	-
Mid	1	2	12	3	4	-	-	22	4	4	2	10
High	20	81	149	47	61	11	12	381	83	29	14	126
Total	21	83	161	50	65	11	12	403	87	33	16	136
<i>Crocidura flavescens</i>												
Low	-	-	15			6				9		
Mid	-	-	7			3				4		
High	1	3	27			14				17		
Total	1	3	49			23				30		
<i>Myosorex varius</i>												
Low	-	-	-			-				-		
Mid	-	-	-			-				-		
High	7	2	33			20				22		
Total	7	2	33			20				22		

* Relative age classes: Juv = juvenile; Subad = subadult; Ad = adult.

Female reproductive status: Imp = imperforate; Perf = perforate; Preg = pregnant; Lact = lactating.

Male reproductive status: Ab = abdominal; S-scrot = semi-scrotal; Scrot = scrotal.

Dashes (-) indicate zero captures or not applicable.

¹ Reproductive status and sex could not be determined for shrew species; therefore, reproductive-status categories are not reported for *Crocidura flavescens* and *Myosorex varius*.

The *A. ineptus* captures comprised subadults and adults, with no juveniles recorded during the study. Adult captures were highest at low elevations, while subadults were recorded at both low and high elevations at very low abundances (Table 5.27). All three age categories of *M. natalensis* were captured during the study. Captures of adults and subadults were highest at high elevations, while juvenile captures were highest at low elevations (Table 5.27). The relative age of *M. natalensis* was significantly influenced by elevation ($B = 51.77$, $\chi^2 = 23.58$, $p < 0.001$; Appendix 22 and 23). Post-hoc comparisons indicated that adult captures were significantly higher at mid elevations than at both low and high elevations, while juvenile captures were significantly lower at high elevations than at mid elevations (Table 5.27; Appendix 23). *Rhabdomys dilectus* captures comprised all three age categories recorded from mid and high elevations, with the highest number of adults reported from high elevations (Table 5.27). Juvenile, subadult and adult of *C. flavescens* captures were primarily

recorded at high elevations, while *M. varius* subadult and adult captures were only recorded at high elevations (Table 5.27).

In *A. ineptus*, the number of females was slightly higher than that of males (Table 5.27). Both sexes were more abundant at lower elevations. Samples sizes for the various reproductive state categories were generally too low to allow meaningful inferences regarding demographic trends. For *M. natalensis*, captures of abdominal and semi-scrotal males were highest at high elevations, while scrotal males were highest at low elevations (Table 5.27). Elevation had a significant association with the reproductive status of *M. natalensis* ($B = 112.43$, $\chi^2 = 36.60$, $p < 0.001$; Appendix 22). Post-hoc comparisons indicated that scrotal male captures were significantly higher at low elevations than at mid elevations (Appendix 24). The imperforate female captures were greatest at low elevations, perforate female captures at high elevations, pregnant females at low elevations, while lactating females were highest at high elevations (Table 5.27).

For both shrew species, more male captures were recorded than females (Table 5.27).

The head and body length value for *A. ineptus* was highest at high elevations while that of *M. natalensis* was highest at mid elevations (Table 5.28). The head and body length values for shrew species, *C. flavescens* were highest at low elevations (Table 5.28). Elevation seemed to have a significant influence on the head and body length values for *A. ineptus*, *M. natalensis* and *C. flavescens* (Table 5.28).

Table 5.28. The outcomes from Kruskal-Wallis tests of species-specific effects of elevational gradients on small mammal size (combined head and body length (mm)) in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Metrics	Head and Body Length (mm) ($\bar{X} \pm SD$)			Z-test	P-value
	Low	Mid	High	DF	
<i>Aethomys ineptus</i>	121.21 $\pm$ 23.05	-	146.00 $\pm$ 16.64	9.01 (2)	0.011 *
<i>Mastomys natalensis</i>	96.94 $\pm$ 21.28	102.38 $\pm$ 20.72	96.77 $\pm$ 19.70	8.76 (2)	0.013 *
<i>Crocidura flavescens</i>	101.67 $\pm$ 11.91	96.43 $\pm$ 6.13	80.58 $\pm$ 21.87	10.88 (2)	0.004 *

* Significant differences when evaluated at $\alpha = 0.05$

The head and body length value for *A. ineptus* was significantly higher at high than at low elevations and showed no significant differences between mid and low or mid and high elevations (Table 5.28; Appendix 26). The head and body mean value for *M. natalensis* was significantly higher at mid than at high elevations and at mid than at low elevations (Table 5.28; Appendix 26). The head and body mean value for the shrew

species, *C. flavescens*, was significantly higher at low than at high elevations and showed no significant differences between high and mid elevation or between mid and low elevations (Table 5.28; Appendix 26).

5.4.2 Slope Aspect

The number of captures for adults for *A. ineptus* were highest on north facing slopes, while that of *M. natalensis* were highest on north-west facing slopes and that of *R. dilectus* were highest on south-east facing slopes (Table 5.29). For the shrew species, *C. flavescens*, the number of captures for adults were highest on west facing slopes, while that of *M. varius* were highest on south-west facing slopes (Table 5.29). The number of captures for subadults of *A. ineptus* were highest on north and south-east facing slopes, while that of *M. natalensis* were highest on west facing slopes and that of *R. dilectus* on south-east facing slopes (Table 5.29). For shrew species, *C. flavescens* and *M. varius*, the number of captures for subadults were highest on south-east facing slopes (Table 5.29). The juvenile captures for *A. ineptus* were not recorded at all on any of the slopes, while those of *M. natalensis* were highest on west facing slopes and those of *R. dilectus* were highest on north-east facing slopes (Table 5.29). For the shrew species, the number of captures for juvenile *C. flavescens* were highest on south-east facing slopes, while no juvenile *M. varius* were captured (Table 5.29). Slope aspect only had a significant association with the age of *M. natalensis* ($B = 74.27$, $\chi^2 = 29.75$, $p = 0.008$) and *R. dilectus* ($B = 60.53$, $\chi^2 = 25.89$, $p = 0.004$) (Appendix 27). For *M. natalensis*, the number of captures for adults were significantly higher on the north-west facing slopes than on the west facing slopes, while the number of captures for juveniles were significantly lower on the south-east facing slopes than on the west facing slopes (Appendix 28). For *R. dilectus*, the number of captures for adults were significantly higher on the north-west facing slope than on the west facing slope (Appendix 28).

Table 5.29. The total captures of focal small mammal species in relation to slope aspect in eThekwinini KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Slope Aspect	Relative Age				Sex							
	Juv	Subad	Ad	Imp	Perf	Preg	Lact	Total	Ab	Males S-scr	Scrot	Total
<i>Aethomys ineptus</i>												
East	-	-	1	-	1	-	-	1	-	-	-	-
West	-	-	3	-	1	-	-	1	1	1	-	2
North	-	1	11	3	3	-	1	7	1	4	-	5
South	-	-	-	-	-	-	-	-	-	-	-	-
North-East	-	-	4	1	1	-	-	2	1	-	1	2
North-West	-	-	2	1	1	-	-	2	-	-	-	-
South-East	-	1	7	-	-	1	3	4	3	1	-	4
South-West	-	-	-	-	-	-	-	-	-	-	-	-
Total	-	2	28	5	7	1	4	17	6	6	1	13
<i>Mastomys natalensis</i>												
East	-	-	4	-	-	-	-	-	2	2	-	4
West	27	43	91	36	32	3	14	85	48	21	7	76
North	1	2	8	3	1	-	1	5	4	-	2	6
South	-	1	1	-	1	-	-	1	-	1	-	1
North-East	2	11	17	5	5	4	1	15	11	3	1	15
North-West	12	21	107	38	15	9	6	68	36	19	17	72
South-East	2	18	40	12	10	0	12	34	16	5	5	26
South-West	4	5	26	10	4	0	1	15	14	5	1	20
Total	48	101	294	104	68	16	35	223	131	56	33	220
<i>Rhabdomys dilectus</i>												
East	-	5	8	4	-	2	-	6	6	1	-	7
West	2	10	19	10	2	1	-	13	15	2	1	18
North	-	-	-	-	-	-	-	-	-	-	-	-
South	-	-	-	-	-	-	-	-	-	-	-	-
North-East	4	9	46	4	19	3	5	31	19	4	5	28
North-West	2	2	54	8	13	-	3	24	23	9	2	34
South-East	2	18	57	13	22	3	3	41	21	10	5	36
South-West	2	8	16	4	10	-	2	16	4	4	2	10
Total	12	52	200	43	66	9	13	131	88	30	15	133
<i>Crociodura flavescens</i>												
East	-	-	6			3				3		
West	-	-	16			8				8		
North	-	-	2			1				1		
South	-	-	4			1				3		
North-East	-	-	4			2				2		
North-West	-	1	9			3				7		
South-East	1	2	6			3				6		
South-West	-	-	2			2				-		
Total	1	3	49			23				30		
<i>Myosorex varius</i>												
East	-	-	-			-				-		
West	-	-	2			1				1		
North	-	-	-			-				-		
South	-	-	-			-				-		
North-East	-	-	9			5				4		
North-West	-	2	7			6				3		
South-East	-	3	10			4				9		
South-West	-	-	12			5				7		
Total	-	5	40			21				24		

* Relative age classes: Juv = juvenile; Subad = subadult; Ad = adult.

Female reproductive status: Imp = imperforate; Perf = perforate; Preg = pregnant; Lact = lactating.

Male reproductive status: Ab = abdominal; S-scr = semi-scrotal; Scrot = scrotal.

Dashes (-) indicate zero captures or not applicable.

¹ Reproductive status and sex could not be determined for shrew species; therefore, reproductive-status categories are not reported for *Crociodura flavescens* and *Myosorex varius*.

The number of captures for female *A. ineptus* were highest on north facing slopes, while that of *M. natalensis* were highest on west facing slopes and that of *R. dilectus* were highest on south-east facing slopes (Table 5.29). For shrew species, *C. flavescens*, the number of captures for females were highest on west facing slopes, while that of *M. varius* were highest on north-west facing slopes (Table 5.29). The number of captures for males for *A. ineptus* were highest on north facing slopes, while that of *M. natalensis* were highest on west facing slopes and that of *R. dilectus* were highest on south-east facing slopes (Table 5.29). The number of captures for males for *C. flavescens* were highest on west facing slopes, while that of *M. varius* were highest on south-east facing slopes than other slope orientations (Table 5.29). Slope aspect did not have a significant association with the sex ratio of small mammal species (Appendix 27 and 30).

The number of captures for *A. ineptus* abdominal males were highest on south-east facing slopes, *M. natalensis* were highest on west facing slopes, and most *R. dilectus* abdominal males were recorded on north-west facing slopes (Table 5.29). The number of captures for semi-scrotal males for *A. ineptus* were highest on north facing slopes while that of *M. natalensis* were highest on west facing slopes and that of *R. dilectus* were highest on south-east facing slopes (Table 5.29). The number of captures for scrotal males for *A. ineptus* were highest on north-east facing slopes while that of *M. natalensis* were highest on north-west facing slopes and that of *R. dilectus* were highest on north-east and south-east facing slopes (Table 5.29). The number of captures for imperforate females for *A. ineptus* were highest on north facing slopes while that of *M. natalensis* were highest on north-west facing slopes and that of *R. dilectus* were highest on south-east facing slopes (Table 5.29).

The number of captures for *A. ineptus* abdominal males were highest on south-east facing slopes, *M. natalensis* were highest on west facing slopes, and most *R. dilectus* abdominal males were recorded on north-west facing slopes (Table 5.29). The number of captures for semi-scrotal males for *A. ineptus* were highest on north facing slopes, while that of *M. natalensis* were highest on west facing slopes and that of *R. dilectus* were highest on south-east facing slopes (Table 5.29). The number of captures for scrotal males for *A. ineptus* were highest on north-east facing slopes, while that of *M. natalensis* were highest on north-west facing slopes and that of *R. dilectus* were highest on north-east and south-east facing slopes (Table 5.29). The number of

captures for imperforate females for *A. ineptus* were highest on north facing slopes, while that of *M. natalensis* were highest on north-west facing slopes and that of *R. dilectus* were highest on south-east facing slopes (Table 5.29). The number of captures for perforate females for *A. ineptus* were highest on north facing slopes, while that of *M. natalensis* were highest on west facing slopes and that of *R. dilectus* were highest on south-east facing slopes (Table 5.29). The number of captures for pregnant females for *A. ineptus* were highest on south-east facing slopes, while that of *M. natalensis* were highest on north-east facing slopes and that of *R. dilectus* were highest on north-east facing slopes (Table 5.29). The number of captures for lactating females for *A. ineptus* were highest on north facing slopes, while that of *M. natalensis* were highest on west facing slopes and that of *R. dilectus* were highest on north-east facing slopes than other slope orientations (Table 5.29). The slope aspect had a significant association with the reproductive status of *M. natalensis* ($B = 178.02$, $\chi^2 = 65.78$, $p = 0.011$) and *R. dilectus* ($B = 147.51$, $\chi^2 = 54.73$, $p = 0.004$) (Appendix 27). For *M. natalensis*, only the number of captures for lactating females was significantly higher on south-east facing slopes than on west facing slopes (Appendix 29). For *R. dilectus*, none of the individual reproductive status categories showed a significant difference between slope orientations at $p < 0.05$ (Appendix 29).

The head and body length values for *A. ineptus* and *M. natalensis* were highest on the south-east facing slopes while that of *R. dilectus* were highest on the north-west facing slopes (Table 5.30). The head and body length values for the shrew species, *C. flavescens*, were higher on the south-west facing slopes while those of *M. varius* were higher on the north-west facing slopes (Table 5.30). The slope aspect had a significant effect on the head and body values for *A. ineptus*, *M. natalensis*, *R. dilectus*, and shrew species, *C. flavescens* and *M. varius* (Table 5.30).

Table 5.30. The outcomes from Kruskal-Wallis tests of species-specific associations of slope aspect with small mammal length (combined head and body length (mm)) in eThekwini KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Species	Head and Body Length (mm) ($\bar{X} \pm \text{SD}$)								Z-test	P-value
	East	North	North-East	North-West	South	South-East	South-West	West	DF	
<i>Aethomys ineptus</i>	-	124.42 $\pm$ 26.17	123.75 $\pm$ 18.87	108.00 $\pm$ 2.83	-	148.13 $\pm$ 10.68	-	104.67 $\pm$ 3.51	13.41 (5)	0.020 *
<i>Mastomys natalensis</i>	96.25 $\pm$ 8.54	102.36 $\pm$ 21.10	103.57 $\pm$ 20.12	100.72 $\pm$ 18.60	87.50 $\pm$ 10.61	71.85 $\pm$ 14.02	106.46 $\pm$ 26.45	93.39 $\pm$ 21.28	23.35 (7)	0.001 *
<i>Rhabdomys dilectus</i>	88.54 $\pm$ 12.53	-	86.14 $\pm$ 16.40	99.07 $\pm$ 13.37	-	90.16 $\pm$ 11.17	89.85 $\pm$ 10.44	86.65 $\pm$ 16.53	37.06 (5)	<0.001 *
<i>Crocidura flavescens</i>	109.00 $\pm$ 11.97	98.50 $\pm$ 3.54	93.25 $\pm$ 11.09	86.90 $\pm$ 20.91	98.25 $\pm$ 14.08	60.00 $\pm$ 18.96	111.50 $\pm$ 4.95	90.56 $\pm$ 9.53	23.98 (7)	0.001 *
<i>Myosorex varius</i>	-	-	66.78 $\pm$ 21.62	85.11 $\pm$ 16.65	-	99.25 $\pm$ 17.54	56.75 $\pm$ 22.05	53.00 $\pm$ 32.53	11.01 (4)	0.026 *

* Significant differences when evaluated at $\alpha = 0.05$

The head and body length value for *A. ineptus* was significantly lower on the west facing slopes than on the south-east facing slopes and significantly higher than those on the east facing slopes (Table 5.30; Appendix 31). The head and body length values were also significantly lower on the north-west facing slopes than on the south-east slopes and significantly higher on the north-west than that on the east facing slopes. The north facing grassland slope had significantly lower head and body length values than the south-east facing slopes (Table 5.30; Appendix 31). The head and body length values for *M. natalensis* were significantly higher on the west facing slopes than on the south-east facing slopes, and significantly lower on the west than those on the north-west, north-east and south-west facing slopes (Table 5.30; Appendix 31). The head and body length values for *R. dilectus* were significantly higher on the north-west facing slopes than on north-east, west, south-west and south-east facing slopes (Table 5.30; Appendix 31). For shrew species, the head and body length values for *C. flavescens* were significantly lower on the south-east slopes than those on the west, north-west, north-east, south, north, east and south-west slopes (Table 5.30; Appendix 31). The head and body length values for *M. varius* were significantly higher on the north-west than west, south-west, and north-east slopes, but the difference between the north-west and south-east facing slopes was not statistically significant (Table 5.30; Appendix 31).

5.4.3 Seasons

The number of captures for adult *A. ineptus* were highest in the spring season while that of *M. natalensis* were highest in the summer season and that of *R. dilectus* were highest in the autumn season (Table 5.31). The number of captures for adults of the shrew species, *C. flavescens* and *M. natalensis* were highest in the autumn season (Table 5.31). The number of captures for subadults for *A. ineptus* were highest in the spring and summer seasons while that of *M. natalensis* and *R. dilectus* were highest in the autumn season (Table 5.31). For *C. flavescens* and *M. varius*, the number of captures for subadults were highest in the autumn season (Table 5.31). Juveniles for *A. ineptus* were not recorded in all seasons while the captures of juveniles for *M. natalensis* were highest in the autumn season and that of *R. dilectus* were highest in the spring season (Table 5.31). For *C. flavescens*, the number of captures for juveniles

were highest in the autumn season and no juveniles were recorded for *M. varius* (Table 5.31).

Table 5.31. The total captures of focal small mammal species in relation to seasons in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Season	Relative Age			Sex								
	Juv	Subad	Ad	Females				Total	Males			Total
				Imp	Perf	Preg	Lact		Ab	S-scrot	Scrot	
<i>Aethomys ineptus</i>												
Autumn	-	-	4	1	1	-	-	2	1	1	-	2
Winter	-	-	5	1	2	-	-	3	1	1	-	2
Spring	-	1	10	3	3	1	-	7	1	2	1	4
Summer	-	1	9	-	1	-	4	5	3	2	-	5
Total	-	2	28	5	7	1	4	17	6	6	1	13
<i>Mastomys natalensis</i>												
Autumn	21	56	73	47	21	2	9	79	63	6	2	71
Winter	7	14	38	20	8	-	1	29	19	8	3	30
Spring	7	20	75	24	21	2	2	49	16	17	20	53
Summer	13	11	108	13	18	12	23	66	33	25	8	66
Total	48	101	294	104	68	16	35	223	131	56	33	220
<i>Rhabdomys dilectus</i>												
Autumn	2	25	73	18	20	-	7	45	41	13	1	55
Winter	3	22	54	18	22	-	-	40	26	3	10	39
Spring	5	3	40	3	12	5	4	24	11	10	3	24
Summer	2	2	33	4	12	4	2	22	10	4	1	15
Total	12	52	200	43	66	9	13	131	88	30	15	133
<i>Crocidura flavescens</i>												
Autumn	1	2	17			4				9		
Winter	-	-	13			8				5		
Spring	-	1	12			3				4		
Summer	-	-	7			8				12		
Total	1	3	49			23				30		
<i>Myosorex varius</i>												
Autumn	-	5	17			2				5		
Winter	-	-	7			3				2		
Spring	-	-	5			6				5		
Summer	-	-	11			10				12		
Total	-	5	40			21				24		

* Relative age classes: Juv = juvenile; Subad = subadult; Ad = adult.

Female reproductive status: Imp = imperforate; Perf = perforate; Preg = pregnant; Lact = lactating.

Male reproductive status: Ab = abdominal; S-scrot = semi-scrotal; Scrot = scrotal.

Dashes (–) indicate zero captures or not applicable.

¹ Reproductive status and sex could not be determined for shrew species; therefore, reproductive-status categories are not reported for *Crocidura flavescens* and *Myosorex varius*.

The seasons only had a significant influence on the age of *M. natalensis* ($B = 79.07$, $\chi^2 = 44.36$, $p < 0.001$), and *R. dilectus* ($B = 47.92$, $\chi^2 = 21.60$, $p < 0.001$) (Appendix 32). Although the model for *M. varius* returned $p = 0.050$, this was not considered statistically significant at the $p < 0.05$ threshold. For *M. natalensis*, the number of captures for adults significantly increased in the summer season and significantly decreased in the autumn season relative to winter (Appendix 33). For *R. dilectus*, the number of captures for adults significantly increased in the spring and summer seasons relative to winter, while that of juveniles was significantly higher in the spring season than in winter (Appendix 33). No significant post hoc seasonal differences

were identified for the age categories of *A. ineptus*, *C. flavescens* or *M. varius* (Appendix 33).

The female captures for *A. ineptus* were highest during the spring season, whereas that of *M. natalensis* and *R. dilectus* peaked in autumn (Table 5.31). Among the shrew species, female *C. flavescens* number of captures were highest in both spring and autumn, while female *M. varius* number of captures were highest in autumn. The male captures of *A. ineptus* were highest in summer, whereas male captures of *M. natalensis* and *R. dilectus* were highest in autumn. Similarly, male captures of the shrew species *C. flavescens* and *M. varius* peaked during the autumn season. Despite these seasonal patterns, seasons did not have a significant influence on the sex of the focal small mammal species captured during the study (Appendix 32 & 35).

The abdominal male captures for *A. ineptus* were highest in summer season while those of *M. natalensis* and *R. dilectus* were highest in the autumn season (Table 5.31). The semi-scrotal male captures for *A. ineptus* were highest in spring and summer seasons while that of *M. natalensis* were highest in the summer season and that of *R. dilectus* were highest in the autumn season (Table 5.31). The number of captures for scrotal males for *A. ineptus* and *M. natalensis* were highest in the spring season, while that of *R. dilectus* were highest in the winter season (Table 5.31). The number of captures of imperforate females for *A. ineptus* were highest in the spring season while that of *M. natalensis* were highest in the autumn season and that of *R. dilectus* were highest in the winter and autumn seasons (Table 5.31). The number of captures for perforate females for *A. ineptus* were highest in the spring season while that of *M. natalensis* were highest in the spring and autumn seasons and that of *R. dilectus* were highest in the winter season (Table 5.31). The number of captures for pregnant females for *A. ineptus* were highest in spring season while that of *M. natalensis* were highest in summer and that of *R. dilectus* were highest in the spring season (Table 5.31). The number of captures for lactating females for *A. ineptus* and *M. natalensis* were highest in the summer season, while that of *R. dilectus* were highest in the autumn season (Table 5.31).

The seasons had a significant association with the reproductive status of *M. natalensis* ($B = 207.47$, $\chi^2 = 119.32$, $p < 0.001$) and *R. dilectus* ($B = 134.63$, $\chi^2 = 62.79$, $p < 0.001$) (Appendix 32). For *M. natalensis*, the number of captures for abdominal males were significantly higher in the autumn than in winter (Appendix 34). The number of captures

of imperforate females were significantly higher in summer than in winter, while the number of captures of pregnant females were significantly higher in autumn and spring than in winter (Appendix 34). The number of captures of lactating females were significantly higher in autumn than in winter (Appendix 34). For *R. dilectus*, the number of captures for abdominal males significantly decreased in the spring season relative to winter, scrotal males were significantly higher in winter than in autumn, spring and summer, and the number of captures of imperforate females were significantly lower in autumn and spring than in winter (Appendix 34). The number of captures for perforate females were significantly lower in autumn and spring than in winter, while the number of captures for pregnant females were significantly higher in spring than in winter (Appendix 34). No significant seasonal differences were identified for the reproductive status categories of *A. ineptus* (Appendix 34).

The head and body length values for *A. ineptus*, *M. natalensis* and *R. dilectus* were highest in the summer while those of *C. flavescens* and *M. varius* were highest in the summer and autumn respectively (Table 5.32). The seasons had a significant influence on the head and body length values for *M. natalensis*, *R. dilectus* and *M. varius* (Table 5.32).

Table 5.32. The outcomes from Kruskal-Wallis tests of species-specific influences of season on small mammal length (combined head and body length (mm)) in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Species	Head and Body Length (mm) ($\bar{X} \pm \text{SD}$)				Z-test	P-value
	Winter	Spring	Summer	Autumn	DF	
<i>Aethomys ineptus</i>	122.80 $\pm$ 25.94	124.73 $\pm$ 25.70	142.70 $\pm$ 17.52	113.75 $\pm$ 21.75	5.11 (3)	0.164
<i>Mastomys natalensis</i>	89.64 $\pm$ 21.25	103.63 $\pm$ 20.80	106.25 $\pm$ 20.44	91.51 $\pm$ 16.55	64.18 (3)	<0.001 *
<i>Rhabdomys dilectus</i>	86.99 $\pm$ 13.92	88.02 $\pm$ 16.57	97.65 $\pm$ 15.60	92.33 $\pm$ 11.77	29.68 (3)	<0.001 *
<i>Crocidura flavescens</i>	82.00 $\pm$ 17.62	88.62 $\pm$ 24.86	96.86 $\pm$ 27.46	90.10 $\pm$ 15.86	4.82 (3)	0.186
<i>Myosorex varius</i>	73.00 $\pm$ 10.25	35.00 $\pm$ 6.28	56.18 $\pm$ 22.69	81.09 $\pm$ 11.27	19.55 (3)	<0.001 *

* Significant differences when evaluated at $\alpha = 0.05$

The head and body length values for *M. natalensis* were significantly higher in the spring than in the winter season, in summer than in the winter season, and in spring and summer than in the autumn season (Table 5.32; Appendix 36). For *R. dilectus*, the head and body length values were significantly higher in the autumn than in the winter season, in summer than in the winter season, in summer than in the spring season, and in summer than in the autumn season (Table 5.32; Appendix 36). For *M. varius*,

the head and body length values were significantly higher in the winter and autumn than in the spring season, and in autumn than in summer (Table 5.32; Appendix 36).

5.4.4 Grassland condition

The number of captures for adults and subadults for all focal small mammal species were highest at primary good grasslands, (Table 5.33). No juveniles were recorded for *A. ineptus* in any of the grasslands, while captures for juvenile *M. natalensis* and *R. dilectus* were highest in primary good grasslands (Table 5.33).

Table 5.33. The total captures of focal small mammal species demographics in relation to grassland condition in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Grassland Condition	Relative Age			Sex								
				Females				Total	Males			Total
	Juv	Subad	Ad	Imp	Perf	Preg	Lact		Ab	S-Scrot	Scrot	
<i>Aethomys ineptus</i>												
Primary Good		2	26	4	6	1	4	15	6	6	1	13
Primary Intermediate	-	-	1	-	1	-	-	1	-	-	-	-
Secondary Good	-	-	-	-	-	-	-	0	-	-	-	-
Secondary Intermediate	-	-	1	1	-	-	-	1	-	-	-	-
Total		2	28	5	7	1	4	17	6	6	1	13
<i>Mastomys natalensis</i>												
Primary Good	35	66	160	64	34	9	28	135	72	28	26	126
Primary Intermediate	-	-	3	-	-	-	-	-	2	1	-	3
Secondary Good	5	6	61	21	10	6	1	38	23	8	3	34
Secondary Intermediate	8	29	70	19	24	1	6	50	34	19	4	57
Total	48	101	294	104	68	16	35	223	131	56	33	220
<i>Rhabdomys dilectus</i>												
Primary Good	6	27	105	18	41	6	8	73	39	15	11	65
Primary Intermediate	-	5	8	4	-	2	-	6	6	1	-	7
Secondary Good	2	9	30	6	15	-	2	23	10	6	2	18
Secondary Intermediate	4	11	57	15	10	1	3	29	33	8	2	43
Total	12	52	200	43	66	9	13	131	88	30	15	133
<i>Crocidura flavescens</i>												
Primary Good	-	2	29			13		13		18		18
Primary Intermediate	-	-	-			-		-		-		-
Secondary Good	1	-	4			2		2		3		3
Secondary Intermediate	-	1	16			8		8		9		9
Total	1	3	49			23		23		30		30
<i>Myosorex varius</i>												
Primary Good	-	3	20			9		9		14		14
Primary Intermediate	-	-	-			-		-		-		-
Secondary Good	-	-	12			5		5		7		7
Secondary Intermediate	-	2	8			7		7		3		3
Total		5	30			21		21		24		24

* Relative age classes: Juv = juvenile; Subad = subadult; Ad = adult.

Female reproductive status: Imp = imperforate; Perf = perforate; Preg = pregnant; Lact = lactating.

Male reproductive status: Ab = abdominal; S-scrot = semi-scrotal; Scrot = scrotal.

Dashes (–) indicate zero captures or not applicable.

¹ Reproductive status and sex could not be determined for shrew species; therefore, reproductive-status categories are not reported for *Crocidura flavescens* and *Myosorex varius*.

The grassland condition only had a significant association with the age of *M. natalensis* ($B = 47.65$, $\chi^2 = 21.01$, $p = 0.002$) (Appendix 37). The adult captures were significantly higher in the secondary good grasslands than in the secondary-intermediate grasslands, while the captures for juveniles and subadults were not significantly influenced by grassland condition (Appendix 38). The female captures for all focal small mammal species were highest on the primary-good grassland patches (Table 5.33). Grassland conditions did not have a significant effect on the sexes of the focal small mammal species (Appendix 39 & 40). The number of captures for all reproductive stages of *A. ineptus*, *M. natalensis* and *R. dilectus* were highest in the primary-good grasslands (Table 5.33). The grassland condition had a significant effect on the reproductive status of *M. natalensis* ($B = 110.7$, $\chi^2 = 36.56$, $p = 0.006$) and *R. dilectus* ($B = 103.8$, $\chi^2 = 36.07$, $p = 0.007$) (Appendix 39). For *M. natalensis*, the number of captures for scrotal males, imperforate and lactating females were significantly higher in the primary-good grasslands while that of pregnant females were significantly higher in the secondary-good grasslands (Appendix 39). For *R. dilectus*, none of the individual post hoc comparisons between grassland conditions were significant, despite the overall significant association between grassland condition and reproductive status (Appendix 39).

The head and body length values for *A. ineptus* were highest in the primary-good grasslands while that of *M. natalensis* and *R. dilectus* were highest on the secondary-good grasslands (Table 5.34). The head and body length values for the shrew species, *C. flavescens*, were highest on the primary-good grasslands while that of *M. varius* were highest in the secondary-intermediate grasslands (Table 5.34).

Table 5.34. The outcomes from Kruskal-Wallis tests of species-specific associations between grassland condition and small mammal length (combined head and body length (mm)) in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Metrics	Head and Body (mm) ($\bar{X} \pm \text{SD}$)				H test (DF)	P-value
	Primary-Good	Primary-Intermediate	Secondary-Good	Secondary-Intermediate		
<i>Aethomys ineptus</i>	128.32 $\pm$ 23.58	-	-	-	0.69 (2)	0.710
<i>Mastomys natalensis</i>	97.97 $\pm$ 20.57	100.00 $\pm$ 5.00	104.26 $\pm$ 19.94	95.64 $\pm$ 20.90	13.78 (2)	0.003 *
<i>Rhabdomys dilectus</i>	88.30 $\pm$ 13.57	88.54 $\pm$ 12.53	95.68 $\pm$ 12.92	92.83 $\pm$ 15.92	13.75 (2)	0.003 *
<i>Crociodura flavescens</i>	90.00 $\pm$ 22.10	-	80.00 $\pm$ 30.14	88.71 $\pm$ 13.30	1.14 (2)	0.565
<i>Myosorex varius</i>	70.04 $\pm$ 16.89	-	56.75 $\pm$ 22.05	79.60 $\pm$ 23.45	6.75 (2)	0.034 *

* Significant differences when evaluated at $\alpha = 0.05$

The head and body length values for *M. natalensis* were significantly higher in the secondary-good than in the secondary-intermediate grasslands and in the secondary-good than in the primary-good grasslands (Table 5.34; Appendix 41). The head and body length value for *R. dilectus* was significantly higher in the secondary-intermediate than primary-good and in the secondary-good than primary-good grasslands (Table 5.34; Appendix 41). The head and body length value for *M. varius* was only significantly higher in the secondary-intermediate than secondary-good grasslands (Table 5.34; Appendix 41).

5.4.5 Regional variation

The number of captures for adult *A. ineptus* was highest in the Kloof region while captures for subadults were recorded in low numbers in the Kloof and Camperdown Rural regions (Table 5.35). No juveniles for this species were recorded in any of the regions. In the case of *M. natalensis*, the largest number of adults and subadults from this population were recorded in the Hillcrest region while captures of juveniles were highest in the Kloof region (Table 5.35). The captures for adults and juveniles for *C. flavescens*, were highest in the Hillcrest region, while captures for subadults were highest in the Camperdown Rural region (Table 5.35).

Table 5.35. The total captures of focal small mammal species demographics in relation to regional variation in eThekwin KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Regions		Relative Age				Sex								
		Juv	Subad	Ad	Imp	Perf	Females Preg	Lact	Total	Males S-scrot			Scrot	Total
<i>Aethomys ineptus</i>														
	Kloof	-	1	18	4	6	-	1	11	2	5	1	8	
	Hillcrest	-	-	1	-	-	-	-	-	1	-	-	1	
	Camperdown	-	1	9	1	1	1	3	6	3	1	-	4	
	Total	-	2	28	5	7	1	4	17	6	6	1	13	
<i>Mastomys natalensis</i>														
	Kloof	25	36	82	36	15	10	13	74	34	16	19	69	
	Hillcrest	21	43	162	49	45	6	10	110	76	31	9	116	
	Camperdown	2	22	50	19	8	-	12	39	21	9	5	35	
	Total	48	101	294	104	68	16	35	223	131	56	33	220	
<i>Rhabdomys dilectus</i>														
	Kloof	-	-	-	-	-	-	-	-	-	-	-	-	
	Hillcrest	4	7	29	11	5	1	-	17	17	6	2	25	
	Camperdown	16	75	128	37	59	10	12	118	68	26	13	107	
	Total	20	82	157	48	64	11	12	135	85	32	15	132	
<i>Crocidura flavescens</i>														
	Kloof	-	-	15		6					9			
	Hillcrest	1	-	23		11					13			
	Camperdown	-	3	11		6					8			
	Total	1	3	49		23					30			
<i>Myosorex varius</i>														
	Kloof	-	-	-		-					-			
	Hillcrest	-	-	-		-					-			
	Camperdown	7	2	33		20					22			
	Total	7	2	33		20					22			

* Relative age classes: Juv = juvenile; Subad = subadult; Ad = adult.

Female reproductive status: Imp = imperforate; Perf = perforate; Preg = pregnant; Lact = lactating.

Male reproductive status: Ab = abdominal; S-scrot = semi-scrotal; Scrot = scrotal.

Dashes (-) indicate zero captures or not applicable.

¹ Reproductive status and sex could not be determined for shrew species; therefore, reproductive-status categories are not reported for *Crocidura flavescens* and *Myosorex varius*.

Only the ages of captured individuals of *C. flavescens* ($B = 14.71$, $\chi^2 = 10.00$, $p = 0.040$) and *M. natalensis* ($B = 45.22$, $\chi^2 = 17.97$, $p = 0.001$) showed a significant relationship with regions (Appendix 42). For *M. natalensis*, the number of captures for adults were highest in the Hillcrest region but were not significantly different from those in the Kloof or Camperdown Rural regions, while that of juveniles was significantly higher in the Kloof region than in the Camperdown Rural region (Appendix 42). However, the parameter estimates for the shrew species, *C. flavescens*, did not show any significant regional differences for age (Appendix 43). Populations of *A. ineptus*, irrespective of sex, were highest in the Kloof region while those of *M. natalensis* were highest in the Hillcrest region (Table 5.35). For shrew species, *C. flavescens*, captures for both females and males were also highest in the Hillcrest region (Table 5.35). The

regions did not have a significant influence on the sexes of small mammal species (Appendix 42 & 43).

The number of captures for abdominal males for *A. ineptus* and *M. natalensis* were highest in the Camperdown Rural region, while scrotal males for *A. ineptus* were highest in the Kloof region and that of *M. natalensis* were highest in the Hillcrest region (Table 5.35). The numbers of scrotal males for *A. ineptus* and *M. natalensis* were highest in the Kloof and Hillcrest regions, respectively (Table 5.35). The numbers of imperforate and perforate females for *A. ineptus* were highest in the Kloof region while that of *M. natalensis* were highest in the Hillcrest region (Table 5.35). The numbers of pregnant females for *A. ineptus* were highest in the Camperdown region while those of *M. natalensis* were highest in the Kloof region (Table 5.35). Numbers of lactating females for *A. ineptus* were highest in the Camperdown region while those of *M. natalensis* were highest in the Kloof region (Table 5.35). Only the reproductive status of *M. natalensis* was significantly influenced by region ($B = 112.36$, $\chi^2 = 39.04$, $p < 0.001$) (Appendix 42). The number of captures for scrotal males for *M. natalensis* were significantly lower in the Hillcrest region (Appendix 44) than in the Kloof region, while the number of captures for pregnant females in the Hillcrest region were not significantly different from those in the Kloof region ($p = 0.051$).

The head and body length values for *A. ineptus* were highest in the Camperdown region while that of *M. natalensis* was highest in the Hillcrest region (Table 5.36). The head and body length value for the shrew species, *C. flavescens*, was highest in the Kloof region (Table 5.36). The regions had a significant effect on the head and body length values for *A. ineptus* and *C. flavescens* (Table 5.36).

Table 5.36. The outcomes from Kruskal-Wallis tests of species-specific associations of regional variation with small mammal length (combined head and body length (mm)) in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Species	Head and Body (mm) ($\bar{X} \pm SD$)			Z-test	P-value
	Kloof	Hillcrest	Camperdown Rural	DF	
<i>Aethomys ineptus</i>	121.21 $\pm$ 23.05	-	146.00 $\pm$ 16.64	9.01 (2)	0.011 *
<i>Mastomys natalensis</i>	96.94 $\pm$ 21.28	99.85 $\pm$ 21.55	97.07 $\pm$ 15.82	1.16 (2)	0.560
<i>Crocidura flavescens</i>	101.67 $\pm$ 11.91	90.25 $\pm$ 11.50	71.93 $\pm$ 27.74	9.75 (2)	0.008 *

* Significant differences when evaluated at $\alpha = 0.05$

The head and body length values for *A. ineptus* were significantly higher in the Camperdown region than in the Kloof region (Table 5.36; Appendix 46). No significant

differences were observed in head and body length values for this species when comparing the Hillcrest and Camperdown and Hillcrest and Kloof regions (Table 5.36; Appendix 46). For *C. flavescens*, the head and body length values were significantly higher in the Kloof than Camperdown region and in the Kloof than Hillcrest region (Table 5.36; Appendix 46). No significant differences were observed for this species when comparing between Camperdown and Hillcrest regions (Table 5.36; Appendix 46).

5.4.6 Habitat management regimes (treatments)

The number of captures for adult *A. ineptus* and *M. natalensis* were highest inside protected areas while that of *R. dilectus* were highest in the control site (Table 5.37). The captures for adults for the shrew species, *C. flavescens* were highest inside protected areas (Table 5.37). The subadult populations for *A. ineptus* were highest inside protected areas and in the control site while that of *M. natalensis* were highest inside protected areas and that of *R. dilectus* were highest in the control site (Table 5.37). No juveniles for the species *A. ineptus* were recorded in any of the treatments while the captures for juvenile *M. natalensis* were highest inside protected areas and those of *R. dilectus* were highest in the control site (Table 5.37).

Table 5.37. The total captures of focal small mammal species demographics in relation to habitat management regimes (treatments) in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

	Relative Age							Sex				
	Juv	Subad	Ad	Females				Total	Males			Total
				Imp	Perf	Preg	Lact		Ab	S-scrot	Scrot	
<i>Aethomys ineptus</i>												
Inside	-	1	18	4	6	-	1	11	2	5	1	8
Outside	-	-	1	-	-	-	-	-	1	-	-	1
Control	-	1	9	1	1	1	3	6	3	1	-	4
Total	-	2	28	5	7	1	4	17	6	6	1	13
<i>Mastomys natalensis</i>												
Inside	29	58	154	49	46	8	17	120	67	33	21	121
Outside	17	21	90	36	14	8	6	64	43	14	7	64
Control	2	22	50	19	8	-	12	39	21	9	5	35
Total	48	101	294	104	68	16	35	223	131	56	33	220
<i>Rhabdomys dilectus</i>												
Inside	-	2	12	2	4	-	-	6	3	3	2	8
Outside	2	9	16	9	1	1	-	11	15	1	-	16
Control	10	41	172	32	61	8	13	114	70	26	13	109
Total	12	52	200	43	66	9	13	131	88	30	15	133
<i>Crocidura flavescens</i>												
Inside	-	-	26			13		13		13		13
Outside	1	-	12			4		4		9		9
Control	-	3	11			6		6		8		8
Total	1	3	49			23		23		30		30
<i>Myosorex varius</i>												
Inside	-	-	-			-		-		-		-
Outside	-	-	-			-		-		-		-
Control	7	2	33			20		20		22		22
Total	7	2	33			20		20		22		22

* * Relative age classes: Juv = juvenile; Subad = subadult; Ad = adult.

Female reproductive status: Imp = imperforate; Perf = perforate; Preg = pregnant; Lact = lactating.

Male reproductive status: Ab = abdominal; S-scrot = semi-scrotal; Scrot = scrotal.

Dashes (–) indicate zero captures or not applicable.

¹ Reproductive status and sex could not be determined for shrew species; therefore, reproductive-status categories are not reported for *Crocidura flavescens* and *Myosorex varius*.

The treatments only had a significant association with the ages of *C. flavescens* ($B = 15.94$, $\chi^2 = 11.26$, $p = 0.024$) and *M. natalensis* ($B = 39.16$, $\chi^2 = 12.12$, $p = 0.017$) (Appendix 47). For *M. natalensis*, the number of captures for juveniles were significantly lower in the control site than outside protected areas (Appendix 48). There was no significant difference in juvenile captures between inside and outside protected areas. However, the parameter estimates for the shrew species, *C. flavescens*, did not show any significant differences for age between the treatments (Appendix 48). Irrespective of sex, captures for *A. ineptus* and *M. natalensis* were highest inside protected areas while those of *R. dilectus* were highest in the control site. Captures for both female and male of the shrew species, *C. flavescens*, were highest inside protected areas (Table 5.37). The treatments did not have a significant influence on

the sex ratio of small mammal species based on the overall regression analysis (Appendix 47 & 50).

The number of captures for abdominal males for *A. ineptus* were highest inside protected areas, while those of *R. dilectus* were highest in the control site and those of *M. natalensis* were highest inside protected areas (Table 5.37). The number of captures for semi-scrotal and scrotal males for *A. ineptus* and *M. natalensis* were highest inside protected areas while those of *R. dilectus* were highest in the control site (Table 5.37). The number of captures for imperforate and perforate females for *A. ineptus* and *M. natalensis* were highest inside protected areas while those of *R. dilectus* were highest in the control site. The number of captures for pregnant and lactating females for *A. ineptus* and *R. dilectus* were higher in the control site than inside and outside protected areas while pregnant female captures for *M. natalensis* were equal inside and outside protected areas and absent in the control site, whereas lactating female captures were highest inside protected areas (Table 5.37).

The treatments had a significant effect on the reproductive status of *M. natalensis* ($B = 97.65$, $\chi^2 = 24.61$, $p = 0.017$) and *R. dilectus* ($B = 76.81$, $\chi^2 = 29.38$, $p = 0.003$) (Appendix 47). For *M. natalensis*, none of the individual post hoc comparisons between treatments for reproductive status were significant (Appendix 49). For *R. dilectus*, abdominal male captures were significantly lower inside than outside protected areas (Appendix 49). The remaining post hoc comparisons for reproductive status were not significant.

The head and body length values for *A. ineptus* were highest in the control site while those of *M. natalensis* were highest outside protected areas and those of *R. dilectus* were highest inside protected areas (Table 5.38). For the shrew species, *C. flavescens*, the head and body length values were highest inside protected areas. The treatments had a significant influence on the head and body length values for all small mammal species (Table 5.38).

Table 5.38. The outcomes from Kruskal-Wallis tests of species-specific associations between habitat management regimes (treatments) and small mammal length (combined head and body length (mm)) in eThekweni KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Species	Head and Body Length ($\bar{X} \pm \text{SD}$)			Z-test	P-value
	Inside Protected Area	Outside Protected Area	Control	DF	
<i>Aethomys ineptus</i>	121.21 $\pm$ 23.05	-	146.00 $\pm$ 16.64	9.01	0.011 *
<i>Mastomys natalensis</i>	95.85 $\pm$ 18.97	104.11 $\pm$ 24.71	97.07 $\pm$ 15.82	12.89	0.002 *
<i>Rhabdomys dilectus</i>	99.21 $\pm$ 14.00	87.59 $\pm$ 18.07	90.53 $\pm$ 13.70	7.52	0.023 *
<i>Crocidura flavescens</i>	97.73 $\pm$ 10.52	88.46 $\pm$ 15.05	71.93 $\pm$ 27.74	8.76	0.013 *

* Significant differences when evaluated at $\alpha = 0.05$

The head and body length values for *A. ineptus* were significantly higher in the control site than inside protected areas and showed no significant differences between the control site and outside protected areas as well as between outside and inside protected areas (Table 5.38; Appendix 51). The head and body length values for *M. natalensis* were significantly higher outside than inside protected areas and were significantly higher outside protected areas than in the control site (Table 5.38; Appendix 51). There were no significant differences in head and body length values for *M. natalensis* between inside protected areas and the control site (Table 5.38; Appendix 51). The head and body length values for *R. dilectus* were significantly higher inside than outside protected areas, and they were also higher inside protected areas than in control sites (Table 5.38; Appendix 51). However, there were no significant differences in head and body length values of this species between outside protected areas and the control site (Table 5.38; Appendix 51). The head and body length values for *C. flavescens* were significantly higher inside protected areas than in the control sites, and there were no significant differences in head and body length values of this species between the control site and sites outside protected areas, or between captures outside and inside protected areas (Table 5.38; Appendix 51).

CHAPTER 6

6.1 DISCUSSION

The KwaZulu-Natal Sandstone Sourveld (KZNSS) grassland is a Critically Endangered grassland ecosystem occurring mainly within the eThekweni Municipal Area (EMA) and extending into the KwaZulu-Natal (KZN) Midlands. Mucina & Rutherford (2006) described the ecosystem as occurring on nutrient-poor, acidic sandstone-derived soils that support diverse plant and animal assemblages, while Nkambule et al. (2016) highlighted its importance within the Maputaland-Pondoland-Albany (MPA) biodiversity hotspot. Naicker et al. (2016) and Boon et al. (2016) reported that >90% of KZNSS landscape has been transformed or fragmented by urbanisation, agriculture, afforestation and invasive alien plants, with only a small proportion formally protected. More recently, McLean et al. (2024, 2025a) showed that the remaining fragments are largely small and isolated, raising concerns regarding their long-term ecological viability.

Within eThekweni, the Durban Metropolitan Open Space System (D'MOSS), described by Boon et al. (2016) and more recently by McLean et al. (2024), plays an important role in conserving remnant KZNSS grassland patches and maintaining ecological connectivity within an urban landscape. However, McLean et al. (2025a, 2025b) showed that many important biodiversity features and conservation opportunities occur outside formally protected areas, highlighting the importance of assessing the conservation value of remnant habitat fragments. Although Drury (2016) and Boon et al. (2016) have given increasing attention to vegetation patterns and floristic composition in KZNSS landscape, information on faunal assemblages, particularly small mammals, remains limited.

Small mammals are widely recognised as sensitive indicators of habitat condition, as demonstrated by de Klerk & Avenant (2017) and Loggins et al. (2019), because they respond rapidly to changes in vegetation structure, resource availability and landscape configuration. Furthermore, previous studies by Els & Kerley (1996), Lagesse & Thondhlana (2016), and Simelane et al. (2018) have shown that small mammal assemblages respond to environmental variables such as elevation, slope aspect, seasonal variation and habitat condition. This study therefore aimed to assess the ecological status and conservation value of remnant KZNSS grassland fragments

using small mammal assemblages by comparing sites inside formally protected areas, outside (D'MOSS) protected areas and a relatively undisturbed reference site. The study further examined whether unprotected and partially protected KZNSS D'MOSS contribute meaningfully to conservation objectives and whether they support habitat specialists or are dominated by disturbance-tolerant generalist species.

The results showed differences in small mammal captures, trap success, Minimum Known Alive (MKA), species richness, and diversity across environmental gradients, including elevation, slope aspect, seasonality, grassland condition, regional variation, and habitat management regimes. These findings underscore the sensitivity of small mammal assemblages, which serve as bio-indicators of ecosystem health, to both fine-scale habitat heterogeneity and the broader landscape context within the KZNSS landscape. They further emphasise the importance of maintaining structural and compositional complexity within remnant grassland patches to support not only dominant generalist species but also less abundant and potentially more vulnerable specialists. Given the critically endangered status of the KZNSS landscape and its continued transformation, sustaining ecological processes that uphold small mammal diversity is essential for overall ecosystem functioning and for informing effective conservation planning. Importantly, the finding that some non-protected fragments supported relatively diverse small mammal assemblages aligns with the work of Jewitt et al. (2015), Boon et al. (2016), and Lagesse & Thondhlana (2016), who demonstrated that conservation success in fragmented landscapes increasingly depends on recognising and managing biodiversity values beyond formally protected areas. The presence of diverse small mammal assemblages within some non-protected fragments therefore suggests that areas outside formally protected areas may still contribute meaningfully to the conservation of the KZNSS landscape, provided that appropriate management and protection measures are implemented. Similar findings have been reported elsewhere in South Africa, where studies by Simelane et al. (2018) and Loggins et al. (2019) have shown that small mammal diversity persists in well-structured, non-protected grassland fragments.

The findings demonstrated that small mammal assemblages in fragmented KZNSS landscape exhibit marked variation in abundance, species richness, diversity, and functional guild composition in response to multi-scale ecological and environmental gradients. This reinforces their value as indicators of ecosystem condition, consistent

with the findings of Avenant (2000) and de Klerk & Avenant (2017). Overall capture success was comparable to that reported by Lagesse & Thondhlana (2016) and Ramahlo et al. (2022) in other southern African grassland studies documenting high small mammal densities in structurally heterogeneous habitats. Similarly, the patterns observed in the present study are consistent with those reported by Pardini et al. (2005), Mortelliti et al. (2010), and Simelane et al. (2018), who demonstrated the sensitivity of small mammal assemblages to habitat structure, fragmentation, and landscape context.

Rodent species dominated the assemblages, with *M. natalensis* and *R. dilectus* forming the core of the assemblage, a pattern consistent with the findings of Monadjem et al. (2015) and Lagesse & Thondhlana (2016), who showed that generalist and disturbance-tolerant species often dominate modified landscapes because of their broad ecological tolerances and high reproductive potential. The predominance of *M. natalensis* (54% of captures) is consistent with the well-documented adaptability and ability of the species to attain high population densities, as reported by Loggins et al. (2019) and Ramahlo et al. (2022). The substantial contribution of *R. dilectus* (28% of captures) reflects its strong association with mesic grassland habitats and its ability to persist across a range of grassland conditions, as reported by Meynard et al. (2012), Rymer & Pillay (2012), and Simelane et al. (2018).

In contrast, rare species such as *P. melanotis* and *Steatomys* spp. accounted for <1% of captures, which is consistent with the patchy distributions and low trappability of specialist or microhabitat-restricted species reported by Monadjem et al. (2015) and Ramahlo et al. (2022). The moderate representation of shrews (*C. flavescens* and *M. varius*), accounting for ~10% of captures, may suggest that sufficient habitat moisture and invertebrate prey persist within the KZNSS landscape. This interpretation is consistent with the findings of Churchfield & Rychlik (2006) and Simelane et al. (2018), who reported that shrew occurrence and persistence are generally associated with mesic habitats, suitable ground cover, and the availability of invertebrate prey.

Functional guild analyses further confirmed the dominance of large omnivores (e.g., *A. ineptus*, *M. namaquensis*, *M. natalensis*), reflecting their greater abundance and, consistent with Monadjem et al. (2015) and Loggins et al. (2019), the ecological versatility of species capable of exploiting diverse food resources. In contrast, small

omnivores and specialist guilds exhibited low capture rates, which may indicate greater sensitivity to habitat fragmentation and modification, as similarly reported by Monadjem et al. (2015), Simelane et al. (2018), and Ramahlo et al. (2022). Differences in trap success and population estimates between shrew species may also reflect species-specific variation in habitat preferences, detectability, and space use, as described by Churchfield & Rychlik (2006) and Simelane et al. (2018). These patterns suggest that maintaining grassland structural and compositional complexity may be important for supporting both dominant generalists and less abundant specialists, a conclusion consistent with studies by Lagesse & Thondhlana (2016), Loggins et al. (2019), and de Klerk & Avenant (2017), which highlighted the sensitivity of small mammal assemblages to habitat condition and landscape change.

6.1.1 The effect of elevational gradients in the KwaZulu-Natal Sandstone Sourveld ecosystem.

Elevation is one of the key drivers of small mammal assemblage structure in the Critically Endangered and highly fragmented KZNSS landscape. McCain (2005), Cáceres et al. (2011), and Richard et al. (2022) showed that elevational gradients influence climate, vegetation structure and disturbance intensity. Across elevational gradients, Fuentes-Montemayor et al. (2019) and Bergstrom et al. (2023) demonstrated that small mammal assemblages are further shaped by interactions between elevation-related environmental conditions, habitat fragmentation and surrounding land use. Similarly, Denys et al. (2009) reported that anthropogenic disturbance can alter small mammal assemblage composition by favouring widespread, disturbance-tolerant species, while Richard et al. (2022) demonstrated that small mammal assemblages respond differentially to environmental gradients such as elevation and microhabitat characteristics.

In general, higher elevations are often characterised by cooler conditions and greater structural complexity. Studies by Torre & Arrizabalaga (2009) in Mediterranean mountain systems in Spain, and by Boahene et al. (2021) and Chirichella et al. (2022) in montane ecosystems in Ghana and Italy, respectively, have shown that such conditions may support species associated with mesic microhabitats and may be associated with greater shrew abundance and assemblage turnover than lower

elevations. Similarly, McCain (2005), Torre & Arrizabalaga (2009), Duan et al. (2024), Cáceres et al. (2011), and Richard et al. (2022) reported that species richness often peaks at mid elevations, where intermediate environmental conditions may provide favourable habitat complexity and resource availability. Climate change may further modify these elevational patterns. McCain (2005) and Chen et al. (2020) suggested that climate warming could drive upslope shifts in species distributions, increase species overlap and competition, and potentially lead to local extinctions of high-elevation specialists. These studies highlight the importance of elevation in shaping small mammal assemblages within the KZNSS landscape and provide context for interpreting the patterns observed in this study.

In this study, elevation had a significant influence on total captures, trap success, MKA, species richness, diversity, and functional guild composition. Total captures, MKA, and species diversity peaked at mid elevations, whereas trap success, species richness, and the abundance of granivores and insectivores were relatively higher at high elevations within the sampled range. In contrast, herbivores, large omnivores, and small omnivores were most abundant at mid elevations, with the lowest abundances generally recorded at low elevations. However, the elevational gradient examined (~450-1 100 m) was relatively small and unlikely to represent distinct climatic zones. Instead, the observed patterns reflect local elevational variation and associated environmental heterogeneity, including differences in topography and habitat structure described by Els & Kerley (1996), vegetation complexity and moisture availability reported by Simelane et al. (2018), and variation in disturbance intensity and surrounding land use documented by Lagesse & Thondhlana (2016) and Ramahlo et al. (2022). Comparable patterns have been reported in other South African studies. For example, Els & Kerley (1996) found that small mammal assemblage structure varied across a relatively small elevational range (~430-580 m) in the Groendal Wilderness Area, where vegetation structure and topographic features were associated with assemblage patterns. Similarly, Taylor et al. (2015) reported increasing species richness with elevation in the Soutpansberg Mountains (~769-1 747 m), although their broader elevational range likely encompassed stronger climatic gradients than those examined in the present study. Simelane et al. (2018) likewise demonstrated that habitat characteristics and local environmental conditions influenced small mammal assemblages across an elevational gradient of ~373-1 571

m in the foothills of the Drakensberg Mountains. These studies suggest that the elevational and functional guild patterns observed in the present study may be more closely associated with local habitat heterogeneity than with broad climatic gradients. Likewise, the lower captures, diversity, and guild abundances recorded at lower elevations may reflect differences in habitat condition, land use, or disturbance, as suggested by Lagesse & Thondhlana (2016) and Ramahlo et al. (2022), although these factors were not directly quantified in the present study. The relatively higher abundances recorded at mid and high elevations may similarly be associated with more favourable habitat conditions and greater resource heterogeneity. Consequently, maintaining habitat heterogeneity and ecological integrity within remnant KZNSS grasslands may be important for supporting diverse small mammal assemblages, functional guilds, and the ecosystem processes they sustain.

Small mammal population estimates and trap success were generally not influenced by elevation, except for species-specific patterns. *Rhabdomys dilectus* exhibited significantly higher captures and trap success at higher elevations, consistent with its association with mesic grasslands and dense vegetation cover described by Meynard et al. (2012) and Rymer & Pillay (2012). Although captures of *C. flavescens* were also highest at higher elevations, irrespective of age class, these differences were not statistically significant. Nevertheless, the species is commonly associated with moist habitats and dense ground cover, as reported by Skinner & Chimimba (2005) and Monadjem et al. (2015). In contrast, *A. ineptus* was most abundant at lower elevations, whereas *M. natalensis* peaked at mid elevations. Several less abundant species, including *O. auratus*, *P. melanotis*, and *Steatomys* spp., were recorded predominantly at mid to high elevations, while *Steatomys* spp. represented a new species record for the higher elevation sites. *Steatomys* spp. could not be identified to species level because the capture-mark-recapture (CMR) study design did not permit the collection of voucher specimens required for definitive taxonomic identification. Monadjem et al. (2015) reported that these species are generally associated with mesic grasslands, dense vegetation, friable soils, or structurally complex habitats. Their occurrence may therefore indicate that these higher elevation grassland fragments retain habitat features important for specialist species, whereas their low abundances at lower elevations may reflect the limited availability of suitable habitat, although habitat characteristics were not directly quantified in the present study. Futuyma & Moreno

(1988) defined specialist species as organisms that utilise a relatively narrow range of resources or occupy a limited range of habitats, whereas generalist species exploit a wide variety of resources and habitats. Devictor et al. (2010) similarly described specialist species as having narrow ecological niches and specific environmental requirements, making them generally more sensitive to habitat alteration and environmental change. In contrast, they characterised generalist species as possessing broad ecological niches that enable them to tolerate greater environmental variability and disturbance. These patterns suggest that local elevational variation and associated habitat conditions may influence species-specific abundance and distribution within the KZNSS landscape, consistent with observations by Els & Kerley (1996), Simelane et al. (2018), Ramahlo et al. (2022), and Bergstrom et al. (2023), who highlighted the importance of habitat structure, environmental heterogeneity, and disturbance in shaping small mammal assemblages.

Only *M. natalensis* exhibited a significant relationship between elevation and population age structure, with juveniles occurring predominantly at lower elevations and adults and subadults more abundant at mid and higher elevations. This pattern may suggest spatial variation in breeding activity, survival, or post-dispersal occupancy, although these processes were not directly examined in the present study. Leirs et al. (1997) demonstrated that population dynamics in African rodents are strongly influenced by demographic processes, including seasonal reproduction and density-dependent survival, which can contribute to spatial variation in population structure. Similarly, reproductive activity appeared to vary with elevation. In *A. ineptus*, males with abdominal testes were most frequently captured at higher elevations, whereas semi-scrotal and scrotal males were more common at lower elevations. Females with imperforate and perforate reproductive tracts predominated at lower elevations, while pregnant and lactating females were more frequently recorded at higher elevations. Likewise, abdominal and semi-scrotal males of *M. natalensis* were more abundant at higher elevations, whereas scrotal males were more common at lower elevations. These patterns may reflect local variation in habitat conditions, resource availability, or breeding phenology rather than direct effects of elevation. Comparable studies by Leirs et al. (1997), Monadjem & Perrin (2003), and Monadjem et al. (2015) demonstrated that demographic structure and reproductive activity in southern African small mammals are strongly influenced by rainfall, temperature,

habitat quality, and food availability. Sex ratios remained relatively stable across elevations, suggesting that local environmental variation may influence abundance and reproductive activity more strongly than sex composition. These findings reinforce the value of small mammals as bio-indicators of habitat quality and ecosystem integrity. As demonstrated by de Klerk & Avenant (2017), small mammals respond rapidly to changes in vegetation structure and habitat condition, while Els & Kerley (1996) and Bergstrom et al. (2023) highlighted the importance of habitat structure, environmental heterogeneity, and land-use change in shaping assemblage composition. Specialist species are generally more sensitive to habitat disturbance than generalists, as reported by Monadjem et al. (2015), Lagesse & Thondhlana (2016), and Ramahlo et al. (2022). The occurrence of specialist species within the higher elevation KZNSS grassland fragments may therefore indicate the conservation value of these habitats. Consequently, identifying and prioritising remnant KZNSS patches that support diverse assemblages and habitat specialists may enhance conservation planning by maintaining ecological integrity and safeguarding habitat for species most vulnerable to landscape transformation.

Significant relationships were observed between the local elevational gradient and species-specific body size. *Aethomys ineptus* individuals were larger at higher elevations, *M. natalensis* attained maximum body size at intermediate elevations, and *C. flavescens* was larger at lower elevations. These contrasting patterns may suggest that body size responses to elevation are species-specific and may reflect differences in habitat associations, resource availability, and physiological tolerances rather than elevation alone. The larger body size of *A. ineptus* at higher elevations is broadly consistent with Bergmann's rule, first proposed by Bergmann (1847), which proposes that larger body sizes may provide thermoregulatory advantages in cooler environments. However, because temperature and thermoregulatory responses were not directly measured in this study, this explanation remains speculative. Likewise, Monadjem et al. (2023) demonstrated that elevation filters rodent and shrew assemblages through species-specific morphological traits, suggesting that different species may respond differently to environmental gradients. Yom-Tov & Geffen (2006) further showed that geographic variation in body size is associated with climatic conditions and resource availability. The patterns observed in the present study may reflect local environmental conditions associated with the KZNSS landscape

elevational gradient, although habitat characteristics, resource availability, and microclimatic conditions were not directly quantified. These findings suggest that elevation may influence body size indirectly through changes in habitat structure and microclimatic conditions, but further studies explicitly measuring these variables would be required to confirm the mechanisms underlying the observed morphometric patterns.

6.1.2 The effect of slope aspect in the KwaZulu-Natal Sandstone Sourveld ecosystem.

Slope aspect strongly influences grassland microclimates by creating warmer, drier conditions on north and west facing slopes and cooler, moister conditions on south and east-facing slopes, thereby affecting vegetation structure and resource availability. Bennie et al. (2006) and Geiger et al. (2009) demonstrated that topographic variation alters local temperature and moisture regimes, which in turn influence habitat conditions. Within the KZNSS landscape, relatively intact mesic grasslands are commonly characterised by *Themeda triandra* and *Tristachya leucothrix*, whereas disturbed or degraded areas often exhibit increased dominance of *Aristida congesta* and *Eragrostis curvula* (Mucina & Rutherford, 2006). Such variation in vegetation structure may enhance habitat heterogeneity by influencing food availability, shelter, and microclimatic conditions for small mammals. Brady & Weil (2016) highlighted the importance of soil moisture in influencing vegetation productivity, while Dorigo et al. (2021) demonstrated that topographic variation can influence small mammal distributions through its effects on habitat conditions. Although these mechanisms were not directly measured in the present study, they provide a plausible explanation for the species-specific responses to slope aspect observed here. Despite extensive research on the ecology and conservation of KZNSS landscape (Hlanguza, 2015; Boon et al., 2016; Naicker et al., 2016; Drury, 2016; Nkambule et al., 2016; Odindi et al., 2016), the influence of slope aspect on small mammal assemblages has received little attention. To our knowledge, this is among the first studies to explicitly examine slope aspect as a driver of small mammal assemblage structure within KZNSS landscape, addressing an important knowledge gap and suggesting that topographic heterogeneity may contribute to patterns of habitat use and species distribution in this threatened ecosystem.

In this study, slope aspect significantly influenced total captures, trap success, MKA, species richness, and functional guild composition, but not species diversity. Total captures and MKA were highest on west facing slopes, suggesting that these slopes may have provided more favourable habitat conditions for some small mammal species. In contrast, trap success and species richness were highest on south-east and south-west facing slopes, indicating that different slope aspect orientations may favour different small mammal species or ecological groups. Bennie et al. (2006) demonstrated that slope aspect alters the amount of solar radiation received, creating differences in local temperature and moisture regimes. Geiger et al. (2009) further showed that these microclimatic differences can influence habitat conditions, while Yang et al. (2020) reported from subtropical mountain forests in China that slope-related microclimates were associated with variation in vegetation structure and resource availability. Similarly, Gong et al. (2008) found that slope aspect influenced productivity and plant species composition in hilly grasslands of Inner Mongolia, whereas Zhu et al. (2016) reported differences in the functional diversity of grassland assemblages on the Loess Plateau associated with slope aspect. In temperate forests of the Garhwal Himalaya, Sharma et al. (2010) observed variation in forest composition, assemblage structure, and soil properties among different slope orientations, while Kimball et al. (2017) demonstrated that slope aspect was associated with differences in drought tolerance in restored plant assemblages. More recently, Nepali et al. (2021) found that slope aspect influenced plant species richness and composition across an elevational gradient in Nepal. Chen et al. (2024) reported that soil aggregate stability and humus development differed among north, east, west, and south facing slopes in alpine areas of southwest China, with north facing slopes generally supporting higher humus content, suggesting that slope orientation may influence soil conditions relevant to habitat quality. Gu et al. (2025) found that slope aspect affected soil quality in temperate agricultural landscapes, and Wang et al. (2026) demonstrated that slope aspect influenced soil protistan functional structure in subtropical forests. These studies suggest that slope aspect has the potential to modify environmental conditions across diverse ecosystems. Therefore, the higher trap success and species richness recorded on south-east and south-west facing slopes in the present study may reflect more favourable microhabitats with greater vegetation cover, food resources, or shelter. However, because vegetation structure,

soil characteristics, and microclimatic variables were not measured directly, these explanations remain speculative.

Typically, north, south, and east facing slopes supported lower total captures and MKA, which may indicate differences in habitat suitability associated with slope-driven environmental conditions. Because studies explicitly investigating the effects of slope aspect on small mammal assemblages in African grasslands are scarce, comparisons in this discussion necessarily draw on research from montane, forest, and grassland ecosystems. While the magnitude of slope effects may differ among ecosystems, the underlying mechanisms linking slope aspect to microclimate, vegetation structure, and habitat quality are broadly applicable and provide a useful ecological framework for interpreting the patterns observed in the present study. Lee et al. (2019) found that slope characteristics were associated with differences in ground vegetation that, in turn, influenced small rodent populations in a temperate forest ecosystem in South Korea. Similarly, Dorigo et al. (2021) reported that topographic variation was associated with differences in habitat conditions and vegetation characteristics that influenced small mammal assemblages in alpine grasslands of the Italian Alps. In the montane forests of the North Uluguru Mountains, Tanzania, Chidodo et al. (2020) found that rodent assemblages varied with habitat characteristics associated with topography, while Shuai et al. (2017) demonstrated in the Qinling Mountains of central China that slope aspect and elevation were associated with differences in small mammal species richness and assemblage structure. These studies suggest that the patterns observed in the present study may reflect the combined influence of the local elevational gradient and slope aspect rather than slope aspect alone. Nevertheless, because the mechanisms underlying these relationships were not measured directly, this interpretation should be considered tentative.

Functional guilds also responded differently to slope aspect, suggesting that topographic variation may influence habitat use and resource availability. Granivores recorded the highest captures on north-east facing slopes, whereas omnivores were most abundant on west facing slopes. Herbivores reached their highest abundance on north-west facing slopes, while insectivores were most abundant on south-east facing slopes. Brady & Weil (2016) explained that topography influences soil moisture, nutrient availability, and vegetation development, all of which affect habitat quality and resource distribution. Likewise, Zhang et al. (2022) showed that slope-related

environmental gradients can alter habitat suitability and species distributions through their effects on vegetation and local environmental conditions. These findings suggest that the guild-specific responses observed in the present study may reflect differences in vegetation structure, food resources, or microhabitat conditions associated with different slope orientations, although these factors were not measured directly. In contrast, herbivores and small omnivores generally recorded relatively low captures and MKA across most slope aspect orientations, which may indicate that these guilds were less responsive to aspect-related environmental gradients or influenced by other ecological factors, such as seasonal resource availability or competition, that were beyond the scope of this study. The findings suggest that slope aspect may indirectly influence small mammal assemblages through its effects on local microclimatic conditions and habitat characteristics, thereby contributing to species-specific and functional guild responses. As slope aspect has received relatively little attention in African grassland small mammal ecology, these findings provide preliminary evidence that topographic heterogeneity may be an important consideration for understanding habitat suitability and conserving KZNSS grassland biodiversity.

The population estimates and trap success of individual species exhibited species-specific responses to slope aspect. *Rhabdomys dilectus* and *M. varius* attained their highest abundances on south-west facing slopes, whereas *A. chrysophilus* was most abundant on north facing slopes and *M. natalensis* peaked on west facing slopes. Trap success mirrored these patterns, suggesting that slope aspect may have influenced both species occurrence and activity. Chidodo et al. (2020) found that rodent assemblages in the North Uluguru Mountains varied with habitat characteristics associated with topography. Zhang et al. (2022) showed that slope-related environmental gradients can influence habitat suitability and species distributions. Similarly, Bennie et al. (2006) demonstrated that slope aspect modifies solar radiation and local microclimates, and Geiger et al. (2009) showed that these differences influence temperature and soil moisture. Yang et al. (2020) further reported that variation in slope-related microclimates can affect vegetation structure and habitat heterogeneity. These studies suggest that the species-specific patterns observed in the present study may reflect differences in vegetation structure, resource availability, or microhabitat conditions associated with different slope orientations, although these variables were not measured directly.

Slope aspect also influenced age structure and reproductive activity. Adult populations of *A. ineptus*, *M. natalensis*, and *R. dilectus* were highest on north, north-west, and south-east facing slopes, while juvenile distributions differed among species. Reproductive activity of *M. natalensis* and *R. dilectus* was also associated with slope aspect, with lactating *M. natalensis* females most abundant on south-east facing slopes. These patterns may indicate that some slope orientations provide more favourable conditions for breeding and recruitment. Bennie et al. (2006) and Geiger et al. (2009) showed that south and south-east facing slopes generally experience cooler temperatures and greater soil moisture retention. Yang et al. (2020) and Zhang et al. (2022) found that these conditions often support greater vegetation cover and habitat complexity. Such conditions may provide improved nesting sites, more stable food resources, and increased protection from predators, potentially benefiting reproductive individuals and younger age classes. However, because vegetation structure, food availability, and predator abundance were not quantified, these explanations remain speculative. Shuai et al. (2017) further demonstrated that slope aspect and elevation interact to influence local environmental conditions and small mammal assemblages, suggesting that the demographic patterns observed in this study may reflect the combined influence of the local elevational gradient and slope aspect rather than slope aspect alone.

In contrast, neither male nor female captures differed significantly among slope orientations, indicating that sex-specific distributions largely reflected overall population patterns rather than direct responses to slope aspect. Although no studies have explicitly examined sex-specific responses to slope aspect in KZNSS grasslands, Dorigo et al. (2021) and Chidodo et al. (2020) showed that topographic variation influences small mammal abundance and assemblage structure by modifying habitat conditions. In the present study, the lack of significant differences between males and females may therefore suggest that slope aspect influenced overall habitat suitability rather than the distribution of either sex. The absence of significant differences between males and females in the present study therefore suggests that slope aspect may influence habitat suitability at the population level, affecting abundance, age structure, and reproductive activity without differentially influencing the distribution of either sex.

Small mammal morphometric traits were significantly influenced by slope aspect. The observed trends showed that the rodent species *A. ineptus* and *M. natalensis* had larger head and body lengths on south-east facing slopes, while *R. dilectus* and *M. varius* tended to be larger on north-west facing slopes. The shrew species, *C. flavescens* was largest on south-west facing slopes. Such variation may be mediated by aspect-driven differences in microclimate, vegetation structure, and resource availability. Bennie et al. (2006) demonstrated that slope aspect influences local patterns of solar radiation, temperature, and moisture, while Yang et al. (2020) and Zhang et al. (2022) showed that these environmental differences can affect vegetation productivity, habitat structure, and resource availability. Although few studies have explicitly examined the effects of slope aspect on small mammal morphology, Shuai et al. (2017) reported that slope aspect and elevation interact to shape environmental gradients associated with small mammal assemblages. Similarly, Chidodo et al. (2020) found that topographic variation influenced rodent assemblages through differences in habitat characteristics, while Dorigo et al. (2021) showed that microhabitat conditions associated with topographic variation affected small mammal occurrence and activity. These studies suggest that the variation in body size and morphometric traits observed in the present study may reflect species-specific responses to habitat quality, energetic constraints, and resource availability associated with particular slope orientations, although these mechanisms were not directly measured. These findings suggest that, within KZNSS landscape, slope aspect may influence not only small mammal abundance and assemblage composition but also morphological variation through its effects on habitat conditions and microclimate. This highlights the importance of incorporating topographic heterogeneity into habitat management and conservation planning.

6.1.3 Seasonal variations on the KwaZulu-Natal Sandstone Sourveld ecosystem.

Seasonal variation is a key driver of small mammal assemblage dynamics, although its effects may be mediated by fine-scale landscape features that influence local microclimatic conditions, as suggested by Guiden & Orrock (2020). Merritt et al. (2003) demonstrated that seasonal changes in rainfall and temperature influence resource availability and habitat conditions, which in turn affect small mammal abundance, reproduction, and activity patterns. In southern African grasslands, including the

KZNSS landscape of the EMA, distinct wet and dry seasons are generally associated with seasonal shifts in small mammal assemblages, with higher abundance and capture rates typically recorded during warmer, wetter periods (Monadjem & Perrin, 2003). Similarly, Ramalho et al. (2022) and Shilereyo et al. (2023) reported that seasonal rainfall and increased habitat productivity were associated with changes in small mammal abundance, capture rates, and assemblage composition in African grassland ecosystems. Merritt et al. (2003) and Monadjem & Perrin (2003) showed that increased food availability during the wet season may improve body condition and support breeding activity, potentially contributing to higher recruitment and population growth in many rodent species. In contrast, cooler and drier winter conditions are often associated with reduced vegetation cover, lower resource availability, and decreased reproductive activity. During these less favourable periods, some species may reduce activity and conserve energy through behavioural and physiological strategies such as fat accumulation, torpor, or other energy-saving mechanisms, as discussed by Genoud (2022). These studies suggest that seasonal climate and associated changes in resource availability are important drivers of small mammal assemblage dynamics in grassland ecosystems, although the magnitude of these responses is likely to vary among species and local habitat conditions.

Seasonal variation significantly influenced small mammal population dynamics in this study, with total captures and MKA differing among seasons, whereas trap success, species richness, and diversity remained unaffected. Higher captures and MKA values during autumn may suggest that this period provided more favourable conditions for small mammal populations, potentially due to increased vegetation productivity, food availability, and habitat cover following seasonal rainfall. Similar relationships between post-rainfall resource availability and small mammal population responses have been reported by Merritt et al. (2003) and Ferrari et al. (2022), although these studies were conducted in temperate forest and Mediterranean agroecosystems. In contrast, winter had the lowest captures and MKA values, which may be associated with reduced food availability, lower vegetation productivity, and decreased reproductive activity. Seasonal declines in small mammal abundance during colder and resource-limited periods have been documented in grassland systems by Briese & Smith (1974), while Stephenson (1994) highlighted the energetic constraints imposed by low temperatures on small mammal activity and energy allocation.

Functional guild responses further demonstrated that seasonal effects differed among trophic groups. Granivores, insectivores, and large omnivores showed peak captures during autumn, whereas small omnivores and herbivores reached higher abundance during spring and summer, respectively. These patterns likely reflect differences in resource requirements and seasonal availability of food resources among guilds. Similar guild-specific responses to seasonal resource variation have been reported in agricultural and grassland landscapes by Butet et al. (2006) and in savanna systems by Ofori et al. (2015). For example, increased autumn abundance of granivores may be associated with greater seed availability following seasonal productivity peaks, as observed in other grassland ecosystems (Jolly et al., 2024). The results indicate that seasonal changes in rainfall, vegetation productivity, and resource availability influence small mammal assemblages by affecting population abundance and guild-specific responses. Although comparable patterns have been reported across different ecosystems, the magnitude and timing of seasonal effects are likely shaped by local climatic conditions, habitat structure, and species-specific ecological requirements.

Seasonal variation did not significantly influence overall small mammal population estimates, but species-specific responses were evident. *Aethomys ineptus* was most abundant during spring, *M. natalensis* during summer, and *R. dilectus* during autumn. The shrews *C. flavescens* and *M. varius* also reached peak abundance in autumn. These differences likely reflect species-specific life-history strategies and responses to seasonal changes in resource availability and habitat conditions. Briesse & Smith (1974) and Merritt et al. (2003) demonstrated that seasonal climatic variation and associated changes in food resources influence breeding activity and population dynamics in small mammals. In southern Africa, *M. natalensis* populations commonly increase during warmer, wetter periods when rainfall promotes vegetation growth, food availability, and reproductive activity (Monadjem & Perrin, 1998). Similarly, the association of *R. dilectus* with structurally complex grassland habitats suggests that increased vegetation cover following seasonal rainfall may contribute to higher abundance during late summer and autumn (Monadjem et al., 2015). Trap success patterns supported these species-specific responses, with *M. natalensis* and *R. dilectus* showing higher captures during autumn, while *C. flavescens* exhibited significantly higher trap success in autumn compared with other seasons. Increased autumn abundance of shrews may be linked to greater invertebrate availability and

improved habitat conditions following summer rainfall, as shrews are strongly dependent on invertebrate prey resources (Skinner & Chimimba, 2005). Similar seasonal responses to rainfall and vegetation productivity have been documented in African small mammal assemblages (Ofori et al., 2015), while studies in agricultural and grassland systems have shown that seasonal changes in vegetation structure and resource availability can drive shifts in small mammal assemblage composition (Butet et al., 2006).

Seasonal patterns were also evident in population structure and reproduction. Adult *M. natalensis* were more frequently captured during summer, coinciding with increased reproductive activity, whereas juvenile *R. dilectus* were more common during spring, corresponding with their breeding period. Briesse & Smith (1974) and Merritt et al. (2003) demonstrated that seasonal variation in reproductive output and recruitment is strongly influenced by environmental conditions, particularly rainfall and resource availability. No juvenile *A. ineptus* were captured during the study, which may be related to its relatively low capture frequency and the reduced likelihood of detecting dependent juveniles that remain within nests and are less active above ground. Similar demographic variation associated with seasonal activity and trapping probability has been reported in small mammal studies (Hotopp et al., 2022). Reproductive activity further reflected species-specific seasonal strategies. *Mastomys natalensis* had more lactating females during summer, consistent with breeding responses to favourable rainfall conditions and increased food availability (Monadjem & Perrin, 1998). In contrast, *R. dilectus* showed pregnant females during spring and lactating females during autumn, suggesting an extended breeding period supported by favourable grassland conditions. Monadjem et al. (2015) reported that *R. dilectus* is strongly associated with mesic grasslands and dense vegetation, habitats that provide suitable cover and food resources. The higher autumn abundance of *C. flavescens* and *M. varius* may similarly reflect seasonal increases in habitat productivity and food availability following summer rainfall. Skinner & Chimimba (2005) described *C. flavescens* as an insectivorous species dependent on abundant invertebrate prey, while Ofori et al. (2015) demonstrated that rainfall-driven increases in habitat productivity influence small mammal abundance in African savanna ecosystems. These findings indicate that seasonal rainfall, vegetation productivity, and resource availability influence small mammal population dynamics, age structure, and

reproductive timing in the KZNSS landscape, while species-specific ecological traits mediate responses to seasonal environmental variation. Seasonal variation did not significantly influence the abundance of males or females in any of the species examined, suggesting that seasonal environmental conditions affected population abundance similarly in both sexes. However, species-specific differences in movement and reproductive behaviour may still influence seasonal capture probabilities (Monadjem et al., 2015).

Morphometric analysis revealed seasonal differences in head and body length for several species. *Mastomys natalensis* and *R. dilectus* exhibited larger body sizes during summer and autumn, possibly reflecting the greater proportion of reproductively active adults or improved body condition associated with increased food availability. Briele & Smith (1974) demonstrated that seasonal rainfall and resource availability influence breeding activity and population dynamics in small mammals, while Merritt et al. (2003) highlighted the importance of seasonal climatic conditions and food availability in influencing individual condition. Similarly, the greater body length observed in *M. varius* during autumn and winter may indicate improved growth under favorable environmental conditions. Ofori et al. (2015) likewise reported that rainfall-driven increases in vegetation productivity influence seasonal patterns of small mammal abundance. These findings suggest that seasonal variation contributes to differences in individual condition and population structure within KZNSS landscape small mammal assemblages, with species-specific life-history traits mediating responses to changing environmental conditions.

6.1.4 The effect of grassland condition on the KwaZulu-Natal Sandstone Sourveld ecosystem.

Grassland condition is widely recognised as an important determinant of small mammal assemblage structure, highlighting the value of small mammals as bio-indicators of habitat quality in the KZNSS landscape. Primary grasslands with good ecological integrity are characterised by dense, structurally complex vegetation and high plant diversity, features that enhance the availability of food resources, shelter, and protection from predators. Delany (1974) emphasised the importance of vegetation cover in creating suitable microhabitats for small mammals, while Churchfield et al. (1997) demonstrated that habitat heterogeneity increases the

availability of ecological niches. Similarly, Méré et al. (2015) showed that structurally diverse grasslands support more diverse small mammal assemblages, and Avenant & Cavallini (2007) together with MacFadyen (2014) reported that vegetation complexity and ecological integrity promote higher small mammal diversity than degraded habitats. Consequently, the occurrence of multiple feeding guilds and both habitat-associated and generalist species is widely regarded as indicative of relatively stable habitat conditions and low levels of disturbance, as noted by Avenant & Cavallini (2007) and de Klerk & Avenant (2017). Conversely, Yarnell et al. (2007) demonstrated that habitat degradation and vegetation simplification reduce assemblage diversity while favouring disturbance-tolerant generalist species. These studies demonstrate that variation in small mammal assemblage composition provides a sensitive measure of grassland condition, a conclusion further supported by Minor & Eichholz (2024).

Grassland condition significantly influenced small mammal total captures, MKA, and species richness, but had no significant effect on trap success or species diversity. Contrary to expectations, primary-good grasslands did not consistently support greater species abundance or richness than the other grassland condition categories. Instead, secondary-intermediate grasslands supported the highest total captures, MKA, and species richness. Most secondary-intermediate grasslands sampled in this study were located within protected areas, particularly GGNR, where reclaimed and restored grasslands are actively managed through prescribed burning, invasive species control, and habitat restoration (Giba Gorge Environmental Precinct Conservation Management Plan, 2016/2017). This suggests that small mammals may benefit from habitats experiencing intermediate levels of disturbance, provided that vegetation structure and habitat heterogeneity are maintained. Churchfield et al. (1997) demonstrated that structurally heterogeneous habitats provide a greater diversity of ecological niches for small mammals, while Yarnell et al. (2007) showed that disturbance history can strongly influence assemblage composition. In grassland ecosystems, Alard et al. (2005) demonstrated that vegetation composition in grasslands reflects both successional processes and previous land-use history, highlighting the lasting influence of disturbance and habitat management on grassland assemblages. Likewise, MacFadyen (2014) highlighted the importance of grassland structure and condition in shaping small mammal assemblages in South African grasslands. At GGNR, these ecological processes are supported through prescribed

burning, invasive species control, and habitat restoration, management interventions that are intended to maintain vegetation structure and biodiversity (Giba Gorge Environmental Precinct Conservation Management Plan, 2016/2017). Consistent with this approach, Little et al. (2015) reported that fire and grazing regimes influence vegetation structure and plant diversity in South African grasslands, whereas Tälle et al. (2018) emphasised the importance of appropriate grassland management for conserving biodiversity in semi-natural grasslands. These findings suggest that well-managed secondary-intermediate grasslands can retain sufficient structural complexity and resource availability to support abundant and species-rich small mammal assemblages despite representing secondary vegetation.

Grassland condition also influenced functional guild composition. Granivores and small omnivores were most abundant in secondary-intermediate grasslands, whereas herbivores peaked in secondary-good grasslands, suggesting guild-specific responses to vegetation composition and food resources. Yarnell et al. (2007) demonstrated that fire, grazing, and rainfall influence small mammal assemblages through their effects on vegetation characteristics. Thoresen et al. (2021) showed that disturbance can alter soil arthropod assemblages by modifying habitat structure and nutrient cycling, potentially affecting insectivorous and omnivorous small mammals. Likewise, Minor & Eichholz (2024) reported that vegetation diversity and structural complexity influence small mammal assemblages in native and restored grasslands. These findings suggest that differences in vegetation structure and resource availability associated with grassland condition shape both overall assemblage characteristics and the distribution of functional guilds.

Species-specific responses demonstrated that small mammal species differed in their responses to grassland condition. The higher abundance of *R. dilectus* and *M. varius* in secondary-intermediate and secondary-good grasslands may reflect their association with structurally heterogeneous habitats. Mérő et al. (2015) reported that semi-disturbed grasslands with patchy vegetation provide a variety of food resources and refuge from predators, thereby supporting diverse small mammal assemblages. Similarly, the greater abundance of insectivores, particularly *C. flavescens*, in secondary-intermediate grasslands may indicate greater invertebrate availability. Churchfield et al. (1997) demonstrated that shrew populations are closely linked to the abundance of soil and litter-dwelling invertebrates, while Sun et al. (2017) showed that

insectivorous small mammals are most abundant in habitats supporting diverse invertebrate assemblages. In contrast, *A. ineptus* and *M. natalensis* were more abundant in primary-good grasslands, most of which occurred within KNR. Skinner & Chimimba (2005) described *A. ineptus* as a species associated with dense vegetation and rocky substrates that provide shelter and nesting sites. Likewise, Loggins et al. (2019) demonstrated that vegetation cover influences habitat use and perceived predation risk in savanna small mammals, while Minor & Eichholz (2024) showed that structurally diverse grasslands support more complex small mammal assemblages. These studies suggest that the intact vegetation structure and rocky terrain characteristic of primary-good grasslands within KNR may have provided favourable habitat conditions for these species. However, because most primary-good grasslands were concentrated within the lower elevation KNR, the observed patterns may also reflect local site characteristics, including elevation and rocky terrain, rather than grassland condition alone.

Grassland condition also influenced demographic patterns, although these effects differed among species. Adult and subadult populations of *A. ineptus*, *M. natalensis*, *R. dilectus*, *C. flavescens*, and *M. varius* were generally highest in primary-good grasslands, while juveniles of *M. natalensis* and *R. dilectus* were also most abundant in these habitats. No juvenile *A. ineptus* were captured. However, only the age structure of *M. natalensis* differed significantly among grassland condition types, with adult numbers significantly higher in secondary-good grasslands. Churchfield et al. (1997) demonstrated that vegetation structure and habitat quality influence small mammal population composition by affecting shelter availability and resource access, while MacFadyen (2014) showed that differences in grassland condition influence demographic responses through changes in habitat suitability and food availability. These findings suggest that habitat condition may influence recruitment and population structure differently among species.

Reproductive activity also varied among grassland condition types. Although reproductive individuals of *A. ineptus*, *M. natalensis*, and *R. dilectus* were generally most abundant in primary-good grasslands, significant differences were detected only for *M. natalensis* and *R. dilectus*. In *M. natalensis*, scrotal males, imperforate females, and lactating females were significantly more abundant in primary-good grasslands, whereas pregnant females were significantly more abundant in secondary-good

grasslands. Similarly, *R. dilectus* had significantly higher numbers of abdominal males and pregnant females in primary-good grasslands. Merritt et al. (2003) demonstrated that favourable habitat conditions and increased food availability improve body condition and reproductive activity in small mammals, while Brieske & Smith (1974) reported that breeding intensity and recruitment increase under favourable environmental conditions. The greater abundance of reproductive individuals in primary-good grasslands may therefore indicate that these habitats provide favourable conditions for breeding, although local environmental characteristics and site-specific factors may also have contributed to these patterns. Grassland condition did not significantly influence the abundance of males or females in any of the species examined, suggesting that both sexes responded similarly to differences in habitat condition.

Morphometric traits, particularly head-body length, also varied with grassland condition. *Mastomys natalensis* and *R. dilectus* attained larger body sizes in secondary-good grasslands, which may reflect differences in habitat quality and resource availability among grassland condition types. MacFadyen (2014) demonstrated that grassland condition influences habitat suitability for small mammals in South African grasslands, while Minor & Eichholz (2024) showed that vegetation structure and habitat complexity shape small mammal responses in native and restored grasslands. In contrast, the insectivorous shrew *C. flavescens* attained larger body sizes in primary-good grasslands. Churchfield et al. (1997) demonstrated that shrew populations are closely associated with habitats that support abundant soil and litter dwelling invertebrates, while Skinner & Chimimba (2005) described *C. flavescens* as an insectivorous species dependent on productive habitats for prey resources. These findings suggest that variation in grassland condition may influence individual growth and body condition through differences in habitat quality and food availability. The observed demographic and morphometric responses reinforce the value of small mammals as bio-indicators of grassland condition and habitat quality in the KZNSS landscape.

6.1.5 Regional variations on the KwaZulu-Natal Sandstone Sourveld ecosystem.

Regional variation in elevation, protected area size, land use, and habitat condition among Kloof (Krantzkloof Nature Reserve (KNR); lower elevation; ~584 ha), Hillcrest

(Giba Gorge Nature Reserve (GGNR); mid elevation; ~400 ha), and Camperdown Rural (Doornrug and Uitkomst Conservancy DUC); higher elevation; ~4-900 ha) is likely to influence small mammal assemblages. Yates et al. (1997) demonstrated that larger habitat patches generally support greater species richness by encompassing a wider diversity of habitats and ecological niches. Caro (2002) highlighted the importance of protected area size in maintaining viable mammal populations. Similarly, Bedoya-Durán et al. (2021) showed that larger, structurally heterogeneous habitats support greater mammalian diversity by providing a broader range of resources and microhabitats. The larger DUC landscape may provide more suitable conditions for diverse small mammal assemblages than the smaller KNR and GGNR reserves. Protected area size also influences fragmentation and edge effects. Fahrig (2003) demonstrated that habitat fragmentation increases isolation and edge influence, while Hurst et al. (2013) showed that edge-related changes in microclimate and habitat quality can negatively affect mammal assemblages.

Smaller reserves such as KNR and GGNR may contain proportionally less core habitat and be more susceptible to edge effects than the larger DUC landscape. Taylor et al. (1993) first proposed that landscape connectivity is fundamental to maintaining ecological processes, and Mortelliti et al. (2010) demonstrated that connectivity plays an important role in sustaining small mammal populations in fragmented landscapes. Reduced connectivity may therefore limit dispersal and population persistence in smaller or more isolated protected areas. In addition to protected area size, regional variation in climate, land use, and habitat quality is likely to influence assemblage composition. Lagesse & Thondhlana (2016) and Ramahlo et al. (2022) demonstrated that land transformation and habitat modification alter small mammal diversity and abundance in southern African ecosystems. Simelane et al. (2018) and Torre et al. (2024) showed that climatic variation associated with elevation influences primary productivity and resource availability, thereby affecting small mammal assemblages. Loggins et al. (2019) further demonstrated that vegetation structure influences habitat use and occupancy in African savannas, highlighting the importance of habitat complexity alongside climatic and anthropogenic factors. These studies indicate that variation in small mammal assemblages across the three study regions is likely driven by interactions among protected area size, habitat heterogeneity, landscape connectivity, elevation, and anthropogenic disturbance.

The regions significantly influenced total captures, trap success, MKA, species richness, and diversity. These regional patterns broadly reflected the elevational patterns observed in this study, suggesting that variation among Kloof, Hillcrest, and Camperdown Rural was, at least in part, associated with the local elevational gradient. These differences suggest that regional variation in small mammal assemblages is influenced not only by elevation but also by protected area size, land-use history, vegetation structure, habitat heterogeneity, and landscape connectivity. McCain (2005) demonstrated that changes in climatic conditions and productivity along elevational gradients influence mammal species richness, supporting the observed differences among the three regions. Similarly, Lagesse & Thondhlana (2016) found that land transformation reduces habitat quality and alters small mammal assemblages in southern African ecosystems. Ramahlo et al. (2022) showed that urbanisation and habitat modification influence small mammal abundance and diversity, suggesting that regional differences in land-use history may also have contributed to the assemblage patterns observed in this study.

Camperdown Rural, situated at the highest elevation and comprising the largest reserve, likely provided a wider range of vegetation types, microhabitats, and food resources than the smaller protected areas at Hillcrest and Kloof. Loggins et al. (2019) demonstrated that structurally complex vegetation increases habitat use by small mammals by providing greater cover and foraging opportunities, while Shilereyo et al. (2023) reported that heterogeneous habitats support greater small mammal species richness through increased habitat and resource diversity. Likewise, Yates et al. (1997) demonstrated that larger habitat patches support more species because they contain a greater diversity of habitats, whereas Fahrig (2003) showed that habitat fragmentation increases edge effects and isolation, reducing habitat quality for many species. Umetsu & Pardini (2007) further demonstrated that fragmented landscapes restrict dispersal and reduce connectivity among populations, processes that may contribute to lower diversity and abundance in the smaller, more isolated protected areas.

These regional differences were also reflected in functional guild composition. Granivorous and insectivorous small mammals were most abundant in Camperdown Rural, suggesting that the larger and more heterogeneous landscape supported greater food availability and habitat diversity. Taylor (1998) demonstrated that

structurally diverse grasslands provide abundant seeds and shelter for granivorous rodents, while Monadjem & Perrin (2003) showed that productive grasslands support higher abundances of both rodent and shrew species through increased food resources and vegetation cover. In contrast, the relatively higher abundance of large omnivores in Hillcrest may reflect moderate habitat disturbance associated with its smaller protected area size and surrounding urban land use. Lagesse & Thondhlana (2016) demonstrated that anthropogenic disturbance often favours habitat generalists capable of exploiting altered environments, while Luna et al. (2017) reported that omnivorous small mammals become more abundant in moderately disturbed habitats because of their dietary flexibility. These studies suggest that interactions among elevation, protected area size, vegetation structure, habitat heterogeneity, and land-use intensity shape both small mammal assemblage structure and functional guild composition, reinforcing the value of small mammals as indicators of habitat quality and landscape condition.

Species-specific population patterns further demonstrated that environmental differences among Kloof, Hillcrest, and Camperdown Rural influenced small mammal assemblage structure. Avenant & Cavallini (2007) and de Klerk & Avenant (2017) highlighted that predictable species-specific responses to environmental variation underpin the value of small mammals as bio-indicators of habitat quality and landscape change. *Mastomys natalensis* is known to respond positively to productive habitats with abundant food resources and moderate disturbance, consistent with the effects of habitat modification on small mammal assemblages reported by Ramahlo et al. (2022). In contrast, Monadjem et al. (2015) reported that *R. dilectus* is strongly associated with dense, structurally complex grasslands, which may explain its higher abundance in the larger, less fragmented Camperdown Rural landscape. Similarly, the relatively high abundance of *C. flavescens* in Hillcrest may reflect its preference for moist habitats with abundant invertebrate prey. Churchfield (1990) demonstrated that shrews are closely associated with moist microhabitats and abundant invertebrates, making them sensitive indicators of fine-scale habitat conditions. These findings suggest that differences in vegetation structure, habitat quality, and food availability contributed to the observed species-specific responses.

Local elevational gradients likely also contributed to differences in assemblage composition by influencing microclimate, vegetation, and resource availability. McCain

(2005) demonstrated that climatic variation along elevational gradients shapes mammal species richness. Lagesse & Thondhlana (2016) showed that land-use history and habitat condition influence small mammal assemblages in South African grasslands. Similarly, Umetsu & Pardini (2007) and Bedoya-Durán et al. (2021) demonstrated that habitat fragmentation, connectivity, and landscape heterogeneity affect small mammal abundance, species richness, and assemblage composition. These interacting environmental factors likely contributed to the regional patterns observed in this study. Demographic patterns also varied among regions. Hillcrest supported higher numbers of reproductively active adult *M. natalensis*, suggesting that this region may have provided favourable conditions for breeding. Pulliam (1988) proposed that high-quality habitats can function as demographic sources where reproduction exceeds mortality, although source-sink dynamics were not directly assessed in the present study. The greater abundance of breeding adults in Hillcrest may therefore be consistent with favourable habitat conditions that support local population growth. In contrast, the higher abundance of juveniles in Kloof during autumn may reflect seasonal recruitment following rainfall, when increased primary productivity enhances food availability. Monadjem et al. (2015) described reproduction in southern African small mammals as being closely associated with favourable environmental conditions and resource availability. Likewise, age and reproduction related patterns in *C. flavescens* may reflect variation in habitat moisture and invertebrate prey availability, consistent with the ecological observations of Churchfield (1990). These findings suggest that regional differences in elevation, habitat structure, and resource availability influenced species composition, reproductive activity, and population structure across the KZNSS landscape.

Body size in small mammals is influenced by environmental gradients, including climate, primary productivity, and habitat quality. Millien et al. (2006) demonstrated that environmental variation can drive morphological differences among mammalian populations, while Yom-Tov & Geffen (2006) showed that body size often varies geographically in response to climatic factors such as temperature and precipitation. In the present study, head and body length differed significantly among regions for *A. ineptus* and *C. flavescens*. *A. ineptus* attained larger body sizes in Camperdown Rural than in Hillcrest and Kloof, whereas *C. flavescens* reached its largest body sizes in Kloof. These regional differences may reflect variation in habitat quality, resource

availability, and local environmental conditions. Monadjem & Perrin (2003) demonstrated that vegetation structure and habitat quality influence small mammal populations, while Millien et al. (2006) linked environmental heterogeneity to morphological variation. The larger body size of *A. ineptus* in Camperdown Rural may therefore be associated with the larger reserve size, greater habitat heterogeneity, and more diverse vegetation structure of this region, which could provide increased foraging opportunities and shelter. Similarly, the larger body size of *C. flavescens* in Kloof may reflect relatively mesic habitat conditions and greater invertebrate prey availability. Churchfield (1990) highlighted the importance of habitat quality and prey resources in shaping shrew ecology, supporting this interpretation. Bedoya-Durán et al. (2021) further demonstrated that structurally complex, resource-rich habitats support diverse and persistent mammal assemblages, suggesting that habitat heterogeneity may also influence individual performance. These findings suggest that climate, habitat quality, and landscape characteristics interact to influence body size variation in small mammals. Consequently, morphometric traits may provide useful indicators of habitat suitability and environmental conditions. Avenant & Cavallini (2007) and de Klerk & Avenant (2017) highlighted the value of small mammals as bio-indicators because their populations and morphological responses are sensitive to environmental change.

6.1.6 The effect of habitat management regimes (treatments) on the KwaZulu-Natal Sandstone Sourveld ecosystem.

Protected areas play a fundamental role in conserving biodiversity, ecosystem processes, and ecological integrity. Rodrigues et al. (2004) identified protected areas as a cornerstone of global conservation strategies, while Dudley (2008) emphasised their role in limiting anthropogenic disturbance and maintaining ecological integrity. Their effectiveness has been widely demonstrated. Gray et al. (2016) reported that biodiversity is generally higher within protected areas than in comparable unprotected landscapes, Santangeli et al. (2022) showed that protected areas maintain suitable habitats for mammals and birds, and Magoulick et al. (2024) found greater mammal recovery within protected areas. However, Bedoya-Durán et al. (2021) demonstrated that conservation outcomes also depend on surrounding habitat quality, while Taylor et al. (1993) and Brudvig et al. (2009) highlighted the importance of habitat connectivity

for maintaining ecological processes across landscapes. In the present study, protected grasslands were characterised by limited grazing and relatively intact vegetation, whereas semi-managed and unprotected D'MOSS grasslands outside protected areas experienced greater grazing, agriculture, fragmentation, and urbanisation. These differences were reflected in small mammal assemblages, with protected areas supporting greater abundance, species richness, and functional diversity. Monadjem & Perrin (2003) demonstrated that structurally complex grasslands provide greater habitat heterogeneity and resource availability for small mammals, whereas Fahrig (2003, 2017, 2020) and Fischer & Lindenmayer (2007) showed that fragmentation and landscape modification reduce habitat quality and favour disturbance-tolerant species.

The minimally disturbed control site consistently supported the highest total captures, trap success, MKA, species richness, and diversity, indicating that relatively intact grassland habitats provide favourable conditions for small mammal assemblages. Similar patterns have been reported by Caro (2002), Lagesse & Thondhlana (2016), Simelane et al. (2018), Shilereyo et al. (2023), and Torre et al. (2024), who demonstrated that protected or structurally heterogeneous habitats generally support greater small mammal abundance and diversity than transformed landscapes. For example, Caro (2002) reported higher small mammal diversity within Katavi National Park than in surrounding human-modified areas, while Lagesse & Thondhlana (2016) found greater species richness inside protected grasslands than in adjacent grazed habitats. Likewise, Loggins et al. (2019) showed that vegetation structure strongly influences small mammal occupancy and activity, whereas Hurst et al. (2013, 2014) demonstrated that habitat modification and edge effects alter assemblage composition. At the landscape scale, Fahrig (2003, 2017), Hernández & Simonetti (2013), Gray et al. (2016), Bedoya-Durán et al. (2021), and Magoulick et al. (2024) collectively demonstrated that fragmentation, land-use change, and reduced habitat connectivity simplify habitat structure and lower biodiversity, while protected and well-connected landscapes generally maintain more diverse mammal assemblages. Taylor et al. (1993), Umetsu & Pardini (2007), and Brudvig et al. (2009) further highlighted the importance of habitat connectivity in maintaining biodiversity across fragmented landscapes. Although vegetation structure, connectivity, and microclimatic conditions were not directly quantified in the present study, the lower diversity and altered

assemblage composition observed in disturbed sites may reflect the combined effects of habitat degradation, fragmentation, and anthropogenic pressures. These findings reinforce the value of small mammals as bio-indicators of grassland condition, with the high diversity and abundance recorded at the minimally disturbed control site likely indicating relatively favourable habitat conditions and greater ecological stability than more disturbed grasslands.

Higher trap success was recorded outside protected areas than within GGNR. This pattern may be associated with the proximity of an adjacent D'MOSS grassland patch that was directly connected to the reserve, rather than being an isolated patch like the other D'MOSS grassland patches outside the protected area. This connectivity may provide additional resources or facilitate small mammal movement between the surrounding habitats and the protected area, potentially influencing the observed pattern. However, landscape connectivity, movement patterns, and dispersal were not directly assessed in this study; therefore, these interpretations should be considered as potential explanations rather than confirmed mechanisms. Crooks & Sanjayan (2006) highlighted the importance of landscape connectivity in maintaining ecological processes, while Umetsu & Pardini (2007), Bedoya-Durán et al. (2021), Brudvig et al. (2009), Brodie et al. (2023), and Cumming (2025) demonstrated that habitat connectivity and ecological spillover can enhance biodiversity beyond reserve boundaries. The higher trap success outside GGNR may reflect connected edge habitats and a permeable landscape matrix rather than protection status alone. Alternatively, differences in vegetation structure, resource availability, or local habitat conditions outside the reserve may also have contributed to these patterns.

Functional guilds also responded differently to habitat treatments, indicating that habitat quality and vegetation structure influence resource use by small mammals. Granivores and insectivores were most abundant at the minimally disturbed control site, consistent with findings by Caro (2002), Lagesse & Thondhlana (2016), Ramahlo et al. (2022), and Shilereyo et al. (2023), who reported that relatively intact habitats support higher abundances of specialist small mammals. Van Deventer & Nel (2006), Kuiper & Parker (2013), and Loggins et al. (2019) demonstrated that dense vegetation provides food resources and refuge from predators, while Paniccia et al. (2022) highlighted the role of habitat structure in maintaining functional diversity. Although vegetation characteristics and food resources were not directly measured, the greater

abundance of granivores and insectivores at the control site may indicate more favourable foraging conditions and greater microhabitat heterogeneity. Large omnivores were most abundant inside protected areas, which may reflect their broader dietary niche and greater tolerance of environmental variation, consistent with the ecological characteristics of generalist species described by Futuyma & Moreno (1988) and Devictor et al. (2010). Herbivores and small omnivores were not significantly influenced by treatment, suggesting broader ecological tolerances or responses to environmental factors not examined in this study. These findings indicate that habitat quality, vegetation complexity, and landscape connectivity, rather than protection status alone, are important drivers of small mammal abundance and functional composition, highlighting the importance of conserving structurally diverse and well-connected grassland landscapes.

The numbers of breeding adult *M. natalensis* were higher inside protected areas than outside protected areas and at the control site. This pattern may suggest that protected habitats provided more favourable conditions for reproduction, potentially through greater vegetation cover, reduced disturbance, or improved resource availability. Leirs et al. (2008), Mulungu et al. (2016), and Shilereyo et al. (2023) similarly reported that reproductive activity in small mammals is often associated with habitat quality and resource availability. Indicators of reproductive activity in *R. dilectus* were highest at the control site, which may reflect this species's ability to utilise minimally disturbed habitats with suitable resources and habitat conditions. Meynard et al. (2012), Rymer & Pillay (2012), and Lagesse & Thondhlana (2016) demonstrated that *R. dilectus* can occupy a range of open and heterogeneous habitats and may respond positively to favourable environmental conditions. The absence of juveniles of *A. ineptus* across treatments may indicate low recruitment during the sampling period, seasonal variation in reproductive activity, or species-specific habitat preferences. However, because recruitment dynamics and breeding phenology were not directly investigated, these explanations remain speculative. Happold & Happold (1992), Monadjem & Perrin (1998), and Caro (2002) highlighted that reproductive patterns and recruitment in small mammals are strongly influenced by environmental conditions and habitat suitability. The observed patterns suggest that habitat quality and resource availability may influence breeding activity and recruitment in small mammals, although additional

studies specifically examining reproductive ecology would be necessary to confirm these relationships.

Morphometric differences among treatments suggested that habitat conditions may influence body size patterns in small mammals. *Mastomys natalensis* attained larger body sizes outside protected areas than inside protected areas and at the control site. One possible explanation is that reduced interspecific competition in disturbed landscapes may allow greater resource availability and contribute to increased body size. However, competitive release was not directly assessed in this study, as competition and dietary overlap were not measured; therefore, this interpretation should be considered a potential explanation requiring further investigation rather than a confirmed mechanism. Caro (2002), Kennis et al. (2008), and Mulungu et al. (2016) reported that human-modified habitats are often characterised by simplified small mammal assemblages dominated by disturbance-tolerant species. As a highly adaptable generalist, *M. natalensis* may benefit from reduced competition where habitat specialists are less abundant. Boutin (1990) proposed that reduced competition can increase per capita resource availability, potentially allowing greater energy allocation to growth. However, because resource availability and interspecific competition were not directly measured, this explanation remains speculative. In contrast, *R. dilectus* and *C. flavescens* attained larger body sizes inside protected areas. Meynard et al. (2012) and Rymer & Pillay (2012) demonstrated that *R. dilectus* is associated with mesic habitats and continuous vegetation cover, while Churchfield (1990) and Churchfield & Rychlik (2006) highlighted the dependence of shrews on moist habitats with abundant invertebrate prey. The larger body sizes observed in protected areas may therefore reflect more favourable habitat conditions, including greater vegetation complexity, stable microclimates, and more consistent food resources, although these factors were not directly quantified in the present study. The contrasting morphometric responses among species suggest that body size is influenced by species-specific ecological traits and habitat requirements. Yom-Tov & Geffen (2006), Meiri & Dayan (2003), and McCain (2005) demonstrated that environmental conditions, climate, and habitat quality contribute to geographic variation in mammalian body size, while Goodman & Rakotondravony (2006), Dammhahn et al. (2012), and Shilereyo et al. (2023) showed that favourable habitat conditions are associated with improved body condition and population performance.

These findings suggest that variation in body size among treatments may reflect differences in habitat quality and resource availability, although further studies measuring vegetation characteristics, food resources, and body condition are needed to confirm these mechanisms.

Habitat fragmentation and degradation influence mammal populations by altering habitat suitability, connectivity, and resource availability. Fahrig (2003, 2017), Umetsu & Pardini (2007), and Panzacchi et al. (2010) demonstrated that fragmented landscapes modify species composition and disproportionately affect habitat-associated species. The present findings therefore suggest that conservation management may influence not only small mammal abundance and diversity but also morphological characteristics and, potentially, population performance. Because small mammals respond rapidly to environmental change, morphometric traits may provide useful indicators of habitat condition and ecosystem integrity. Lindenmayer & Likens (2010) highlighted the value of biological indicators for detecting ecological change, while van Buren & Jarzyna (2022) demonstrated that functional composition in small mammal assemblages changes in response to environmental change. Although body condition, resource availability, and competition were not directly measured, the species-specific morphometric patterns observed in this study may represent early indicators of habitat quality across protected and human-modified landscapes. The occurrence and abundance of *P. melanotis*, *O. auratus*, *A. ineptus*, and *Steatomys* spp. further highlight the potential value of habitat-associated small mammals as bio-indicators of ecosystem integrity. These species were classified as habitat specialists based on published ecological information rather than direct measurements of resource use or niche breadth. Lindenmayer & Likens (2010) recognised specialist species as sensitive ecological indicators because their narrow habitat requirements result in rapid responses to vegetation change and habitat fragmentation, a pattern also reported by Shilereyo et al. (2023) and Viteri et al. (2023). Their generally low abundances are consistent with findings by Monadjem & Perrin (2003), Hurst et al. (2013), and Loggins et al. (2019), who showed that habitat-associated small mammals are often more sensitive to environmental disturbance than generalist species.

Poemys melanotis was recorded only inside protected areas, suggesting a dependence on relatively intact and structurally complex grasslands, consistent with the observations of Monadjem & Perrin (2003) and Lagesse & Thondhlana (2016).

Otomys auratus occurred across all habitat treatments but was more abundant inside protected areas, which may reflect its association with mesic habitats such as wet grasslands, seepages, and riparian vegetation, as described by Kuiper & Parker (2013), Lagesse & Thondhlana (2016), and Loggins et al. (2019). *Aethomys ineptus* also occurred across all treatments but reached its highest abundance inside protected areas, possibly indicating an association with structurally complex habitats, whereas *Steatomys* spp. was recorded only at the minimally disturbed control site, which may suggest a dependence on intact grasslands or particular soil conditions. Because microhabitat characteristics and resource availability were not directly quantified, these interpretations remain speculative.

In contrast, the widespread occurrence of *M. natalensis* and *R. dilectus* across all habitat treatments reflects their broader ecological tolerances and ability to persist in modified landscapes, as reported by Hurst et al. (2013), Lagesse & Thondhlana (2016), and Loggins et al. (2019). These findings reinforce the value of specialist small mammals as indicators of habitat quality and ecosystem integrity. Lindenmayer & Likens (2010) proposed that declines in specialist species can provide early warning of ecosystem degradation, while Hurst et al. (2014) and Gray et al. (2016) demonstrated that changes in small mammal assemblages closely track habitat condition. Although further studies examining habitat requirements and demographic responses are needed, monitoring specialist small mammals may provide a practical approach for assessing grassland condition and conservation effectiveness.

CHAPTER 7

7.1 CONCLUSION

This study provides a comprehensive assessment of how small mammal assemblages respond to ecological and environmental variability within the highly fragmented and KZNSS landscape of the EMA. The findings suggest that small mammal assemblages may be influenced by a complex interaction of environmental factors, including a relatively modest local elevational gradient, slope aspect, seasonal variation, grassland condition, regional variation, and habitat management practices. Although the elevational gradient examined in this study was relatively limited, elevation nevertheless appeared to influence assemblage structure, species composition, reproductive patterns, demographic characteristics, and morphometric traits. Slope aspect also appeared to play an important role in structuring small mammal assemblages, potentially through its effects on microclimatic conditions, vegetation structure, and resource availability. Because slope aspect remains relatively understudied in African grassland ecosystems, these findings may contribute important insights into its ecological significance and highlight the need for further investigation of this variable in future biodiversity studies. More broadly, the results suggest that habitat quality, structural heterogeneity, and landscape context may be important determinants of small mammal diversity and persistence within KZNSS landscape. Protected and minimally disturbed grasslands generally supported higher abundances of habitat-associated and specialist species, whereas disturbed and fragmented habitats tended to be dominated by more disturbance-tolerant generalists such as *M. natalensis* and *R. dilectus*. These contrasting responses suggest that small mammal assemblages may provide useful indications of ecosystem condition and habitat integrity. From a conservation perspective, the findings indicate that maintaining habitat quality, structural heterogeneity, and connectivity among remnant grassland patches within the D'MOSS landscape may be important for supporting diverse and functionally intact small mammal assemblages. The generally low abundances and restricted distributions of several habitat-associated species further suggest that specialist species may be particularly vulnerable to continued habitat modification and fragmentation. Overall, this study suggests that small mammals have considerable potential as bioindicators of ecological condition within KZNSS landscape. Changes in their abundance, assemblage composition, functional

structure, and morphometric characteristics may provide early indications of ecological change and could therefore assist in monitoring ecosystem health, informing restoration initiatives, and guiding conservation planning within this threatened ecosystem.

7.2 Limitations of the Study and Future Research

Despite providing important insights into the responses of small mammal assemblages within the Critically Endangered KZNSS landscape, of the EMA, this study has several limitations that should be considered when interpreting the findings. Although sampling was conducted across multiple sites and habitat management regimes, the study was restricted to the EMA, which may limit the broader applicability of the results to KZNSS grasslands elsewhere. Furthermore, while key environmental variables such as elevation, slope aspect, seasonality, grassland condition, regional variation and habitat management regimes were examined, fine-scale habitat characteristics, including vegetation composition, grass height, litter depth, soil properties, and microhabitat availability, were not quantified and may explain additional variation in small mammal assemblages. The reliance on live trapping may also have underestimated cryptic, trap-shy, or low-density species, while the relatively short sampling period may not have captured longer-term population dynamics associated with climatic variability. In addition, although slope aspect emerged as an important ecological driver, microclimatic variables such as soil moisture, temperature, and humidity were not measured directly. Future research should therefore adopt long-term, multi-scale approaches that incorporate detailed habitat and microclimatic measurements, complementary sampling techniques, and broader geographic coverage to improve understanding of the mechanisms underlying species-specific responses and further strengthen the use of small mammals as bioindicators of ecosystem condition. From a conservation perspective, the findings suggest that maintaining and restoring the remaining KZNSS grassland remnants within the D'MOSS network, improving connectivity between fragmented habitat patches, and implementing appropriate fire, grazing, and invasive alien plant management are likely to support diverse and functionally intact small mammal assemblages. Incorporating long-term small mammal monitoring into biodiversity management programmes may also provide a

practical means of assessing ecosystem condition and evaluating the effectiveness of conservation interventions within this Critically Endangered ecosystem.

7.3 Management Recommendations

Based on the findings of this study, conservation management within the KZNSS landscape should prioritise the maintenance and restoration of remaining grassland remnants within the D'MOSS network and surrounding landscapes. Conserving habitat heterogeneity and improving connectivity among fragmented grassland patches are likely to support diverse and functionally intact small mammal assemblages, particularly habitat-associated and specialist species that appear sensitive to habitat modification and fragmentation. Management practices that maintain indigenous grassland structure, including ecologically appropriate fire and grazing regimes and the control of invasive alien plant species, are also likely to enhance habitat quality and promote biodiversity persistence. Furthermore, conservation planning should consider environmental factors such as local elevational gradients and slope aspect, as these variables appear to influence small mammal assemblages and may help identify important habitat refugia within fragmented landscapes. Finally, incorporating long-term, seasonally representative monitoring of small mammal assemblages into biodiversity management programmes would provide a practical means of assessing ecosystem condition, evaluating the effectiveness of management interventions, and informing adaptive conservation strategies within this Critically Endangered ecosystem.

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CHAPTER 9

APPENDICES

Appendix 1: The overall small mammal species captured per season per trapline in the Krantzklou Nature Reserve from July 2018-June 2019.

Season	Species	Traplines										Total
		1	2	3	4	5	6	7	8	9	10	
Winter	<i>Mastomys natalensis</i>	5	1	2	7							15
Winter	<i>Aethomys ineptus</i>			1	3							4
Winter	<i>Crocidura flavescens</i>		1	1							1	3
		Total=5	Total=2	Total=4	Total=10						Total=1	Total: 22
Spring	<i>Mastomys natalensis</i>	12		8	10			2		3		35
Spring	<i>Aethomys ineptus</i>	1	7	3								11
Spring	<i>Crocidura flavescens</i>										2	2
Spring	<i>Mus minutoides</i>						2					2
Spring	<i>Lemniscomys rosalia</i>	1		1								2
		Total=14	Total=7	Total=12	Total=10		Total=2	Total=2		Total=3	Total=2	Total: 52
Summer	<i>Mastomys natalensis</i>	5		14	13			7	11	3	1	54
Summer	<i>Aethomys ineptus</i>		3									3
Summer	<i>Crocidura flavescens</i>					4						4
Summer	<i>Lemniscomys rosalia</i>								1			1
Summer	<i>Otomys auratus</i>				1							1
		Total=5	Total=3	Total=14	Total=14	Total=4		Total=7	Total=12	Total=3	Total=1	Total:63
Autumn	<i>Mastomys natalensis</i>	7	1	7	10		1	2	8	2	1	39
Autumn	<i>Mus minutoides</i>						1					1
Autumn	<i>Aethomys ineptus</i>		4									4
Autumn	<i>Crocidura flavescens</i>		1	1	1	1					1	5
		Total=7	Total=6	Total=8	Total=11	Total=1	Total=2	Total=2	Total=8	Total=2	Total=2	Total:49
Overall Captures												Total:186

Appendix 2: The overall small mammal species captured per season per trapline at Giba Gorge Nature Reserve from July 2018-June 2019.

Season	Species	Traplines										Total
		1	2	3	4	5	6	7	8	9	10	
Winter	<i>Mastomys natalensis</i>	1		14	11	4				1		31
Winter	<i>Micaelamys namaquensis</i>				2	6			1			9
Winter	<i>Rhabdomys dilectus</i>			3		1					9	13
Winter	<i>Crocidura flavescens</i>	1				1					4	6
Winter	<i>Otomys auratus</i>			3						1		4
Winter	<i>Dendromous melanotis</i>					1						1
Winter	<i>Mus minutoides</i>					1						1
		Total=2		Total=20	Total=13	Total=14			Total=1	Total=2	Total=13	Total=65
Spring	<i>Mastomys natalensis</i>	3	3	8	11	14		1	4	1	8	53
Spring	<i>Rhabdomys dilectus</i>	2		1							11	14
Spring	<i>Crocidura flavescens</i>	3	1		2	1						7
Spring	<i>Micaelamys namaquensis</i>				1	1						2
Spring	<i>Dendromous melanotis</i>				1	3						4
Spring	<i>Lemniscomys rosalia</i>		1									1
		Total=8	Total=5	Total=9	Total=15	Total=19		Total=1	Total=4	Total=1	Total=19	Total: 81
Summer	<i>Mastomys natalensis</i>		4	13	5	8		2	14		2	57
Summer	<i>Rhabdomys dilectus</i>				3						1	4
Summer	<i>Crocidura flavescens</i>				1						1	2
Summer	<i>Aethomys ineptus</i>									1		1
			Total=4	Total=13	Total=9	Total=8		Total=2	Total=14	Total=10	Total=4	Total: 64
Autumn	<i>Mastomys natalensis</i>	1	4	8	5	15		6	15	22	2	78
Autumn	<i>Rhabdomys dilectus</i>			3		1		1		1	3	9
Autumn	<i>Crocidura flavescens</i>	2			4			2			2	10
Autumn	<i>Micaelamys namaquensis</i>				1	2						3
Autumn	<i>Dendromous melanotis</i>				1							1
Autumn	<i>Lemniscomys rosalia</i>		1						1			2
Autumn	<i>Otomys auratus</i>			3						1	1	5
		Total=3	Total=5	Total=14	Total=11	Total=18		Total=9	Total=16	Total=24	Total=8	Total:108
Overall Captures												Total: 318

Appendix 3: The overall small mammal species captured per season per trapline at Doornrug and Uitkomst from July 2018-June 2019.

		Traplines										
Season	Species	1	2	3	4	5	6	7	8	9	10	Total
Winter	<i>Mastomys natalensis</i>	-	4	2		4	1	1				12
Winter	<i>Rhabdomys dilectus</i>	9	14		2	1	3	7	2	22	3	63
Winter	<i>Crocidura flavescens</i>	1	1							2	2	6
Winter	<i>Myosorex varius</i>		1					1		5		7
Winter	<i>Steatomys</i> spp.		1								1	2
Winter	<i>Aethomys ineptus</i>								1			2
		Total=10	Total=21	Total=2	Total=2	Total=5	Total=5	Total=9	Total=3	Total=29	Total=6	Total=92
Spring	<i>Mastomys natalensis</i>		3		6		2		1	-	1	13
Spring	<i>Rhabdomys dilectus</i>		2	1	6	3	1	5	4	14	1	37
Spring	<i>Crocidura flavescens</i>		2				1					3
Spring	<i>Myosorex varius</i>		1					3		1		5
Spring	<i>Aethomys ineptus</i>	1			2							3
		Total=1	Total=8	Total=1	Total=14	Total=3	Total=4	Total=10	Total=5	Total=15	Total=2	Total=63
Summer	<i>Mastomys natalensis</i>	1	5	4	10		1		1			22
Summer	<i>Rhabdomys dilectus</i>	4	4	4	2	2	3	2	3	6	4	34
Summer	<i>Crocidura flavescens</i>		1									1
Summer	<i>Lemniscomys rosalia</i>				2							2
Summer	<i>Myosorex varius</i>	1				1	3	4		2		11
Summer	<i>Micaelamys namaquensis</i>			1								1
Summer	<i>Otomys auratus</i>									1		1
Summer	<i>Aethomys ineptus</i>	1			4							5
		Total=7	Total=10	Total=9	Total=18	Total=3	Total=7	Total=6	Total=4	Total=9	Total=4	Total=77
Autumn	<i>Mastomys natalensis</i>	2	5	3	17		4	1	1			33
Autumn	<i>Rhabdomys dilectus</i>	4	16	7	15	4	11	14	5	16	1	93
Autumn	<i>Crocidura flavescens</i>	1	1	1				2				5
Autumn	<i>Lemniscomys rosalia</i>											
Autumn	<i>Myosorex varius</i>	3	4	2	4		3	4		2		22
		Total=10	Total=26	Total=13	Total=36	Total=4	Total=18	Total=21	Total=6	Total=18	Total=1	Total=153
Overall Captures												Total=385

Appendix 4: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests on metrics associated with the effect of elevational gradients on small mammal captures within eThekwinini KwaZulu-Natal Sandstone Sourveld Grasslands (July 2018 to June 2019).

Elevation	Elevation	Z-test	P-value
Total captures	Low - Mid	-18.20	0.055*
	Low - High	21.83	0.002*
	Mid - High	3.63	0.685
Trap success	Low - Mid	-8.01	0.400
	Low - High	24.30	<0.001*
	Mid - High	16.29	0.069
MKA	Low - Mid	-16.61	0.080
	Low - High	21.34	0.003*
	Mid - High	4.73	0.598
Species richness	Low - Mid	-14.6	0.115
	Low - High	30.08	<0.001*
	Mid - High	15.48	0.076
Species diversity	Low - High	27.92	<0.001*
	Low - Mid	-33.01	<0.001*
	High - Mid	-5.09	0.553

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 5: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests on functional guild metrics for total captures and minimum known alive values associated with the effect of elevation gradients on small mammal captures within KwaZulu-Natal Sandstone Sourveld Grasslands in eThekwinini (July 2018 to June 2019).

Metrics	Functional guilds	Elevation	Z-test	P-value
Total Captures	Granivores	Low – Mid	-11.23	0.190
		Low – High	47.26	<0.001*
		Mid – High	36.03	<0.001*
	Herbivores	High – Low	-1.70	0.677
		High – Mid	-17.30	<0.001*
		Low – Mid	-15.60	0.004*
	Insectivores	Mid – Low	0.09	0.992
		Mid – High	19.69	0.011*
		Low – High	19.60	0.001*
	Large Omnivores	High – Low	-5.27	0.452
		High – Mid	-20.92	0.018*
		Low – Mid	-15.65	0.096
	Small Omnivores	Mid – Low	2.99	0.406
		Mid – High	4.01	0.237
		Low – High	1.02	0.703
MKA	Granivores	Low – Mid	-12.10	0.158
		Low – High	46.97	<0.001*
		Mid – High	34.87	<0.001*
	Herbivores	Mid – Low	1.61	0.845
		Mid – High	20.98	0.007*
		Low – High	19.36	0.002*
	Insectivores	High – Low	-5.27	0.452
		High – Mid	-20.92	0.018*
		Low – Mid	-15.65	0.096
	Large Omnivores	High – Low	-6.69	0.338
		High – Mid	-20.83	0.018*
		Low – Mid	-14.14	0.131
	Small Omnivores	Mid – Low	2.99	0.406
		Mid – High	4.01	0.237
		Low – High	1.02	0.703

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 6: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests on associated with small mammals captures relative to slope aspect across KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Slope Aspect	Z-test	P-value
Total captures	South – North	5.38	0.800
	South – East	6.50	0.746
	South – North West	30.26	0.096
	South – North East	34.75	0.083
	South – South East	-46.09	0.017*
	South – South West	-46.38	0.029*
	South – West	-60.4	0.001*
	North – East	1.13	0.943
	North – North West	-24.89	0.064
	North – North East	-29.38	0.064
	North – South East	-40.72	0.007*
	North – South West	-41.00	0.018*
	North – West	-55.03	<0.001*
	East – North West	-23.76	0.037*
	East – North East	-28.25	0.046*
	East – South East	-39.59	0.003*
	East – South West	-39.88	0.012*
	East – West	-53.9	<0.001*
	North West – North East	44.88	0.694
	North West – South East	-15.83	0.123
	North West – South West	-16.11	0.231
	North West – West	-30.14	0.002*
	North East – South East	-11.34	0.392
	North East – South West	-11.63	0.463
	North East – West	-25.65	0.043*
	South East – South West	-0.28	0.985
	South East – West	-14.31	0.219
	South West – West	-14.03	0.334
Trap success	North – North West	-12.21	0.364
	North – East	18.58	0.241
	North – South	-26.75	0.209
	North – West	-30.20	0.038*
	North – South West	-34.06	0.050*
	North – North East	-42.04	0.008*
	North – South East	-53.78	<0.001*
	North West – East	6.37	0.578
	North West – South	-14.54	0.425
	North West – West	-17.99	0.059*
	North West – South West	-21.85	0.105
	North West – North East	29.83	0.009*
	North West – South East	-41.57	<0.001*
	East – South	-8.17	0.684
	East – West	-11.62	0.360
	East – South West	-15.48	0.329
	East – North East	-23.46	0.098
	East – South East	-35.20	0.008*
	South – West	-3.45	0.856
	South – South West	-7.31	0.731
	South – North East	15.29	0.446
	South – South East	-27.03	0.164
	West – South West	3.86	0.791
	West – North East	11.84	0.351
	West – South East	23.58	0.043*
	South West – North East	7.98	0.615
	South West – South East	19.72	0.190
	North East – South East	-11.74	0.376
MKA	South – East	6.13	0.760
	South – North	11.38	0.592
	South – North West	31.48	0.083
	South – North East	35.88	0.073

Metrics	Slope Aspect	Z-test	P-value
Species richness	South – South East	-46.06	0.017*
	South – South West	-51.13	0.016*
	South – West	-61.5	0.001*
	East – North	-5.25	0.740
	East – North West	-25.35	0.026*
	East – North East	-29.75	0.036*
	East – South East	-39.94	0.003*
	East – South West	-45.00	0.004*
	East - West	-55.38	<0.001*
	North – North West	-20.10	0.134
	North – North East	-24.50	0.122
	North – South East	-34.69	0.021*
	North – South West	-39.75	0.022*
	North – West	-50.13	<0.001*
	North West – North East	4.40	0.700
	North West – South East	-14.59	0.155
	North West – South West	-19.65	0.143
	North West – West	-30.03	0.002*
	North East – South East	-10.19	0.442
	North East – South West	-15.25	0.335
	North East – West	-25.63	0.043*
	South East – South West	-5.06	0.736
	South East – West	-15.44	0.184
	South West – West	-10.38	0.474
	South – North	0.25	0.990
	South – East	3.58	0.854
	South – North East	11.67	0.550
	South – North West	13.78	0.437
	South – South West	-21.56	0.298
	South – West	-37.18	0.045*
	South – South East	-37.38	0.048*
	North – East	3.33	0.829
	North – North East	-11.42	0.460
	North – North West	-13.53	0.302
	North – South West	-21.31	0.208
	North – West	-36.93	0.009*
	North – South East	-37.13	0.011*
	East – North East	-8.08	0.558
	East – North West	-10.19	0.360
	East – South West	-17.98	0.244
	East – West	-33.59	0.007*
	East – South East	-33.79	0.009*
	North East – North West	-2.11	0.850
	North East – South West	-9.9	0.522
	North East – West	-25.51	0.039*
	North East – South East	-25.71	0.047*
	North West – South West	-7.79	0.552
	North West – West	-23.40	0.012*
	North West – South East	-23.60	0.018*
Species diversity	South West – West	-15.61	0.270
	South West – South East	15.81	0.280
	West – South East	0.20	0.986
	South – North	1.38	0.946
	South – East	10.29	0.592
	South – West	-15.88	0.383
	South – North West	21.49	0.218
	South – South East	-23.50	0.206
	South – North East	28.63	0.136
	South – South West	-38.63	0.058*
	North – East	8.92	0.557
	North – West	-14.50	0.297
	North – North West	-20.11	0.118

Metrics	Slope Aspect	Z-test	P-value
	North – South East	-22.13	0.124
	North – North East	-27.25	0.073
	North – South West	-37.25	0.025*
	East – West	-5.58	0.646
	East – North West	-11.20	0.306
	East – South East	-13.21	0.298
	East – North East	-18.33	0.177
	East – South West	-28.33	0.062
	West – North West	5.61	0.538
	West – South East	7.63	0.494
	West – North East	12.75	0.294
	West – South West	22.75	0.102
	North West – South East	-2.01	0.838
	North West – North East	7.14	0.514
	North West – South West	-17.14	0.183
	South East – North East	5.13	0.686
	South East – South West	-15.13	0.293
	North East – South West	-10.00	0.510

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 7: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests on functional guild metrics for total captures and minimum known alive values associated with the effect of slope aspect on small mammal captures within KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Functional guilds	Slope aspect	Z-test	P-value
Total Captures	Granivores	North – East	20.38	0.154
		South – East	20.38	0.259
		North – South	0.00	1.000
		North – West	-20.40	0.119
		North – North West	-25.88	0.033*
		North – North East	-27.08	0.058*
		North – South West	-33.50	0.032*
		North – South East	-48.72	<0.001*
		South – West	-20.40	0.234
		South – North West	25.88	0.115
		South – North East	27.08	0.134
		South – South West	-33.50	0.080
		South – South East	-48.72	0.005*
		East – West	-0.03	0.998
		East – North West	-5.50	0.593
		East – North East	-6.71	0.599
		East – South West	-13.13	0.358
		East – South East	-28.34	0.018*
		West – North West	5.48	0.523
		West – North East	6.68	0.558
		West – South West	13.10	0.317
		West – South East	28.32	0.007*
		North West – North East	1.21	0.907
		North West – South West	-76.25	0.529
		North West – South East	-22.84	0.014*
		North East – South West	-6.42	0.653
		North East – South East	-21.64	0.070
		South West – South East	15.22	0.261
	Herbivores	East – North	0.00	1.000
		East – South	0.00	1.000
		East – South West	-7.19	0.430

Metrics	Functional guilds	Slope aspect	Z-test	P-value
		East – South East	-8.00	0.294
		East – North West	-9.24	0.160
		East – North East	-9.58	0.240
		East – West	-11.50	0.115
		North – South	0.00	1.000
		North – South West	-7.19	0.472
		North – South East	-8.00	0.355
		North – North West	-9.24	0.232
		North – North East	-9.58	0.293
		North – West	-11.50	0.169
		South – South West	-7.19	0.557
		South – South East	-8.00	0.474
		South – North West	9.24	0.378
		South – North East	9.58	0.406
		South – West	-11.50	0.293
		South West – South East	0.81	0.925
		South West – North West	2.05	0.791
		South West – North East	2.40	0.793
		South West – West	-4.31	0.606
		South East – North West	1.24	0.834
		South East – North East	1.58	0.836
		South East – West	-3.50	0.601
		North West – North East	0.35	0.958
		North West – West	-2.26	0.679
		North East – West	-1.92	0.793
	Insectivores	East – North	-1.71	0.901
		East – North West	-5.63	0.570
		East – West	-15.31	0.164
		East – North East	-18.96	0.124
		East – South West	-24.15	0.079
		East – South East	-26.02	0.024*
		East – South	-28.21	0.105
		North – North West	-3.93	0.737
		North – West	-13.60	0.281
		North – North East	-17.25	0.210
		North – South West	-22.44	0.137
		North – South East	-24.31	0.063
		North – South	-26.50	0.151
		North West – West	-9.68	0.241
		North West – North East	13.33	0.179
		North West – South West	-18.51	0.113
		North West – South East	-20.39	0.022*
		North West – South	-22.58	0.153
		West – North East	3.65	0.740
		West – South West	8.84	0.484
		West – South East	10.71	0.289
		West – South	12.90	0.435
		North East – South West	-5.19	0.706
		North East – South East	-70.62	0.540
		North East – South	-9.25	0.595
		South West – South East	1.88	0.886
	Large Omnivores	South West – South	4.06	0.826
		South East – South	2.19	0.897
		East – South	-0.92	0.963
		East – North East	-15.13	0.280
		East – North West	-29.80	0.008*
		East – South West	-30.17	0.054*
		East – North	-33.35	0.033*
		East – South East	-39.29	0.003*
		East – West	-64.29	<0.001*
		South – North East	14.21	0.473
		South – North West	28.89	0.108

Metrics	Functional guilds	Slope aspect	Z-test	P-value
MKA	Small Omnivores	South – South West	-29.25	0.164
		South – North	32.44	0.123
		South – South East	-38.38	0.045*
		South – West	-63.38	<0.001*
		North East – North West	-14.68	0.194
		North East – South West	-15.04	0.337
		North East – North	18.23	0.244
		North East – South East	-24.17	0.065
		North East – West	-49.17	<0.001*
		North West – South West	-0.36	0.978
		North West – North	3.55	0.789
		North West – South East	-9.49	0.350
		North West – West	-34.49	<0.001*
		South West – North	3.19	0.853
		South West – South East	9.13	0.539
		South West – West	-34.13	0.017*
		North – South East	-5.94	0.689
		North – West	-30.94	0.031*
		South East – West	-25.00	0.030*
		North – East	9.96	0.097
		North East – East	9.96	0.063
		North West – East	9.96	0.021*
		South – East	9.96	0.189
		South East – East	9.96	0.047*
		South West – East	9.96	0.097
		North – North East	0.00	1.000
		North – North West	0.00	1.000
		North – South	0.00	1.000
		North – South East	0.00	1.000
		North – South West	0.00	1.000
		North – West	-12.03	0.029*
		North East – North West	0.00	1.000
		North East – South	0.00	1.000
		North East – South East	0.00	1.000
		North East – South West	0.00	1.000
		North East – West	-12.03	0.012*
		North West – South	0.00	1.000
		North West – South East	0.00	1.000
		North West – South West	0.00	1.000
		North West – West	-12.03	<0.001*
		South – South East	0.00	1.000
		South – South West	0.00	1.000
		South – West	-12.03	0.095
		South East – South West	0.00	1.000
		South East – West	-12.03	0.006*
		South West – West	-12.03	0.029*
		East – West	-2.07	0.667
	Granivores	North – East	19.96	0.162
		South – East	19.96	0.269
		North – South	0.00	1.000
		North – West	-21.30	0.104
		North – North West	-25.74	0.034*
		North – North East	-26.63	0.062
		North – South West	-33.94	0.030*
		North – South East	-48.38	<0.001*
		South – West	-21.30	0.214
		South – North West	25.74	0.117
		South – North East	26.63	0.140
		South – South West	-33.94	0.076
		South – South East	-48.38	0.006*
		East – West	-1.34	0.906
		East – North West	-5.78	0.575

Metrics	Functional guilds	Slope aspect	Z-test	P-value
	Herbivores	East – North East	-6.67	0.602
		East – South West	-13.98	0.327
		East – South East	-28.42	0.017*
		West – North West	4.44	0.604
		West – North East	5.33	0.641
		West – South West	12.64	0.334
		West – South East	27.08	0.010*
		North West – North East	0.89	0.931
		North West – South West	-8.20	0.498
		North West – South East	-22.64	0.014*
		North East – South West	-7.31	0.608
		North East – South East	-21.75	0.069
		South West – South East	14.44	0.286
		East – North	0.00	1.000
		East – South	0.00	1.000
		East – South East	-4.00	0.587
		East – South West	-7.25	0.410
		East – North West	-9.31	0.144
		East – North East	-9.67	0.219
		East – West	-11.60	0.099
		North – South	0.00	1.000
		North – South East	-4.00	0.632
		North – South West	-7.25	0.452
		North – North West	-9.31	0.213
		North – North East	-9.67	0.272
		North – West	-11.60	0.150
		South – South East	-4.00	0.710
		South – South West	-7.25	0.539
		South – North West	9.31	0.358
		South – North East	9.67	0.385
		South – West	-11.60	0.272
		South East – South West	-3.25	0.697
		South East – North West	5.31	0.354
		South East – North East	5.67	0.441
		South East – West	-7.60	0.240
		South West – North West	2.06	0.783
		South West – North East	2.42	0.784
		South West – West	-4.35	0.590
	Insectivores	North West – North East	0.36	0.955
		North West – West	-2.29	0.665
		North East – West	-1.93	0.784
		East – North	-1.63	0.906
		East – North West	-4.63	0.641
		East – West	-15.58	0.157
		East – North East	-18.96	0.123
		East – South West	-24.13	0.079
		East – South East	-25.47	0.027*
		East – South	-29.38	0.091
		North – North West	-3.00	0.797
		North – West	-13.95	0.268
		North – North East	-17.33	0.207
		North – South West	-22.50	0.135
		North – South East	-23.84	0.068
		North – South	-27.75	0.132
		North West – West	-10.95	0.184
		North West – North East	14.33	0.148
		North West – South West	-19.50	0.095
		North West – South East	-20.84	0.019*
		North West – South	-24.75	0.117
		West – North East	3.38	0.758
		West – South West	8.55	0.497

Metrics	Functional guilds	Slope aspect	Z-test	P-value
		West – South East	9.89	0.327
		West – South	13.80	0.403
		North East – South West	-5.17	0.707
		North East – South East	-6.51	0.571
		North East – South	-10.42	0.549
		South West – South East	1.34	0.918
		South West – South	5.25	0.776
		South East – South	3.91	0.817
	Large Omnivores	East – South	-2.33	0.906
		East – North East	-14.00	0.316
		East – North West	-29.81	0.008*
		East – South West	-30.40	0.052*
		East – North	-33.77	0.031*
		East – South East	-35.33	0.007*
		East – West	-59.58	<0.001*
		South – North East	11.67	0.555
		South – North West	27.48	0.126
		South – South West	-28.06	0.180
		South – North	31.44	0.133
		South – South East	-33.00	0.084
		South – West	-57.25	0.002*
		North East – North West	-15.81	0.160
		North East – South West	-16.40	0.294
		North East – North	19.77	0.205
		North East – South East	-21.33	0.102
		North East – West	-46.58	<0.001*
		North West – South West	-0.59	0.965
		North West – North	3.96	0.765
		North West – South East	-5.53	0.585
		North West – West	-29.78	0.001*
		South West – North	3.38	0.844
		South West – South East	4.94	0.739
		South West – West	-29.19	0.041*
		North – South East	-1.56	0.916
		North – West	-25.81	0.071
	Small Omnivores	South East – West	-24.25	0.035*
		North – East	9.96	0.097
		North East – East	9.96	0.063
		North West – East	9.96	0.021*
		South – East	9.96	0.189
		South East – East	9.96	0.047*
		South West – East	9.96	0.097
		North – North East	0.00	1.000
		North – North West	0.00	1.000
		North – South	0.00	1.000
		North – South East	0.00	1.000
		North – South West	0.00	1.000
		North – West	-12.03	0.029*
		North East – North West	0.00	1.000
		North East – South	0.00	1.000
		North East – South East	0.00	1.000
		North East – South West	0.00	1.000
		North East – West	-12.03	0.012*
		North West – South	0.00	1.000
		North West – South East	0.00	1.000
		North West – South West	0.00	1.000
		North West – West	-12.03	<0.001*
		South – South East	0.00	1.000
		South – South West	0.00	1.000
		South – West	-12.03	0.095
		South East – South West	0.00	1.000
		South East – West	-12.03	0.006*

Appendix 7 continued

Metrics	Functional guilds	Slope aspect	Z-test	P-value
		South West – West	-12.03	0.029*
		East – West	-2.07	0.667

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 8: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests on metrics associated with small mammals captures relative to seasons across KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Seasons	Z-test	P-value
Total captures	Winter – Spring	2.42	0.787
	Winter – Summer	13.48	0.132
	Winter – Autumn	23.30	0.009*
	Spring – Summer	-11.07	0.217
	Spring – Autumn	20.88	0.020*
	Summer – Autumn	9.82	0.273
Trap success	Spring – Winter	-6.03	0.501
	Spring – Summer	-8.75	0.330
	Spring – Autumn	11.82	0.188
	Winter – Summer	2.72	0.762
	Winter – Autumn	5.78	0.519
	Summer – Autumn	3.07	0.733
MKA	Winter – Spring	6.28	0.483
	Winter – Summer	14.27	0.111
	Winter – Autumn	23.92	0.008*
	Spring – Summer	-7.98	0.372
	Spring – Autumn	17.63	0.049*
	Summer – Autumn	9.65	0.281
Species richness	Winter – Spring	3.12	0.721
	Winter – Summer	3.12	0.721
	Winter – Autumn	18.03	0.039*
	Spring – Autumn	14.92	0.088
	Summer – Autumn	14.92	0.088
	Spring – Summer	0.00	1.000
Species diversity	Summer – Spring	7.70	0.370
	Summer – Winter	-11.67	0.174
	Summer – Autumn	15.70	0.067
	Spring – Winter	-3.97	0.644
	Spring – Autumn	8.00	0.351
	Winter – Autumn	4.03	0.638

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 9: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests on functional guild metrics for total captures and minimum known alive values associated with the effect of seasons on small mammal captures within KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Functional guilds	Seasons	Z-test	P-value
Total captures	Granivores	Summer – Spring	0.42	0.959
		Summer – Winter	-0.63	0.937
		Summer – Autumn	7.22	0.372
		Spring – Winter	-0.22	0.979
		Spring – Autumn	6.80	0.400

Metrics	Functional guilds	Seasons	Z-test	P-value
MKA	Herbivores	Winter – Autumn	6.58	0.415
		Winter – Spring	1.82	0.725
		Winter – Summer	3.73	0.469
		Winter – Autumn	7.85	0.128
		Spring – Summer	-1.92	0.710
		Spring – Autumn	6.03	0.242
	Insectivores	Summer – Autumn	4.12	0.425
		Spring – Summer	-0.25	0.974
		Spring – Winter	-3.90	0.616
		Spring – Autumn	17.65	0.023*
		Summer – Winter	-3.65	0.639
		Winter – Autumn	13.75	0.077
	Large Omnivores	Winter – Spring	9.07	0.306
		Winter – Summer	13.28	0.134
		Winter – Autumn	19.12	0.031*
		Spring – Summer	-4.22	0.634
		Spring – Autumn	10.05	0.256
		Summer – Autumn	5.83	0.510
	Small Omnivores	Summer – Winter	-2.03	0.549
		Summer – Autumn	3.90	0.250
		Summer – Spring	6.07	0.074
		Winter – Autumn	1.87	0.582
		Winter – Spring	4.03	0.234
		Autumn – Spring	-2.17	0.523
	Granivores	Spring – Summer	-0.30	0.970
		Spring – Winter	-2.32	0.774
		Spring – Autumn	9.12	0.259
		Summer – Winter	-2.02	0.803
		Summer – Autumn	8.82	0.275
		Winter – Autumn	6.80	0.400
	Herbivores	Winter – Spring	1.68	0.735
		Winter – Autumn	3.88	0.439
		Winter – Summer	6.00	0.228
		Spring – Autumn	2.20	0.661
		Spring – Summer	-4.32	0.386
		Autumn – Summer	-2.12	0.673
	Insectivores	Summer – Spring	0.10	0.990
		Summer – Winter	-3.13	0.687
		Summer – Autumn	16.57	0.033*
		Spring – Winter	-3.03	0.697
		Spring – Autumn	16.47	0.034*
		Winter – Autumn	13.43	0.084
	Large Omnivores	Winter – Spring	13.50	0.126
		Winter – Summer	18.65	0.035*
		Winter – Autumn	23.38	0.008*
		Spring – Summer	-5.15	0.560
		Spring – Autumn	9.88	0.263
		Summer – Autumn	4.73	0.592
	Small Omnivores	Summer – Winter	-2.03	0.549
		Summer – Autumn	3.90	0.250
		Summer – Spring	6.07	0.074
		Winter – Autumn	1.87	0.582
		Winter – Spring	4.03	0.234
		Autumn – Spring	-2.17	0.523

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 10: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests on metrics associated with small mammals captures relative to grassland conditions across KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Grassland Condition	Z-Test	P-value
Total captures	Secondary Good – Primary Intermediate	12.57	0.498
	Secondary Good – Primary Good	14.94	0.057*
	Secondary Good – Secondary Intermediate	-29.68	0.002*
	Primary Intermediate – Primary Good	2.37	0.895
	Primary Intermediate – Secondary Intermediate	-17.10	0.361
	Primary Good – Secondary Intermediate	-14.74	0.076
Trap success	Primary Intermediate – Secondary Good	-5.11	0.783
	Primary Intermediate – Primary Good	11.73	0.512*
	Primary Intermediate – Secondary Intermediate	-17.75	0.344
	Secondary Good – Primary Good	6.63	0.400
	Secondary Good – Secondary Intermediate	-12.64	0.191
	Primary Good – Secondary Intermediate	-6.02	0.470
MKA	Secondary Good – Primary Intermediate	6.30	0.734
	Secondary Good – Primary Good	13.74	0.080
	Secondary Good – Secondary Intermediate	-28.82	0.003*
	Primary Intermediate – Primary Good	7.44	0.677
	Primary Intermediate – Secondary Intermediate	-22.52	0.229
	Primary Good – Secondary Intermediate	-15.08	0.069
Species richness	Secondary Good – Primary Good	13.32	0.082
	Secondary Good – Primary Intermediate	20.77	0.251
	Secondary Good – Secondary Intermediate	-34.87	<0.001*
	Primary Good – Primary Intermediate	-7.45	0.669
	Primary Good – Secondary Intermediate	-21.56	0.008*
	Primary Intermediate – Secondary Intermediate	-14.10	0.440
Species diversity	Primary Good – Secondary Good	-2.23	0.767
	Primary Good – Secondary Intermediate	-11.37	0.153
	Primary Good – Primary Intermediate	-29.14	0.089
	Secondary Good – Secondary Intermediate	-9.14	0.323
	Secondary Good – Primary Intermediate	26.91	0.130
	Secondary Intermediate – Primary Intermediate	17.77	0.322

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 11: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests on functional guild metrics for total captures and minimum known alive values associated with the effect of grassland conditions on small mammal captures within KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Functional guilds	Grassland condition	Z-test	P-value
Total captures	Granivores	Primary Good – Secondary Good	-5.44	0.443
		Primary Good – Secondary Intermediate	-24.19	0.001*
		Primary Good – Primary Intermediate	-43.17	0.007*
		Secondary Good – Secondary Intermediate	-18.75	0.031*
		Secondary Good – Primary Intermediate	37.73	0.024*
		Secondary Intermediate – Primary Intermediate	18.98	0.261
	Herbivores	Primary Intermediate – Secondary Intermediate	-2.40	0.824
		Primary Intermediate – Secondary Good	-4.75	0.656
		Primary Intermediate – Primary Good	11.09	0.281
		Secondary Intermediate – Secondary Good	2.35	0.672
		Secondary Intermediate – Primary Good	8.69	0.069

Metrics	Functional guilds	Grassland condition	Z-test	P-value
MKA	Insectivores	Secondary Good – Primary Good	6.34	0.161
		Primary Intermediate – Secondary Good	-14.23	0.377
		Primary Intermediate – Primary Good	23.84	0.125
		Primary Intermediate – Secondary Intermediate	-32.31	0.047*
		Secondary Good – Primary Good	9.61	0.159
		Secondary Good – Secondary Intermediate	-18.08	0.031*
	Large Omnivores	Primary Good – Secondary Intermediate	-8.47	0.241
		Secondary Good – Primary Intermediate	2.57	0.888
		Secondary Good – Primary Good	24.01	0.002*
		Secondary Good – Secondary Intermediate	-27.78	0.004*
		Primary Intermediate – Primary Good	21.44	0.225
		Primary Intermediate – Secondary Intermediate	-25.21	0.174
	Small Omnivores	Primary Good – Secondary Intermediate	-3.77	0.646
		Primary Good – Primary Intermediate	0.00	1.000
		Primary Good – Secondary Good	-4.27	0.152
		Primary Good – Secondary Intermediate	-10.02	0.001*
		Primary Intermediate – Secondary Good	-4.27	0.543
		Primary Intermediate – Secondary Intermediate	-10.02	0.158
	Granivores	Secondary Good – Secondary Intermediate	-5.75	0.115
		Primary Good – Secondary Good	-6.20	0.381
		Primary Good – Secondary Intermediate	-24.00	0.001*
		Primary Good – Primary Intermediate	-42.02	0.009*
		Secondary Good – Secondary Intermediate	-17.80	0.041*
		Secondary Good – Primary Intermediate	35.82	0.032*
	Herbivores	Secondary Intermediate – Primary Intermediate	18.02	0.286
		Primary Intermediate – Secondary Intermediate	-2.42	0.816
		Primary Intermediate – Secondary Good	-4.85	0.639
		Primary Intermediate – Primary Good	10.06	0.311
		Secondary Intermediate – Secondary Good	2.44	0.653
		Secondary Intermediate – Primary Good	7.65	0.098
	Insectivores	Secondary Good – Primary Good	5.21	0.239
		Primary Intermediate – Secondary Good	-14.36	0.373
		Primary Intermediate – Primary Good	23.74	0.126
		Primary Intermediate – Secondary Intermediate	-32.44	0.046*
		Secondary Good – Primary Good	9.39	0.169
		Secondary Good – Secondary Intermediate	-18.08	0.031*
	Large Omnivores	Primary Good – Secondary Intermediate	-8.70	0.228
		Secondary Good – Primary Intermediate	1.07	0.953
		Secondary Good – Primary Good	19.07	0.014*
		Secondary Good – Secondary Intermediate	-24.32	0.011*
		Primary Intermediate – Primary Good	18.00	0.307
		Primary Intermediate – Secondary Intermediate	-23.25	0.208
	Small Omnivores	Primary Good – Secondary Intermediate	-5.25	0.521
		Primary Good – Primary Intermediate	0.00	1.000
		Primary Good – Secondary Good	-4.27	0.152
		Primary Good – Secondary Intermediate	-10.02	0.001*
		Primary Intermediate – Secondary Good	-4.27	0.543
		Primary Intermediate – Secondary Intermediate	-10.02	0.158
		Secondary Good – Secondary Intermediate	-5.75	0.115

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 12: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests on metrics associated with small mammals captures relative to regional variation across KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Regions	Z-Test	P-value
Total captures	Kloof – Hillcrest	18.25	0.019*
	Kloof – Camperdown	23.60	0.002*
	Hillcrest – Camperdown	5.35	0.490
Trap success	Kloof – Hillcrest	15.40	0.048*
	Kloof – Camperdown	25.06	0.001*
	Hillcrest – Camperdown	9.66	0.214
MKA	Kloof – Hillcrest	17.05	0.028*
	Kloof – Camperdown	23.26	0.003*
	Hillcrest – Camperdown	6.21	0.423
Species richness	Kloof – Hillcrest	17.94	0.018*
	Kloof – Camperdown	34.49	<0.001*
	Hillcrest – Camperdown	16.55	0.029*
Species diversity	Kloof – Hillcrest	21.50	0.004*
	Kloof – Camperdown	36.89	<0.001*
	Hillcrest – Camperdown	15.39	0.038*

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 13: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests on functional guild metrics for total captures and minimum known alive values associated with the effect of regional variation on small mammal captures within KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Functional guilds	Regions	Z-test	P-value
Total captures	Granivores	Kloof – Hillcrest	15.81	0.024*
		Kloof – Camperdown	60.69	<0.001*
		Hillcrest – Camperdown	44.88	<0.001*
	Herbivores	Camperdown – Kloof	-1.11	0.803
		Camperdown – Hillcrest	-7.48	0.094
		Kloof – Hillcrest	6.36	0.154
	Insectivores	Kloof – Hillcrest	6.75	0.317
		Kloof – Camperdown	22.61	<0.001*
		Hillcrest – Camperdown	15.86	0.019*
	Large Omnivores	Camperdown – Kloof	-12.25	0.110
		Camperdown – Hillcrest	-24.43	0.001*
		Kloof – Hillcrest	12.18	0.112
MKA	Small Omnivores	Camperdown – Kloof	-2.99	0.309
		Camperdown – Hillcrest	-6.01	0.041*
		Kloof – Hillcrest	3.03	0.303
	Granivores	Kloof – Hillcrest	16.70	0.017*
		Kloof – Camperdown	59.80	<0.001*
		Hillcrest – Camperdown	43.10	<0.001*
	Herbivores	Camperdown – Kloof	-2.75	0.523
		Camperdown – Hillcrest	-9.23	0.033*
		Kloof – Hillcrest	6.48	0.135
	Insectivores	Kloof – Hillcrest	6.00	0.373
		Kloof – Camperdown	22.24	<0.001*
		Hillcrest – Camperdown	16.24	0.016*
	Large Omnivores	Camperdown – Kloof	-13.04	0.088
		Camperdown – Hillcrest	-23.11	0.003*
		Kloof – Hillcrest	10.08	0.188
	Small Omnivores	Camperdown – Kloof	-2.99	0.309
		Camperdown – Hillcrest	-6.01	0.041*
		Kloof – Hillcrest	3.03	0.303

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 14: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests on metrics associated with small mammals captures relative to habitat management regimes (treatments) across KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Treatments	Z-test	P-value
Total captures	Outside – Inside	23.93	0.002*
	Outside – Control	26.44	<0.001*
	Inside – Control	2.51	0.746
Trap success	Inside – Outside	-11.45	0.141
	Inside – Control	23.09	0.003*
	Outside – Control	11.64	0.134
MKA	Outside – Inside	24.18	0.002*
	Outside – Control	26.83	<0.001*
	Inside – Control	2.65	0.732
Species richness	Outside – Inside	24.79	0.001*
	Outside – Control	37.91	<0.001*
	Inside – Control	13.13	0.083
Species diversity	Outside – Inside	1.83	0.806
	Outside – Control	27.05	<0.001*
	Inside – Control	25.23	<0.001*

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 15: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests on functional guild metrics for total captures and minimum known alive values associated with the effect of habitat management regimes (treatments) on small mammal captures within KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Functional guilds	Treatments	Z-test	P-value
Total captures	Granivores	Inside – Outside	-0.09	0.990
		Inside – Control	52.83	<0.001*
		Outside – Control	52.74	<0.001*
	Herbivores	Control – Outside	-2.55	0.568
		Control – Inside	-6.04	0.176
		Outside – Inside	3.49	0.435
	Insectivores	Outside – Inside	11.90	0.078
		Outside – Control	25.19	<0.001*
		Inside – Control	13.29	0.049*
	Large Omnivores	Control – Outside	-1.24	0.872
		Control – Inside	-35.44	<0.001*
		Outside – Inside	34.20	<0.001*
	Small Omnivores	Control – Outside	-2.99	0.309
		Control – Inside	-6.01	0.041*
		Outside – Inside	3.03	0.303
MKA	Granivores	Outside – Inside	0.23	0.974
		Outside – Control	51.56	<0.001*
		Inside – Control	51.34	<0.001*
	Herbivores	Control – Outside	-4.20	0.330
		Control – Inside	-7.75	0.074
		Outside – Inside	3.55	0.414
	Insectivores	Outside – Inside	11.70	0.082
		Outside – Control	25.09	<0.001*
		Inside – Control	13.39	0.047*
	Large Omnivores	Control – Outside	-4.53	0.554
		Control – Inside	-31.63	<0.001*
		Outside – Inside	27.10	<0.001*
	Small Omnivores	Control – Outside	-2.99	0.309
		Control – Inside	-6.01	0.041*
		Outside – Inside	3.03	0.303

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 16: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests of species-specific tests in relation to the effect of elevational gradients on small mammal populations in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Species	Elevation	Z-test	P-value
Population estimates	<i>Aethomys ineptus</i>	Mid-High	-4.91	0.275
		Mid-Low	9.14	0.055
		High-Low	4.23	0.234
	<i>Crocidura flavescens</i>	Low-High	6.98	0.077
		Low-Mid	-11.64	0.030*
		High-Mid	-4.66	0.359
	<i>Mastomys natalensis</i>	High-Low	5.44	0.404
		High-Mid	25.43	0.002*
		Low-Mid	-19.99	0.022*
	<i>Rhabbomys dilectus</i>	Low-Mid	-9.25	0.251
		Low-High	-37.92	<0.001*
		Mid-High	-28.67	<0.001*
Trap success	<i>Aethomys ineptus</i>	Mid – High	5.02	0.344
		Mid – Low	8.08	0.151
		High – Low	-3.06	0.466
	<i>Crocidura flavescens</i>	Mid – Low	1.75	0.816
		Mid – High	7.13	0.313
		Low – High	5.38	0.336
	<i>Mastomys natalensis</i>	High – Low	-3.61	0.603
		High – Mid	-20.28	0.021*
		Low – Mid	-16.68	0.073
	<i>Rhabbomys dilectus</i>	Low-Mid	-11.28	0.188
		Low-High	47.24	<0.001*
		Mid-High	35.97	<0.001*

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 17: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests of species-specific tests in relation to the effect of slope aspect on small mammal populations in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Species	Aspect	Z-test	P-value
Population estimates	<i>Aethomys ineptus</i>	North – East	4.88	0.375
		North East – East	4.88	0.321
		South – East	4.88	0.483
		South West – East	4.88	0.375
		West – East	4.88	0.268
		North – North East	0.00	1.000
		North – South	0.00	1.000
		North – South West	0.00	1.000
		North – West	0.00	1.000
		North – North West	-1.55	0.740
		North – South East	-11.22	0.031*
		North East – South	0.00	1.000
		North East – South West	0.00	1.000
		North East – West	0.00	1.000
		North East – North West	-10.55	0.696
		North East – South East	-11.22	0.015*
		South – North West	1.55	0.806
		South West – North West	1.55	0.740
		West – North West	1.55	0.638
		South – South West	0.00	1.000
		South – West	0.00	1.000
		South – South East	-11.22	0.096
		South West – South East	11.22	0.031*

Appendix 17 continued...		West – South East	11.22	0.005	
		South West – West	0.00	1.000	
Metrics	Species	Aspect	Z-test	P-value	
	<i>Crocidura flavescens</i>	North West – East	3.33	0.402	
		North West – South East	-9.67	0.007*	
		East – South East	-6.34	0.168	
		North – East	4.67	0.596	
		South – East	4.67	0.675	
		South East – East	4.67	0.527	
		North – South	0.00	1.000	
		North – South East	0.00	1.000	
		North – South West	-7.50	0.437	
		North – North East	-9.33	0.289	
		North – West	-9.45	0.242	
		North – North West	-10.67	0.154	
		South – North East	9.33	0.402	
		South East – North East	9.33	0.205	
		South – North West	10.67	0.292	
		South East – North West	10.67	0.063	
		South – South East	0.00	1.000	
		South – South West	-7.50	0.526	
		South – West	-9.45	0.371	
		South East – South West	-7.50	0.369	
		South East – West	-9.45	0.144	
		East – South West	-2.83	0.748	
		East – North East	-4.67	0.554	
		East – West	-4.78	0.497	
		East – North West	-6.00	0.346	
		South West – North East	1.83	0.835	
		South West – West	-1.95	0.809	
		South West – North West	3.17	0.672	
		North East – West	-0.12	0.987	
		North East – North West	-1.33	0.834	
		West – North West	1.22	0.819	
	<i>Mastomys natalensis</i>	South East – North East	3.52	0.750	
		South East – North	4.72	0.681	
		South East – North West	7.51	0.353	
		South East – East	7.51	0.471	
		South East – South West	-11.18	0.352	
		South East – West	-16.5	0.080	
		South East – South	23.8	0.151	
		North East – North	1.19	0.925	
		North East – North West	-3.99	0.684	
		North East – East	3.99	0.735	
		North East – South West	-7.66	0.562	
		North East – West	-12.98	0.235	
		North East – South	-20.28	0.246	
		North – North West	-2.79	0.786	
		North – East	2.80	0.819	
		North-South West	-6.46	0.634	
		North – West	-11.78	0.299	
		North – South	-19.08	0.282	
		North West – East	0.00	1.000	
		North West – South West	-3.67	0.735	
		North West – West	-8.99	0.256	
		North West – South	-16.29	0.302	
		East – South West	-3.67	0.772	
		East – West	-8.99	0.382	
		East – South	-16.29	0.340	
		South West – West	-5.32	0.655	
		South West – South	12.62	0.486	
		West – South	7.30	0.658	
		<i>Myosorex varius</i>	North – East	14.88	0.093
			South – East	14.88	0.184
			West – East	14.88	0.036*

Appendix 17 continued...		North – South	0.00	1.000
		North – West	0.00	1.000
Metrics	Species	Aspect	Z-test	P-value
Trap success	<i>Rhabdomys dilectus</i>	North – South East	-3.41	0.685
		North – North West	-6.29	0.402
		North – North East	-14.96	0.091
		North – South West	-22.00	0.023
		South – North East	14.96	0.181
		West – North East	14.96	0.035*
		South – North West	6.29	0.536
		West – North West	6.29	0.236
		South – West	0.00	1.000
		South – South East	-3.41	0.753
		South – South West	-22.00	0.064
		West – South East	3.41	0.600
		West – South West	22.00	0.007*
		South East – North West	2.88	0.615
		South East – East	11.47	0.121
		South East – North East	11.55	0.119
		South East – South West	-18.59	0.027*
		North West – East	8.59	0.178
		North West – North East	8.67	0.174
		North West – South West	-15.71	0.036*
		East – North East	-0.08	0.992
		East – South West	-7.13	0.421
		North East – South West	-7.04	0.426
		North – East	20.58	0.115
		South – East	20.58	0.213
		North – South	0.00	1.000
		North – West	-4.95	0.679
		North – North West	-24.18	0.029
		North – South West	-24.75	0.084
		North – North East	-25.18	0.058
		North – South East	-33.56	0.007*
		South – West	-4.95	0.752
		South – North West	24.18	0.107
		South – South West	-24.75	0.158
		South – North East	25.18	0.132
		South – South East	-33.56	0.036*
		West – East	15.63	0.135
		West – North West	19.23	0.015*
		West – South West	19.8	0.098
		West – North East	20.23	0.060
		West – South East	28.61	0.003*
		East – North West	-3.60	0.703
		East – South West	-4.17	0.750
		East – North East	-4.60	0.700
		East – South East	-12.98	0.235
		North West – South West	-0.57	0.959
		North West – North East	1.00	0.918
		North West – South East	-9.38	0.269
		South West – North East	0.43	0.974
		South West – South East	8.81	0.477
		North East – South East	-8.38	0.455
	<i>Aethomys ineptus</i>	South – East	4.67	0.694
		South West – East	4.67	0.619
		South – North	24.69	0.050*
		South West – North	24.69	0.016*
		South – North East	9.79	0.409
		South West – North East	9.79	0.296
		South – North West	4.34	0.687
		South West – North West	4.34	0.586
		South – South West	0.00	1.000
		South – West	-6.00	0.594
		South – South East	-18.47	0.108

	South West – West	-6.00	0.485
Appendix 17 continued...	South West – South East	18.47	0.038*

Metrics	Species	Aspect	Z-test	P-value
		North West – East	0.33	0.961
		North West – West	-1.66	0.768
		North West – North East	5.45	0.420
		North West – South East	-14.13	0.020*
		North West – North	20.35	0.011
		East – West	-1.33	0.859
		East – North East	-5.13	0.541
		East – South East	-13.80	0.079
		East – North	-20.02	0.033*
		West – North East	3.79	0.613
		West – South East	12.47	0.070
		West – North	18.69	0.030*
		North East – South East	-8.68	0.269
		North East – North	14.90	0.112
		South East – North	6.22	0.484
	<i>Crocidura flavescens</i>	South West – North West	3.60	0.734
		South West – East	3.94	0.753
		South West – North	7.06	0.606
		South West – North East	7.06	0.572
		South West – South East	18.63	0.116
		South West – West	-21.04	0.066
		South West – South	35.31	0.035*
		North West – East	0.34	0.970
		North West – North	3.46	0.744
		North West – North East	3.46	0.701
		North West – South East	-15.03	0.064
		North West – West	-17.44	0.020
		North West – South	-31.71	0.027*
		East – North	-3.13	0.803
		East – North East	-3.13	0.780
		East – South East	-14.69	0.160
		East – West	-17.10	0.087
		East – South	-31.38	0.047
		North – North East	0.00	1.000
		North – South East	-11.56	0.330
		North – West	-13.98	0.223
		North – South	-28.25	0.092
		North East – South East	-11.56	0.269
		North East – West	-13.98	0.162
		North East – South	-28.25	0.074
		South East – West	-2.41	0.793
		South East – South	16.69	0.276
		West – South	14.28	0.342
	<i>Mastomys natalensis</i>	East – South	-5.17	0.792
		East – North East	-13.71	0.323
		East – North	-13.73	0.376
		East – South West	-27.85	0.072
		East – North West	-31.27	0.005*
		East – South East	-34.57	0.008*
		East – West	-58.92	<0.001*
		South – North East	8.54	0.663
		South – North	8.56	0.681
		South – South West	-22.69	0.275
		South – North West	26.10	0.143
		South – South East	-29.41	0.121
		South – West	-53.75	0.004*
		North East – North	0.02	0.999
		North East – South West	-14.15	0.362
		North East – North West	-17.56	0.116
		North East – South East	-20.87	0.108
		North East – West	-45.21	<0.001
		North – South West	-14.13	0.406

		North – North West	-17.54	0.183
		North – South East	-20.84	0.156
Appendix 17 continued...				
Metrics	Species	Aspect	Z-test	P-value
<i>Myosorex varius</i>		North – West	-45.19	0.001
		South West – North West	3.41	0.795
		South West – South East	6.72	0.648
		South West – West	-31.06	0.029*
		North West – South East	-3.31	0.742
		North West – West	-27.65	0.003*
		South East – West	-24.34	0.033*
		East – North	0.00	1.000
		East – North East	0.00	1.000
		East – North West	0.00	1.000
		East – South	0.00	1.000
		East – South East	0.00	1.000
		East – South West	0.00	1.000
		East – West	0.00	1.000
		North – North East	0.00	1.000
		North – North West	0.00	1.000
		North – South	0.00	1.000
		North – South East	0.00	1.000
		North – South West	0.00	1.000
		North – West	0.00	1.000
		North East – North West	0.00	1.000
		North East – South	0.00	1.000
		North East – South East	0.00	1.000
		North East – South West	0.00	1.000
		North East – West	0.00	1.000
		North West – South	0.00	1.000
		North West – South East	0.00	1.000
		North West – South West	0.00	1.000
		North West – West	0.00	1.000
		South – South East	0.00	1.000
		South – South West	0.00	1.000
		South – West	0.00	1.000
		South East – South West	0.00	1.000
		South East – West	0.00	1.000
		South West – West	0.00	1.000
<i>Rhabdomys dilectus</i>		North – East	20.67	0.148
		South – East	20.67	0.252
		North – South	0.00	1.000
		North – West	-21.38	0.102
		North – North West	-25.29	0.037*
		North – North East	-26.88	0.060
		North – South West	-33.94	0.030*
		North – South East	-48.69	<0.001*
		South – West	-21.38	0.212
		South – North West	25.29	0.123
		South – North East	26.88	0.137
		South – South West	-33.94	0.076
		South – South East	-48.69	0.005*
		East – West	-0.71	0.951
		East – North West	-4.62	0.653
		East – North East	-6.21	0.627
		East – South West	-13.27	0.352
		East – South East	-28.02	0.019*
		West – North West	3.91	0.648
		West – North East	5.50	0.630
		West – South West	12.56	0.337
		West – South East	27.31	0.009*
		North West – North East	1.59	0.877
		North West – South West	-8.65	0.475
		North West – South East	-23.40	0.011*
		North East – South West	-7.06	0.621
		North East – South East	-21.81	0.068
		South West – South East	14.75	0.276

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 18: The Dunn's post-hoc test outcomes from Kruskal-Wallis tests of species-specific tests in relation to the seasons of seasons on small mammal populations in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Species	Seasons	Z-test	P-value
Population estimates	<i>Aethomys ineptus</i>	Winter – Summer	0.08	0.979
		Winter – Autumn	0.12	0.970
		Winter – Spring	2.00	0.520
		Summer – Autumn	0.03	0.991
		Summer – Spring	1.92	0.538
		Autumn – Spring	-1.88	0.545
	<i>Crocidura flavescens</i>	Summer – Autumn	0.17	0.973
		Summer – Winter	-2.00	0.691
		Summer – Spring	2.13	0.668
		Autumn – Winter	-1.83	0.716
		Autumn – Spring	-1.97	0.693
		Winter – Spring	0.14	0.978
	<i>Mastomys natalensis</i>	Spring – Autumn	7.58	0.310
		Spring – Winter	-8.66	0.254
		Spring – Summer	-15.19	0.042
		Autumn – Winter	-1.08	0.880
		Autumn – Summer	-7.61	0.278
		Winter – Summer	6.53	0.361
	<i>Myosorex varius</i>	Autumn – Spring	-1.82	0.717
		Autumn – Summer	-2.02	0.687
		Autumn – Winter	-7.97	0.112
		Spring – Summer	-0.20	0.968
		Spring – Winter	-6.15	0.219
		Summer – Winter	-5.95	0.235
	<i>Rhabdomys dilectus</i>	Autumn – Summer	-2.97	0.688
		Autumn – Spring	-4.64	0.534
		Autumn – Winter	-4.84	0.516
		Summer – Spring	1.67	0.823
		Summer – Winter	-1.88	0.801
		Spring – Winter	-0.21	0.978
Trap success	<i>Aethomys ineptus</i>	Autumn – Summer	-5.87	0.269
		Autumn – Spring	-7.88	0.137
		Autumn – Winter	-9.45	0.075
		Summer – Spring	2.02	0.704
		Summer – Winter	-3.58	0.499
		Spring – Winter	-1.57	0.768
	<i>Crocidura flavescens</i>	Summer – Spring	5.53	0.434
		Summer – Winter	-11.30	0.110
		Summer – Autumn	18.50	0.009*
		Spring – Winter	-5.77	0.415
		Spring – Autumn	12.97	0.057
		Winter – Autumn	7.20	0.309
	<i>Mastomys natalensis</i>	Winter – Spring	13.25	0.131
		Winter – Summer	18.55	0.034*
		Winter – Autumn	25.13	0.004*
		Spring – Summer	-5.30	0.546
		Spring – Autumn	11.88	0.175
		Summer – Autumn	6.58	0.453
	<i>Myosorex varius</i>	Autumn – Spring	0.00	1.000
		Autumn – Summer	0.00	1.000
		Autumn – Winter	0.00	1.000
		Spring – Summer	0.00	1.000
		Spring – Winter	0.00	1.000
		Summer – Winter	0.00	1.000
	<i>Rhabdomys dilectus</i>	Spring – Summer	-0.73	0.928
		Spring – Winter	-1.95	0.809
		Spring – Autumn	8.72	0.280
		Summer – Winter	-1.22	0.880
		Summer – Autumn	7.98	0.323
		Winter – Autumn	6.77	0.402

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 19: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests of species-specific tests in relation to the effect of grassland conditions on small mammal populations in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Species	Grassland Condition	Z-test	P-value
Population estimates	Aethomys ineptus	Secondary Intermediate – Secondary Good	2.21	0.509
		Secondary Intermediate – Primary Good	2.81	0.331
		Secondary Intermediate – Primary Intermediate	14.63	0.025
		Secondary Good – Primary Good	0.59	0.829
	Crociodura flavescens	Secondary Good – Primary Intermediate	12.41	0.054
		Primary Good – Primary Intermediate	-11.82	0.057
		Primary Good – Secondary Good	-4.63	0.291
		Primary Good – Secondary Intermediate	-8.26	0.074
	Mastomys natalensis	Primary Good – Primary Intermediate	-10.10	0.310
		Secondary Good – Secondary Intermediate	-3.63	0.499
		Secondary Good – Primary Intermediate	5.46	0.596
		Secondary Intermediate – Primary Intermediate	1.83	0.860
	Myosorex varius	Secondary Good – Primary Intermediate	1.52	0.925
		Secondary Good – Primary Good	1.68	0.793
		Secondary Good – Secondary Intermediate	-10.26	0.190
		Primary Intermediate – Primary Good	0.16	0.992
	Rhabdomys dilectus	Primary Intermediate – Secondary Intermediate	-8.74	0.589
		Primary Good – Secondary Intermediate	-8.58	0.201
		Primary Good – Secondary Intermediate	-1.76	0.704
		Primary Good – Secondary Good	-6.97	0.113
Trap success	Aethomys ineptus	Primary Good – Primary Intermediate	-40.97	<0.001*
		Secondary Intermediate – Secondary Good	5.21	0.334
		Secondary Intermediate – Primary Intermediate	39.21	<0.001*
		Secondary Good – Primary Intermediate	34.00	0.001*
	Crociodura flavescens	Primary Good – Secondary Good	-6.96	0.296
		Primary Good – Secondary Intermediate	-15.55	0.023*
		Primary Good – Primary Intermediate	-48.59	<0.001*
		Secondary Good – Secondary Intermediate	-8.59	0.289
	Mastomys natalensis	Secondary Good – Primary Intermediate	41.64	0.007*
		Secondary Intermediate – Primary Intermediate	33.04	0.032*
		Secondary Good – Secondary Intermediate	-4.90	0.392
		Secondary Good – Primary Good	12.29	0.008*
	Myosorex varius	Secondary Good – Primary Intermediate	14.00	0.202
		Secondary Intermediate – Primary Good	7.39	0.133
		Secondary Intermediate – Primary Intermediate	9.10	0.412
		Primary Good – Primary Intermediate	-1.71	0.872
	Rhabdomys dilectus	Primary Intermediate – Secondary Good	-6.61	0.652
		Primary Intermediate – Primary Good	19.32	0.171
		Primary Intermediate – Secondary Intermediate	-25.77	0.082
		Secondary Good – Primary Good	12.71	0.041*
	Aethomys ineptus	Secondary Good – Secondary Intermediate	-19.16	0.012*
		Primary Good – Secondary Intermediate	-6.45	0.325
		Primary Intermediate – Secondary Good	-3.75	0.836
		Primary Intermediate – Primary Good	18.95	0.279
	Crociodura flavescens	Primary Intermediate – Secondary Intermediate	-26.35	0.151
		Secondary Good – Primary Good	15.2	0.048*
		Secondary Good – Secondary Intermediate	-22.6	0.017*
		Primary Good – Secondary Intermediate	-7.41	0.362
	Mastomys natalensis	Primary Good – Primary Intermediate	0.00	1.000
		Primary Good – Secondary Good	0.00	1.000
		Primary Good – Secondary Intermediate	0.00	1.000
		Primary Intermediate – Secondary Good	0.00	1.000
	Myosorex varius	Primary Intermediate – Secondary Intermediate	0.00	1.000
		Secondary Good – Secondary Intermediate	0.00	1.000
		Primary Good – Secondary Good	-5.71	0.420
		Primary Good – Secondary Intermediate	-24.26	0.001*
	Rhabdomys dilectus	Primary Good – Primary Intermediate	-44.16	0.006*
		Secondary Good – Secondary Intermediate	-18.55	0.033*
		Secondary Good – Primary Intermediate	38.45	0.021*

Appendix 19 continued...

Metrics	Species	Grassland Condition	Z-test	P-value
		Secondary Intermediate – Primary Intermediate	19.90	0.239

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 20: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests of species-specific tests in relation to the effect of regional variation on small mammal populations in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Species	Regions	Z-test	P-value
Population estimates	<i>Aethomys ineptus</i>	Hillcrest – Kloof	-1.55	0.565
		Hillcrest – Camperdown	5.95	0.027*
		Kloof – Camperdown	4.40	0.102
	<i>Crocidura flavescens</i>	Kloof – Camperdown	6.78	0.116
		Kloof – Hillcrest	9.45	0.029*
		Camperdown – Hillcrest	-2.68	0.538
	<i>Mastomys natalensis</i>	Camperdown – Kloof	-12.12	0.044*
		Camperdown – Hillcrest	-15.04	0.022*
		Kloof – Hillcrest	2.92	0.653
	<i>Rhabdomys dilectus</i>	Kloof – Hillcrest	6.87	0.289
		Kloof – Camperdown	51.00	<0.001*
		Hillcrest – Camperdown	44.13	<0.001*
Trap success	<i>Aethomys ineptus</i>	Hillcrest – Kloof	-9.48	0.039*
		Hillcrest – Camperdown	10.33	0.025*
		Kloof – Camperdown	0.85	0.853
	<i>Crocidura flavescens</i>	Kloof – Camperdown	0.94	0.878
		Kloof – Hillcrest	6.26	0.307
		Camperdown – Hillcrest	-5.33	0.385
	<i>Mastomys natalensis</i>	Camperdown – Kloof	-9.63	0.205
		Camperdown – Hillcrest	-22.18	0.004*
		Kloof – Hillcrest	12.55	0.098
	<i>Rhabdomys dilectus</i>	Kloof – Hillcrest	16.33	0.020*
		Kloof – Camperdown	60.18	<0.001*
		Hillcrest – Camperdown	43.85	<0.001*

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 21: The Dunn's *post-hoc* test outcomes from Kruskal-Wallis tests of species-specific tests in relation to the effect of habitat management regimes (treatments) on small mammal populations in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Species	Treatments	Z-test	P-value
Population estimates	<i>Aethomys ineptus</i>	Inside – Outside	-1.55	0.565
		Inside – Control	5.95	0.027*
		Outside – Control	4.40	0.102
	<i>Crocidura flavescens</i>	Inside – Control	5.14	0.236
		Inside – Outside	-5.99	0.167
		Control – Outside	-0.85	0.844
	<i>Mastomys natalensis</i>	Control – Outside	-12.69	0.045*
		Control – Inside	-13.94	0.024*
		Outside – Inside	1.25	0.846
	<i>Rhabdomys dilectus</i>	Inside – Outside	-2.03	0.755
		Inside – Control	48.67	<0.001*
		Outside – Control	46.64	<0.001*
Trap success	<i>Aethomys ineptus</i>	Outside – Inside	9.48	0.039*
		Outside – Control	10.33	0.025*
		Inside – Control	0.85	0.853
	<i>Crocidura flavescens</i>	Outside – Control	4.40	0.473

Appendix 21 continued...		Outside – Inside	13.19	0.031*
Metrics	Species	Treatments	Z-test	P-value
	<i>Mastomys natalensis</i>	Control – Inside	-8.79	0.152
		Control – Outside	-4.19	0.581
		Control – Inside	-27.61	<0.001*
	<i>Rhabdomys dilectus</i>	Outside – Inside	23.43	0.002*
		Outside – Inside	0.03	0.997
		Outside – Control	52.03	<0.001*
		Inside – Control	52.00	<0.001*

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 22: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of elevational gradients on small mammal age, reproductive status and sex in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Species	B-coefficient	χ^2	DF	P-value
Age	<i>Aethomys ineptus</i>	4.2	0.36	2	0.836
	<i>Crocidura flavescens</i>	9.38	4.52	4	0.340
	<i>Mastomys natalensis</i>	51.77	23.58	4	<0.001*
Reproductive status	<i>Aethomys ineptus</i>	35.4	13.34	12	0.345
	<i>Mastomys natalensis</i>	112.43	36.59	12	<0.001*
Sex	<i>Aethomys ineptus</i>	7.89	1.73	2	0.421
	<i>Crocidura flavescens</i>	9.61	0.11	2	0.946
	<i>Mastomys natalensis</i>	17.26	0.94	2	0.625

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 23: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of elevational gradients on small mammal age in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Age	Elevation	B	Std. Error	Wald	DF	P-value
<i>Aethomys ineptus</i>	Adults	Intercept	18.75	1.03	333.02	1	<0.001*
		Low	-15.86	0.00	-	1	-
		High	-16.55	1.47	126.44	1	<0.001*
		Mid	Ob	-	-	0	-
	a The reference category is: Sub-adults						
<i>Crocidura flavescens</i>	b This parameter is set to zero because it is redundant.						
	Adults	Intercept	20.92	0.61	1.181.433	1	<0.001*
		Low	0.00	7.794.656	0.00	1	1.000
		Mid	0c	-	-	0	-
		High	-18.72	0.00	.	1	.
	Juveniles	Intercept	-1.10	1.16	0.91	1	0.341
		Low	0.00	0.00	-	1	-
		Mid	0c	-	-	0	-
		High	0.00	0.00	-	1	-
	a The reference category is: Sub-adults						
	B Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.						
	c This parameter is set to zero because it is redundant.						
<i>Mastomys natalensis</i> *	Adults	Intercept	1.82	0.27	45.75	1	<0.001*
		Low	-1.00	0.34	8.87	1	0.003*
		Mid	Ob	-	-	0	-
		High	-0.99	0.32	9.56	1	0.002*
	Juveniles	Intercept	-0.21	0.37	0.31	1	0.578
		Low	-0.16	0.46	0.12	1	0.730
		Mid	Ob	-	-	0	-
		High	-1.38	0.51	7.35	1	0.007*

a The reference category is: Sub-adults

b This parameter is set to zero because it is redundant.

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 24: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of elevational gradients on small mammal reproductive status in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Reproductive status	Elevation	B	Std. Error	Wald	DF	P-value
<i>Aethomys ineptus</i>	Abdominal	Intercept	17.04	2.82	0.00	1	0.995
		High	-15.94	2.82	0.00	1	0.995
		Low	-17.96	2.82	0.00	1	0.995
		Mid	0c	-	-	0	-
	Scrotal	Intercept	-1.79	1.10	2.68	1	0.102
		High	-12.69	1.40	0.00	1	0.993
		Low	0.18	0.00	-	1	-
		Mid	0c	-	-	0	-
	Imperforate	Intercept	-0.18	0.67	0.07	1	0.786
		High	0.18	1.57	0.01	1	0.907
		Low	-0.04	0.00	-	1	-
		Mid	0c	-	-	0	-
	Perforate	Intercept	0.15	0.61	0.07	1	0.799
		High	-0.15	1.54	0.01	1	0.920
		Low	0.03	0.00	-	1	-
		Mid	0c	-	-	0	-
	Pregnant	Intercept	-1.79	12.61	0.00	1	1.000
		High	1.79	12.61	0.00	1	1.000
		Low	-13.37	12.65	0.00	1	0.999
		Mid	0c	-	-	0	-
	Lactating	Intercept	-0.41	6.76	0.00	1	1.000
		High	10.50	6.76	0.00	1	1.000
		Low	-10.20	6.76	0.00	1	1.000
		Mid	0c	-	-	0	-

a The reference category is: Semi-scrotal

b Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.

c This parameter is set to zero because it is redundant.

<i>Mastomys natalensis</i>	Abdominal	Intercept	1.10	0.30	13.58	1	<0.001*
		High	-0.37	0.39	0.91	1	0.341
		Low	-0.35	0.43	0.66	1	0.417
		Mid	0b	-	-	0	-
	Scrotal	Intercept	-1.10	0.52	4.53	1	0.033*
		High	0.08	0.65	0.01	1	0.905
		Low	1.27	0.62	4.23	1	0.040*
		Mid	0b	-	-	0	-
	Imperforate	Intercept	0.85	0.31	7.54	1	0.006*
		High	-0.57	0.41	1.96	1	0.161
		Low	-0.04	0.43	0.01	1	0.933
		Mid	0b	-	-	0	-
	Perforate	Intercept	0.24	0.35	0.47	1	0.494
		High	0.07	0.43	0.03	1	0.870
		Low	-0.30	0.50	0.36	1	0.546
		Mid	0b	-	-	0	-
	Pregnant	Intercept	-1.10	0.52	4.53	1	0.033*
		High	-2.12	1.14	3.44	1	0.064
		Low	0.63	0.66	0.92	1	0.337
		Mid	0b	-	-	0	-
	Lactating	Intercept	-1.32	0.56	5.52	1	0.019*
		High	0.99	0.64	2.39	1	0.122
		Low	1.11	0.68	2.72	1	0.099
		Mid	0b	-	-	0	-

a The reference category is: Semi-scrotal

b This parameter is set to zero because it is redundant.

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 25: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of elevational gradients on small mammal sex in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Sex	Elevation	B	Std. Error	Wald	DF	P-value
Aethomys ineptus	Females	Intercept	-17.24	0.47	1.38	1	<0.001*
		Low	17.55	0.00	-	1	-
		Mid	0b	-	-	0	-
		High	17.64	0.80	491.93	1	<0.001*
a The reference category is: Males							
b This parameter is set to zero because it is redundant.							
Crociodura flavescens	Females	Intercept	-0.29	0.76	0.14	1	0.706
		Low	-0.12	0.93	0.02	1	0.899
		Mid	0b	-	-	0	-
		High	0.09	0.85	0.01	1	0.912
a The reference category is: Males							
b This parameter is set to zero because it is redundant.							
Mastomys natalensis	Females	Intercept	-0.03	0.18	0.03	1	0.860
		Low	0.21	0.24	0.77	1	0.381
		Mid	0b	-	-	0	-
		High	0.03	0.23	0.02	1	0.894
a The reference category is: Males							
b This parameter is set to zero because it is redundant.							

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 26: The outcomes from Kruskal-Wallis of species-specific tests in relation to the effect of elevational gradients on small mammal head and body (mm) in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Elevation	Z-test	P-value
<i>Aethomys ineptus</i>	Mid - Low	6.53	0.469
	Mid - High	16.10	0.080
	Low - High	9.57	0.005*
<i>Crociodura flavescens</i>	High-Mid	-11.608	0.072
	High-Low	-15.156	0.002*
	Mid-Low	3.548	0.616
<i>Mastomys natalensis</i>	High - Low	-10.077	0.487
	High - Mid	-43.161	0.004*
	Low - Mid	-33.084	0.034*

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 27: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of slope aspect on small mammal age, reproductive status and sex in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Species	B-coefficient	χ^2	DF	P-value
Age	<i>Aethomys ineptus</i>	5.57	1.78	5	0.878
	<i>Crocidura flavescens</i>	17.2	11.08	14	0.680
	<i>Mastomys natalensis</i>	74.27	29.75	14	0.008*
	<i>Myosorex varius</i>	12.92	7.82	4	0.099
	<i>Rhabdomys dilectus</i>	60.53	25.89	10	0.004*
Reproductive status	<i>Aethomys ineptus</i>	54.82	0.00	0	1.000
	<i>Mastomys natalensis</i>	178.02	65.78	42	0.011*
	<i>Rhabdomys dilectus</i>	147.51	54.73	30	0.004*
Sex	<i>Aethomys ineptus</i>	13.43	4.30	5	0.507
	<i>Crocidura flavescens</i>	21.45	5.56	7	0.592
	<i>Mastomys natalensis</i>	36.55	8.98	7	0.254
	<i>Myosorex varius</i>	15.77	3.24	4	0.519
	<i>Rhabdomys dilectus</i>	28.3	3.87	5	0.568

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 28: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of slope aspect on small mammal age in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Southveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Age	Slope aspect	B	Std. Error	Wald	DF	P-value
<i>Aethomys ineptus</i>	Adults	Intercept	18.75	5.89	0	1	0.997
		East	0.00	0.00	-	1	-
		North	-16.35	5.89	0	1	0.998
		North-East	0.00	8.33	0	1	1.000
		North-West	0.00	10.20	0	1	1.000
		South-East	-16.80	5.89	0	1	0.998
		West	0c	-	-	0	-
a The reference category is: Sub-adults							
b Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.							
c This parameter is set to zero because it is redundant.							
<i>Crocidura flavescens</i>	Adults	Intercept	18.41	2.42	0.00	1	0.994
		East	0.00	4.72	0.00	1	1.000
		North	0.00	6.45	0.00	1	1.000
		North-East	0.00	5.53	0.00	1	1.000
		North-West	-16.21	2.42	0.00	1	0.995
		South	0.00	5.53	0.00	1	1.000
		South-East	-17.31	2.42	0.00	1	0.994
		South-West	0.00	6.45	0.00	1	1.000
		West	0c	-	-	0	-
	Juveniles	Intercept	-1.10	4.44	0.00	1	1.000
		East	0.00	9.25	0.00	1	1.000
		North	0.00	0.00	-	1	-
		North-East	0.00	10.88	0.00	1	1.000
		North-West	-16.09	6.98	0.00	1	0.998
		South	0.00	10.88	0.00	1	1.000
		South-East	0.41	4.44	0.00	1	1.000
		South-West	0.00	0.00	-	1	-
		West	0c	-	-	0	-
a The reference category is: Sub-adults							
b Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.							
c This parameter is set to zero because it is redundant.							
<i>Mastomys natalensis</i>	Adults	Intercept	0.75	0.19	16.41	1	<0.001*
		East	18.00	5.88	0.00	1	0.998
		North	0.64	0.81	0.62	1	0.433
		North-East	-0.31	0.43	0.54	1	0.464
		North-West	0.88	0.30	8.46	1	0.004*

Species	Age	Slope aspect	B	Std. Error	Wald	DF	P-value
	Juveniles	South	-0.75	1.43	0.28	1	0.599
		South-East	0.05	0.34	0.02	1	0.885
		South-West	0.90	0.52	2.96	1	0.085
		West	0c	-	-	0	-
		Intercept	-0.47	0.25	3.59	1	0.058*
		East	-0.28	10.36	0.00	1	1.000
		North	-0.23	1.25	0.03	1	0.855
		North-East	-1.24	0.81	2.36	1	0.125
		North-West	-0.09	0.44	0.05	1	0.829
		South	-18.37	0.00	-	1	-
		South-East	-1.73	0.79	4.87	1	0.027*
		South-West	0.24	0.71	0.12	1	0.735
		West	0c	-	-	0	-

a The reference category is: Sub-adults.

b Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.

c This parameter is set to zero because it is redundant.

<i>Myosorex varius</i>	Adults	North-East	0.00	8.39	0.00	1	1.000
		North-West	-19.02	1.04	335.97	1	<0.001*
		South-East	-19.06	0.00	-	1	-
		South-West	0.00	7.27	0.00	1	1.000
		West	0c	-	-	0	-

a The reference category is: Sub-adults

b Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.

c This parameter is set to zero because it is redundant.

<i>Rhodomys dilectus</i>	Adults	Intercept	0.64	0.39	2.70	1	0.100
		East	-0.17	0.69	0.06	1	0.804
		North-East	0.99	0.53	3.43	1	0.064
		North-West	2.65	0.82	10.50	1	0.001*
		South-East	0.51	0.48	1.16	1	0.282
		South-West	0.05	0.58	0.01	1	0.930
		West	0b	-	-	0	-
	Juveniles	Intercept	-1.61	0.78	4.32	1	0.038*
		East	-18.69	0.00	-	1	-
		North-East	0.80	0.98	0.66	1	0.415
		North-West	1.61	1.27	1.62	1	0.203
		South-East	-0.59	1.08	0.30	1	0.585
		South-West	0.22	1.11	0.04	1	0.840
		West	0b	-	-	0	-

a The reference category is: Sub-adults

b This parameter is set to zero because it is redundant.

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 29: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of slope aspect on small mammal reproductive status in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Reproductive status	Slope aspect	B	Std. Error	Wald	DF	P-value
<i>Aethomys ineptus</i>	Abdominal	Intercept	0.00	1.45	0.00	1	1.000
		East	0.00	17.74	0.00	1	1.000
		North	-1.32	1.81	0.53	1	0.467
		North-East	12.53	0.00	-	1	-
		North-West	0.00	7.44	0.00	1	1.000
		South-East	1.11	1.94	0.33	1	0.565
		West	0b	-	-	0	-
	Scrotal	Intercept	-4.07	7.91	0.27	1	0.607

Species	Reproductive status	Slope aspect	B	Std. Error	Wald	DF	P-value
<i>Mastomys natalensis</i>	Imperforate	East	2.28	34.01	0.00	1	0.947
		North	0.06	8.75	0.00	1	0.995
		North-East	43.75	0.00	-	1	-
		North-West	2.28	15.78	0.02	1	0.885
		South-East	0.61	10.15	0.00	1	0.952
		West	0b	-	-	0	-
		Intercept	-2.46	3.65	0.45	1	0.501
		East	2.28	18.90	0.02	1	0.904
		North	2.14	3.73	0.33	1	0.566
		North-East	14.37	0.00	-	1	-
	Perforate	North-West	5.78	6.40	0.82	1	0.366
		South-East	0.61	4.74	0.02	1	0.897
		West	0b	-	-	0	-
		Intercept	-0.03	1.46	0.00	1	0.982
		East	5.06	12.63	0.16	1	0.689
		North	-0.32	1.66	0.04	1	0.846
		North-East	12.81	0.00	-	1	-
		North-West	3.15	5.47	0.33	1	0.565
		South-East	-1.48	3.00	0.24	1	0.622
		West	0b	-	-	0	-
	Pregnant	Intercept	-4.07	7.91	0.27	1	0.607
		East	2.28	34.01	0.00	1	0.947
		North	0.06	8.75	0.00	1	0.995
		North-East	2.28	0.00	-	1	-
		North-West	2.28	15.78	0.02	1	0.885
		South-East	4.66	8.03	0.34	1	0.562
		West	0b	-	-	0	-
		Intercept	-2.68	4.05	0.44	1	0.508
		East	2.28	20.18	0.01	1	0.910
		North	1.27	4.21	0.09	1	0.763
	Lactating	North-East	2.28	0.00	-	1	-
		North-West	2.28	9.11	0.06	1	0.803
		South-East	3.97	4.24	0.88	1	0.349
		West	0b	-	-	0	-
	a The reference category is: Semi-scrotal						
	b This parameter is set to zero because it is redundant.						
	Abdominal	Intercept	0.83	0.26	9.98	1	0.002*
		East	-0.83	1.03	0.64	1	0.424
		North	15.48	1.73	0.00	1	0.930
		North-East	0.47	0.70	0.45	1	0.501
		North-West	-0.19	0.39	0.24	1	0.627
		South	-17.85	4.98	0.00	1	0.997
		South-East	0.34	0.58	0.34	1	0.559
		South-West	0.20	0.58	0.12	1	0.728
		West	0c	-	-	0	-
		Intercept	-1.10	0.44	6.34	1	0.012*
	Scrotal	East	-17.52	7.80	0.00	1	0.998
		North	16.71	1.73	0.00	1	0.992
		North-East	0.00	1.23	0.00	1	1.000
		North-West	0.99	0.55	3.23	1	0.072
		South	-17.31	9.92	0.00	1	0.999
		South-East	1.10	0.77	2.04	1	0.153
		South-West	-0.51	1.18	0.19	1	0.665
		West	0c	-	-	0	-
		Intercept	0.54	0.28	3.85	1	0.050*
		East	-18.01	4.39	0.00	1	0.997
	Imperforate	North	15.48	1.73	0.00	1	0.993

Species	Reproductive status	Slope aspect	B	Std. Error	Wald	DF	P-value	
	Perforate	North-East	-0.03	0.78	0.00	1	0.971	
		North-West	0.15	0.39	0.15	1	0.695	
		South	-17.80	5.59	0.00	1	0.997	
		South-East	0.34	0.60	0.32	1	0.574	
		South-West	0.15	0.61	0.06	1	0.801	
		West	0c	-	-	0	-	
		Intercept	0.42	0.28	2.25	1	0.134	
		East	-18.32	5.43	0.00	1	0.997	
		North	14.50	1.73	0.00	1	0.993	
		North-East	0.09	0.78	0.01	1	0.909	
		North-West	-0.66	0.45	2.18	1	0.140	
		South	-0.42	1.44	0.09	1	0.770	
	Pregnant	South-East	0.27	0.62	0.20	1	0.659	
		South-West	-0.64	0.73	0.79	1	0.376	
		West	0c	-	-	0	-	
		Intercept	-1.95	0.62	9.94	1	0.002*	
		East	-17.40	0.00	-	1	-	
		North	0.69	3.68	0.00	1	1.000	
		North-East	2.23	0.98	5.17	1	0.023*	
		North-West	1.20	0.74	2.64	1	0.104	
		South	-17.18	0.00	-	1	-	
		South-East	-14.10	1.36	0.00	1	0.992	
		South-West	-14.51	1.67	0.00	1	0.993	
		Lactating	West	0c	-	-	0	-
	Intercept		-0.41	0.35	1.38	1	0.240	
	East		-18.15	7.57	0.00	1	0.998	
	North		15.32	1.73	0.00	1	0.993	
	North-East		-0.69	1.21	0.33	1	0.565	
	North-West		-0.75	0.58	1.65	1	0.199	
	South		-17.94	9.64	0.00	1	0.999	
	South-East		1.28	0.63	4.08	1	0.043*	
	South-West		-1.20	1.15	1.10	1	0.294	
	West		0c	-	-	0	-	
a The reference category is: Semi-scrotal								
b Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.								
c This parameter is set to zero because it is redundant.								
<i>Rhabdomys dilectus</i>	Abdominal	Intercept	2.02	0.75	7.16	1	0.007*	
		East	-0.22	1.32	0.03	1	0.865	
		North-East	-0.46	0.93	0.24	1	0.624	
		North-West	-1.08	0.85	1.61	1	0.205	
		South-East	-1.27	0.85	2.27	1	0.132	
		South-West	-2.02	1.03	3.81	1	0.051*	
	Scrotal	West	0b	-	-	0	-	
		Intercept	-0.69	1.23	0.32	1	0.571	
		East	-17.78	0.00	-	1	-	
		North-East	0.92	1.40	0.43	1	0.512	
		North-West	-0.81	1.45	0.31	1	0.577	
		South-East	0.00	1.34	0.00	1	1.000	
	Imperforate	South-West	0.00	1.50	0.00	1	1.000	
		West	0b	-	-	0	-	
		Intercept	1.61	0.78	4.32	1	0.038*	
		East	-0.22	1.36	0.03	1	0.870	
		North-East	-1.61	1.05	2.36	1	0.125	
		North-West	-1.73	0.91	3.57	1	0.059*	
		South-East	-1.35	0.88	2.34	1	0.126	
		South-West	-1.61	1.05	2.36	1	0.125	
		West	0b	-	-	0	-	

Species	Reproductive status	Slope aspect	B	Std. Error	Wald	DF	P-value
	Perforate	Intercept	0.00	1.00	0.00	1	1.000
		East	-17.00	4.90	0.00	1	0.997
		North-East	1.56	1.14	1.86	1	0.172
		North-West	0.37	1.09	0.11	1	0.736
		South-East	0.79	1.07	0.54	1	0.461
		South-West	0.92	1.16	0.62	1	0.430
		West	0b	-	-	0	-
	Pregnant	Intercept	-0.69	1.23	0.32	1	0.571
		East	1.39	1.73	0.64	1	0.423
		North-East	0.41	1.44	0.08	1	0.779
		North-West	-17.86	3.57	0.00	1	0.996
		South-East	-0.51	1.39	0.14	1	0.713
		South-West	-17.94	5.55	0.00	1	0.997
		West	0b	-	-	0	-
	Lactating	Intercept	-17.50	4.14	0.00	1	0.997
		East	-1.12	0.00	-	1	-
		North-East	17.73	4.14	0.00	1	0.997
		North-West	16.41	4.14	0.00	1	0.997
		South-East	16.30	4.14	0.00	1	0.997
		South-West	16.81	4.14	0.00	1	0.997
		West	0b	-	-	0	-

a The reference category is: Semi-scrotal

b This parameter is set to zero because it is redundant.

c Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 30: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of slope aspect on small mammal sex in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Sex	Slope aspect	B	Std. Error	Wald	DF	P-value
<i>Aethomys ineptus</i>	Females	Intercept	-0.69	1.23	0.32	1	0.571
		East	19.92	0.00	-	1	-
		North	1.03	1.36	0.58	1	0.448
		North-East	0.69	1.58	0.19	1	0.661
		North-West	19.92	0.00	-	1	-
		South-East	0.69	1.41	0.24	1	0.624
		West	0b	-	-	0	-
	a The reference category is: Males						
	b This parameter is set to zero because it is redundant.						
<i>Crocidura flavescens</i>	Females	Intercept	0.00	0.50	0.00	1	1.000
		East	0.00	0.96	0.00	1	1.000
		North	0.00	1.50	0.00	1	1.000
		North-East	0.00	1.12	0.00	1	1.000
		North-West	-0.85	0.85	0.99	1	0.320
		South	-1.10	1.26	0.76	1	0.383
		South-East	-0.69	0.87	0.64	1	0.423
		South-West	19.23	0.00	-	1	-
		West	0b	-	-	0	-
	a The reference category is: Males						
	b This parameter is set to zero because it is redundant.						
<i>Mastomys natalensis</i>	Females	Intercept	0.11	0.16	0.50	1	0.478
		East	-	0.00	-	1	-
		North	20.32	0.07	0.01	1	0.910

Species	Sex	Slope aspect	B	Std. Error	Wald	DF	P-value
		North-East	0.29	0.41	0.53	1	0.468
		North-West	-0.17	0.23	0.53	1	0.465
		South	-0.11	1.42	0.01	1	0.937
		South-East	0.16	0.31	0.26	1	0.608
		South-West	-0.40	0.38	1.13	1	0.288
		West	0b	-	-	0	-
		a The reference category is: Males					
b This parameter is set to zero because it is redundant.							
<i>Myosorex varius</i>	Females	Intercept	0.00	1.41	0.00	1	1.000
		North-East	0.22	1.57	0.02	1	0.887
		North-West	0.69	1.58	0.19	1	0.661
		South-East	-0.81	1.54	0.28	1	0.598
		South-West	-0.34	1.53	0.05	1	0.826
		West	0b	-	-	0	-
		a The reference category is: Males					
b This parameter is set to zero because it is redundant.							
<i>Rhabdomys dilectus</i>	Females	Intercept	-0.33	0.36	0.80	1	0.371
		East	0.17	0.67	0.07	1	0.797
		North-East	0.43	0.45	0.91	1	0.340
		North-West	0.05	0.45	0.01	1	0.915
		South-East	0.46	0.43	1.12	1	0.289
		South-West	0.80	0.54	2.15	1	0.143
		West	0b	-	-	0	-
a The reference category is: Males							
b This parameter is set to zero because it is redundant.							
* Significant differences when evaluated at $\alpha = 0.05$							

Appendix 31: The Dunn's outcomes from Kruskal-Wallis of species-specific tests in relation to the effect of slope aspect on small mammal head and body length (mm) in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Slope aspect	Z-test	P-value
<i>Aethomys ineptus</i>	West - North-West	1.92	0.811
	West - North-East	6.54	0.329
	West - North	7.54	0.183
	West - South-East	16.42	0.006*
	West - East	23.17	0.022*
	North-West - North-East	4.63	0.543
	North-West - North	5.63	0.401
	North-West - South-East	-14.50	0.037*
	North-West - East	21.25	0.048*
	North-East - North	1.00	0.844
	North-East - South-East	-9.88	0.066
	North-East - East	16.63	0.090
	North - South-East	-8.88	0.027*
	North - East	15.63	0.087
	South-East - East	6.75	0.468
<i>Crocidura flavescens</i>	South-East - West	-15.74	0.014*
	South-East - North-West	17.91	0.012*
	South-East - North-East	19.24	0.038*
	South-East - South	21.86	0.018*
	South-East - North	26.61	0.027*
	South-East - East	34.61	<0.001*
	South-East - South-West	-38.61	0.001*
	West - North-West	2.18	0.727
	West - North-East	3.50	0.685
	West - South	6.13	0.478
	West - North	10.88	0.347
	West - East	18.88	0.011*

Species	Slope aspect	Z-test	P-value
<i>Mastomys natalensis</i>	West - South-West	22.88	0.048*
	North-West - North-East	1.33	0.885
	North-West - South	-3.95	0.665
	North-West - North	8.70	0.467
	North-West - East	16.70	0.036*
	North-West - South-West	-20.70	0.083
	North-East - South	-2.63	0.810
	North-East - North	7.38	0.581
	North-East - East	15.38	0.123
	North-East - South-West	-19.38	0.147
	South - North	4.75	0.722
	South - East	12.75	0.201
	South - South-West	-16.75	0.210
	North - East	8.00	0.525
	North - South-West	-12.00	0.437
	East - South-West	-4.00	0.751
	South - West	-51.05	0.575
	South - East	69.75	0.529
	South - South-East	-94.13	0.306
	South - North-West	106.15	0.244
	South - North-East	110.97	0.235
	South - North	121.86	0.215
	South - South-West	-132.69	0.154
	West - East	18.70	0.773
	West - South-East	43.08	0.026*
	West - North-West	55.10	<0.001*
	West - North-East	59.92	0.019*
	West - North	70.82	0.076
	West - South-West	81.64	<0.001*
	East - South-East	-24.38	0.712
	East - North-West	-36.40	0.575
	East - North-East	-41.22	0.545
	East - North	-52.11	0.485
	East - South-West	-62.94	0.351
	South-East - North-West	12.03	0.543
	South-East - North-East	16.84	0.556
	South-East - North	27.74	0.509
	South-East - South-West	-38.56	0.157
	North-West - North-East	4.82	0.852
	North-West - North	15.71	0.695
	North-West - South-West	-26.54	0.273
	North-East - North	10.90	0.809
	North-East - South-West	-21.72	0.495
	North - South-West	-10.82	0.807
<i>Myosorex varius</i>	West - South-West	2.88	0.774
	West - North-East	7.19	0.483
	West - South-East	10.02	0.315
	West - North-West	20.75	0.043*
	South-West - North-East	4.32	0.455
	South-West - South-East	7.14	0.174
	South-West - North-West	17.88	0.002*
	North-East - South-East	-2.83	0.619
	North-East - North-West	-13.56	0.028*
	South-East - North-West	10.73	0.059*
<i>Rhabdomys dilectus</i>	North-East - East	1.17	0.960
	North-East - West	-5.73	0.735
	North-East - South-West	-6.65	0.711
	North-East - South-East	-13.24	0.316
	North-East - North-West	-74.75	<0.001*
	East - West	-4.57	0.856
	East - South-West	-5.48	0.833
	East - South-East	-12.07	0.598
	East - North-West	-73.58	0.002*
	West - South-West	0.92	0.964

Appendix 31 continued...

Species	Slope aspect	Z-test	P-value
	West - South-East	7.51	0.644
	West - North-West	69.01	<0.001*
	South-West - South-East	6.59	0.703
	South-West - North-West	68.10	<0.001*
	South-East - North-West	61.51	<0.001*

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 32: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of seasons on small mammal age, reproductive status in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

	Species	B-coefficient	χ^2	DF	P-value
Age	<i>Aethomys ineptus</i>	5.3	1.49	3	0.684
	<i>Crocidura flavescens</i>	11.46	5.08	6	0.533
	<i>Mastomys natalensis</i>	79.07	44.36	6	<0.001*
	<i>Myosorex varius</i>	11.04	7.81	3	0.050*
	<i>Rhabdomys dilectus</i>	47.92	21.6	6	<0.001*
Reproductive status	<i>Aethomys ineptus</i>	46.76	19.84	18	0.342
	<i>Mastomys natalensis</i>	207.47	119.32	18	<0.001*
	<i>Rhabdomys dilectus</i>	134.63	62.79	18	<0.001*
Sex	<i>Aethomys ineptus</i>	10.21	0.5	3	0.920
	<i>Crocidura flavescens</i>	14.48	2.69	3	0.441
	<i>Mastomys natalensis</i>	21.26	0.84	3	0.840
	<i>Myosorex varius</i>	12.46	1.6	3	0.659
	<i>Rhabdomys dilectus</i>	22.97	4.78	3	0.189

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 33: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of seasons on small mammal age in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Age	Species	B	Std. Error	Wald	DF	P-value
<i>Aethomys ineptus</i>	Adults	Intercept	20.75	1.05	387.46	1	<0.001*
		Winter	0b	-	-	0	-
		Spring	-18.45	1.49	153.88	1	<0.001*
		Summer	-18.55	0.00	-	1	-
		Autumn	0.00	0.00	-	1	-
a The reference category is: Sub-adults							
b This parameter is set to zero because it is redundant.							
<i>Crocidura flavescens</i>	Adults	Intercept	19.92	5.60	0.00	1	0.997
		Winter	0c	-	-	0	-
		Spring	-17.43	5.60	0.00	1	0.998
		Summer	0.00	8.58	0.00	1	1.000
		Autumn	-17.78	5.60	0.00	1	0.997
	Juveniles	Intercept	-1.10	9.46	0.00	1	1.000
		Winter	0c	-	-	0	-
		Spring	-17.32	13.76	0.00	1	0.999
		Summer	0.00	0.00	-	1	-
		Autumn	0.41	9.46	0.00	1	1.000

a The reference category is: Sub-adults

Species	Age	Species	B	Std. Error	Wald	DF	P-value
b Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.							
c This parameter is set to zero because it is redundant.							
Mastomys natalensis	Adults	Intercept	1.00	0.31	10.20	1	0.001*
		Winter	0b	-	-	0	-
		Spring	0.32	0.40	0.65	1	0.421
		Summer	1.29	0.45	8.35	1	0.004*
		Autumn	-0.73	0.36	4.16	1	0.041*
	Juveniles	Intercept	-0.69	0.46	2.24	1	0.134
		Winter	0b	-	-	0	-
		Spring	-0.36	0.64	0.31	1	0.576
		Summer	0.86	0.62	1.94	1	0.164
		Autumn	-0.29	0.53	0.30	1	0.587
a The reference category is: Sub-adults							
b This parameter is set to zero because it is redundant.							
Myosorex varius	Adults	Intercept	20.27	0.51	1.587.082	1	<0.001*
		Winter	0c	-	-	0	-
		Spring	0.00	0.00	-	1	-
		Summer	0.00	7.59	0.00	1	1.000
		Autumn	-19.04	0.00	-	1	-
a The reference category is: Sub-adults							
b Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.							
c This parameter is set to zero because it is redundant.							
Rhabdomys dilectus	Adults	Intercept	0.90	0.25	12.60	1	<0.001*
		Winter	0b	-	-	0	-
		Spring	1.69	0.65	6.78	1	0.009*
		Summer	1.91	0.77	6.11	1	0.013*
		Autumn	0.17	0.34	0.26	1	0.613
	Juveniles	Intercept	-1.99	0.62	10.48	1	0.001*
		Winter	0b	-	-	0	-
		Spring	2.50	0.96	6.87	1	0.009*
		Summer	1.99	1.17	2.88	1	0.090
		Autumn	-0.53	0.96	0.31	1	0.578
a The reference category is: Sub-adults							
b This parameter is set to zero because it is redundant.							

* Significant differences when evaluated at α = 0.05

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 34: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of seasons on small mammal reproductive status in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Reproductive status	Seasons	B	Std. Error	Wald	DF	P-value
<i>Aethomys ineptus</i>	Abdominal	Intercept	0.00	1.41	0.00	1	1.000
		Autumn	0.00	2.00	0.00	1	1.000
		Spring	-0.69	1.87	0.14	1	0.711
		Summer	0.41	1.68	0.06	1	0.810
		Winter	0b	-	-	0	-
	Scrotal	Intercept	-18.98	7.56	0.00	1	0.998
		Autumn	-0.19	0.00	-	1	-
		Spring	18.28	7.56	0.00	1	0.998
		Summer	-0.49	0.00	-	1	-
		Winter	0b	-	-	0	-
	Imperforate	Intercept	0.00	1.41	0.00	1	1.000
		Autumn	0.00	2.00	0.00	1	1.000

Species	Reproductive status	Seasons	B	Std. Error	Wald	DF	P-value		
	Perforate	Spring	0.41	1.68	0.06	1	0.81		
		Summer	-17.86	5.34	0.00	1	0.997		
		Winter	0b	-	-	0	-		
		Intercept	0.69	1.23	0.32	1	0.571		
		Autumn	-0.69	1.87	0.14	1	0.711		
	Pregnant	Spring	-0.29	1.53	0.04	1	0.851		
		Summer	-1.39	1.73	0.64	1	0.423		
		Winter	0b	-	-	0	-		
		Intercept	-18.98	7.56	0.00	1	0.998		
		Autumn	-0.19	0.00	-	1	-		
	Lactating	Spring	18.28	7.56	0.00	1	0.998		
		Summer	-0.49	0.00	-	1	-		
		Winter	0b	-	-	0	-		
		Intercept	-17.59	0.87	412.60	1	<0.001		
		Autumn	-0.19	7.25	0.00	1	1.000		
		Spring	-0.05	4.79	0.00	1	1.000		
		Summer	18.28	0.00	-	1	-		
		Winter	0b	-	-	0	-		
a The reference category is: Semi-scrotal									
b This parameter is set to zero because it is redundant.									
<i>Mastomys natalensis</i>	Abdominal	Intercept	0.87	0.42	4.21	1	0.040*		
		Autumn	1.49	0.60	6.13	1	0.013*		
		Spring	-0.93	0.55	2.87	1	0.090		
		Summer	-0.59	0.50	1.39	1	0.238		
		Winter	0b	-	-	0	-		
	Scrotal	Intercept	-0.98	0.68	2.10	1	0.147		
		Autumn	-0.1180.	1.06	0.01	1	0.912		
		Spring	1.14	0.75	2.31	1	0.129		
		Summer	-0.16	0.79	0.04	1	0.841		
		Winter	0b	-	-	0	-		
	Imperforate	Intercept	0.92	0.42	47.98	1	0.028		
		Autumn	1.14	0.60	3.59	1	0.058		
		Spring	-0.57	0.53	1.19	1	0.276		
		Summer	-1.57	0.54	8.45	1	0.004		
		Winter	0b	-	-	0	-		
	Perforate	Intercept	0.00	0.50	0.00	1	1.000		
		Autumn	1.25	0.68	3.38	1	0.066		
		Spring	0.21	0.60	0.13	1	0.723		
		Summer	-0.33	0.59	0.31	1	0.576		
		Winter	0b	-	-	0	-		
	Pregnant	Intercept	-18.40	0.35	2.744.566	1	<0.001*		
		Autumn	17.30	0.89	378.83	1	<0.001*		
		Spring	16.26	0.83	387.49	1	<0.001*		
		Summer	17.66	0.00	-	1	-		
		Winter	0b	-	-	0	-		
	Lactating	Intercept	-2.08	1.06	3.84	1	0.050*		
		Autumn	2.49	1.18	4.40	1	0.036*		
		Spring	-0.06	1.30	0.00	1	0.963		
		Summer	2.00	1.10	3.30	1	0.069		
		Winter	0b	-	-	0	-		
a The reference category is:Semi-scrotal									
b This parameter is set to zero because it is redundant.									
<i>Rhabdomys dilectus</i>	Abdominal	Intercept	2.16	0.61	12.54	1	<0.001*		
		Autumn	-1.01	0.69	2.16	1	0.142		
		Spring	-2.06	0.75	7.57	1	0.006*		
		Summer	-1.24	0.85	2.14	1	0.143		
		Winter	0b	-	-	0	-		
	Scrotal	Intercept	1.20	0.66	3.35	1	0.067		
		Autumn	-3.77	1.23	9.41	1	0.002*		
		Spring	-2.41	0.93	6.69	1	0.010*		

Species	Reproductive status	Seasons	B	Std. Error	Wald	DF	P-value
	Imperforate	Summer	-2.59	1.30	3.99	1	0.046*
		Winter	0b	-	-	0	-
		Intercept	1.79	0.62	8.26	1	0.004*
		Autumn	-1.47	0.72	4.12	1	0.042*
		Spring	-3.00	0.91	10.92	1	<0.001*
	Perforate	Summer	-1.79	0.94	3.61	1	0.057*
		Winter	0b	-	-	0	-
		Intercept	1.99	0.62	10.48	1	0.001*
		Autumn	-1.56	0.71	4.82	1	0.028*
		Spring	-1.81	0.75	5.83	1	0.016*
	Pregnant	Summer	-0.89	0.84	1.12	1	0.290
		Winter	0b	-	-	0	-
		Intercept	-18.26	0.71	667.02	1	<0.001*
		Autumn	-1.14	4.519.666	0.00	1	1.000
		Spring	17.57	0.89	385.84	1	<0.001*
	Lactating	Summer	18.26	0.00	-	1	-
		Winter	0b	-	-	0	-
		Intercept	-17.90	4.438.060	0.00	1	0.997
		Autumn	17.28	4.438.060	0.00	1	0.997
		Spring	16.98	4.438.060	0.00	1	0.997
		Summer	17.20	4.438.061	0.00	1	0.997
		Winter	0b	-	-	0	-

a The reference category is: Semi-scrotal

b This parameter is set to zero because it is redundant.

c Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 35: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of seasons on small mammal sex in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Sex	Seasons	B	Std. Error	Wald	DF	P-value
<i>Aethomys ineptus</i>	Females	Intercept	0.41	0.91	0.20	1	0.657
		Winter	0b	-	-	0	-
		Spring	0.15	1.11	0.02	1	0.889
		Summer	-0.41	1.11	0.13	1	0.715
		Autumn	-0.41	1.35	0.09	1	0.765

a The reference category is: Males

b This parameter is set to zero because it is redundant.

<i>Crocidura flavescens</i>	Females	Intercept	-0.81	0.60	1.82	1	0.177
		Winter	0b	-	-	0	-
		Spring	1.28	0.83	2.39	1	0.122
		Summer	0.52	0.97	0.29	1	0.590
		Autumn	0.41	0.76	0.29	1	0.591

a The reference category is: Males

b This parameter is set to zero because it is redundant.

<i>Mastomys natalensis</i>	Females	Intercept	-0.03	0.26	0.02	1	0.896
		Winter	0b	-	-	0	-
		Spring	-0.05	0.33	0.02	1	0.892
		Summer	0.13	0.31	0.16	1	0.690
		Autumn	0.17	0.31	0.30	1	0.586

a The reference category is: Males

b This parameter is set to zero because it is redundant.

Appendix 35 continued...

Species	Sex	Seasons	B	Std. Error	Wald	DF	P-value
<i>Myosorex varius</i>	Females	Intercept	-0.92	0.84	1.20	1	0.273
		Winter	0b	-	-	0	-
		Spring	1.32	1.24	1.14	1	0.286
		Summer	1.10	1.03	1.13	1	0.287
		Autumn	0.73	0.94	0.61	1	0.435
a The reference category is: Males							
b This parameter is set to zero because it is redundant.							
<i>Rhabdomys dilectus</i>	Females	Intercept	0.03	0.23	0.01	1	0.910
		Winter	0b	-	-	0	-
		Spring	-0.03	0.37	0.01	1	0.945
		Summer	0.59	0.41	2.04	1	0.153
		Autumn	-0.27	0.30	0.78	1	0.378
a The reference category is: Males							
b This parameter is set to zero because it is redundant.							
* Significant differences when evaluated at α = 0.05							

Appendix 36: The outcomes from Kruskal-Wallis of species-specific tests in relation to the effect of seasons on small mammal head and body length (mm) in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Seasons	Z-test	P-value
<i>Aethomys ineptus</i>	Autumn - Winter	-3.15	0.593
	Autumn - Spring	-3.71	0.470
	Autumn - Summer	-10.10	0.052*
	Winter - Spring	0.56	0.907
	Winter - Summer	6.95	0.148
	Spring - Summer	-6.40	0.095
<i>Crocidura flavescens</i>	Winter-Autumn	7.15	0.193
	Winter-Spring	10.08	0.096
	Winter-Summer	14.43	0.046*
	Autumn-Spring	-2.93	0.595
	Autumn-Summer	-7.28	0.283
	Spring-Summer	-4.36	0.547
<i>Mastomys natalensis</i>	Winter - Autumn	15.65	0.426
	Winter - Spring	89.44	<0.001*
	Winter - Summer	118.77	<0.001*
	Autumn - Spring	-73.79	<0.001*
	Autumn - Summer	-103.12	<0.001*
	Spring - Summer	-29.33	0.082

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 37: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of grassland conditions on small mammal age, reproductive status and sex in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

	Species	B-coefficient	χ^2	DF	P-value
Age	<i>Aethomys ineptus</i>	3.5	0.93	2	0.629
	<i>Crocidura flavescens</i>	11.7	5.42	4	0.247
	<i>Mastomys natalensis</i>	47.65	21.01	6	0.002*
	<i>Myosorex varius</i>	8.823	3.575	2	0.167
	<i>Rhabdomys dilectus</i>	29.731	4.563	6	0.601
Reproductive status	<i>Aethomys ineptus</i>	23.1	6.72	12	0.875
	<i>Mastomys natalensis</i>	110.7	36.56	18	0.006*
	<i>Rhabdomys dilectus</i>	103.8	36.07	18	0.007*
Sex	<i>Aethomys ineptus</i>	6.2	2.38	2	0.304
	<i>Crocidura flavescens</i>	9.5	0.14	2	0.931
	<i>Mastomys natalensis</i>	21.5	5.67	3	0.129
	<i>Myosorex varius</i>	12	2.88	2	0.237
	<i>Rhabdomys dilectus</i>	23.4	6.12	3	0.106

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 38: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of grassland conditions on small mammal age in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Age	Grassland condition	B	Std. Error	Wald	DF	P-value
<i>Aethomys ineptus</i>	Adults	Intercept	18.75	0.73	652.82	1	<0.001*
		Primary-Good	-16.18	0.00	-	1	-
		Primary-Intermediate	0.00	0.00	-	1	-
		Secondary-Good	0b	-	-	0	-
a The reference category is: Sub-adults							
b This parameter is set to zero because it is redundant.							
<i>Crocidura flavescens</i>	Adults	Intercept	2.77	1.03	7.24	1	0.007*
		Primary-Good	-0.10	1.26	0.01	1	0.938
		Secondary-Good	17.29	7.45	0.00	1	0.998
		Secondary-Intermediate	0c	-	-	0	-
	Juveniles	Intercept	-18.40	7.45	0.00	1	0.998
		Primary-Good	-0.09	10.44	0.00	1	1.000
		Secondary-Good	37.07	0.00	-	1	-
		Secondary-Intermediate	0c	-	-	0	-

a The reference category is: Sub-adults

b Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.

c This parameter is set to zero because it is redundant.

<i>Mastomys natalensis</i>	Adults	Intercept	0.88	0.22	15.92	1	<0.001*
		Primary-Good	0.00	0.27	0.00	1	0.987
		Primary-Intermediate	18.86	9.22	0.00	1	0.998
		Secondary-Good	1.44	0.48	8.92	1	0.003*

Appendix 38 continued...

Species	Age	Grassland condition	B	Std. Error	Wald	DF	P-value
		Secondary-Intermediate	0c	-	-	0	-
	Juveniles	Intercept	-1.29	0.40	10.40	1	0.001*
		Primary-Good	0.65	0.45	2.10	1	0.147
		Primary-Intermediate	0.54	0.00	-	1	-
		Secondary-Good	1.11	0.73	2.32	1	0.127
		Secondary-Intermediate	0c	-	-	0	-

a The reference category is: Sub-adults

b Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.

c This parameter is set to zero because it is redundant.

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 39: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of grassland condition on small mammal reproductive status in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Reproductive status	Grassland condition	B	Std. Error	Wald	DF	P-value
<i>Aethomys ineptus</i>	Abdominal	Intercept	0.00	0.58	0.00	1	1.000
		Primary-Good	0.00	0.00	-	1	-
		Primary-Intermediate	0.00	8.39	0.00	1	1.000
		Secondary-Intermediate	0c	-	-	0	-
	Scrotal	Intercept	-1.79	1.08	2.75	1	0.097
		Primary-Good	0.00	0.00	-	1	-
		Primary-Intermediate	0.00	0.00	-	1	-
		Secondary-Intermediate	0c	-	-	0	-
	Imperforate	Intercept	18.84	6.03	0.00	1	0.998
		Primary-Good	-19.24	6.03	0.00	1	0.997
		Primary-Intermediate	-19.02	10.72	0.00	1	0.999
		Secondary-Intermediate	0c	-	-	0	-
	Perforate	Intercept	0.15	0.58	0.07	1	0.789
		Primary-Good	-0.15	0.00	-	1	-
		Primary-Intermediate	17.36	5.49	0.00	1	0.997
		Secondary-Intermediate	0c	-	-	0	-
	Pregnant	Intercept	-1.79	1.08	2.75	1	0.097
		Primary-Good	0.00	0.00	-	1	-
		Primary-Intermediate	0.00	0.00	-	1	-
		Secondary-Intermediate	0c	-	-	0	-
	Lactating	Intercept	-0.41	0.65	0.40	1	0.530
		Primary-Good	0.00	0.00	-	1	-
		Primary-Intermediate	0.00	9.51	0.00	1	1.000
		Secondary-Intermediate	0c	-	-	0	-

a The reference category is: Semi-scrotal

b Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing. c This parameter is set to zero because it is redundant.

Appendix 39 continued...

Species	Reproductive status	Grassland condition	B	Std. Error	Wald	DF	P-value
<i>Mastomys natalensis</i>	Abdominal	Intercept	0.58	0.29	4.13	1	0.042*
		Primary-Good	0.36	0.36	1.00	1	0.318
		Primary-Intermediate	0.11	1.26	0.01	1	0.930
		Secondary-Good	0.47	0.50	0.90	1	0.343
		Secondary-Intermediate	0b	-	-	0	-
	Scrotal	Intercept	-1.56	0.55	8.02	1	0.005*
		Primary-Good	1.48	0.61	5.85	1	0.016*
		Primary-Intermediate	-16.66	9.03	0.00	1	0.999
		Secondary-Good	0.58	0.87	0.44	1	0.508
		Secondary-Intermediate	0b	-	-	0	-
	Perforate	Intercept	0.23	0.31	0.58	1	0.447
		Primary-Good	-0.04	0.40	0.01	1	0.921
		Primary-Intermediate	-17.73	6.29	0.00	1	0.998
		Secondary-Good	-0.01	0.57	0.00	1	0.985
		Secondary-Intermediate	0b	-	-	0	-
	Imperforate	Intercept	0.00	0.32	0.00	1	1.000
		Primary-Good	0.83	0.40	4.36	1	0.037*
		Primary-Intermediate	-17.07	5.09	0.00	1	0.997
		Secondary-Good	0.97	0.53	3.35	1	0.067
		Secondary-Intermediate	0b	-	-	0	-
	Pregnant	Intercept	-2.94	1.03	8.24	1	0.004*
		Primary-Good	1.81	1.10	2.73	1	0.098
		Primary-Intermediate	-16.00	0.00	-	1	-
		Secondary-Good	2.66	1.16	5.25	1	0.022*
		Secondary-Intermediate	0b	-	-	0	-
	Lactating	Intercept	-1.15	0.47	6.06	1	0.014*
		Primary-Good	1.15	0.54	4.57	1	0.033*
		Primary-Intermediate	-17.01	8.77	0.00	1	0.998
		Secondary-Good	-0.93	1.16	0.64	1	0.424
		Secondary-Intermediate	0b	-	-	0	-

a The reference category is: Semi-scrotal

b This parameter is set to zero because it is redundant.

c Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.

<i>Rhabdomys dilectus</i>	Abdominal	Intercept	1.42	0.39	12.93	1	<0.001*
		Primary-Good	-0.46	0.50	0.86	1	0.354
		Primary-Intermediate	0.38	1.15	0.11	1	0.745
		Secondary-Good	-0.91	0.65	1.95	1	0.163
		Secondary-Intermediate	0b	-	-	0	-
	Scrotal	Intercept	-1.39	0.79	3.08	1	0.080
		Primary-Good	1.08	0.89	1.48	1	0.224
		Primary-Intermediate	-17.09	0.00	-	1	-
		Secondary-Good	0.29	1.14	0.06	1	0.800
		Secondary-Intermediate	0b	-	-	0	-
	Imperforate	Intercept	0.63	0.44	2.06	1	0.151
		Primary-Good	-0.45	0.56	0.64	1	0.426
		Primary-Intermediate	0.76	1.20	0.40	1	0.528
		Secondary-Good	-0.63	0.73	0.75	1	0.386
		Secondary-Intermediate	0b	-	-	0	-

Appendix 39 continued...

Species	Sex	Grassland condition	B	Std. Error	Wald	DF	P-value
		Primary-Good	-1.29	0.81	2.52	1	0.112
		Secondary-Good	-1.18	0.91	1.71	1	0.191
		Secondary-Intermediate	0b	-	-	0	-
		a The reference category is: Males					
b This parameter is set to zero because it is redundant.							
<i>Rhabdomys dilectus</i>	Females	Intercept	-0.45	0.24	3.50	1	0.062
		Primary-Good	0.57	0.30	3.69	1	0.055
		Primary-Intermediate	0.30	0.61	0.24	1	0.623
		Secondary-Good	0.90	0.40	5.01	1	0.025*
		Secondary-Intermediate	0b	-	-	0	-
		a The reference category is: Males					
b This parameter is set to zero because it is redundant.							
* Significant differences when evaluated at α = 0.05							

Appendix 41: The outcomes from Kruskal-Wallis of species-specific tests in relation to the effect of grassland conditions on small mammal head and body (mm) in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Grassland conditions	Z-test	P-value
<i>Aethomys ineptus</i>	Secondary-Intermediate - Primary-Good	5.71	0.522
	Secondary-Intermediate - Primary-Intermediate	20.00	0.107
	Primary-Good - Primary-Intermediate	-14.29	0.110
<i>Crocidura flavescens</i>	Secondary-Good - Secondary-Intermediate	-1.64	0.835
	Secondary-Good - Primary-Good	5.77	0.438
	Secondary-Intermediate - Primary-Good	4.14	0.375
<i>Mastomys natalensis</i>	Secondary-Intermediate - Primary-Good	25.30	0.085
	Secondary-Intermediate - Primary-Intermediate	38.38	0.608
	Secondary-Intermediate - Secondary-Good	72.10	<0.001*
	Primary-Good - Primary-Intermediate	-13.08	0.860
	Primary-Good - Secondary-Good	-46.80	0.006*
	Primary-Intermediate - Secondary-Good	-33.72	0.655
<i>Myosorex varius</i>	Secondary-Good - Primary-Good	6.14	0.189
	Secondary-Good - Secondary-Intermediate	-14.58	0.009*
	Primary-Good - Secondary-Intermediate	-8.44	0.089
<i>Rhabdomys dilectus</i>	Primary-Intermediate - Primary-Good	6.00	0.786
	Primary-Intermediate - Secondary-Intermediate	-40.27	0.080
	Primary-Intermediate - Secondary-Good	-41.10	0.091
	Primary-Good - Secondary-Intermediate	-34.27	0.002*
	Primary-Good - Secondary-Good	-35.10	0.010*
	Secondary-Intermediate - Secondary-Good	0.83	0.956

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 42: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of regional variation on small mammal age, reproductive status and sex in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

	Species	B-coefficient	χ^2	DF	P-value
Age	<i>Aethomys ineptus</i>	4.2	0.36	2	0.836
	<i>Crocidura flavescens</i>	14.71	10	4	0.04 *
	<i>Mastomys natalensis</i>	45.22	17.97	4	0.001 *
Reproductive status	<i>Aethomys ineptus</i>	35.4	13.34	12	0.345
	<i>Mastomys natalensis</i>	112.36	39.04	12	<0.001 *
Sex	<i>Aethomys ineptus</i>	7.89	1.73	2	0.421
	<i>Crocidura flavescens</i>	10.04	0.13	2	0.937
	<i>Mastomys natalensis</i>	17.33	1.29	2	0.526

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 43: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of regional variation on small mammal age in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Age	Regions	B	Std. Error	Wald	DF	P-value
<i>Aethomys ineptus</i>	Adults	Intercept	2.89	1.03	7.92	1	0.005*
		Kloof	0b	.	.	0	.
		Hillcrest	15.86	0.00	.	1	.
		Camperdown	-0.69	1.47	0.22	1	0.638

a The reference category is: Sub-adults

b This parameter is set to zero because it is redundant.

<i>Crocidura flavescens</i>	Adults	Intercept	19.92	5.19	0.00	1	0.997
		Kloof	0c	.	.	0	.
		Hillcrest	-0.03	6.77	0.00	1	1.000
		Camperdown	-18.62	5.19	0.00	1	0.997
	Juveniles	Intercept	-1.10	8.59	0.00	1	1.000
		Kloof	0c	.	.	0	.
		Hillcrest	17.85	9.62	0.00	1	0.999
		Camperdown	-19.08	0.00	.	1	.

a The reference category is: Sub-adults

b Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.

c This parameter is set to zero because it is redundant.

<i>Mastomys natalensis</i>	Adults	Intercept	0.82	0.20	16.95	1	<0.001*
		Kloof	0b	.	.	0	.
		Hillcrest	0.50	0.26	3.65	1	0.056*
		Camperdown	0.00	0.33	0.00	1	0.995
	Juveniles	Intercept	-0.37	0.26	1.96	1	0.161
		Kloof	0b	.	.	0	.
		Hillcrest	-0.35	0.37	0.89	1	0.344
		Camperdown	-2.03	0.78	6.74	1	0.009*

a The reference category is: Sub-adults

b This parameter is set to zero because it is redundant.

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 44: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of regional variation on small mammal reproductive status in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Reproductive status	Regions	B	Std. Error	Wald	DF	P-value
<i>Aethomys ineptus</i>	Abdominal	Intercept	-0.92	0.84	1.20	1	0.273
		Camperdown	2.02	1.43	2.00	1	0.158
		Hillcrest	17.96	4.348.363	0.00	1	0.997
		Kloof	0c	-	-	0	-
	Scrotal	Intercept	-1.61	1.10	2.16	1	0.142
		Camperdown	-12.87	1.396.601	0.00	1	0.993
		Hillcrest	-0.18	0.00	-	1	-
		Kloof	0c	-	-	0	-
	Imperforate	Intercept	-0.22	0.67	0.11	1	0.739
		Camperdown	0.22	1.57	0.02	1	0.887
		Hillcrest	0.04	7.010.157	0.00	1	1.000
		Kloof	0c	-	-	0	-
	Perforate	Intercept	0.18	0.61	0.09	1	0.763
		Camperdown	-0.18	1.54	0.01	1	0.906
		Hillcrest	-0.03	6.364.276	0.00	1	1.000
		Kloof	0c	-	-	0	-
	Pregnant	Intercept	-15.16	875.61	0.00	1	0.986
		Camperdown	15.16	875.61	0.00	1	0.986
		Hillcrest	13.37	0.00	-	1	-
		Kloof	0c	-	-	0	-
	Lactating	Intercept	-1.61	1.10	2.16	1	0.142
		Camperdown	2.71	1.59	2.90	1	0.089
		Hillcrest	1.20	7.529.995	0.00	1	1.000
		Kloof	0c	-	-	0	-

a The reference category is: Semi-scrotal

b Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.

c This parameter is set to zero because it is redundant.

<i>Mastomys natalensis</i>	Abdominal	Intercept	0.75	0.30	6.18	1	0.013*
		Camperdown	0.09	0.50	0.04	1	0.852
		Hillcrest	0.14	0.37	0.15	1	0.700
		Kloof	0b	-	-	0	-
	Scrotal	Intercept	0.17	0.34	0.26	1	0.613
		Camperdown	-0.76	0.65	1.35	1	0.245
		Hillcrest	-1.41	0.51	7.68	1	0.006*
		Kloof	0b	-	-	0	-
	Imperforate	Intercept	0.81	0.30	7.28	1	0.007*
		Camperdown	-0.06	0.50	0.02	1	0.899
		Hillcrest	-0.35	0.38	0.87	1	0.350
		Kloof	0b	-	-	0	-
	Perforate	Intercept	-0.07	0.36	0.03	1	0.857
		Camperdown	-0.05	0.60	0.01	1	0.930
		Hillcrest	0.44	0.43	1.04	1	0.308
		Kloof	0b	-	-	0	-

Appendix 44 continued...

Species	Reproductive status	Regions	<i>B</i>	Std. Error	Wald	<i>DF</i>	<i>P</i> -value
	Pregnant	Intercept	-0.47	0.40	1.36	1	0.244
		Camperdown	-20.84	0.00	.	1	.
		Hillcrest	-1.17	0.60	3.80	1	0.051
		Kloof	0b	.	.	0	.
	Lactating	Intercept	-0.21	0.37	0.31	1	0.578
		Camperdown	0.50	0.58	0.74	1	0.391
		Hillcrest	-0.92	0.52	3.14	1	0.076
		Kloof	0b	.	.	0	.

a The reference category is: Semi-scrotal

b This parameter is set to zero because it is redundant.

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 45: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of regional variation on small mammal sex in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Sex	Regions	<i>B</i>	Std. Error	Wald	<i>DF</i>	<i>P</i> -value
<i>Aethomys ineptus</i>	Females	Intercept	0.32	0.47	0.47	1	0.493
		Kloof	0b	-	-	0	-
		Hillcrest	-19.55	0.00	-	1	-
		Camperdown	0.09	0.80	0.01	1	0.913

a The reference category is: Males

b This parameter is set to zero because it is redundant.

<i>Crociodura flavescens</i>	Females	Intercept	-0.41	0.53	0.59	1	0.442
		Kloof	0b	-	-	0	-
		Hillcrest	0.24	0.67	0.13	1	0.721
		Camperdown	0.12	0.76	0.02	1	0.876

a The reference category is: Males

b This parameter is set to zero because it is redundant.

<i>Mastomys natalensis</i>	Females	Intercept	0.18	0.17	1.18	1	0.278
		Kloof	0b	-	-	0	-
		Hillcrest	-0.24	0.21	1.21	1	0.272
		Camperdown	-0.07	0.29	0.07	1	0.796

a The reference category is: Males

b This parameter is set to zero because it is redundant.

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 46: The outcomes from Kruskal-Wallis of species-specific tests in relation to the effect of regional variation on small mammal head and body (mm) in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Regions	Z-test	P-value
<i>Aethomys ineptus</i>	Hillcrest - Kloof	-6.53	0.469
	Hillcrest - Camperdown	16.10	0.080
	Kloof - Camperdown	9.57	0.005*
<i>Crocidura flavescens</i>	Camperdown-Hillcrest	-7.51	0.148
	Camperdown-Kloof	-17.76	0.002*
	Hillcrest-Kloof	-10.25	0.044*
<i>Mastomys natalensis</i>	Camperdown - Kloof	-2.82	0.878
	Camperdown - Hillcrest	-14.81	0.387
	Kloof - Hillcrest	11.99	0.380

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 47: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of habitat management regimes (treatments) on small mammal age, reproductive status and sex in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Metrics	Species	B-coefficient	χ^2	DF	P-value
Age	<i>Aethomys ineptus</i>	4.2	0.36	2	0.836
	<i>Crocidura flavescens</i>	15.94	11.26	4	0.024 *
	<i>Mastomys natalensis</i>	39.16	12.12	4	0.017 *
Reproductive status	<i>Rhabdomys dilectus</i>	23.56	5.5	4	0.240
	<i>Aethomys ineptus</i>	35.4	13.34	12	0.345
	<i>Mastomys natalensis</i>	97.65	24.61	12	0.017 *
	<i>Rhabdomys dilectus</i>	76.81	29.38	12	0.003 *
Sex	<i>Aethomys ineptus</i>	7.89	1.73	2	0.421
	<i>Crocidura flavescens</i>	11.07	1.33	2	0.514
	<i>Mastomys natalensis</i>	16.08	0.08	2	0.962
	<i>Rhabdomys dilectus</i>	14.14	1.44	2	0.487

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 48: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of habitat management regimes (treatments) on small mammal age in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Age	Treatments	B	Std. Error	Wald	DF	P-value
<i>Aethomys ineptus</i>	Adults	Intercept	18.75	1.03	333.02	1	<0.001*
		Inside	-15.86	0.00	.	1	.
		Outside	0b	.	.	0	.
		Control	-16.55	1.47	126.44	1	<0.001*
a The reference category is: Sub-adults							
b This parameter is set to zero because it is redundant.							
<i>Crocidura flavescens</i>	Adults	Intercept	20.36	6.55	0.00	1	0.998
		Inside	0.05	8.42	0.00	1	1.000
		Outside	0c	.	.	0	.
		Control	-19.06	6.55	0.00	1	0.998
	Juveniles	Intercept	17.87	6.55	0.00	1	0.998
		Inside	-18.97	12.45	0.00	1	0.999
		Outside	0c	.	.	0	.
		Control	-37.90	0.00	.	1	.

a The reference category is: Sub-adults

b Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.

c This parameter is set to zero because it is redundant.

<i>Mastomys natalensis</i>	Adults	Intercept	1.46	0.24	36.06	1	<0.001*
		Inside	-0.48	0.29	2.78	1	0.095
		Outside	0b	.	.	0	.
		Control	-0.63	0.35	3.24	1	0.072
	Juveniles	Intercept	-0.21	0.33	0.42	1	0.517
		Inside	-0.48	0.40	1.47	1	0.226
		Outside	0b	.	.	0	.
		Control	-2.19	0.81	7.33	1	0.007*

a The reference category is: Sub-adults

b This parameter is set to zero because it is redundant.

<i>Rhabdomys dilectus</i>	Adults	Intercept	0.58	0.42	1.91	1	0.167
		Inside	1.22	0.87	1.96	1	0.162
		Outside	0b	.	.	0	.
		Control	0.86	0.45	3.62	1	0.057*
	Juveniles	Intercept	-1.50	0.78	3.70	1	0.054*
		Inside	-17.68	0.00	.	1	.
		Outside	0b	.	.	0	.
		Control	0.09	0.86	0.01	1	0.914

a The reference category is: Sub-adults

b This parameter is set to zero because it is redundant.

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 49: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of habitat management regimes (treatments) on small mammal reproductive status in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Reproductive status	Treatments	B	Std. Error	Wald	DF	P-value
<i>Aethomys ineptus</i>	Abdominal	Intercept	17.04	2.82	0.00	1	0.995
		Inside	-17.96	2.82	0.00	1	0.995
		Outside	0c	-	-	0	-
		Control	-15.94	2.82	0.00	1	0.995
	Scrotal	Intercept	-1.79	1.10	2.68	1	0.102
		Inside	0.18	0.00	-	1	-
		Outside	0c	-	-	0	-
		Control	-12.69	1.40	0.00	1	0.993
	Imperforate	Intercept	-0.18	0.67	0.07	1	0.786
		Inside	-0.04	0.00	-	1	-
		Outside	0c	-	-	0	-
		Control	0.18	1.57	0.01	1	0.907
	Perforate	Intercept	0.15	0.61	0.07	1	0.799
		Inside	0.03	0.00	-	1	-
		Outside	0c	-	-	0	-
		Control	-0.15	1.54	0.01	1	0.920
	Pregnant	Intercept	-1.79	12.61	0.00	1	1.000
		Inside	-13.37	12.65	0.00	1	0.999
		Outside	0c	-	-	0	-
		Control	1.79	12.61	0.00	1	1.000
	Lactating	Intercept	-0.41	6.76	0.00	1	1.000
		Inside	-1.20	6.76	0.00	1	1.000
		Outside	0c	-	-	0	-
		Control	1.50	6.76	0.00	1	1.000

a The reference category is: Semi-scrotal

b Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.

c This parameter is set to zero because it is redundant.

<i>Mastomys natalensis</i>	Abdominal	Intercept	1.12	0.31	13.30	1	<0.001*
		Inside	-0.41	0.37	1.23	1	0.268
Appendix 49 continued...							

Species	Reproductive status	Treatments	B	Std. Error	Wald	DF	P-value
		Outside	0b	-	-	0	-
		Control	-0.28	0.50	0.30	1	0.585
	Scrotal	Intercept	-0.69	0.46	2.24	1	0.134
		Inside	0.24	0.54	0.20	1	0.656
		Outside	0b	-	-	0	-
		Control	0.11	0.73	0.02	1	0.884
	Imperforate	Intercept	0.94	0.32	8.99	1	0.003*
		Inside	-0.55	0.39	2.01	1	0.156
		Outside	0b	-	-	0	-
		Control	-0.20	0.51	0.15	1	0.700
	Perforate	Intercept	0.00	0.38	0.00	1	1.000
		Inside	0.33	0.44	0.57	1	0.452
		Outside	0b	-	-	0	-
		Control	-0.12	0.62	0.04	1	0.848
	Pregnant	Intercept	-0.56	0.44	1.59	1	0.207
		Inside	-0.86	0.59	2.09	1	0.148
		Outside	0b	-	-	0	-
		Control	-20.75	0.00	-	1	-
	Lactating	Intercept	-0.85	0.49	3.02	1	0.082
		Inside	0.18	0.57	0.10	1	0.748
		Outside	0b	-	-	0	-
		Control	1.14	0.66	2.98	1	0.084

a The reference category is:Semi-scrotal

b This parameter is set to zero because it is redundant.

<i>Rhabdomys dilectus</i>	Abdominal	Intercept	2.71	1.03	6.88	1	0.009*
		Inside	-2.71	1.32	4.23	1	0.040*
		Outside	0b	-	-	0	-
		Control	-1.72	1.06	2.64	1	0.104
		Intercept	-16.91	4.70	0.00	1	0.997
		Inside	16.51	4.70	0.00	1	0.997
		Outside	0b	-	-	0	-
		Control	16.22	4.70	0.00	1	0.997
	Imperforate	Intercept	2.20	1.05	4.35	1	0.037*
		Inside	-2.60	1.39	3.48	1	0.062
		Outside	0b	-	-	0	-
		Control	-1.99	1.09	3.35	1	0.067
	Perforate	Intercept	0.00	1.41	0.00	1	1.000
		Inside	0.29	1.61	0.03	1	0.858
		Outside	0b	-	-	0	-
		Control	0.85	1.43	0.35	1	0.552
	Pregnant	Intercept	0.00	1.41	0.00	1	1.000
		Inside	-19.08	8.03	0.00	1	0.998
		Outside	0b	-	-	0	-
		Control	-1.18	1.47	0.64	1	0.423
	Lactating	Intercept	-17.05	0.34	2.520.385	1	<0.001*
		Inside	-1.66	6.68	0.00	1	1.000
		Outside	0b	-	-	0	-
		Control	16.36	0.00	-	1	-

a The reference category is: Semi-scrotal

b This parameter is set to zero because it is redundant.

c Floating point overflow occurred while computing this statistic. Its value is therefore set to system missing.

* Significant differences when evaluated at $\alpha = 0.05$

Appendix 50: The outcomes of multinomial logistic regression for species-specific tests in relation to the effect of habitat management regimes (treatments) on small mammal sex in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Sex	Treatments	B	Std. Error	Wald	DF	P-value
<i>Aethomys ineptus</i>	Females	Intercept	-17.24	0.47	1.38	1	<0.001*
		Inside	17.55	0.00	-	1	-
		Outside	0b	-	-	0	-
		Control	17.64	0.80	491.93	1	<0.001*
a The reference category is: Males							
b This parameter is set to zero because it is redundant.							
<i>Crocidura flavescens</i>	Females	Intercept	-0.81	0.60	1.82	1	0.177
		Inside	0.81	0.72	1.28	1	0.258
		Outside	0b	-	-	0	-
		Control	0.52	0.81	0.42	1	0.517
a The reference category is: Males							
b This parameter is set to zero because it is redundant.							
<i>Mastomys natalensis</i>	Females	Intercept	0.03	0.18	0.03	1	0.860
		Inside	0.01	0.22	0.00	1	0.963
		Outside	0b	-	-	0	-
		Control	0.08	0.29	0.07	1	0.792
a The reference category is: Males							
b This parameter is set to zero because it is redundant.							
<i>Rhabdomys dilectus</i>	Females	Intercept	-0.38	0.39	0.92	1	0.339
		Inside	0.09	0.67	0.02	1	0.896
		Outside	0b	-	-	0	-
		Control	0.44	0.41	1.12	1	0.291
a The reference category is: Males							
b This parameter is set to zero because it is redundant.							
* Significant differences when evaluated at $\alpha = 0.05$							

Appendix 51: The outcomes from Kruskal-Wallis of species-specific tests in relation to the effect of habitat management regimes (treatments) on small mammal head and body length (mm) in KwaZulu-Natal Sandstone Sourveld Grasslands in eThekweni (July 2018 to June 2019).

Species	Treatments	H-test	P-value
<i>Aethomys ineptus</i>	Outside - Inside	6.53	0.469
	Outside - Control	16.10	0.080
	Inside - Control	9.57	0.005*
<i>Crocidura flavescens</i>	Control-Outside	-5.31	0.371
	Control-Inside	-14.53	0.005*
	Outside-Inside	9.21	0.079
<i>Mastomys natalensis</i>	Inside - Control	7.00	0.681
	Inside - Outside	-49.49	<0.001*
	Control - Outside	-42.49	0.023*
<i>Rhabdomys dilectus</i>	Outside - Control	11.24	0.470
	Outside - Inside	65.46	0.009*
	Control - Inside	-54.23	0.010*

* Significant differences when evaluated at $\alpha = 0.05$