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Small mammal responses to Scarp Forest Restoration in the Maputoland-Pondoland-Albany Hotspot, South Africa

by

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Abstract

Restoration ecology is a relatively new field. Although a range of attributes have been used to assess restoration success, they have not been standardised across studies. Recently, three main ecological attributes have been identified as key measures to standardise the assessment of restoration success: species diversity, vegetation structure and ecological processes. However few studies have combined more than two of these ecological attributes when assessing restoration success. The aim of this study was to apply these three ecological attributes to determine whether Scarp Forest restoration has been successful from the perspective of small mammals at the Buffelsdraai Landfill Site, Durban, South Africa. I assessed the response of small mammals to Scarp Forest restoration at 2, 4 and 6 year post-restoration periods. I surveyed small mammals every three months for one year in three restoration treatments (2010, 2012 and 2014 restored), as well as in surrounding sugarcane and riverine forest sites. At each site I measured the vegetation structure and small mammal diversity. Additionally, I conducted stable isotope analysis on vegetation and invertebrate samples to compile a baseline database of potential prey, and compared these data with the stable isotope composition of hair and tissue samples collected from rodents and shrews to analyse the trophic structure of the small mammal assemblages. In support for the prediction that vegetation structure should increase in complexity at restored sites, tree species richness, density and height were higher at the 2010 restored than more recently restored sites; and grass height and percentage cover were highest at 2012 restored sites. Except, forb and grass species richness were higher at newly restored sites. Second, rodent abundance was higher at the 2010 restored sites than the 2012 and 2014 restored sites and sugarcane sites. However, shrew abundance and species richness were not significantly different among the study sites. Third, carbon and nitrogen isotopic composition of rodent hairs suggest that these species utilised resources associated with the 2010 restored sites rather than those associated with recently restored sites, sugarcane sites and forests. Further, the stable isotope ratios of carbon and nitrogen in *Mastomys natalensis*' tissues showed that these rodents predominantly utilised resources associated with the 2010 restored sites irrespective of the tissue that was analysed. Conversely, carbon and nitrogen isotopic composition of shrew hairs suggest that these species foraged at the sites where they were captured. Taken together, my results suggest that at Buffelsdraai, the restoration efforts have ensured progressive succession in the scarp forest after 10 years, at least from the perspective of most small mammals.

Preface

The field and lab work described in this dissertation was carried out in the School of Life Sciences, University of KwaZulu-Natal, Westville Campus, from July 2015 to July 2017, under the supervision of Dr. Dalene Vosloo and Prof. M.C. Schoeman.

This study represents original work by the author and has not otherwise been submitted in any form for any degree or diploma to any tertiary institution. Where use has been made of the work of others it is duly acknowledged in the text.

Declaration

I, Angelique Tiara Lazarus declare that

1. The research reported in this dissertation, except where otherwise indicated, is my original research.

2. This dissertation has not been submitted for any degree or examination at any other university.

3. This dissertation does not contain other persons' data, pictures, graphs or other information, unless specifically acknowledged being sources from other persons.

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CHAPTER 1

INTRODUCTION

1.1.Human land use and restoration ecology

Although in recent years the growth rates of world agricultural production and crop yields has slowed, species richness has declined by approximately 8.1% on average globally, mainly as a result of large increases in croplands and pastures (McGill, 2015). At the same time, urban areas are expanding rapidly worldwide (Grimm *et al.*, 2008). These phenomena are positively related to an exponentially increasing human population (Kowarik, 2011). Human land use affects ecosystems in different ways, for example, removal or conversion of vegetation, pollution of air, soil and water, and habitat fragmentation (Grimm *et al*., 2008). Agricultural and urban lands are inhabited by fewer species, fewer organisms, and smaller organisms than undisturbed areas (Newbold *et al.*, 2015). Currently, anthropogenically modified land occupies the greatest proportion of the Earth's terrestrial surface (Grimm *et al.*, 2008). On the other hand, areas allowed to recover after human land use - like reforested lands - can compare in terms of biodiversity as well as areas that hadn't been touched (Newbold *et al.*, 2015).

Restoration ecology aims to understand the processes necessary to rehabilitate ecosystems that have been degraded by anthropogenic activities (Society for Ecological Restoration International Science and Policy Working Group, 2004). Globally, it is an increasingly important conservation priority (Holl *et al.*, 2000). One of the most important goals of restoration ecology is to create self-sustainable ecosystems that are resilient to disturbance. Restoration projects are often implemented at broad spatial scales because of large degraded areas (Lamb *et al.*, 2005), the complexity of the drivers of degradation (Holl *et al.*, 2000), and the relevant ecological processes that occur at a landscape scale (Kalies *et al.*, 2012). However, at these spatial scales it is also challenging to accurately quantify restoration ecology success (Bell *et al*., 2008).

To quantify restoration success, at least three major ecological attributes need to be measured (Ruiz-Jaen & Aide, 2005). First, vegetation structure such as vegetation cover, plant density and biomass (Walters, 2000; Wilkins *et al.*, 2003), which predicts plant succession (Wilkins *et al.*, 2003), Second, species diversity of the faunal taxa, including abundance and species richness (McCoy & Mushinsky, 2002). Third, ecosystem processes

involving the focal taxa and the restored vegetation, for example herbivory or insectivory (Rhoades *et al*., 1998; Ruiz-Jaen & Aide, 2005). Although studies have assessed one or two of these ecosystem attributes, to the best of my knowledge, no study has applied all three.

1.2. Using stable isotope analysis to assess diet consumption of animal assemblages

Isotopes are forms of chemical elements that have the same number of protons and electrons, yet different masses (Dawson *et al*., 2002; Crawford *et al*., 2008). Stable isotopes are isotopes that do not decay and therefore differ from radioisotopes (Young *et al*., 2010). Because stable isotopes have different masses, they react differentially in environmental and physiological processes (Crawford *et al*., 2008), and differ in abundance (Post, 2002). Isotopic fractionation is a process that results in the lighter, more common isotopes proceeding through chemical or physical reactions, whereas the heavier, less common isotopes remain behind (O'Brien *et al*., 2000), results in predictable changes in isotopic ratios. These changes can be measured in vegetation and/or animal tissue samples using an isotopic mass spectrometer (McKinney *et al.*, 1950).

Understanding animal foraging preferences in degraded versus restored landscapes is key in quantifying restoration success (Hobson, 1999; West *et al*., 2006; Cernusak & Hutley, 2011). One way to estimate foraging patterns is to investigate the variation in the ratios of the stable isotopes C, H, N, O and S in the consumer's tissue. These stable isotope ratios reflect the food they consumed (Petersen & Fry, 1987; Samelius *et al*., 2007; Crawford *et al*., 2008).

Different animal tissues have different turnover rates (Crawford *et al*., 2008), and synthesise food intake at different temporal scales (Bearhop *et al*., 2003; Rubenstein & Hobson, 2004). Thus, isotope analysis of different tissues provides different spatial and temporal dimensions of the animal's diet and movement from a single sampling event. This analysis, in turn, can indicate whether there has been a strengthening of ecological processes across a restored landscape (Crawford *et al*., 2008).

1.2.1 Carbon and Nitrogen distributions in nature

Elements play various roles in physical and biochemical processes, hence isotopes can be utilised to answer a suite of ecological questions (Fry, 2006). Stable isotopes most commonly used in ecological studies are carbon (C), nitrogen (N), hydrogen (H), oxygen

(O), sulphur (S) and strontium (Sr). C and N isotopes are the main elements used in diet studies (Petersen & Fry, 1987; Hobson, 1999; Kelly, 2000).

Stable C isotope ratios are essentially tracers of various C sources in the food web. In terrestrial ecosystems, photosynthetic metabolism of plants mediates variation in stable C isotopes ($\delta^{13}C$) (Lajtha & Michener, 1994). For example, $\delta^{13}C$ values decrease with increased latitude (Rubenstein & Hobson, 2004; Fry, 2006), because the proportions of C_3 and C_4 plants change (Abelson & Hoering, 1961). Stable N isotope ratios ($\delta^{15}N$) are used to reconstruct food webs, partly because they increase at each trophic level (Fry, 2006); $\delta^{15}N$ will usually increase $2^0/_{00}$ - $4^0/_{00}$ at each trophic level (Kelly, 2000). This predictable pattern means that hypotheses regarding the diet of consumers and resource partitioning can be tested (Caut *et al.*, 2009).

1.2.2. Ecological applications of stable isotope analysis

The diets of vertebrate species often show seasonal or long term variation (Dalerum & Angerbjörn, 2005). Thus, to understand population dynamics of species requires a good understanding of seasonal variation in food sources (Reid *et al.*, 1997).

Traditional methods of analysing diet include stomach content analysis, identification of faeces content or direct observations of feeding habits (Monadjem *et al.*,1997). However, these methods have limitations including bias towards less digestible materials in stomach content analysis, and untraceable materials in scat. (Crawford *et al.*, 2008; Codron *et al.*, 2015). Traditional methods have collected valuable data. However, some are invasive and may introduce bias in the results. For example, scat analysis does not include all material digested by individuals, only remnants of material are identified which requires a great deal of skill, and is time consuming (Soininen *et al.*, 2009).

Stable isotope analysis may be a better method to analyse trophic niches of species because stable isotopes in animal tissues reflect the average dietary record for the consumer, and eliminate common problems associated with traditional dietary studies (DeNiro & Epstein, 1981). There are three approaches to the use of stable isotopes in understanding temporal diet variation. First, samples from the same tissue that has been sampled over time can be compared to assess long and short term variation. Second, different sections of tissue with progressive growth can be compared because they maintain isotopic values in chronological order (Hobson, 1993; Bearhop *et al.*, 2003; Schwertl *et al.*, 2003). Third, tissues with different turnover rates can be compared to investigate diet over different time periods

because each tissue will integrate elements over time scales specific to its metabolic rate. The most common tissues for such studies are blood, liver, muscle and bone (Howland *et al.*, 2003). However, there are few studies that have applied this technique to investigate temporal change in diets.

1.3. Small mammals as biological indicators of reforestation

Rodents and shrews are considered valuable indicators of habitat integrity (Avenant, 2005; Kryštufek *et al.,* 2008). They fulfil key ecological roles, for example linking primary producers and secondary consumers through prey and predator relations (Perrin & Bodbijl, 2001; Skinner & Chimimba, 2005; Avenant & Cavallini, 2007). In fact, they are important prey for predators such as medium sized mammals and birds of prey (Avenant, 2005; Block *et al.,* 2005; Kalies *et al.,* 2012). Further, they alter the amount of biomass in ecosystems (Avenant *et al.,* 2008; Habtamu & Bekele, 2013) through consumption of vegetation (Keesing, 2000), and are commonly referred to as ecosystem engineers (Avenant & Cavallini, 2007). Small mammals are important for nutrient recycling as they process vegetation, disperse spores and seeds (Kalies *et al.,* 2012), and aerate soils while digging (Avenant & Cavallini, 2007). Further, small mammals respond quickly to disturbance and therefore are good indicators of changes in environments or vegetation structure (Avenant, 2011: Kalies *et al.,* 2012). Previous studies that have investigated restoration success focused on changes in abundance, density and diversity of small mammals. For example, Converse *et al.,* (2006) found that restoration of pine forests resulted in an increase in small mammal densities. Ground cover (shrub vegetation and woody cover) were the most important predictors of small mammal densities at restored sites. Further, there were species-specific responses to changes at the restored forest patches. Vegetation structure often has meaningful impact on South African rodent community structure. For example, Bond *et al.,* (1980) suggested that foliage profiles, ground cover estimates and a horizontal diversity index were better descriptors of rodent habitat than floristic descriptions. Ferreira & Van Aarde (1996) found that small mammal community composition was best explained by species-specific habitat preferences, e.g. Otomys irrotatus was captured in sites with tall long grass, while Mus minutoides avoided such sites (Armstrong and Hensbergen, 1996). Ferreira & Van Aarde (1999) also found that highest vegetation, shrub and herb height, litter depth, number of shrub stems, woody profile index and average shrub height significantly influenced species densities. Similarly, Els & Kerley (1996) found horizontal and vertical foliage density, and shrub canopy cover were best small mammal indicators. Kerley (1992)

found a positive association between plant and rock cover, specifically plant cover at an intermediate height (40 – 60cm) and small mammal diversity (Kerley, 1992). Avenant & Cavallini (2007), split species into ecological groups according to their grazing value, specifically decreaser species or increaser species (Van Oudtshoorn, 1994).

Many forest animals including small mammals, rely on the resources provided by particular structural features of forests (Grove, 2002), partly because of the complex structural attributes of forests which provide a range of foraging and sheltering options (Walters, 2000; Wilkins *et al.,* 2003). The rate at which reforestation returns structural complexity to land previously used for agriculture, is an important determinant of the value of reforested sites to wildlife. Therefore, it is expected that newly restored sites that represent recently planted trees, will support a lower richness and abundance of forest wildlife than restoration plants in long restored sites. Similarly, Ferreira & van Aarde (1996), found lower small mammal diversity in younger restored sites, compared to older restored sites. However, most restoration studies focused solely on small mammal diversity, they did not consider how small mammals utilised the restored landscape as consumers (Hurst *et al.,* 2013; Lamani *et al.,* 2014).

Isotopic gradients are well characterised in terrestrial ecosystems (Hobson, 1999). Stable isotope analysis has been used to quantify the diet of invasive small mammal species, particularly on islands (Hobson *et al.*, 1999; Drever *et al*., 2000; Major *et al*., 2007). For example, on islands the diet of introduced Norway rats, *Rattus norvegicus,* included a high proportion of seabirds (Hobson *et al.*, 1999). The dietary niche of this species was correlated to the size of the island and weather conditions (Stapp & Polis, 2003). Yet Major *et al*., (2007) found that Norway rats had variable diets on islands, and were able to survive when their preferred diet resources declined. Additionally, dietary niche breadth of small mammals has been quantified using stable isotope analysis. For example, based on carbon and nitrogen isotope ratios, individual specialisation was evident in the dusky-footed woodrat, *Neotoma fuscipes* (McEachern *et al*., 2006). Furthermore, Codron *et al*., (2015) found that synoptic rodent species in African savanna habitats occupied isotopically distinct trophic niches, and suggested that competitive exclusion was the driver of these dietary patterns.

Previous studies on African small mammal assemblages have demonstrated the significance of such an approach (Symes *et al.,* 2013, Codron *et al.,* 2015, Robb *et al.,* 2016). Stable isotope analysis is particularly useful because stable carbon and nitrogen isotopes separate most of the potential food sources of a consumer according to their affiliation to either a C_3 or C_4 food web (Bearhop *et al.,* 2003), and their enrichment in $\delta^{15}N$ and $\delta^{14}N$ (Stapp & Polis,

2003). Additionally, stable isotopes can be used as an endogenous marker, because isotopic composition integrate over the period of the tissue growth (Voigt *et al.*, 2003). It is therefore possible to obtain insights into temporal aspects of their feeding behaviour and movements.

Thus, carbon and nitrogen isotopic ratios are ideal to investigate the diets of small mammals in restored landscapes, specifically whether there is a strengthening of ecological processes over time.

1.4. Study aims, objectives and predictions

The aim of this study was to assess the success of scarp forest restoration, from the perspective of small mammals, at the eThekwini Municipality Community Reforestation sites, Buffelsdraai Landfill site.

My objectives were to:

1. Measure the vegetation structure in plots.
2. Survey small mammals during the wet and dry seasons for one year in: three treatments representing different starting times of scarp forest restoration (2, 4 and 6 year periods); sugarcane representing the original land cover prior to forest restoration; and riverine forest representing a forest habitat comparison.
3. Quantify the observed and expected richness of rodents and shrews at each study site using species richness indices (Gotelli & Colwell, 2001), and compare species richness among study sites using sample-based rarefaction curves (Gotelli & Colwell, 2001).
4. Quantify the diet of small mammal assemblages within and across study sites, using stable isotope analysis of hair and tissue samples collected from rodents and shrews. I focused on C and N stable isotopes because differences in carbon isotopes can be used to assess foraging location (Hobson, 1999; Rubenstein & Hobson, 2004), while differences in nitrogen isotopes are used to determine trophic level and diet composition (Dahl *et al.*, 2003; Quilfeldt *et al.*, 2005). C and N isotope analysis is helpful when trying to understand what is integrated within the tissues of an animal from its diet.

I tested the hypothesis that scarp forest restoration will result in more complex vegetation structure, increased small mammal diversity, and strengthening of ecological processes,

specifically the trophic links between the small mammals as consumers and the restored vegetation.

I predicted that:

1. Complexity of vegetation structure will increase among restored sites.
2. Rodent and shrew species will exhibit species-specific responses to increased levels of forest restoration; more specifically relative abundance and species richness of generalist species such as *Mastomys natalensis* and *Suncus lixus* that thrive on disturbance should be higher than those of species such as *Aethomys ineptus* and *Mus minutoides* that are more specialist and sensitive to disturbance.
3. Small mammal abundance and species richness will increase with increased age of forest restoration.
4. The C/N isotope ratios of small mammal assemblages will be more closely associated with older forest restored sites than young restored and sugarcane sites.

CHAPTER 2

METHODS

2.1 Study Site

I sampled small mammals for three weeks during two wet seasons (November 2015 and February 2016) and two dry seasons (May 2016 and July 2016) at the eThekwini Municipality Community Reforestation Site, Buffelsdraai Landfill site, in KwaZulu-Natal province, South Africa (29.6911° S, 31.0500° E). The study site covered an area of 520 ha, at an altitude of ca. 231 m. The climate was warm and temperate with an annual mean temperature of 22.5° C and an annual mean rainfall of 110 mm, a mean temperature of 24.4° C and a mean rainfall of 113 mm during the wet season, and a mean temperature of 19.3° C and a mean rainfall of 44 mm during the dry season (South African Weather Service 2015; www.weathersa.co.za/climate).

The vegetation at Buffelsdraai Landfill site comprised originally sugarcane plantations and patches of riparian forest along drainage lines. Replanting of indigenous trees began in 2009 (Fig. 1). At the time of sampling, the bufferzone was characterised by a mosaic of vegetation types. Most sites had a dominant grassy ground layer (*Panicum* and *Themeda* species) and an upper layer of woody plants that included *Acacia karoo, A. nilotica, A. sieberiana, Erythrina lysistemon, Millettia grandis and Syzygium cordatum.* These indigenous species were chosen for replanting because they are characteristic species present in a Scarp Forest (Mugwedi *et al.,* 2017). Planting commenced in 2009 and ended in 2016. Approximately 51 indigenous tree species were planted. At least seven species were randomly selected and planted at sites during each planting event – these included *Acacia species, Erythrina lysistemon, Dalbergia obovata, Syzygium cordatum, Vangueria infausta and Strelitzia nicolai.* Planting took place throughout the years, mostly during the growing season (between November and February), and after heavy rain events.

All restored sites were former low productive dryland sugarcane fields. The remainder of the buffer zone comprised of patches of indigenous forest, woodlands and grasslands. The vegetation is broadly classified as KwaZulu-Natal Coastal Belt. The topography of the study area is base rich and hard. The dominant geology within the site is the Dwyka Tillite. Because Buffelsdraai was previously sugar cane fields, I assumed that the substrate did not differ significantly among the study sites. Visual observations supported this assumption.

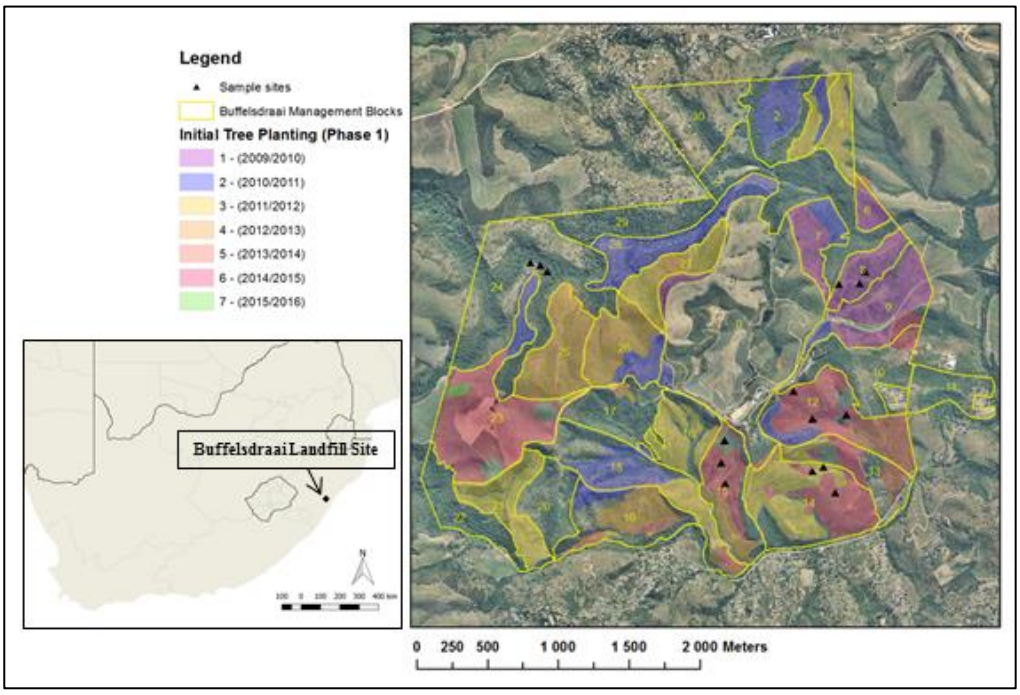

Figure 1: Buffelsdraai Restoration Site

The bufferzone was classified into six different reforested areas (2009-2010, 2010-2011, 2011-2012, 2012-2013, 2013-2014, 2014-2015), riparian forest and sugarcane. I sampled three replicates of five plots (i.e. 15 plots in total): riparian forest; sugarcane; reforested 2009-2010; reforested 2011-2012; and reforested 2013-2014.

2.2. Vegetation structure

Vegetation structure data were collected by conducting step-point 50m line transects, with vegetation data recorded at 5m intervals (Codron *et al.*, 2015). At each interval a 2 x 2m grid was placed and vegetation structure was measured for species that fell within the grid. Species recorded were identified using van Oudtshoon (1992), Van Wyk & Van Wyk (1997) and Koekemoer *et al.*, (2013). Tree height was measured from the base of the tree at ground level to the highest point using a measuring tape (for trees greater than 3m an estimation method was used by holding a 1m ruler against the tree and estimating how many times the ruler would fit the length of the tree), similarly grass height was measured from the base of the stem to the highest point using a measuring tape (Kanowski *et al.*, 2003). Area of canopy cover for trees and grasses were visually estimated as a percentage (Kanowski *et al.*, 2003). Tree density was determined by counting the number of trees within each plot.

2.3. Small mammal sampling

In each plot, I set 25 Sherman live traps (H.B. Sherman Traps, Orlando, Florida), in a 5 x 5 grid formation with one trap per station placed at 10 m intervals (Kalies *et al.*, 2012). I used large and small-sized traps with dimensions 23 x 7.5 x 9 cm and 16 x 5 x 6cm, respectively, which were placed alternatively. I used two sizes of Sherman traps to accommodate for rodent and shrew species that differ in body size (King *et al.*, 2014). Where possible, traps were placed in trees to accommodate arboreal species (Lamani, 2004). I sampled small mammals at each plot for four consecutive nights every season. I checked traps at dawn and baited at dusk with oats and peanut butter (Rautenbach *et al.*, 2014).

Caught individuals were identified to species level using Taylor (1998). Each captured individual was sexed, and their sexual condition recorded (female imperforate, female perforate, male non-scrotal, male scrotal, male sub-scrotal) (Monadjem & Perrin, 2003). I weighed each individual with a Pesola scale (to the nearest 0.5 g). I measured total body length, tail length and head and body length using a metal ruler. I measured right fore-paw and hind-paw lengths (with and without nails) as well as right ear lengths using electronic callipers rounded off to the nearest two decimal places. I tagged caught individuals with individually marked ear-tags for identification and released them at the point of capture (Witmer *et al.*, 2014), however recaptured numbers were too low to use in population estimates. If caught individuals were recaptured, their ear tag number was recorded and they were released at the point of capture. However individuals recaptured during sampling period July 2016 were collected as voucher specimens. Two voucher specimens (one male and one female) of each species at each plot were taken to confirm field identification of species; voucher specimens were stored at the Durban Natural Science Museum. From each voucher specimen, the following tissues and organs were collected for stable isotope analyses: whole blood (split into red blood cells and plasma), liver, hair and bone (see section 2.5 below). All sampling procedures were approved by the Animal Ethics Committee of the University of KwaZulu - Natal (AREC/066//015).

2.4. Stable isotope data collection and treatment

All stable isotope data treatment performed according to protocol provided by the Stable Isotope Laboratory, Mammal Research Institute, University of Pretoria.

2.4.1. Small mammal hair and tissue samples

I collected hair samples from the lower dorsal area from every animal captured. Hair was pulled out using a pair of forceps (Podlesak *et al.,* 2008), and placed in individual vials. In cases where an individual was recaptured a hair sample was recollected, regardless of sample period and site. The whole hair was analysed in all cases. In the laboratory I placed samples in test tubes and degreased the hair by soaking them in a 2:1 ethanol/chloroform mixture, while agitated in an ultra-sonic bath for 20 minutes. The ethanol/chloroform mixture was then poured off and samples were left to dry overnight at 60˚C. Samples were stored at room temperature until stable isotope analysis.

I collected whole blood from voucher specimens. Samples were collected in microcentrifuge tubes and spun in a mini-centrifuge for three minutes at 2000 x g to separate red blood cells and plasma. Red blood cells were removed using a pipette and placed in clean microcentrifuge tubes. Whole blood was stored if separation into red blood cells and plasma was not successful. Red blood cells were stored in liquid nitrogen in the field. In the laboratory, red blood cell samples were removed from the liquid nitrogen, dried overnight at 70˚C and ground to a fine powder using a mortar and pestle and stored at room temperature until analysis

I collected liver samples from voucher specimens. Liver samples were degreased for 20 minutes in a test-tube within a mixture of 2:1 ethanol/chloroform mixture in an ultra-sonic bath. After the mixture was poured off, samples were dried overnight in an oven at 60˚C, and then ground to a fine powder using a mortar and pestle and stored at room temperature until stable isotope analysis

I removed the femur bone from each voucher specimen. Bone collagen was analysed in this study, therefore bone samples were demineralised in 0.5 % HCl for 36 hours at 58˚C. I treated samples with three sequential 2h hour soaks in a 2:1 chloroform/ethanol mixture to remove lipids. Thereafter, I rinsed samples in deionised water and lypholised them for 48 hours. Once samples were dried I removed the shaft of the bone which was ground to a fine powder using a mortar and pestle and stored at room temperature until stable isotope analysis.

2.4.2. Vegetation samples

In each plot (Fig 1), I collected vegetation along a 40m transect line at 20m intervals at three stations. At each station I laid a 2 x 2m grid and recorded and collected the dominant vegetation types (forbs, grasses, trees and sugarcane). I placed plants between folded sheets

of newspaper and pieces of cardboard sheets in plant presses to protect specimens and ventilate water vapour. I stored plant presses in a cabinet. Plants were collected in November 2015 and again in May 2016. In the laboratory, I selected leaves, stems and fruit from tree samples; seeds, leaves and stems from grass samples; stems and leaves from forb samples; and stems and leaves from sugarcane samples which were oven-dried overnight at 60˚C to remove tissue water. Once samples were dried plant parts of each vegetation group (trees, grasses, forbs and sugarcane) were pooled, and ground to a fine powder using a mortar and pestle and stored at room temperature until stable isotope analysis (Dammhahn & Goodman, 2014). To standardise plant collections between seasons I pooled samples for analysis rather than using a specific part of the plant because I wanted to include plant material that would be available to small mammal assemblages during both dry and wet seasons.

2.4.3. Invertebrate samples

In each plot (Fig. 1), I laid four pitfall traps using recyclable materials (plastic 125 ml bottle, paper funnels, sticks and polystyrene sheets) in a 10 x 10m grid formation for three consecutive nights. To limit insects from escaping or crawling out of pitfall traps, I placed a funnel at the entrance of each trap. Additionally, I used polystyrene covers to avoid rain and debris collecting in traps. I checked pitfall traps daily. Sampling of invertebrates occurred during November 2015 and May 2016. Invertebrates were identified to Order, and data were pooled per site. Collected specimens were humanely euthenised in a freezer. In the laboratory samples were oven-dried overnight at 60˚C to remove tissue water. Once samples were dried they were ground to a fine powder using a mortar and pestle and stored at room temperature until analysis (Dammhahn & Goodman, 2014). Additionally, earthworms compromise part of shrew species' diet (Taylor, 1998), therefore shallow pits were dug, 25 x 25 cm and 10 cm in depth along a 40m transect line at 20m intervals at three stations (Decaens & Jimenez, 2002). However, no earthworms were found.

2.4.4. Stable isotope analysis sample preparation

Ground vegetation sampled were weighed to 1.00 - 1.20 mg. Ground hair, red blood cells, bone and insect samples were weighed to 0.50 - 0.60 mg. The amount of sample that must be weighed is dependent on the amount of carbon and nitrogen present in the dry tissue (Voigt *et al.,* 2003). Because live plant material contains more water than animal material, more ground sample is required for stable isotope analysis (Hobson, 1999).

I placed samples in Costech 3.5 x 5 mm pressed tin capsules (Codron *et al.*, 2015). Samples were analysed for 13C/12C and 15N/14N isotope ratios. Samples were combusted at 1000°C in a reactor packed with chromium oxide and silvered copper oxide. Following combustion, oxides were removed in a reduction reactor (reduced copper at 650°C), and the resultant CO_2 and N_2 gases separated on a Carbosieve GC column (65°C, 65 mL/min) before entering the stable light isotope for mass spectrometry analyses.

Isotopic analysis was performed on a Flash EA 1112 Series coupled to a Delta V Plus stable light isotope ratio mass spectrometer via a ConFlo IV system (all equipment supplied by Thermo Fischer, Bremen, Germany), housed at the Stable Isotope Laboratory, Mammal Research Institute, University of Pretoria.
A laboratory running standard (Merck Gel: $\delta^{13}C$ = -20.57‰, $\delta^{15}N$=6.8‰, C%=43.83, N%=14.64) and a blank sample was run after every 12 samples with unknown C and N isotopic values. Every 12^{th} sample was a replicate of the 11^{th} sample to test the reproducibility of results. All results were referenced to Vienna Pee-Dee Belemnite for C isotope values, and to air for N isotope values. Results were expressed in delta notation and per mille scale using the standard equation:

$$\delta X(‰) = [(R_{sample}-R_{standard})/R_{standard}-1]x1000$$

where X= ^{15}N or ^{13}C and R represents $^{15}N/^{14}N$ or $^{13}C/^{12}C$ respectively (Darimont and Reimchen, 2002).

2.5. Statistical analyses

2.5.1. Vegetation response to reforestation

Richness data were square root-transformed; tree density, tree height and grass height data were log-transformed; and tree canopy cover and grass canopy cover data were arcsine-transformed. I tested the transformed data for normality using the Shapiro Wilks test, and homoscedasticity of variances using the Levene's test. To assess the influence of season and study site on forb, grass and tree species richness, tree density, grass and tree height, and grass and tree canopy cover, I performed Two-Way ANOVAs with season and study sites as factors. All statistical analyses were conducted using R (v.3.2.2.0, R Core Team, 2015).

2.5.2. Diversity index

I calculated the Simpsons diversity index of small mammal assemblages at study sites using EstimateS (version 8.2, Colwell, 2009). I used this index because, it measures dominance by identifying the probability that two individuals will be belong to the same species, incorporating both species richness and abundance (Magurran, 2004). Additionally, this index has been widely used, allowing for comparisons with other studies (Makundi *et al.,* 2010; King *et al.,* 2014).

2.5.3. Completeness of small mammal inventory

I calculated expected species richness for each treatment using two species richness estimators: Chao 1 and Jacknife 1 indices (Gotelli & Colwell, 2001), using EstimateS (version 8.2, Colwell, 2009). Chao 1 provides a robust estimation of the minimum species richness, whereas Jacknife 1 reduces the bias of the estimator by removing subsets of the data and recalculating the estimator with the reduced sample (Colwell *et al.,* 2004). These species richness estimators have been shown to perform well in datasets with a limited number of samples (Walther & Morand, 1998). To assess the completeness of the inventories, I calculated the ratio between the observed and expected richness based on the species richness estimators (Schoeman & Jacobs, 2011). Percentage completeness of sampling effort (%) was calculated as:

% completeness = Observed species x 100 / value of the species richness estimator.

To compare rodent and shrew species richness between the plots, I calculated sample-based rarefaction curves (Gotelli & Colwell, 2001) using EstimateS (version 8.2, Colwell, 2009). Sample based rarefaction curves standardise comparisons of species richness among assemblages, assuming random sampling of taxonomically similar individuals that are randomly distributed (Gotelli & Colwell, 2001).

2.5.4. Response of small mammals to reforestation

Richness and abundance data were square root-transformed, and diversity data log-transformed, to meet the assumptions of normality using the Shapiro Wilks test, and homoscedasticity of variances using the Levene's test. To assess the influence of season and study site on rodent and shrew abundance, species richness and diversity, I performed Two-Way ANOVAs with season and study sites as factors, and Tukey post hoc tests with multiple

comparison tests when significant. All statistical analyses were conducted using R (v.3.2.2.0, R Core Team, 2015).

2.5.5. Diet composition of rodents and shrews

Even after transformation, the carbon and nitrogen isotope data were not normally distributed. Thus, I ranked the data, and performed Two-Way ANOVAs, to test the influence of season and study sites on carbon and nitrogen isotope values of rodent and shrew hair samples, and *M. natalensis* tissue samples, and Tukey post hoc tests with multiple comparison tests when significant. Four tissue samples from *M. natalensis* were analysed: bone, hair, liver and red blood cells. In all cases, dry and wet season data were treated separately. All statistical analyses were conducted using R (v.3.2.2.0, R Core Team, 2015).

I assessed the relative contribution of isotopic plant and invertebrate categories in the diets of rodent and shrew species at different study sites by applying a Bayesian isotope mixing model using the package SIAR version 4.1.3. (Parnell *et al.,* 2010; Jackson *et al.,* 2011) in R (v.3.2.2.0, R Core Team, 2015). SIAR produces a range of solutions concerning the contribution of each food source to a consumer's diet, incorporating many sources of variability and multiple dietary sources (Robb *et al.,* 2016). Forbs, grasses, tree material and invertebrates were entered as individual food items. Raw stable isotope data were corrected with diet-tissue fractionation values. There are many factors that influence diet-tissue fractionation values, ranging from food type to inter species variation (Tiezen *et al.,* 1983; Fry, 1988; Ambrose, 1991; Hobson *et al.,* 1993; Bearhop *et al.,* 2002; Ogden *et al.,* 2004; Cherel *et al.,* 2005; Podlesak & McWilliams, 2006; Miller *et al.,* 2008; Symes *et al.,* 2013). Because I did not determine species-specific diet-tissue fractionation factors under laboratory conditions, I used derived diet-tissue fractionation values for $\delta^{13}C$ of 3.5 ‰ for bone (DeNiro & Epstein, 1981), 3.0‰ for hair (MacAvoy *et al.,* 2012; Symes *et al.,* 2013), and 1.6 ‰ for liver (MacAvoy *et al.,* 2005); and derived diet-tissue fractionation values for $\delta^{15}N$ of 4.4‰ for bone (DeNiro & Epstein), 2.7‰ for hair (Galleti *et al.,* 2016), and 5.0 ‰ for liver (MacAvoy *et al.,* 2005). Diet-tissue fractionation values have not been derived for red blood cells in smaller mammals, therefore I used 1‰ for carbon and 3‰ for nitrogen (DeNiro & Epstein 1978, 1981).

Prior to running models, dietary sources were checked for isotopic separation. The isotope values of the four vegetation groups did differ significantly and therefore were included as separate entities. Diet composition was examined at the population level (calculating the

mean value for the proportion of each food source for all individuals in the area). Additionally, standard ellipse area (SEA) was calculated for hair samples of rodents and shrews; bone, hair, liver and red blood cell tissues of *M. natalensis*; and combined dietary sources for each site. This provided measures of isotopic niche widths of the rodent and shrew populations. Further, I calculated the mean distance to centroid as the mean Euclidean distance of each individual of a population to the $\delta^{13}C$ - $\delta^{15}N$ as an estimator of the population isotopic diversity, and the mean nearest neighbour distance which reveals the packing of individuals in the two-dimensional space.

These were produced using the program SIAR by fitting a standard ellipse to the bivariate (carbon and nitrogen) data using maximum likelihood estimators (Robb *et al.*, 2016).

CHAPTER 3

RESULTS

3.1 Differences in vegetation structure among study sites

To test normality of data, Shapiro-Wilk tests were used and to test homogeneity of variance Levene's tests were used. Where assumptions were violated data were log transformed and tested again - the assumptions for parametric tests were met (Table 1, 2).

Table 1. Shapiro-Wilk test of differences in forb, tree and grass species richness, tree density, tree height and site of tree canopy cover among five study sites (2014 restored, 2012 restored, 2010 restored, forest and sugarcane sites) at the Buffelsdraai Landfill Site between November 2015 and July 2016.

	W	**p - value**
Species richness		
Forb	0.765	0.084
Grass	0.792	0.061
Tree	0.901	0.072
Tree density	0.825	0.092
Tree height	0.864	0.075
Canopy cover	0.932	0.095
Grass height	0.894	0.094
Grass % cover	0.872	0.081

Table 2. Levene's Test of equality of variance in forb, tree and grass species richness, tree density, tree height and site of tree canopy cover among five study sites (2014 restored, 2012 restored, 2010 restored, forest and sugarcane sites) at the Buffelsdraai Landfill Site between November 2015 and July 2016.

	df	F-value	p-value
Species Richness			
Forb	9	0.20	0.991
Grass	9	0.19	0.993
Tree	9	0.91	0.537
Tree density	9	0.68	0.714
Tree height	9	1.03	0.451
Canopy cover	9	1.03	0.456
Grass height	9	0.69	0.705
Grass % cover	9	0.52	0.845

I found significant differences in forb species richness between seasons (Table 3). Tukey HSD post hoc test showed that forb species richness was significantly higher during the wet season than the dry season ($p = 0.004$, Fig. 2A). Forb species richness also differed significantly among sites (Table 3), with Tukey HSD post hoc tests showing that forb species richness was significantly higher at 2014 restored sites than at 2012 restored ($p = 0.014$, Fig. 2B), 2010 restored ($p = 0.041$, Fig. 2B), forest ($p = 0.001$, Fig. 2B) and sugarcane ($p = 0.058$, Fig. 2B); and at sugarcane sites than at forest ($p = 0.0389$, Fig. 2B). I found no significant interactions between forb species richness, season and sites (Table 3).

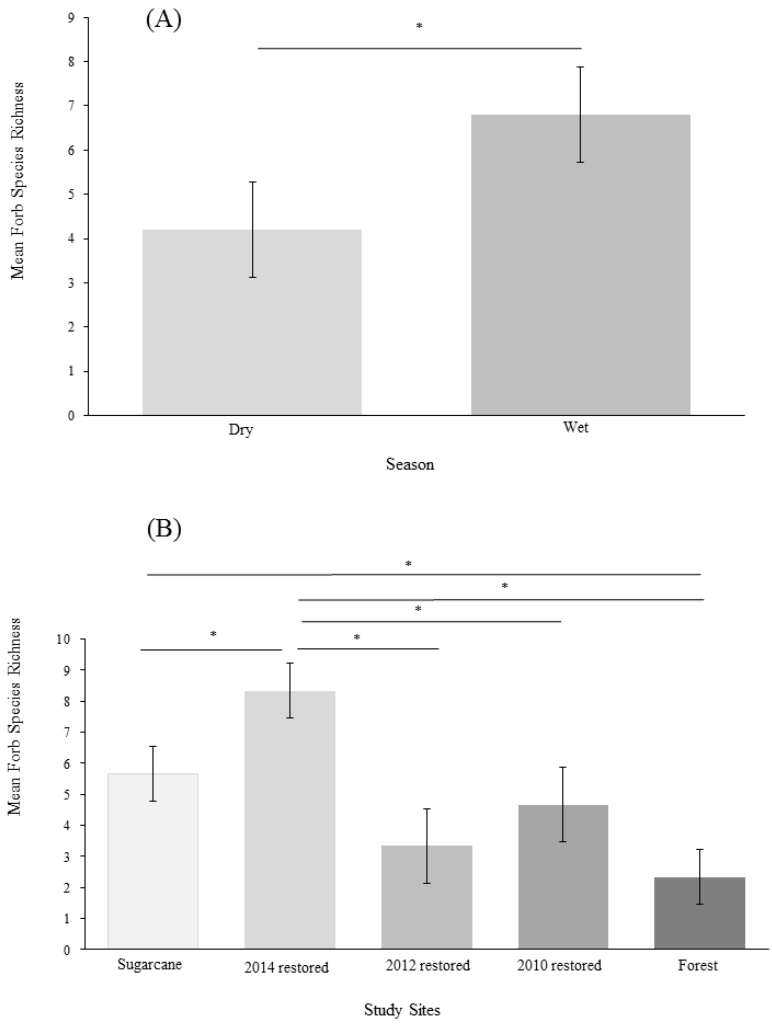

Figure 2. Mean (±SD) forb species richness (A) during wet and dry seasons, and (B) at five study sites of the Buffelsdraai Landfill Site between November 2015 and July 2016**.** Asterisks indicate significant differences ($p < 0.05$).

I found significant differences in grass species richness between seasons (Table 3). Tukey HSD post hoc tests showed that grass species richness was significantly higher during the wet season than the dry season ($p < 0.001$, Fig. 3A). Grass species richness differed significantly among sites (Table 3), with Tukey HSD post hoc tests showing that grass species richness was significantly higher at 2014 restored sites than at forest ($p = 0.004$, Fig. 3B) and sugarcane ($p = 0.014$, Fig. 3B); at 2012 restored sites than at sugarcane ($p = 0.054$, Fig. 3B), forest ($p = 0.003$, Fig. 3B); and at 2010 restored sites than at forest ($p = 0.011$, Fig. 3B). I found no significant interactions between grass species richness, season and sites (Table 3).

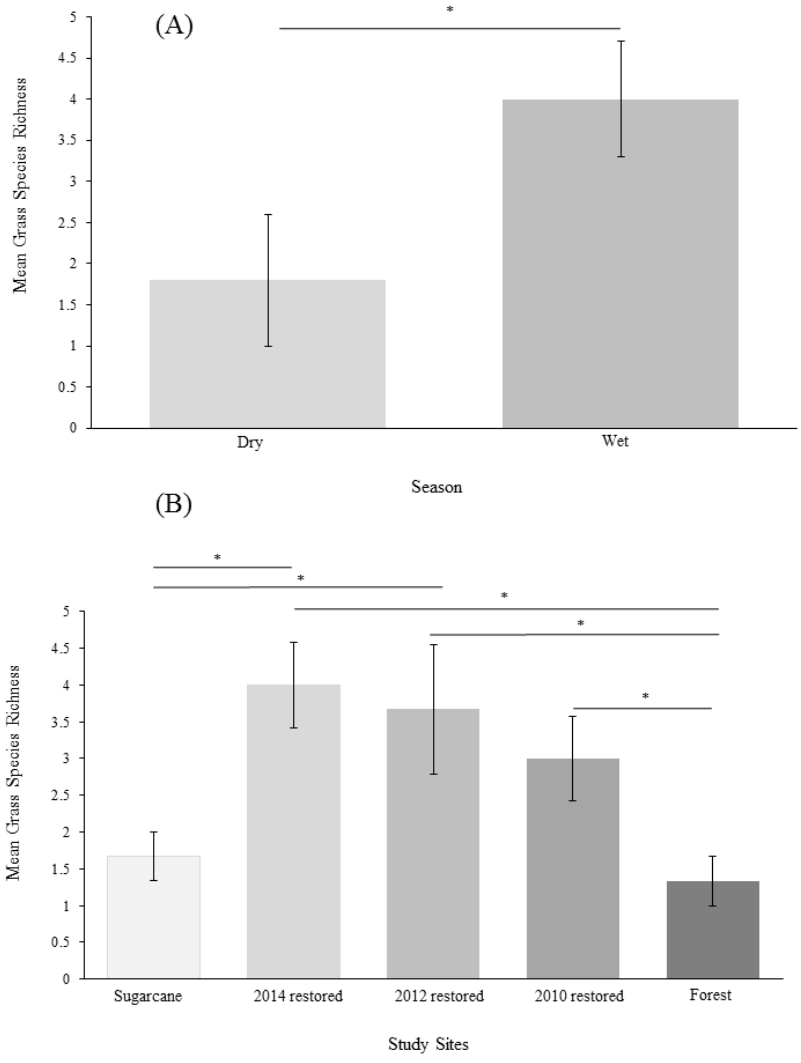

Figure 3. Mean (±SD) grass species richness (A) during wet and dry seasons, and (B) at five study sites of the Buffelsdraai Landfill Site between November 2015 and July 2016**.** Asterisks indicate significant differences ($p < 0.05$).

I found no significant differences in tree species richness between seasons (Table 3). Tree species richness differed significantly among sites (Table 3), with Tukey HSD post hoc tests showing that tree species richness was significantly lower at sugarcane sites than 2014 restored ($p < 0.001$, Fig. 4B), 2012 restored ($p < 0.001$, Fig. 4B), 2010 restored ($p < 0.001$, Fig. 4B) and forest ($p = 0.002$, Fig. 4B). I found no significant interactions between tree species richness, season and sites (Table 3).

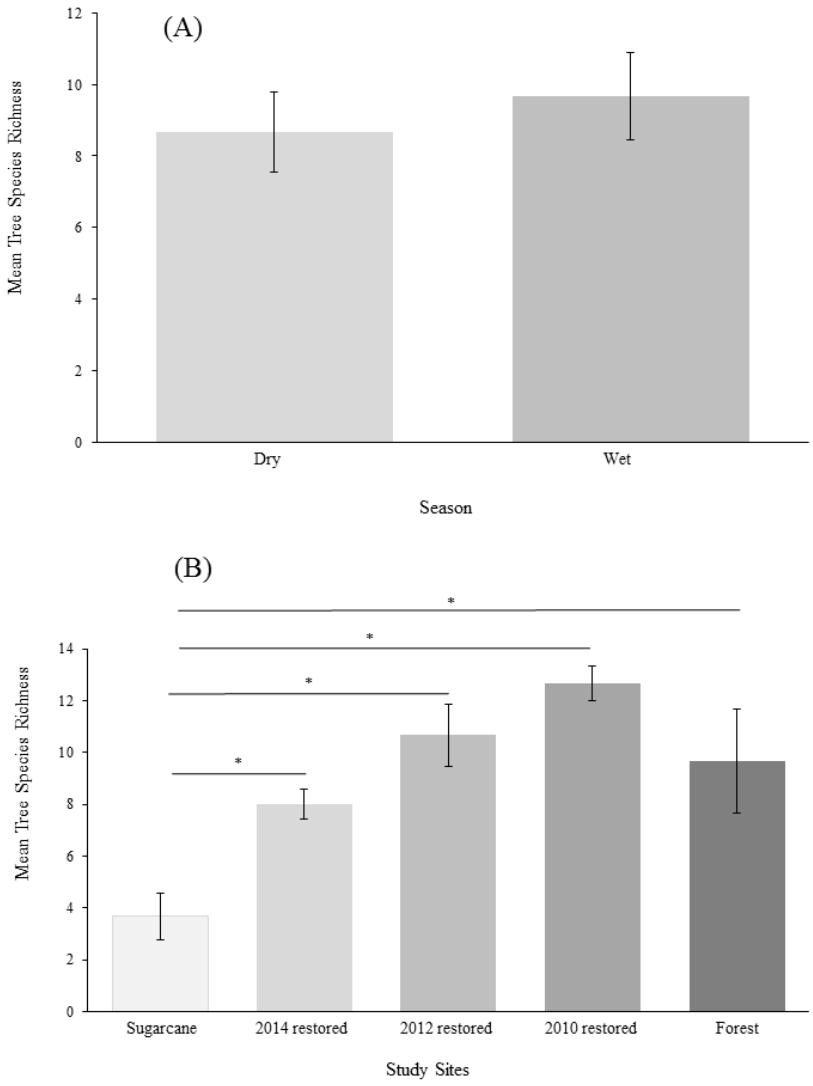

Figure 4. Mean (±SD) tree species richness (A) during wet and dry seasons, and (B) at five study sites of the Buffelsdraai Landfill Site between November 2015 and July 2016**.** Asterisks indicate significant differences ($p < 0.05$).

I found no significant differences in tree density between seasons (Table 3). I found significant differences in tree density among sites (Table 3), with Tukey HSD post hoc tests showing that tree density was significantly lower in sugarcane sites than at 2010 restored ($p < 0.001$, Fig. 5B), 2012 restored ($p < 0.001$, Fig. 5B) and 2014 restored ($p < 0.001$, Fig. 5B); at forest sites than at 2010 restored ($p < 0.001$, Fig. 5B), 2012 restored ($p < 0.001$, Fig. 5B) and 2014 restored ($p = 0.004$, Fig. 5B). Tree density was also significantly higher at forest sites than at sugarcane ($p = 0.006$, Fig. 5B). I found no significant interactions between tree density, season and sites (Table 3).

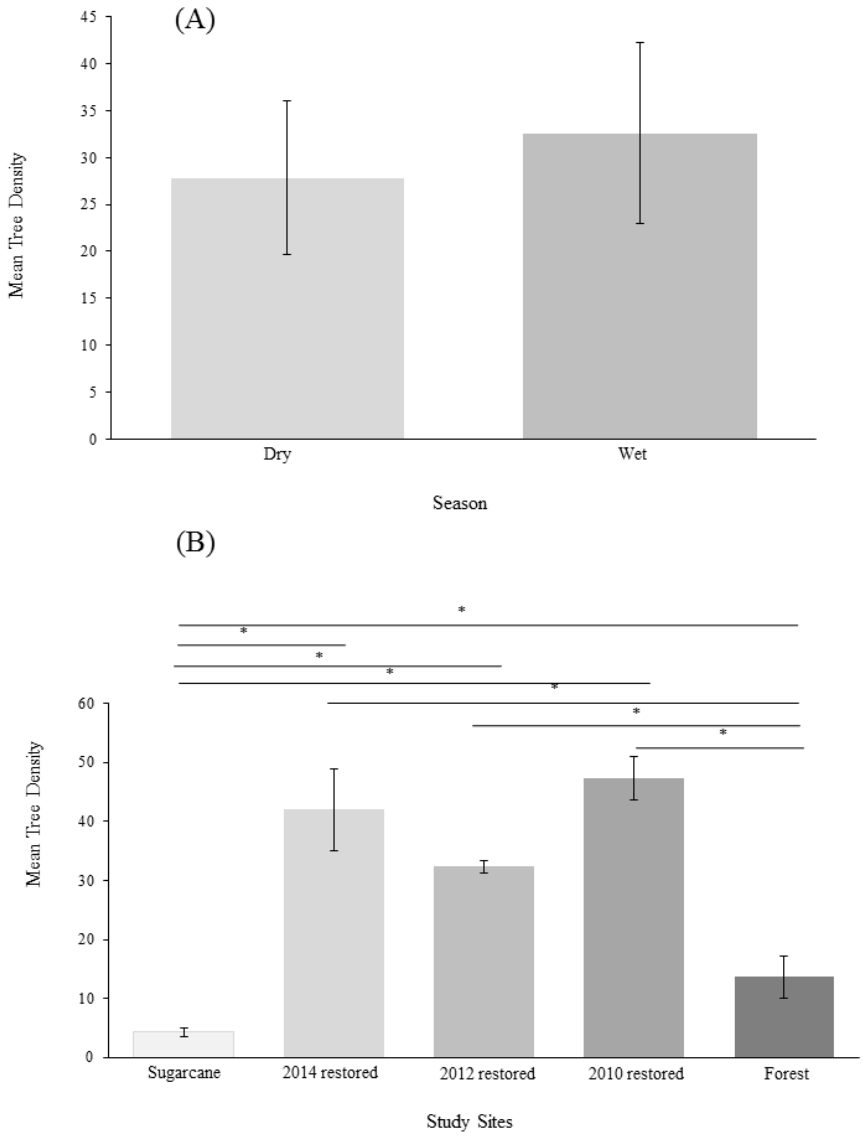

Figure 5. Mean (±SD) tree density (A) during wet and dry seasons, and (B) at five study sites of the Buffelsdraai Landfill Site between November 2015 and July 2016**.** Asterisks indicate significant differences ($p < 0.05$).

I found no significant differences in tree height between seasons (Table 3). Tree height differed significantly among sites (Table 3), with Tukey HSD post hoc tests showing that tree height was significantly higher at forest sites than at 2010 restored ($p < 0.001$, Fig. 6B), 2012 restored ($p < 0.001$, Fig. 6B), 2014 restored ($p < 0.001$, Fig. 6B) and sugarcane ($p = 0.002$, Fig. 6B); at 2010 restored sites than at 2014 restored ($p = 0.011$, Fig. 6B) and sugarcane ($p = 0.002$ Fig. 6B); and at 2012 restored sites than at 2014 restored ($p = 0.014$, Fig. 6B) and sugarcane ($p = 0.003$, Fig. 6B). I found no significant interactions between tree height, season and sites (Table 3).

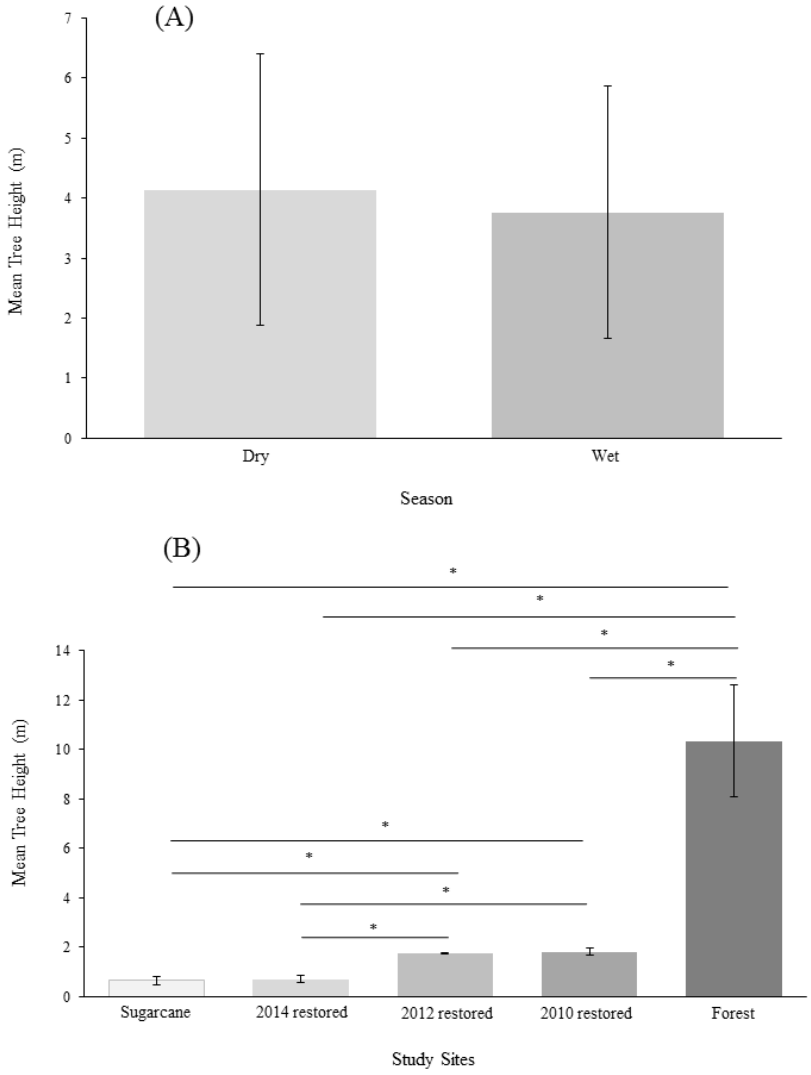

Figure 6. Mean (±SD) tree height (A) during wet and dry seasons, and (B) at five study sites of the Buffelsdraai Landfill Site between November 2015 and July 2016**.** Asterisks indicate significant differences ($p < 0.05$).

I found no significant differences in site of tree canopy between seasons (Table 3). Site of tree canopy cover differed significantly among sites (Table 3), with Tukey HSD post hoc tests showing that site of tree canopy was significantly higher at forest sites than at 2012 restored ($p = 0.003$, Fig. 7B), 2014 restored ($p = 0.003$, Fig. 7B), 2010 restored ($p = 0.002$, Fig. 7B) and sugarcane ($p < 0.001$, Fig. 7B); at 2010 restored sites than at 2012 restored ($p = 0.002$, Fig. 7B), 2014 restored ($p = 0.002$, Fig. 7B) and sugarcane ($p = 0.002$, Fig. 7B); and at 2012 restored sites than at 2014 restored ($p = 0.002$, Fig. 7B) and sugarcane ($p < 0.001$, Fig. 7B). I found no significant interactions between site of tree canopy, season and sites (Table 3).

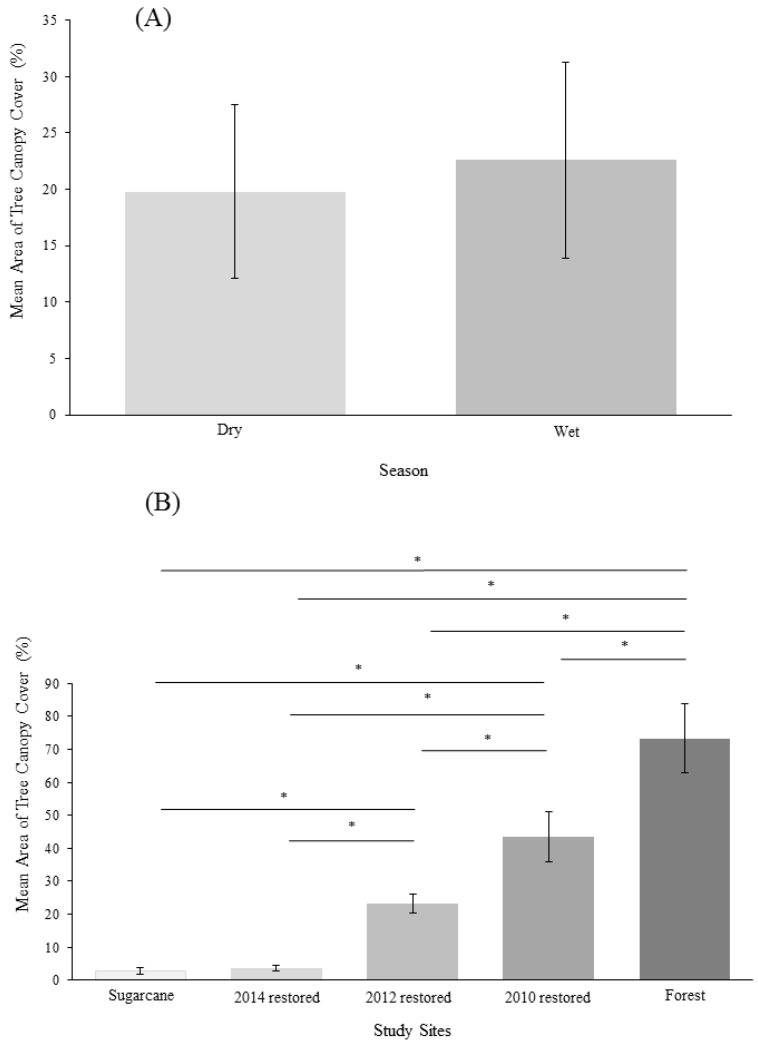

Figure 7. Mean (±SD) tree canopy (A) during wet and dry seasons, and (B) at five study sites of the Buffelsdraai Landfill Site between November 2015 and July 2016**.** Asterisks indicate significant differences ($p < 0.05$).

I found no significant differences in grass height between seasons (Table 3). Grass height differed significantly among sites (Table 3), with Tukey HSD post hoc tests showing that grass height was significantly lower at forest sites that at 2010 restored ($p = 0.005$, Fig. 8B), 2012 restored ($p = 0.007$, Fig. 8B), 2014 restored ($p = 0.024$, Fig. 8B), and sugarcane ($p = 0.018$, Fig. 8B). Grass height was significantly higher at 2012 restored sites than at 2014 restored ($p = 0.008$, Fig. 8B). I found no significant interactions between grass height, season and sites (Table 3).

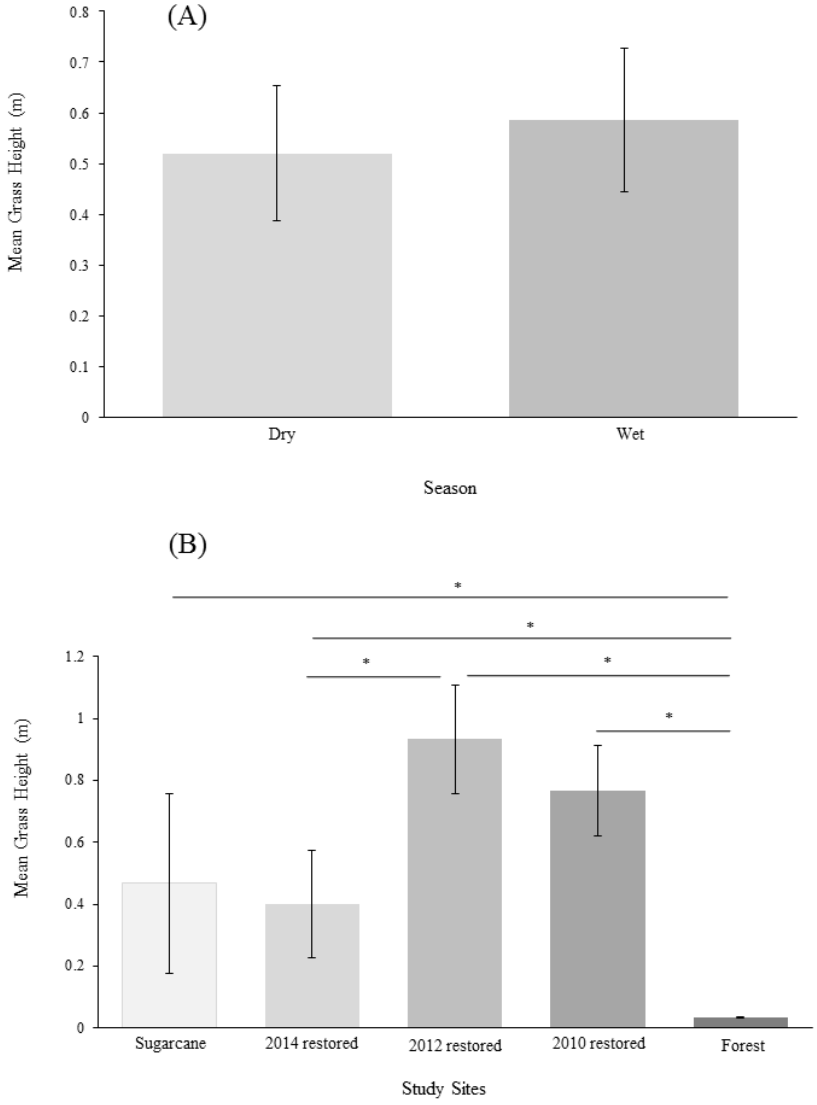

Figure 8. Mean (±SD) grass height (A) during wet and dry seasons, and (B) at five study sites of the Buffelsdraai Landfill Site between November 2015 and July 2016**.** Asterisks indicate significant differences ($p < 0.05$).

I found significant differences in grass percentage cover between seasons (Table 3). Tukey HSD post hoc tests showed that grass % cover was significantly higher during the wet season than the dry season (p = 0.003, Fig. 9A). Grass percentage cover also differed significantly among sites (Table 3), with Tukey HSD post hoc tests showing that grass percentage cover was significantly higher at 2012 restored sites than at 2010 restored (p = 0.009, Fig. 9B), 2014 restored (p = 0.048, Fig. 9B), forests (p = 0.003, Fig. 9B), and sugarcane (p = 0.002, Fig. 9B); 2010 restored sites than at forests (p = 0.010, Fig. 9B), sugarcane (p = 0.008, Fig. 9B), and 2014 restored (p = 0.021, Fig. 9B); and at 2014 restored sites than at sugarcane (p = 0.024, Fig. 9B), and forests (p = 0.017, Fig. 9B). I found no significant interactions between grass percentage cover, season and sites (Table 3).

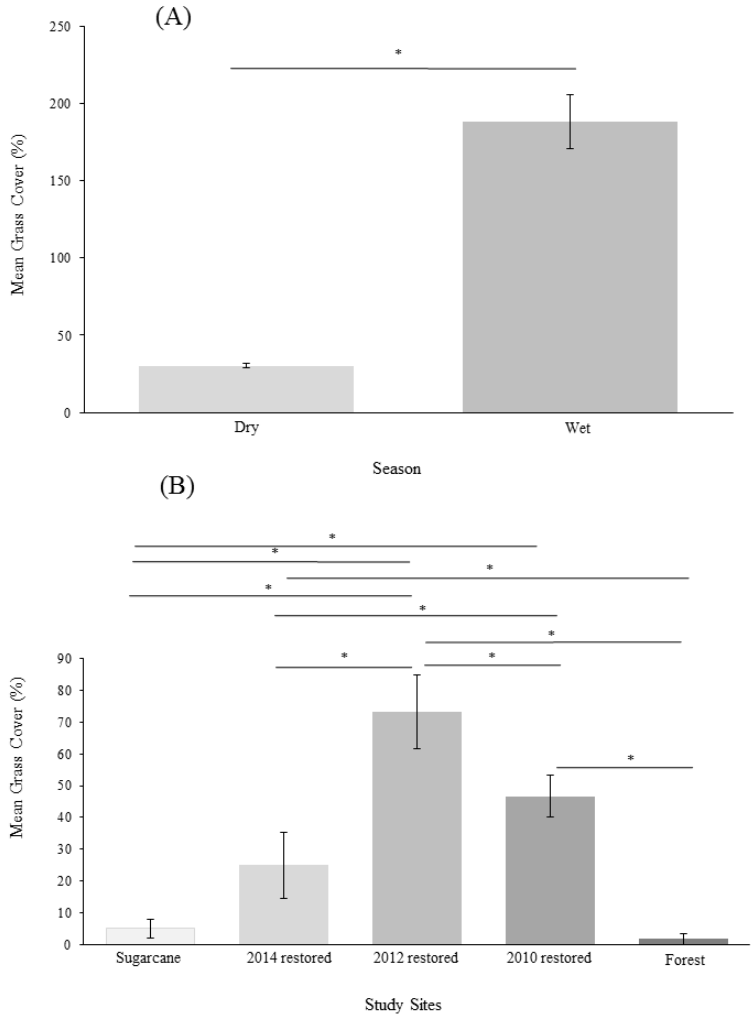

Figure 9. Mean (±SD) grass percentage cover (A) during wet and dry seasons, and (B) at five study sites of the Buffelsdraai Landfill Site between November 2015 and July 2016**.** Asterisks indicate significant differences (p < 0.05).

Table 3. Statistical results of two-way ANOVAs of differences in species richness of forbs, grasses, and trees; tree density, tree height and canopy cover ; and grass height and canopy cover between sites and seasons of the Buffelsdraai Landfill Site between November 2015 and July 2016. Significant p-values are shown in bold.

	Season			Site			Season:Site		
	df	**F-value**	**p-value**	**df**	**F-value**	**p-value**	**df**	**F-value**	**p-value**
Species richness									
Forb	1	10.71	**0.003**	4	6.32	**0.002**	4	0.06	0.993
Grass	1	18.22	**< 0.001**	4	8.58	**< 0.001**	4	0.64	0.639
Tree	1	1.59	0.223	4	16.09	**< 0.001**	4	0.09	0.984
Tree density	1	2.75	0.113	4	55.02	**< 0.001**	4	0.58	0.680
Tree height	1	15.87	**< 0.001**	4	45.57	**< 0.001**	4	1.40	0.279
Canopy cover	1	1.61	0.210	4	50.91	**< 0.001**	4	0.35	0.842
Grass height	1	1.31	0.579	4	7.27	**< 0.001**	4	2.01	0.999
Grass % cover	1	6.47	**0.005**	4	10.25	**< 0.001**	4	1.23	0.998

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3.2 Completeness of small mammal inventory

During 720 trapping hours, I captured 210 small mammals, representing 12 species, classified to 11 genera and three families: Muridae (n=189), Gliridae (n=3) and Soricidae (n=18). The most numerous species caught was *Mastomys natalensis* (n=165) representing 79 % of all captures, followed by *Crocidura cyanea* (n=12), *Lemniscomys rosalia* (n=12), *Steatomys pratensis* (n=6), *Crocidura flavescens* (n=5), *Aethomys ineptus* (n=2), *Dendromus melanotis* (n=2), *Grammomys dolichurus* (n=2), *Graphiurus murinus* (n=1), *Mus minutoides* (n=1), *Otomys auratus* (n=1) and *Suncus infinitesimus* (n=1).

Sample-based rarefaction curves of rodents reached asymptotes, and indicated that species richness was higher in forests, than restored sites and sugarcane sites (Figure 10). At identical sampling efforts (cumulative trapping days = 45), species richness was 5 (SD ± 1.91) at forest sites, 4 (SD ± 1.54) at 2012 restored sites, 4 (SD ± 1.30) at 2014 restored sites, 3 (SD ± 1.47) at sugarcane sites and 3 (SD ± 1.27) at 2010 restored sites.

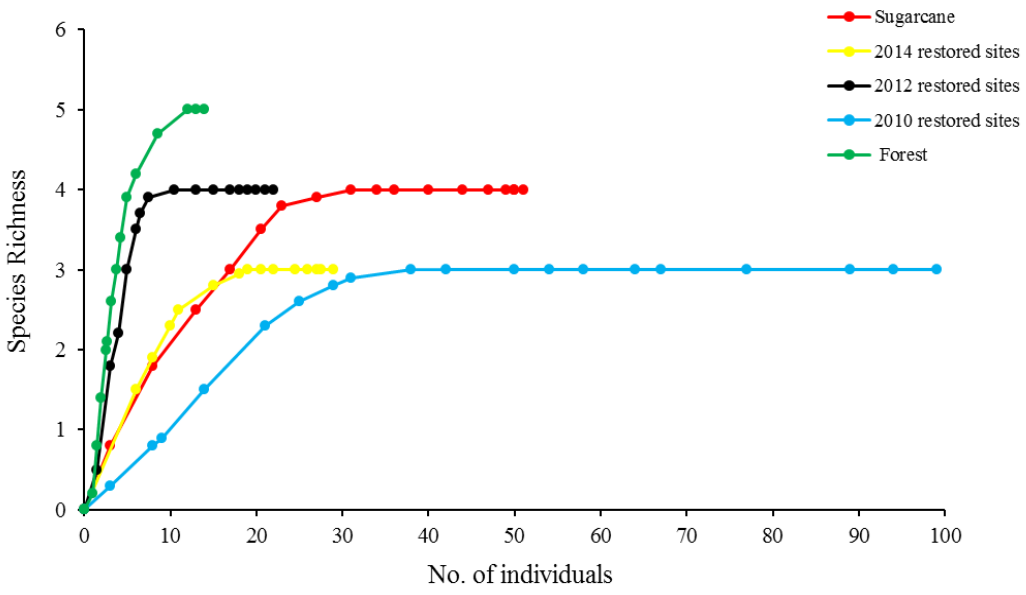

Figure 10. Rarefaction curves of rodent species captured at five study sites of the Buffelsdraai Landfill Site, between November 2015 and July 2016.

Sample-based rarefaction curves of shrews did not reach asymptotes at restored sites (Figure 11). Shrew species richness was higher in 2012 restored sites, than other restored sites, sugarcane sites and forests. At identical sampling efforts (cumulative trapping days = 38), species richness was 2 (SD ± 0.61) at sugarcane sites, 3 (SD ± 0.79) at 2012 restored sites, 2 (SD ± 0.35) at 2014 restored sites and 2 (SD ± 0.61) at 2010 restored sites.

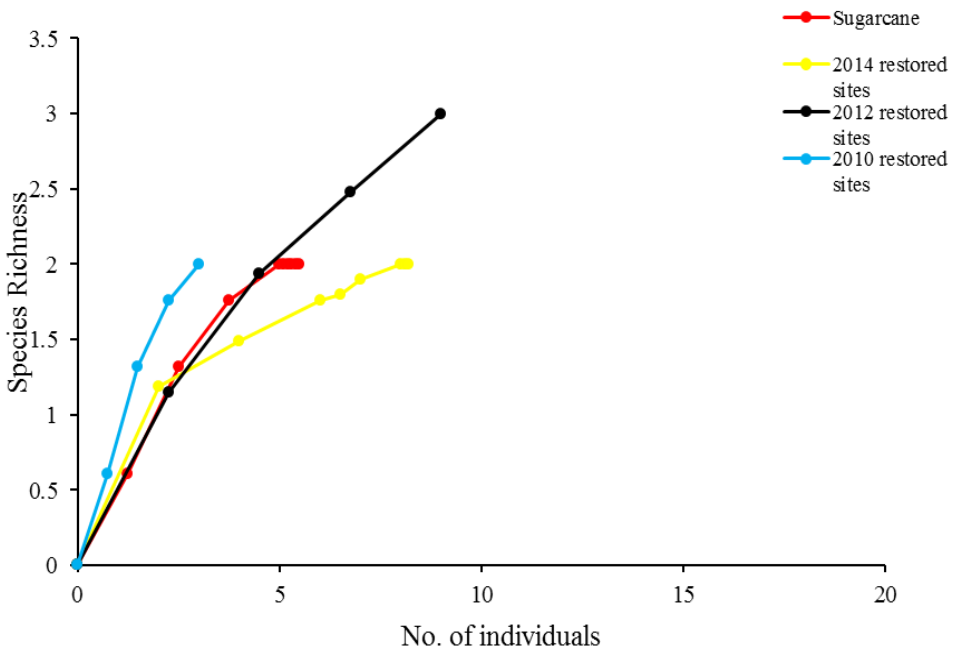

Figure 11. Rarefaction curves of shrew species captured at four study sites of the Buffelsdraai Landfill Site, between November 2015 and July 2016.

The Chao 1 richness estimator indicated that species inventories for rodents were > 82% complete for all study sites (Table 4). The Jacknife 1 richness estimator indicated that all sites were > 72% complete, except sugarcane sites where the species richness estimator showed 62% completeness and forest sites where the species richness estimator showed 60% completeness (Table 4).

The Chao 1 richness estimator indicated that species inventories for shrews were > 70% complete for sites (Table 5). The species richness estimator showed that 2010 restored sites were 100% complete. The Jacknife 1 richness estimator indicated that all sites were > 65% complete (Table 5).

Table 4. Observed species (Obs spp) and expected species richness of rodent assemblages based on Chao 1 and Jacknife 1 richness estimators at five study sites of the Buffelsdraai Landfill Site.

Study Site	Obs spp	Chao 1	% Completeness	Jacknife 1	% Completeness
Sugarcane	4	4.8	83.3	6.5	61.5
2014 restored	3	3.3	90.9	3.8	79.0
2012 restored	4	4.2	95.2	5.5	72.7
2010 restored	3	3.2	93.8	3.7	81.1
Forest	5	6.1	82	8.4	59.5

Table 5. Observed species (Obs spp) and expected species richness of shrew assemblages based on Chao 1 and Jacknife 1 richness estimators at four study sites of the Buffelsdraai Landfill Site.

Study Site	Obs spp	Chao 1	% Completeness	Jacknife 1	% Completeness
Sugarcane	2	2.9	69.0	2.5	80.0
2014 restored	2	2.8	71.4	3.1	64.5
2012 restored	3	3.7	81.1	3.9	76.9
2010 restored	2	2	100.0	3	66.7

3.3 Response of small mammals to restoration

Mastomys natalensis was captured at all study sites. *Lemniscomys rosalia*, *C. cyanea* and *C. flavescens* were captured at all restoration and sugarcane sites, but not at forest sites. *Steatomys pratensis* was captured at 2010 restored sites, sugarcane sites and forest sites. *Dendromus melanotis* was captured at 2014 and 2012 restored sites. *Grammomys dolichurus*, *G. murinus* and *A. ineptus* were captured at forest sites only. The rodent *M. minutoides* and the shrew *S. infinitesimus* were captured at the 2012 restored sites only. *Otomys auratus* was captured at sugarcane sites only (Table 6).

Table 6. Seasonal abundance of rodent and shrew species captured at five different study sites at the Buffelsdraai Landfill Site between November 2015 and July 2016.

	Wet season					Dry season				
	Study sites									
	Sugarcane	2014 restored	2012 restored	2010 restored	Forest	Sugarcane	2014 restored	2012 restored	2010 restored	Forest
Aethomys ineptus	0	0	0	0	2	0	0	0	0	0
Crocidura cyanea	0	2	1	0	0	2	3	3	1	0
Crocidura flavescens	0	0	0	0	0	2	1	1	1	0
Dendromus melanotis	0	1	0	0	0	0	0	1	0	0
Grammomys dolichurus	0	0	0	0	0	0	0	0	0	2
Graphiurus murinus	0	0	0	0	1	0	0	0	0	0
Lemniscomys rosalia	1	1	1	1	0	2	2	0	4	0
Mus minutoides	0	0	0	0	0	0	0	1	0	0
Mastomys natalensis	23	2	3	17	1	19	21	7	72	0
Otomys auratus	0	0	0	0	0	1	0	0	0	0
Suncus infinitesimus	0	0	0	0	0	0	0	1	0	0
Steatomys pratensis	0	0	0	0	0	2	0	0	3	2

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A Shapiro-Wilk test was used to determine normality of data and a Levene's test was used to determine homogeneity of variance. Where assumptions were violated data were log transformed and assumptions of parametric tests were re-tested - the assumptions were met (Table 7, 8).

Table 7. Shapiro-Wilk tests of differences in rodent and shrew abundance, species richness and diversity among five study sites at the Buffelsdraai Landfill Site between November 2015 and July 2016.

	W	**p - value**
Rodent		
Abundance	0.9405	0.1387
Species Richness	0.7094	0.1756
Diversity	0.8390	0.0653
Shrew		
Abundance	0.2638	0.2563
Species Richness	0.6004	0.0757
Diversity	0.9405	0.1387

Table 8. Levene's Tests of equality of variances in rodent and shrew abundance, species richness and diversity among five study sites at the Buffelsdraai Landfill Site between November 2015 and July 2016.

	df	**F-value**	**p-value**
Rodent			
Abundance	9	1.06	0.433
Species Richness	9	0.14	0.998
Diversity	9	0.73	0.612
Shrew			
Abundance	9	1.18	0.357
Species Richness	9	1.04	0.446
Diversity	7	1.08	0.796

I found significant differences in rodent abundance between seasons (Table 10). Tukey HSD post hoc tests showed that rodent abundance was significantly higher during the dry season than the wet season ($p = 0.001$, Fig. 12A). Rodent abundance also differed significantly among sites (Table 10). Tukey HSD post hoc tests showed that rodent abundance was significantly higher at 2010 restored sites than at 2012 restored ($p < 0.001$, Fig. 12B), 2014 restored ($p < 0.001$, Fig. 12B), sugarcane ($p = 0.008$, Fig. 12B) and forest ($p < 0.001$, Fig. 12B) sites. Additionally I found significant interactions between rodent abundance, season and sites: rodent abundance was significantly higher at 2010 restored sites in the dry season than at the other sites and seasons (Table 10).

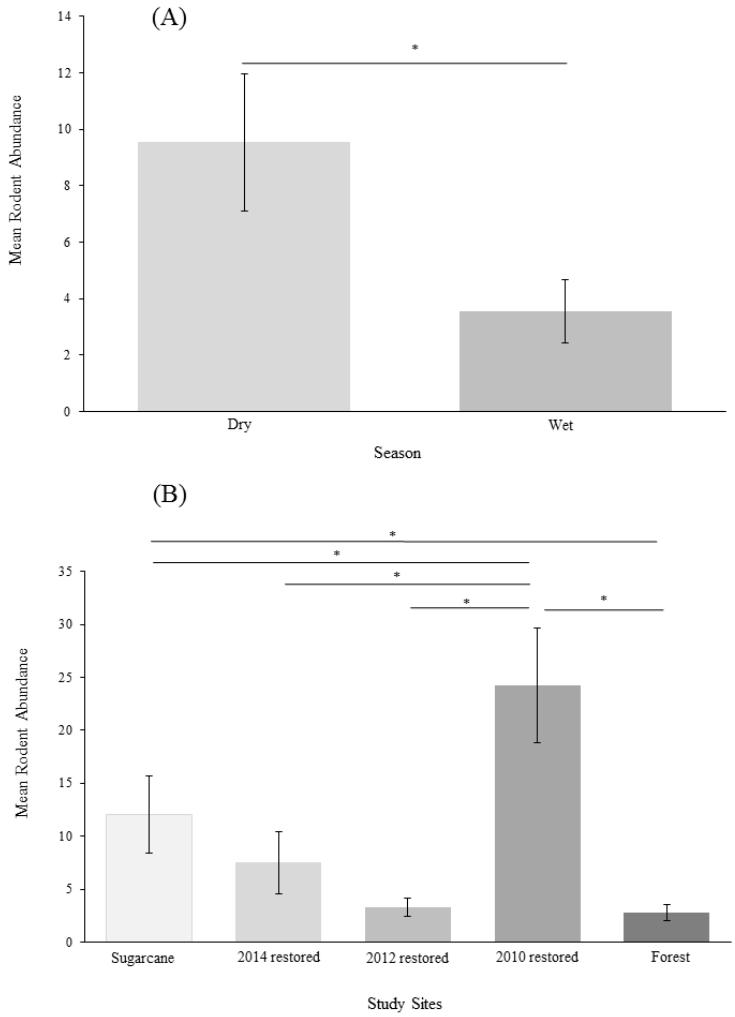

Figure 12. Mean (±SD) rodent abundance (A) during wet and dry seasons, and (B) at five study sites of the Buffelsdraai Landfill Site between November 2015 and July 2016**.** Asterisks indicate significant differences ($p < 0.05$).

I found significant differences in shrew abundance between seasons, but not among sites (Table 13). Tukey HSD post hoc tests showed that shrew abundance was significantly higher during the dry season than the wet season ($p = 0.025$, Fig. 13A). There were no significant interactions between shrew abundance, season and sites (Table 10).

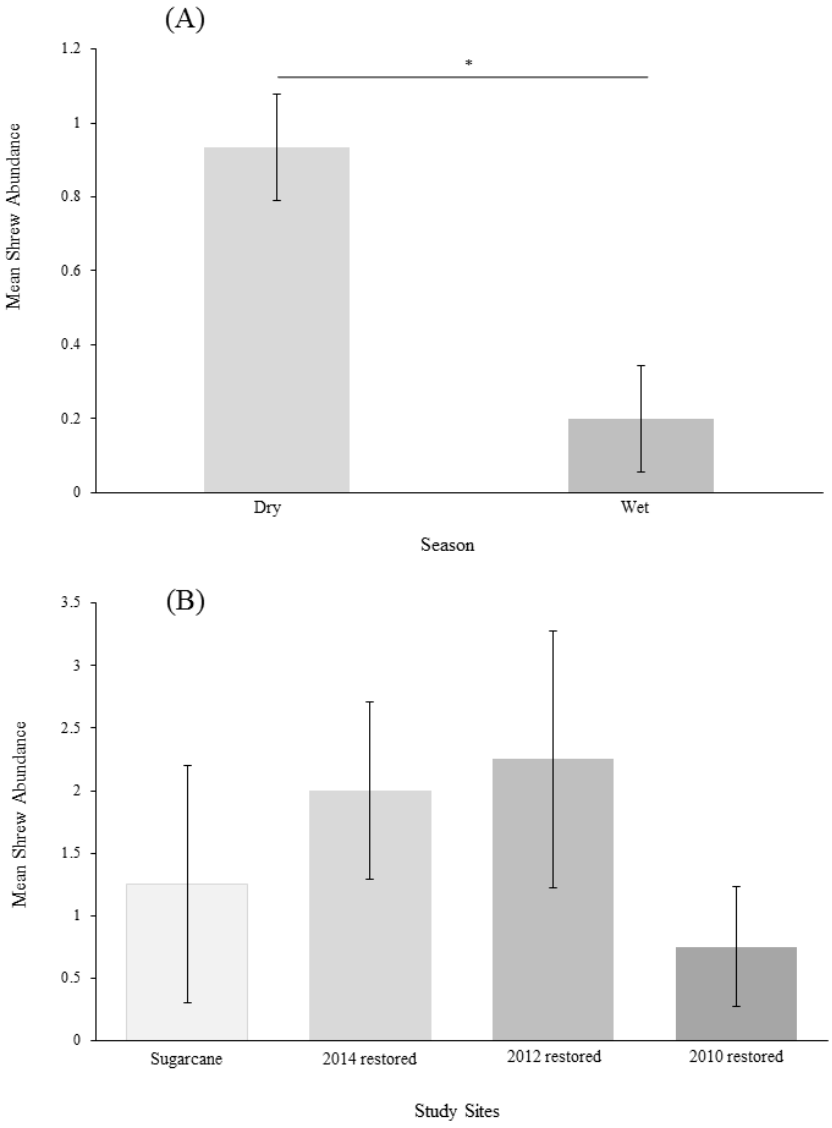

Figure 13. Mean (±SD) shrew abundance (A) during wet and dry seasons, and (B) at five study sites of the Buffelsdraai Landfill Site between November 2015 and July 2016**.** Asterisks indicate significant differences ($p < 0.05$).

I found no significant differences in rodent species richness between seasons, sites and the interaction between season and sites (Fig. 14, Table 10). However, I found significant differences in shrew species richness between seasons, but not among sites or interactions between sites and seasons (Table 10). Tukey HSD post hoc tests showed that shrew species richness was significantly higher during the dry season than the wet season ($p = 0.039$, Fig. 15A).

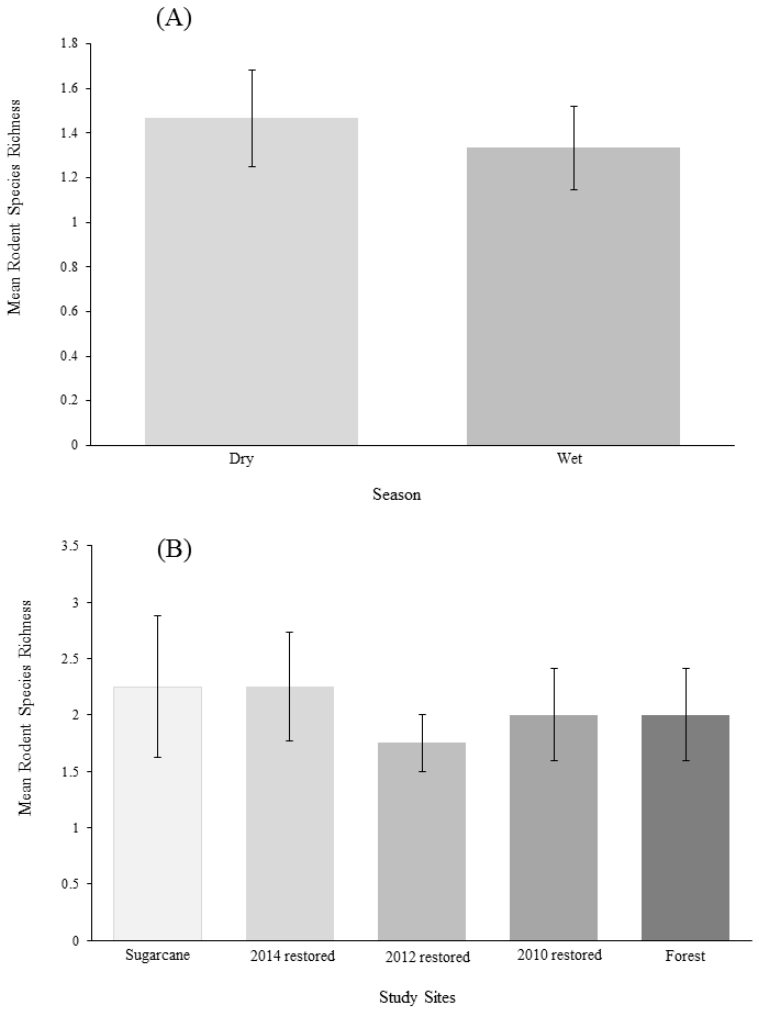

Figure 14. Mean (±SD) rodent species richness (A) during wet and dry seasons, and (B) at five study sites of the Buffelsdraai Landfill Site between November 2015 and July 2016**.** Asterisks indicate significant differences ($p < 0.05$).

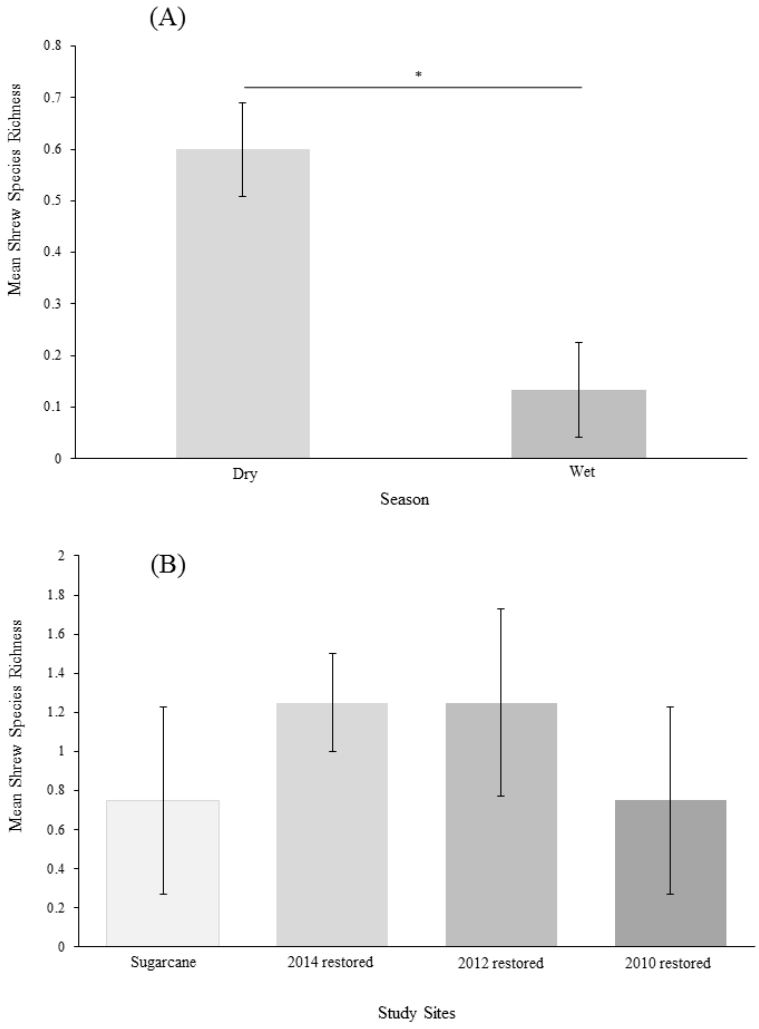

Figure 15. Mean (±SD) shrew species richness (A) during wet and dry seasons, and (B) at five study sites of the Buffelsdraai Landfill Site between November 2015 and July 2016**.** Asterisks indicate significant differences ($p < 0.05$).

I found significant differences in rodent diversity between seasons (Table 10), with Tukey HSD post hoc tests showing that rodent diversity was significantly higher during the dry season than the wet season ($p = 0.022$, Fig. 16A). Rodent diversity also differed significantly among sites (Table 10). Tukey HSD post hoc tests showed that rodent diversity was significantly higher at forest sites than at 2010 restored ($p < 0.001$, Fig. 16B), 2012 restored ($p = 0.003$, Fig. 16B), 2014 restored ($p = 0.001$, Fig. 16B) and sugarcane ($p < 0.001$, Fig. 16B) sites. Additionally, I found significant interactions between rodent diversity, season and sites: rodent diversity was significantly higher in 2010 restored sites in the dry season than at other sites and seasons (Table 10).

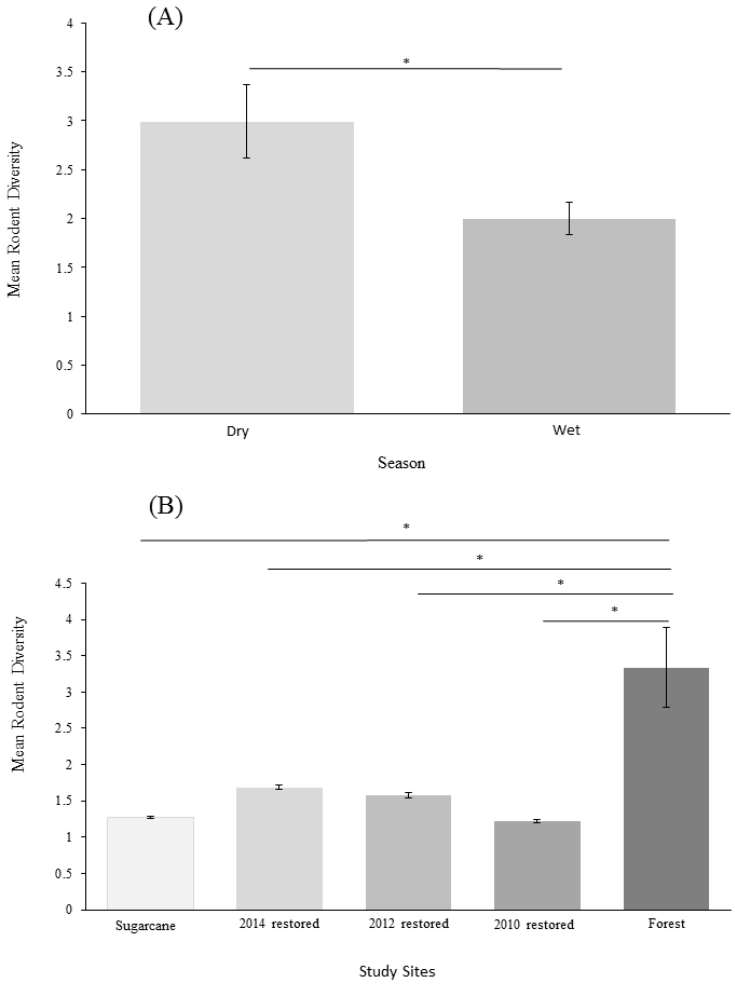

Figure 16. Mean (±SD) rodent diversity (A) during wet and dry seasons, and (B) at five study sites of the Buffelsdraai Landfill Site between November 2015 and July 2016**.** Asterisks indicate significant differences ($p < 0.05$).

I found significant differences in shrew diversity between seasons (Table 10), with Tukey HSD post hoc tests showing that shrew diversity was significantly higher during the dry season than the wet season ($p = 0.008$, Fig. 17A). Shrew diversity differed significantly among sites (Table 10), with Tukey HSD post hoc tests showing that shrew diversity was significantly higher at sugarcane sites than at 2014 restored ($p = 0.003$, Fig.15B) and 2012 restored ($p = 0.006$, Fig. 17B), at 2010 restored sites than at 2014 restored ($p = 0.002$, Fig. 17B) and 2012 restored ($p = 0.035$, Fig. 17B) sites, and at 2012 restored sites than at 2014 restored sites ($p = 0.002$, Fig. 17B). I found no significant interactions between shrew diversity, season and sites (Table 10).

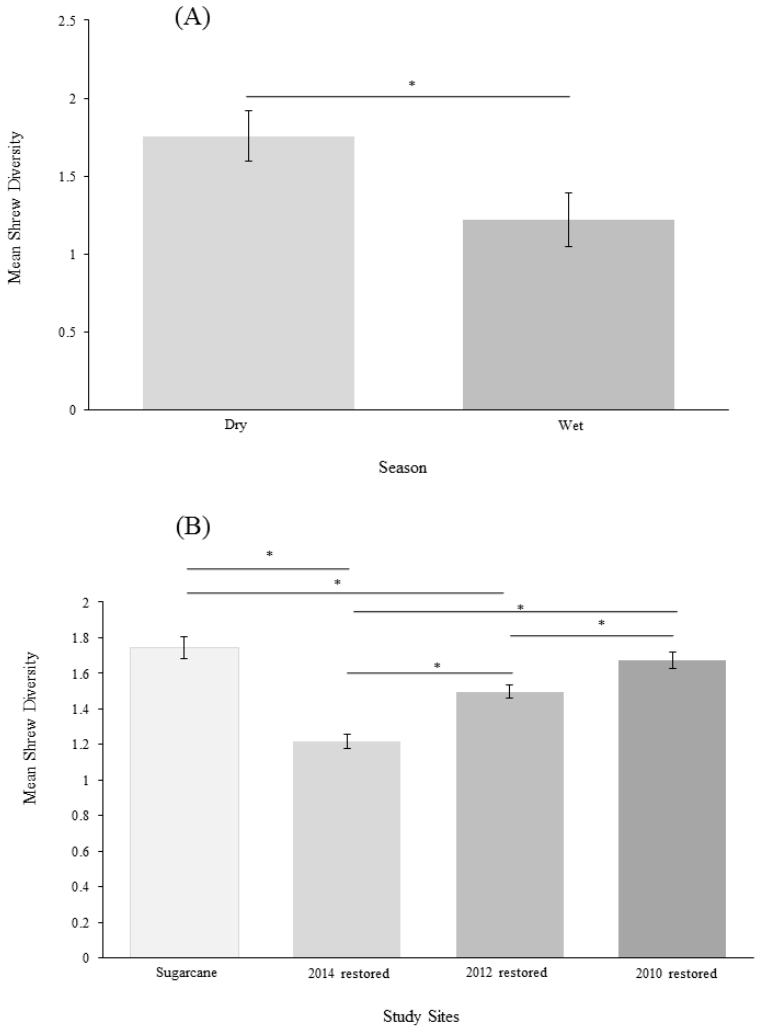

Figure 17. Mean (±SD) shrew diversity (A) during wet and dry seasons, and (B) at five study sites of the Buffelsdraai Landfill Site between November 2015 and July 2016**.** Asterisks indicate significant differences ($p < 0.05$).

Table 9. Simpson's diversity index of rodent and shrew assemblages at study sites of the Buffelsdraai Landfill Site between November 2015 and July 2016.

Study sites	Simpsons diversity index
Rodents	
Sugarcane	0.25
2014 restored	0.68
2012 restored	0.31
2010 restored	0.28
Forest	0.63
Shrews	
Sugarcane	0.43
2014 restored	0.23
2012 restored	0.27
2010 restored	0.48

Table 10. Statistical results based on two-way ANOVAs of differences in rodent and shrew abundance, species richness and diversity between sites and seasons at the Buffelsdraai Landfill Site between November 2015 and July 2016. Significant p-values are shown in bold.

	Season			Site			Season:Site		
	df	**F-value**	**p-value**	**df**	**F-value**	**p-value**	**df**	**F-value**	**p-value**
Rodent									
Relative abundance	1	22.01	**< 0.001**	4	16.58	**< 0.001**	4	9.10	**< 0.001**
Species richness	1	0.19	0.667	4	0.55	0.703	4	0.55	0.703
Diversity	1	7.35	**0.022**	4	38.75	**< 0.001**	4	7.64	**0.004**
Shrew									
Relative abundance	1	4.03	**0.045**	4	1.13	0.319	4	0.78	0.533
Species richness	1	4.90	**0.039**	4	1.60	0.213	4	0.40	0.806
Diversity	1	12.20	**0.008**	3	42.71	**< 0.001**	3	0.09	0.096

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3.4. Stable isotope composition of small mammals in response to restoration.

Plant and invertebrate samples were analysed to provide a habitat baseline of isotopic variation. I collected and analysed 128 plant and 101 invertebrate samples in the wet season, and 113 plant and 89 invertebrate samples in the dry season. Plant samples collected at study sites across Buffelsdraai Landfill Site had a mean $\delta^{13}C$ of - 25.30 $^{0}/_{00}$ (range - 36.18 $^{0}/_{00}$ – - 13.14 $^{0}/_{00}$) and a mean $\delta^{15}N$ of 0.27 $^{0}/_{00}$ (range - 3.90 $^{0}/_{00}$ – 5.51 $^{0}/_{00}$). Mean $\delta^{13}C$ of invertebrate samples collected at study sties: -20.74 $^{0}/_{00}$ (range - 32.64 $^{0}/_{00}$ – - 24.83 $^{0}/_{00}$), and mean $\delta^{15}N$ of invertebrate samples: 4.27 $^{0}/_{00}$ (range - 1.01 $^{0}/_{00}$ – 9.30 $^{0}/_{00}$).

$\delta^{13}C$ values of rodent hair indicated that the group consumed food items across the whole C_3-C_4 spectrum of terrestrial vegetation (Table 12). There were significant differences in the carbon and nitrogen composition of rodent hairs between seasons (Table 13). Tukey HSD post hoc tests showed that carbon composition were more negative during the wet season than the dry season ($p < 0.001$, Fig. 18), likely due to the abundance of C_4 plants present in the wet season. Tukey HSD post hoc tests showed that nitrogen composition of rodent hairs were higher in the dry season than the wet season ($p < 0.001$, Fig. 18). Nitrogen but not carbon composition of rodent hairs differed significantly among sites (Table 13). Tukey HSD post tests showed that nitrogen composition of rodent hairs were lower at 2010 restored sites than 2012 restored ($p < 0.001$, Fig. 18), 2014 restored ($p < 0.001$, Fig. 18), forests ($p = 0.035$, Fig. 18), and sugarcane ($p < 0.001$, Fig. 18) sites. I found no significant interactions between the carbon and nitrogen composition of rodent hair, season and sites (Table 13).

In the wet season, the total overall isotopic niche occupied by rodents was greater at 2010 restored sites than other restored sites, forests and sugarcane sites (Fig. 18A). Rodent species aggregation in the $\delta^{13}C$ - $\delta^{15}N$ plot was high (Table 12; Fig. 18A). The lowest average $\delta^{13}C$ values were recorded for *G. murinus* whereas the highest $\delta^{13}C$ values were recorded for *M. natalensis* (Table 14).

Rodents had narrower $\delta^{13}C$ ranges in the dry season than wet season (Table 12). The total overall isotopic niche occupied by rodents was also greater at 2010 restored sites than at other restored sites, forests and sugarcane sites (Fig. 18B). Rodent species aggregation in the $\delta^{13}C$ - $\delta^{15}N$ plot was high (Table 12). Among the species captured in the dry season, *G. dolichurus* had the lowest $\delta^{13}C$ values whereas the highest $\delta^{13}C$ values were recorded for *O. auratus* (Table 14; Fig. 18B).

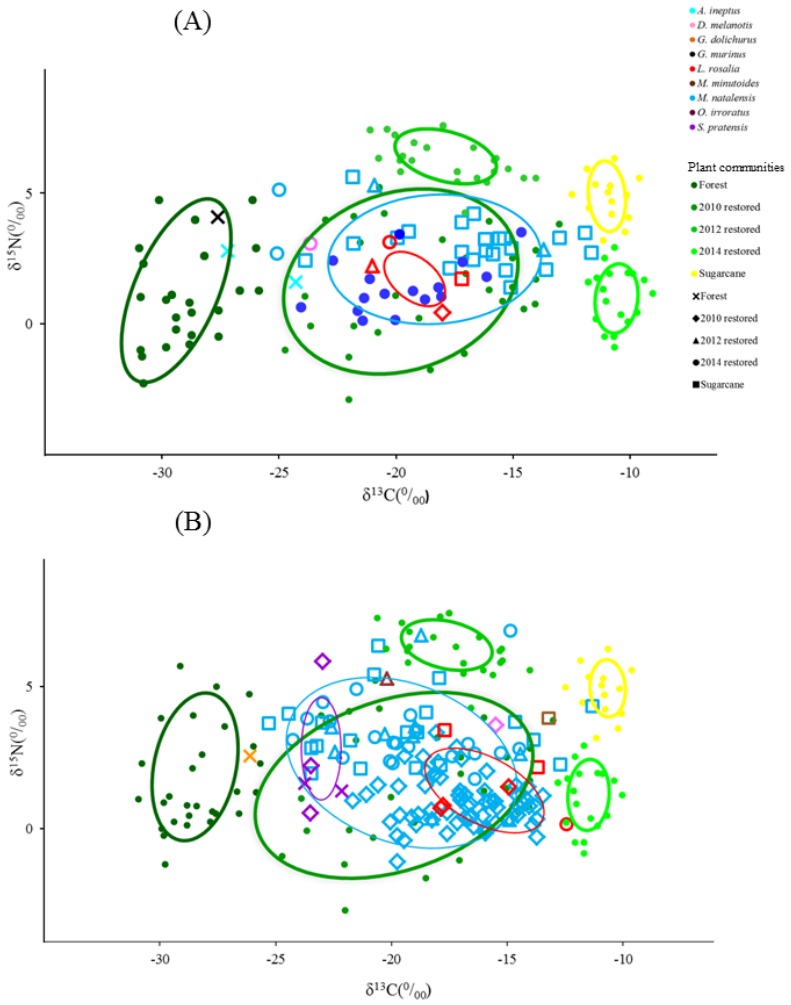

Figure 18. Standard ellipses for rodent's main sources of food based on hair collected at 2010, 2012 and 2014 restored sites, sugarcane sites and forest sites of the Buffelsdraai Landfill Site between November 2015 and July 2016 during the (A) wet season and (B) dry season. Individuals caught at the same study site are depicted in the same symbol (symbols depicted in legend), and species are coded by colour. Plant communities are delineated as ellipses (colours according to legend).

Using SIAR analysis I estimated dietary contributions based on rodent hair samples for rodent species within each study site during the wet and dry seasons to investigate whether species' relative consumption differed among restored sites, sugarcane sites and forests.

According to the Bayesian model at forest sites during the wet season, *A. ineptus* fed mostly on tree material and *G. murinus* fed mostly on invertebrates. During the dry season, *G. dolichurus* fed mostly on grasses (Fig. 19, Table 16).

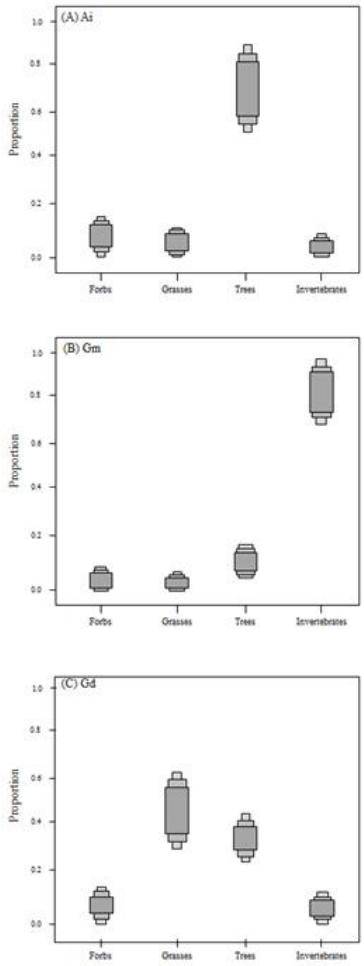

Figure 19. Relative proportions of isotopically distinct categories of prey in the diet of (A) *A. ineptus*, (B) *G. murinus* at forest sites during the wet season, and (C) *G. dolichurus* at forest sites during the dry season, as determined by a Bayesian isotopic mixing model. Box plots show the relative proportions for each food source with 95% (dark grey), 75%, 25% (medium grey) and 5% (light grey) confidence intervals.

At 2010 restored sites during the wet season, *A. ineptus, L. rosalia* and *M. natalensis* fed mostly on tree material, and *D. melanotis* fed predominantly on grasses. During the dry season, *D. melanotis, L. rosalia, M. minutoides, O. auratus* and *S. pratensis* fed mostly on grasses, and *M. natalensis* fed mostly on tree material (Fig. 20, Table 16).

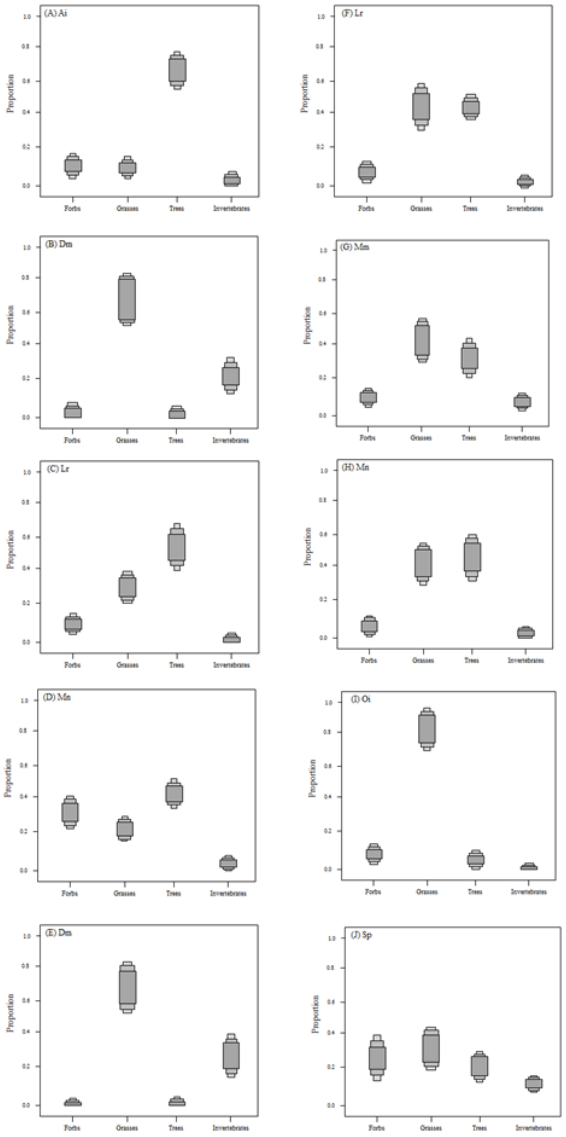

Figure 20. Relative proportions of isotopically distinct categories of prey in the diet of (A) *A. ineptus*, (B) *D. melanotis*, (C) *L. rosalia*, (D) *M. natalensis* at 2010 restored sites during the wet season, and (E) *D. melanotis*, (F) *L. rosalia*, (G) *M. minutoides*, (H) *M. natalensis*, (I) *O. auratus* and (J) *S. pratensis* at 2010 restored sites during the dry season, as determined by a Bayesian isotopic mixing model. Box plots show the relative proportions for each food source with 95% (dark grey), 75%, 25% (medium grey) and 5% (light grey) confidence intervals.

At 2012 restored sites during the dry season, *M. natalensis* fed mostly on tree material (Fig. 21, Table 16). At sugarcane sites during the wet season, *M. natalensis* fed mostly on grasses, and forbs during the dry season (Fig. 22, Table 16).

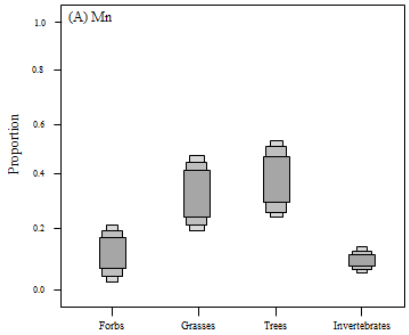

Figure 21. Relative proportions of isotopically distinct categories of prey in the diet of (A) *M.. natalensis* at 2012 restored sites during the dry season, as determined by a Bayesian isotopic mixing model. Box plots show the relative proportions for each food source with 95% (dark grey), 75%, 25% (medium grey) and 5% (light grey) confidence intervals.

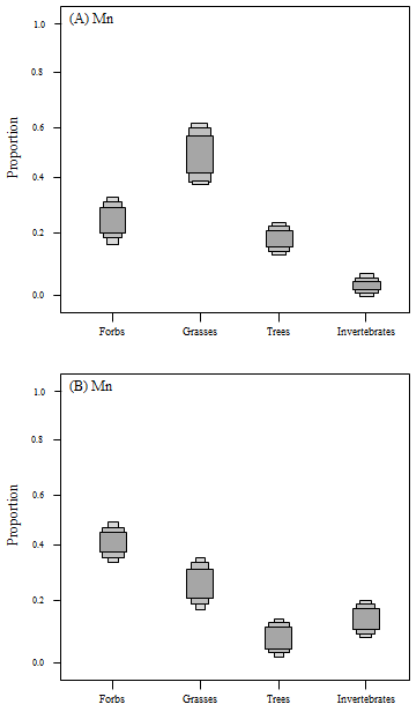

Figure 22. Relative proportions of isotopically distinct categories of prey in the diet of *M. natalensis* at sugarcane sites during the (A) wet season, and (B) dry season, as determined by a Bayesian isotopic mixing model. Box plots show the relative proportions for each food source with 95% (dark grey), 75%, 25% (medium grey) and 5% (light grey) confidence intervals.

In total 165 *M. natalensis* individuals were sampled at all restoration sites, forests and sugarcane sites. I therefore analysed the carbon and nitrogen isotopic values and C/N isotopic ratios of the bone, hair, liver and red blood cells of *M. natalensis* to investigate how this generalist species utilised the sites across the Buffelsdraai landscape.

I found significant differences in carbon and nitrogen composition of *M. natalensis* bone between seasons (Table 13). Tukey HSD post hoc tests showed that carbon composition of *M. natalensis* bone were higher during the wet season than the dry season ($p < 0.001$), and nitrogen composition of *M. natalensis* bone were higher during the wet season than the dry season ($p = 0.039$). Carbon but not nitrogen composition of *M. natalensis* bone differed significantly among sites (Table 13). Tukey HSD post hoc tests showed that carbon composition of *M. natalensis* bone were significantly higher at 2014 restored sites than 2010 restored sites and sugarcane sites ($p<0.05$). I found no significant interactions between *M. natalensis* bone, season and sites (Table 13). In the wet season $\delta^{13}C$ values of *M. natalensis* bone had narrower ranges than in the dry season (Table 12; Fig. 23A). Aggregation in the $\delta^{13}C$ - $\delta^{15}N$ plot was high (Table 12; Fig. 23A).

I found significant differences in carbon and nitrogen composition of *M. natalensis* hair between seasons (Table 13). Tukey HSD post hoc tests showed that carbon composition of *M. natalensis* hair were higher during the wet season than the dry season ($p = 0.002$), and nitrogen composition of *M. natalensis* hair were higher during the wet season than the dry season ($p < 0.001$). I found no significant differences in carbon and nitrogen composition of *M. natalensis* hair among sites (Table 13). I found no significant interactions between *M. natalensis* hair, season and sites (Table 13). In the wet season $\delta^{13}C$ values of *M. natalensis* hair had narrower ranges than in the dry season (Table 12: Fig. 23C). Aggregation in the $\delta^{13}C$ - $\delta^{15}N$ plot was high (Table 12; Fig. 23C).

I found significant differences in nitrogen but not carbon composition of *M. natalensis* liver between seasons (Table 13). Tukey HSD post hoc tests showed that nitrogen composition of *M. natalensis* liver were higher during the wet season than the dry season ($p < 0.001$). I found no significant differences in carbon and nitrogen composition of *M. natalensis* liver among sites (Table 13). I found no significant interactions between *M. natalensis* liver, season and sites (Table 13). In the wet season $\delta^{13}C$ values of *M. natalensis* liver were narrower ranges than in the dry season (Table 12: Fig. 23E). Aggregation in the $\delta^{13}C$ - $\delta^{15}N$ plot was higher in the dry season than during the wet season (Table 12; Fig. 23E).

I found significant differences in carbon but not nitrogen composition of *M. natalensis* red blood cells (RBC) between seasons (Table 13). Tukey HSD post hoc tests showed that

carbon composition of *M. natalensis* RBC were higher during the dry season than the wet season ($p < 0.001$). I found no significant differences in carbon and nitrogen composition of *M. natalensis* RBC among sites (Table 13). I found no significant interactions between *M. natalensis* RBC's season and sites (Table 13).. In the wet season $\delta^{13}C$ values of *M. natalensis* RBC's has narrower ranges than in the dry season (Table 12: Fig. 23G). Aggregation in the $\delta^{13}C$ - $\delta^{15}N$ plot was higher in the dry season than during the wet season (Table 12Fig. 23G).

According to carbon and nitrogen isotope composition of *M. natalensis* bone, I found indirect evidence of dispersal movements of *M. natalensis* between 2010 and 2012 habitat types, with many individuals having values outside the non-outlier range of the habitat in which they were trapped (Fig. 23). The overall isotopic niche based on carbon and nitrogen isotope composition of *M. natalensis* bone were highly correlated to the carbon and nitrogen isotopic composition of 2010 restored sites, (Fig. 23A, 23B). Individuals were captured at all restoration sites, forests and sugarcane, however their carbon and nitrogen isotopic composition aggregated within those of 2010 restored sites. This pattern was true for all tissue types (Fig. 23). The overall isotopic niche of carbon and nitrogen composition of *M. natalensis* hair was highly correlated to the carbon and nitrogen isotopic composition of 2010 restored sites (Fig. 23C, 23D). During the dry season individuals captured at 2010 and 2014 restored sites aggregated within 2014 restored sites. The overall isotopic niche of carbon and nitrogen composition of *M. natalensis* liver was highly correlated to the carbon and nitrogen isotopic signature of 2010 restored sites, (Fig. 23E, 23F), with few outliers of individuals captured at sugarcane sites. The overall isotopic niche of carbon and nitrogen composition of *M. natalensis* RBC was highly correlated to the carbon and nitrogen isotopic composition of 2010 restored sites, (Fig. 23G, 23H), however during the wet season the isotopic niche of carbon nitrogen composition of *M. natalensis* RBC was most similar to those of 2012 restored sites. SIAR analysis revealed that the most important food source for *M. natalensis* was vegetation from the 2010 restored sites. This pattern was true for all tissues types.

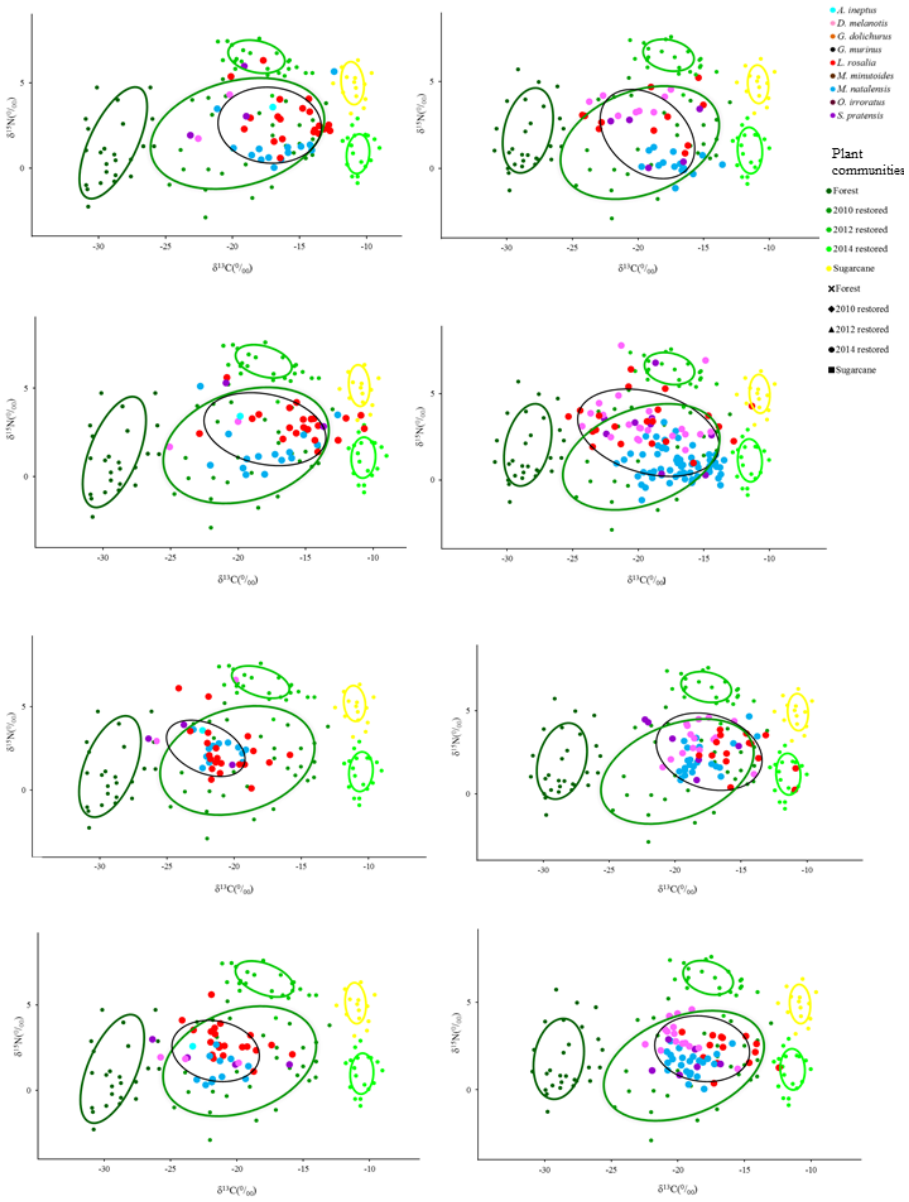

Figure 23. Standard ellipses for different tissues of *M. natalensis* individuals (A) bone collected during the wet season and (B) bone collected during the dry season; (C) hair collected during the wet season and (D) hair collected during the dry season; (E) liver collected during the wet season and (F) liver collected during the dry season; and (G) red blood cells collected during the wet season and (H) red blood cells collected during the dry season, in relation to the isotopic composition of plant communities at 2010, 2012 and 2014 restored sites, sugarcane sites and forest sites between November 2015 and July 2016. Individuals captured at the same study site are depicted in the same colour. Plant communities are largely delineated standard ellipses (colours according to legend).

According to the SIAR analysis, there were no changes in *M. natalensis'* diet over time, because there were no changes in isotopic niches between tissues (Fig. 24, Table 17).

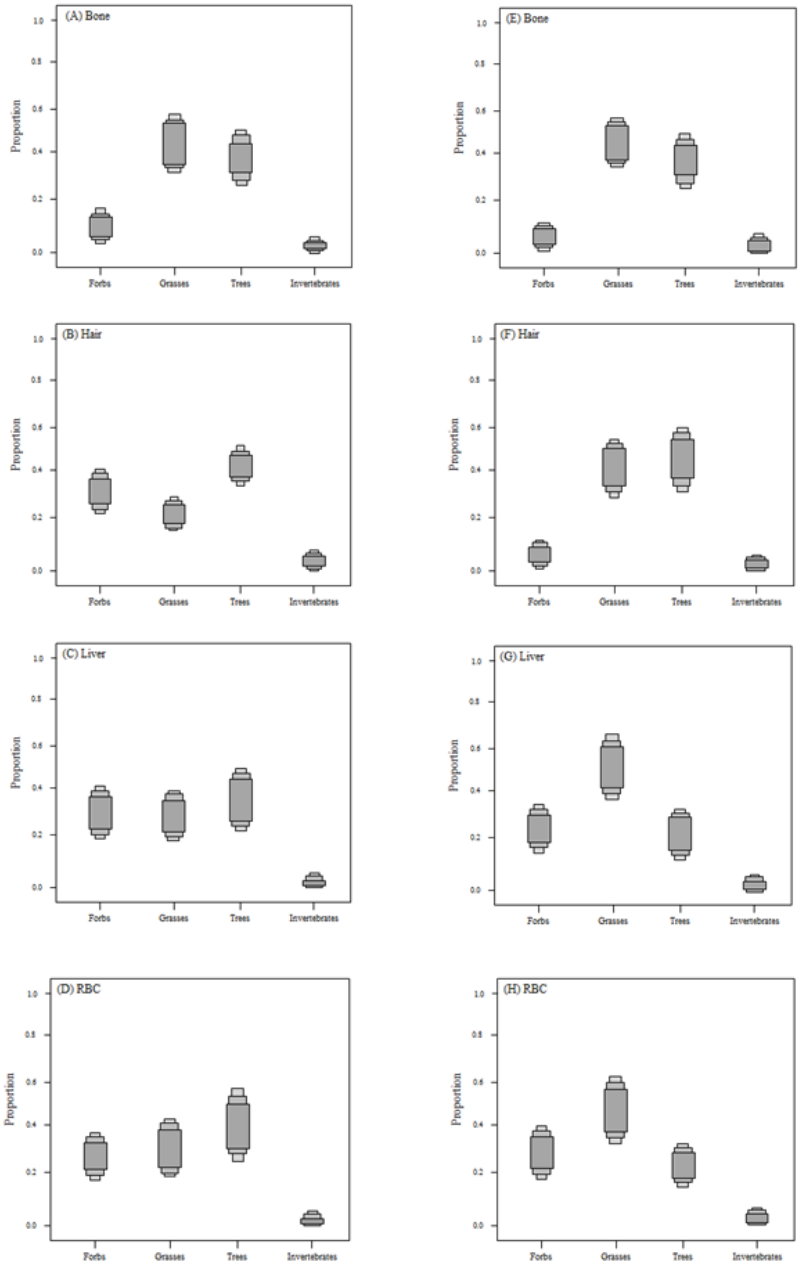

Figure 24. Relative proportions of isotopically distinct categories of prey in the diet of *M. natalensis* tissues at 2010 restored sites during the (A - D) wet season, and (E - H) dry season, as determined by a Bayesian isotopic mixing model. Box plots show the relative proportions for each food source with 95% (dark grey), 75%, 25% (medium grey) and 5% (light grey) confidence intervals.

Shrew hair had a range of $\delta^{15}N$ values: (1.07 ‰ - - 6.93 ‰) indicating that the group consumes predominately protein-rich insect food. I found no significant differences in carbon and nitrogen composition between seasons (Table 13). Carbon, but not nitrogen composition differed significantly among sites (Table13). Tukey HSD post hoc tests showed that carbon composition of shrew hair were significantly higher at 2014 restored sites than 2012 restored ($p = 0.003$, Fig. 25) and sugarcane ($p = 0.020$, Fig. 25) sites. I found no significant interactions between the carbon and nitrogen composition of shrew hair, season and sites (Table 13).

In the wet season, only one species, *C. cyanea*, was captured. The isotopic niche occupied by this species correlated to the site at which individuals were captured (Table 14; Fig. 25A).

Shrew hair had wider $\delta^{15}N$ ranges in the dry season compared to the wet season (Table 15). The isotopic niche occupied by species correlated to the site at which individuals were captured (Table 14; Fig. 25B). Species aggregation within the $\delta^{13}C$ - $\delta^{15}N$ plot was relatively low (Table 15; Fig. 25B). Two new species were captured in the dry season: *C. flavescens* which had higher $\delta^{13}C$ values than *S. infinitesimus* (Table 14; Fig. 25B).

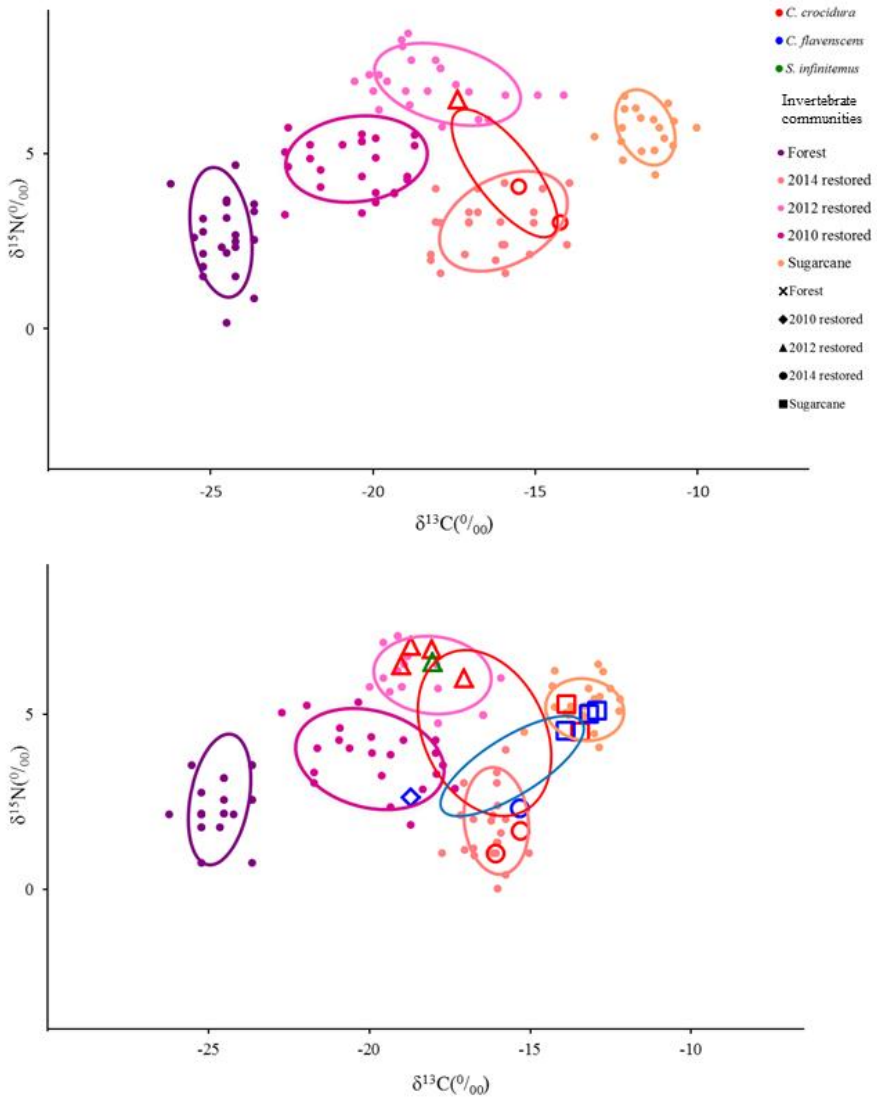

Figure 25. Standard ellipses for shrew's main sources of food based on hair collected at 2010, 2012 and 2014 restored sites, sugarcane sites and forest sites of the Buffelsdraai Landfill Site between November 2015 and July 2016 during the A) wet season and B) dry season. Individuals caught at the same study site are depicted in the same symbol (symbols depicted in legend). Invertebrate communities are delineated as ellipses (colours according to legend).

Using SIAR analysis I estimated dietary contributions based on shrew hair samples for shrew species within each study site during the wet and dry seasons to uncover whether species' relative consumption differed among restored sites and sugarcane.

According to the Bayesian model, irrespective of site, captured shrew species displayed diets with negligible differences, *C. cyanea*, *C. flavescens* and *S. infinitesimus* fed exclusively on invertebrates (Fig. 26, Table 18).

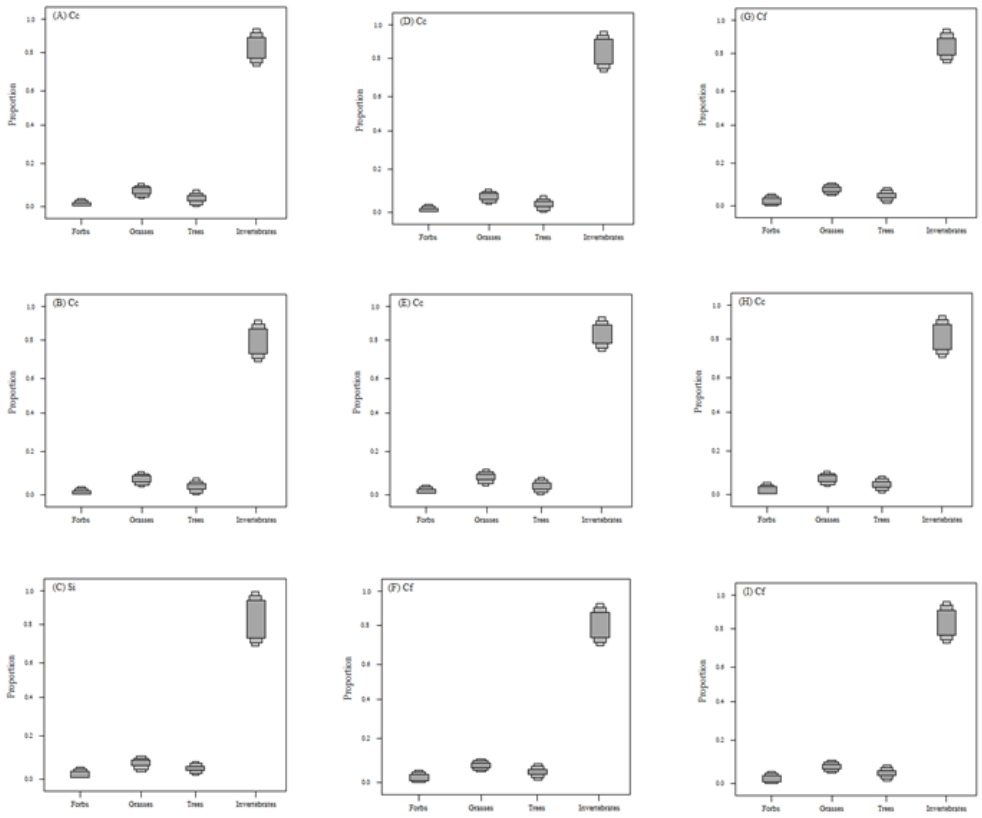

Figure 26. Relative proportions of isotopically distinct categories of prey in the diet *of C. cyanea* at 2012 restored sites (A) during the wet season, (B) during the dry season, (C) *S. infinitesimus* at 2012 restored sites during the dry season, *C. cyanea* at 2014 restored sites (D) during the wet season, (E) during the dry season, (D) *C. flavescens* at 2014 restored sites during the dry season, (G) *C. flavescens* at 2010 restored sites during the dry season, (H) C. cyanea at sugarcane sites during the dry season, and (I) *C. flavescens* at sugarcane sites during the dry season at Buffelsdraai Landfill Site during, as determined by a Bayesian isotopic mixing model. Box plots show the relative proportions for each food source with 95% (dark grey), 75%, 25% (medium grey) and 5% (light grey) confidence intervals.

Table 11. Test of rank equality of variances in carbon and nitrogen isotopic values of rodent and shrew hair, and *M. natalensis* tissues, between wet and dry seasons at sugarcane sites, 2014 restored, 2012 restored, 2010 restored and forest sites, of the Buffelsdraai Landfill Site between November 2015 and July 2016.

	Isotopes	**df**	**F-value**	**p-value**
Rodent hair	Carbon	10	0.911	0.524
	Nitrogen	10	0.637	0.781
Shrew hair	Carbon	5	0.807	0.568
	Nitrogen	5	0.397	0.841
M.natalensis				
Bone	Carbon	8	1.39	0.214
	Nitrogen	8	1.23	0.294
Hair	Carbon	9	0.99	0.452
	Nitrogen	9	0.822	0.597
Liver	Carbon	8	1.50	0.168
	Nitrogen	8	1.26	0.274
RBC	Carbon	6	0.85	0.565
	Nitrogen	6	0.79	0.604

Table 12: Layman metrics and $\delta^{13}C$ ranges for rodent hair and *Mastomys natalensis* tissues, between wet and dry seasons at sugarcane sites, 2014 restored, 2012 restored, 2010 restored and forest sites, of the Buffelsdraai Landfill Site between November 2015 and July 2016.

				DCC (‰)		NND (‰)	
	Season	**Tissue**	**δ13C range (‰)**	**Mean**	**Range**	**Mean**	**Aggregation in $\delta^{13}C - \delta^{15}N$ plot**
Rodent	Wet	Hair	-27.61 - -11.65	6.25	6.21 -6.58	0.63	0.74
	Dry	Hair	-26.06 - -11.33	6.36	6.32 – 6.4	0.62	0.83
M. natalensis	Wet	Bone	-22.55 - -12.73	7.52	7.31 – 7.68	0.41	0.63
	Dry	Bone	-24.23 - -13.56	7.48	7.26 – 7.80	0.31	0.58
	Wet	Hair	-25.08 - -11.33	5.89	5.71 – 5.98	0.74	0.65
	Dry	Hair	-25.23 - -10.65	5.76	5.62 – 5.85	0.65	0.61
	Wet	Liver	-26.37 - -15.83	4.72	4.49 – 4.96	0.46	0.34
	Dry	Liver	-22.29 - -10.92	4.68	4.52 – 4.79	0.44	0.65
	Wet	RBC	-26.37 - -16.03	4.41	4.28 – 4.73	0.63	0.46
	Dry	RBC	-22.45 - -12.46	4.48	4.29 – 4.64	0.30	0.62

DCC – distance to community centroid
NND – nearest neighbour distance

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Table 13. Two-way ANOVAs testing the differences in carbon and nitrogen isotopic values of rodent and shrew hair, and *M. natalensis* tissues, between seasons and sites at the Buffelsdraai Landfill Site between November 2015 and July 2016. Significant p-values are shown in bold.

		Season			Site			Season: Site		
	Isotopes	**df**	**F-value**	**p-value**	**df**	**F-value**	**p-value**	**df**	**F-value**	**p-value**
Rodent hair	Carbon	1	4.84	**0.029**	5	4.97	0.685	4	4.39	0.092
	Nitrogen	1	11.26	**< 0.001**	5	30.99	**< 0.001**	4	0.73	0.573
Shrew hair	Carbon	1	0.98	0.342	3	9.18	**0.002**	1	0.15	0.707
	Nitrogen	1	2.10	0.175	3	2.69	0.098	1	0.23	0.960
M. natalensis										
Bone	Carbon	1	21.12	**< 0.001**	4	4.04	0.065	3	2.67	0.053
	Nitrogen	1	4.41	**0.039**	4	20.24	0.125	3	2.55	0.063
Hair	Carbon	1	10.06	**0.002**	5	2.84	0.180	3	3.69	0.0821
	Nitrogen	1	11.60	**< 0.001**	5	36.97	0.091	3	1.04	0.376
Liver	Carbon	1	0.96	0.330	4	7.44	0.068	3	3.33	0.223
	Nitrogen	1	18.29	**< 0.001**	4	26.28	0.074	3	2.32	0.062
RBC	Carbon	1	32.43	**< 0.001**	3	2.55	0.129	2	3.08	0.102
	Nitrogen	1	2.83	0.131	3	11.34	0.003	2	0.68	0.540

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Table 14: Mean $\delta^{13}C$ and $\delta^{15}N$ isotope values of hair samples collected from rodent and shrew species between wet and dry seasons at sugarcane sites, 2014 restored, 2012 restored, 2010 restored and forest sites, of the Buffelsdraai Landfill Site between November 2015 and July 2016.

Species	**Season**	**$\delta^{13}C$ (Mean) $^{0}/_{00}$**	**$\delta^{15}N$ (Mean) $^{0}/_{00}$**
Rodents			
Aethomys ineptus	Wet	-25.27	2.17
Dendromus melanotis	Wet	-23.86	3.07
Graphiurus murinus	Wet	-27.61	4.77
Lemniscomys rosalia	Wet	-18.17	1.87
Mastomys natalensis	Wet	-17.18	2.45
Dendromus melanotis	Dry	-13.20	3.88
Grammomys dolichurus	Dry	-26.09	2.53
Lemniscomys rosalia	Dry	-15.73	1.45
Mastomys natalensis	Dry	-18.00	2.04
Mus minutoides	Dry	-15.47	3.62
Otomys auratus	Dry	-13.21	5.26
Steatomys pratensis	Dry	-23.48	1.52
Shrews			
Crocidura cyanea	Wet	-15.81	4.00
Crocidura cyanea	Dry	-15.79	7.47
Crocidura flavescens	Dry	-13.08	5.42
Suncus infinitesimus	Dry	-18.03	6.50

Table 15: Layman metrics and $\delta^{13}C$ ranges for shrew hair, between wet and dry seasons at sugarcane sites, 2014 restored, 2012 restored, 2010 restored and forest sites, of the Buffelsdraai Landfill Site between November 2015 and July 2016.

	Season	Tissue	δ15N range (‰)	DCC (‰) Mean	DCC (‰) Range	NND (‰) Mean	Aggregation in $\delta^{13}C - \delta^{15}N$ plot
Shrew	Wet	Hair	1.07 – 6.93	2.31	1.84 – 2.57	2.09	0.41
	Dry	Hair	3.03 – 6.52	2.23	1.02 – 2.65	2.84	0.44

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Table 16. Relative proportions of isotopically distinct categories of prey in the diet of rodent species at forest, 2010 restored, 2012 restored and sugarcane sites during the wet and dry seasons, as determined by a Bayesian mixing model.

Season	Site	Species	Food Sources											
			Forbs			Grasses			Trees			Invertebrates		
			Mode %	Lower 95%	Upper 95%	Mode %	Lower 95%	Upper 95%	Mode %	Lower 95%	Upper 95%	Mode %	Lower 95%	Upper 95%
Wet	Forest	Ai	9.8	2.6	13.6	7.4	1.4	10.7	67.8	52.2	82.4	4.4	0.2	8.6
	Forest	Gm	4.4	0.6	7.3	3.6	0.7	6.2	12.5	8.8	17.1	82.8	73.4	97.9
	2010	Ai	12.8	8.6	16.6	11.4	6.4	14.7	65.8	58.2	77.4	1.4	0.2	11.6
	2010	Dm	3.3	0.4	5.8	70.4	55.6	80.3	1.6	0.4	3.7	21.7	17.3	32.4
	2010	Lr	11.9	7.3	14.4	30..2	22.5	37.6	54.4	42.3	63.8	1.2	0.1	2.3
	2010	Mn	32.1	22.6	38.5	21.5	16.4	27.2	42.8	36.1	58.7	4.8	0.7	7.9
	SC	Mn	25.1	18.6	32.2	50.8	38.6	60.2	19.6	17.1	22.6	2.8	0.5	3.9
Dry	Forest	Gd	8.5	2.2	12.3	43.7	34.6	57.8	36.6	25.6	41.4	7.7	3.4	11.3
	2010	Dm	0.2	0.1	1.3	69.4	55.6	81.3	0.2	0.1	1.7	27.7	17.3	38.4
	2010	Lr	9.9	3.6	12.8	43.5	34.5	57.4	43.4	38.6	51.4	2.3	0.1	3.7
	2010	Mm	10.7	6.5	13.9	42.6	31.5	57.4	31.8	22.7	40.9	8.6	3.5	11.7
	2010	Mn	7.6	1.1	12.6	41.1	30.6	53.2	45.8	36.6	58.9	2.8	0.7	3.9
	2010	Oi	9.3	4.4	16.1	83.5	72.4	96.6	5.8	2.2	8.9	0.7	0.1	1.9
	2010	Sp	26.7	17.2	36.7	32.5	21.8	41.6	20.1	15.5	28.9	12.3	16.2	8.1
	2012	Mn	13.6	6.1	18.6	32.1	21.6	43.2	39.8	24.6	51.9	11.8	8.7	13.9
	SC	Mn	41.8	37.6	45.9	26.1	18.6	37.2	9.8	4.7	13.9	15.4	10.1	19.6

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Table 17. Relative proportions of isotopically distinct categories of prey in the diet of *M. natalensis* tissues at 2010 restored sites during the wet and dry seasons, as determined by a Bayesian mixing model.

Tissue type	Season	Food Sources											
		Forbs			Grasses			Trees			Invertebrates		
		Mode %	Lower 95%	Upper 95%	Mode %	Lower 95%	Upper 95%	Mode %	Lower 95%	Upper 95%	Mode %	Lower 95%	Upper 95%
Bone	Wet	10.2	3.5	14.4	42.8	34.6	56.7	38.2	27.5	48.7	1.1	0.0	3.2
	Dry	6.3	1.2	11.3	46.1	37.3	57.2	37.6	26.4	45.1	1.6	0.0	3.8
Hair	Wet	32.1	22.6	38.5	21.5	16.4	27.2	42.8	36.1	58.7	4.8	0.7	7.9
	Dry	7.6	1.1	12.6	41.1	30.6	53.2	45.8	36.6	58.9	2.8	0.7	3.9
Liver	Wet	30.4	19.9	38.7	29.6	19.3	38.1	36.7	22.7	44.5	0.5	0.0	2.3
	Dry	23.7	17.2	32.4	52.1	38.6	61.5	20.8	13.9	31.6	0.9	0.0	4.3
RBC	Wet	27.1	18.2	36.1	30.4	19.8	40.7	39.9	27.4	45.6	0.7	0.0	1.4
	Dry	28.2	19.4	38.7	49.3	38.8	60.0	20.9	17.6	34.1	1.2	0.0	3.1

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Table 18. Relative proportions of isotopically distinct categories of prey in the diet of shrew species at 2010 restored, 2012 restored, 2014 restored and sugarcane sites during the wet and dry seasons, as determined by a Bayesian mixing model.

Tissue type	Season	Food Sources											
		Forbs			Grasses			Trees			Invertebrates		
		Mode %	Lower 95%	Upper 95%	Mode %	Lower 95%	Upper 95%	Mode %	Lower 95%	Upper 95%	Mode %	Lower 95%	Upper 95%
Bone	Wet	10.2	3.5	14.4	42.8	34.6	56.7	38.2	27.5	48.7	1.1	0.0	3.2
	Dry	6.3	1.2	11.3	46.1	37.3	57.2	37.6	26.4	45.1	1.6	0.0	3.8
Hair	Wet	32.1	22.6	38.5	21.5	16.4	27.2	42.8	36.1	58.7	4.8	0.7	7.9
	Dry	7.6	1.1	12.6	41.1	30.6	53.2	45.8	36.6	58.9	2.8	0.7	3.9
Liver	Wet	30.4	19.9	38.7	29.6	19.3	38.1	36.7	22.7	44.5	0.5	0.0	2.3
	Dry	23.7	17.2	32.4	52.1	38.6	61.5	20.8	13.9	31.6	0.9	0.0	4.3
RBC	Wet	27.1	18.2	36.1	30.4	19.8	40.7	39.9	27.4	45.6	0.7	0.0	1.4
	Dry	28.2	19.4	38.7	49.3	38.8	60.0	20.9	17.6	34.1	1.2	0.0	3.1

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CHAPTER 4

DISCUSSION

In this study, I measured three major ecological attributes to investigate forest-restoration success at Buffelsdraai Landfill Site. I measured the vegetation structure at sites, quantified the expected species richness and diversity of rodents and shrews at each site; and investigated the diet and trophic structure of small mammal assemblages within and among sites, using stable isotope analysis of hair and tissue samples collected from rodents and shrews. I found mixed support for the three proposed hypotheses. Complexity of vegetation structure increased with age of restoration sites. Rodent and shrew species exhibited species-specific responses to increased levels of forest restoration; abundance of the generalist *M. natalensis* was higher than those of *A. ineptus* and *M. minutoides* that are more sensitive to disturbance. Small mammal abundance but not species richness increased with increased age of forest restoration. The C/N isotope ratios of small mammal assemblages were closely associated with older, well-established restored sites.

4.1. Vegetation structure of restored sites

I found evidence for successional changes in vegetation at the restored sites. The 2010 restored sites had the highest tree species richness and tree density. Additionally, the 2010 restored sites had significantly greater tree heights than the 2014 restored sites and a greater tree canopy cover than the other restored sites. The vegetation structure at 2010 restored sites had good canopy cover and less ground cover, similar to forests. Structural complexity of these restoration sites can be attributed to the high density of diverse trees that vary in growth rates and canopy cover, as well as the dominant grassy ground cover that was present prior to planting events. To restore the forest at Buffelsdraai, the goal was to produce a closed canopy within a few years of establishment, which would suppress grasses and other shrubs, and maintain a microclimate that facilitates the recruitment of mature forest trees (Kooyman, 1991; Parrotta, 1995; Parrotta and Knowles, 1999). This was largely achieved at the 2010 restored sites, because they had the longest time to establish themselves and suppress grasses. Reference forests had the highest tree height and canopy cover, and the lowest forb and grass species richness, and grass height and cover, indicating that the restored sites lack mature forest trees.

The 2012 restored sites had the highest grass height and cover probably because more time is required for the planted trees to suppress grasses similarly to the 2010 restored sites. Further, 2012 restored sites had greater tree height and canopy cover than 2014 restored sites. The 2014 restored sites had the highest forb and grass species richness, yet lowest grass height and canopy cover, because trees planted across these sites are young and are not established and these sites are recently disturbed and recovering.

Newly restored forests are unlikely to quickly develop into a complex forest on land that was used for agricultural production for an extended period of time (Wade *et al.,* 2008). Rehabilitation of human-disturbed land can take more than 7 years. For example, following the abandonment of coffee plantations, it took 30 – 40 years for forests to become similar to mature forests in Puerto Rico (Zimmerman *et al.*, 2007). Similarly, only after 35 years of limestone forest restoration in Vietnam more than 30 species of rare and endemic mammal species were recorded (Poffenberger, 2006). A forest restoration project in Tanzania took 18 years before the landscape was restored to a state where the community could continue their pastoralist practices (Monela *et al.,* 2004). Nicolas *et al.,* (2009) found that restoration of vegetation structure, which involved clear-cutting of cultivated lands and planting of seedlings aided by natural vegetation, was evident 10 – 34 years post-restoration. A restoration project in South Africa found that the highest diversity of small mammals was in 8-11 year old rehabilitated sites, suggesting that coastal dune forest restoration was successful (Ferreira & Van Aarde, 1996).Clearly there is great variability in the amount of time required for replanted forests to be considered mature forests (Aide *et al.,* 2000).

4.2 Does diversity of small mammals increase in response to reforestation?

A total of nine rodent species belonging to the family Muridae were captured at Buffelsdraai Landfill Site. Muridae is the largest mammal family worldwide and is represented in southern Africa by 64 species from 25 genera (Skinner and Chimimba, 2005). Based on the Chao 2 and Jacknife 2 species richness estimators, my species inventories for rodents at the restored sites and sugarcane fields were fairly complete (59% - 96%). The low estimate for completeness (~60%) of the forest sites can be attributed to the high number of singletons and doubletons (n = 5; *G. murinus* and *G. dolichurus*) because the richness estimators are strongly influenced by rare species in the assemblages (Gotelli & Colwell, 2001).

Individuals which are trapped more frequently than others are referred to as trap happy. Trap happy animals such as *Mastomys natalensis* are animals that recall rewards (bait) upon

capture and are captured repeatedly (Schradin & Pillay, 2004). This may explain why *M.natalensis* was the most commonly captured species in my study. By contrast, species like *G. murinus* and *G. dolichurus* are considered trap shy (Rautenbach *et al.,* 2013), they learn to avoid traps once they are captured. This may be the reason I seldomly captured these species at Buffelsdraai. However, Avenant & Cavallini (2007) found that during winter when resources were scarce trap shy species often entered traps to eat the bait.

As predicted, rodent abundance was significantly higher at 2010 restored sites than newly restored sites and sugarcane sites. The 2010 restored sites had some grassy layers with well-established tree species, providing ample food resources (Habtamu & Bekele, 2013), microhabitats (Kryštufek *et al.,* 2008) and protection from predators for rodent species (Avenant & Cavallini, 2007). Two rodent species represented most of the captures at the Buffelsdraai landfill Site: *M. natalensis* and *L. rosalia*. *Mastomys natalensis* often dominates rodent assemblages in southern Africa (Monadjem, 1997; Caro, 2001; Avenant, 2002; Monadjem and Perrin 2003; Sluydts *et al.,* 2009). This highly adaptable species (Meester *et al.,* 1979, Smith *et al.,* 2002) is widely distributed, and has a wide habitat tolerance (van Deventer & Nel, 2006) and generalist diet (Monadjem, 1997; Rowe-Rowe, 1995; Mulungu *et al.,* 2011). Further, *M. natalensis* has a high breeding rate with short intervals between exceptionally large litters (Coetzee, 1975; Leirs *et al.,* 1993). It is among the most pervasive and successful invasive mammals in Africa (Leirs, 1995; Sluydts *et al.,* 2009), and is responsible for major changes in ecological communities in areas where they have been introduced (Mwanjabe *et al.,* 2002). Its presence is indicative of habitat disturbance (Kneidinger, 2008; Avenant, 2011; MacFadyen *et al.,* 2012).

In contrast, I found no significant differences in rodent species richness among study sites. At identical sampling efforts rodent species richness was highest at forest sites, and lowest at 2010 and 2012 restored sites. Forests often have high rodent species richness (Ecke *et al.,* 2002; Williams *et al.,* 2002), probably because they are structurally complex environments, with a greater number of trophic and habitat niches available (Tews *et al.,* 2004). Two rare species, *G. dolichurus* and *G. murinus* (Delcros *et al.,* 2015) were only captured in forest sites. Both species are well adapted to forest environments. *G. murinus* is an arboreal species, that nests in tree holes and rock crevices (Wirminghaus & Perrin, 1993; Skinner & Smithers, 1990; Lamani, 2014) that are located well off the ground (Lamani, 2014). It forages solitary in trees searching for fruits (Skinner & Chimimba, 2005), beetles (Baxter *et al.,* 2005) and tiny lizards (Lamani, 2014). The species uses high densities of arboreal connections to forage effectively (Kaplan, 1995). Indeed, this species avoids isolated trees,

and the colonisation of this species is highly dependent on wooded corridors (Madikiza *et al.,* 2010). *Grammomys dolichurus* requires thick vegetation (Monadjem *et al.,* 2015), and has been well documented in forests (Bayliss *et al.,* 2014). However, this species is not considered a forest specialist as it has also been found in habitats with thick herbaceous cover and shrublands (Ralaizafisoloarivony *et al.,* 2014). Rodent forest communities are negatively influenced by disturbance in the surrounding landscapes, which might explain why other common forest-dwelling species such as *Grammomys cometes*, were not captured in this study (Happold, 1975; Malcolm & Ray, 2000).

Rodent assemblage structure depends strongly on local disturbances and the structure and history of the surrounding landscape (Malcolm & Ray, 2000). In southern Africa recently restored sites are often dominated by *M. natalensis (*Meester *et al.,* 1979; Ferreira & van Aarde, 1999). Additionally, recently restored sites exhibit lower rodent diversity than older restored sites (Ferreira & van Aarde, 1996). Low rodent species richness at recently restored sites indicates high level of disturbance, and species composition comprises mainly opportunistic species, with specialist species largely absent (Mbugua, 2002). Fire events play important roles in small mammal community dynamics. Small mammal populations often decline post-fire (Sutherland & Dickman, 1999; Fuller & Perrin, 2001; Letnic & Dickman, 2005). These declines in abundance have been linked to changes in vegetation structure (Monadjem & Perrin, 2003), specifically reduced vegetation cover , increased predation risk (Sutherland & Dickman, 1999), and reduces the availability of food (Yarnell *et al.,* 2007). Therefore local fire regimes should be considered in future analyses. Although the 2010, 2012 and 2014 restored sites in this study were subject to different fire management practices and differed in vegetation structure, there were no significant differences in rodent richness, perhaps because there were no differences in microclimate (Stevens & Husband, 1998; Osbourne *et al.,* 2005; Püttker *et al.,* 2008), yet this was not tested. Similarly, Hurst *et al.,* (2013) found no significant differences in rodent diversity between restored and sugarcane sites.

Contrary to predictions, rodent species composition at all restoration sites were similar to the rodent composition of the original land cover, sugarcane. Caro (2001) found that small mammal species richness was lower within a national park in western Tanzania compared to agricultural sites outside the reserve. Additionally, Jeffery (1977) found that the removal of forests for agricultural use resulted in an increase in diversity and abundance of rodents. These studies suggest that agricultural practices may be beneficial to certain rodent species. One reason may be because predator abundance is lower in agricultural sites (Caro, 2001).

By contrast, Hurst *et al.,* (2013) found that sugarcane sites had lower rodent species richness than restored sites. Specifically, sugarcane sites were dominated by *M. natalensis* and *L. rosalia*. Agricultural practices may have minimal effects on generalist and herbivorous species, but negatively affect more specialised rodent species (Atkeson & Johnson, 1979; Wretenberg *et al.,* 2006). On the other hand, Van Aarde *et al.,* (1996), found that the rodent species composition at restored sites was most similar to species composition at unaltered sites, suggesting restoration sites at Buffelsdraai Landfill Site are not yet completely restored. To better understand rodent species composition at restored sites, more data on species-specific habitat preferences, movement between unaltered and restored sites and interactions between species are necessary (Ferreira & Van Aarde, 2000).

Three shrew species from two genera were captured at Buffelsdraai Landfill Site. Seventeen shrew species from four genera belonging to the family Soricidae are found in southern Africa (Skinner & Chimimba, 2005). Thirteen of those species are found in KwaZulu-Natal. Based on the species richness estimators, species inventories for shrews at the restored sites and sugarcane fields were fairly complete (64% - 100%). Shrews were captured at all sites except forests. At identical sampling efforts species richness was highest at the 2012 restored sites (n = 3 spp), and lowest at the 2010 restored sites (n = 2 spp). Contrary to predictions, there were no significant differences in shrew abundance among study sites.

The shrew species that represented most of the captures was *C. cyanea*. This species often dominates southern African assemblages (Monadjem, 1997; Avenant, 2002). *Crocidura cyanea* has a wide habitat tolerance, is predominantly nocturnal and terrestrial (Happold & Happold, 2013), and selects habitats with dense ground cover that provides shelter from predators (Dickman, 1995). Additionally, habitats with dense ground cover increase their access to preferred types of prey as they are able to forage through leaf litter easily (Dickman, 1995).

The second most common shrew species captured, *C. flavescens,* is commonly associated with habitats modified by humans (Rowe-Rowe & Meester, 1982). In support, *C. flavescens* was trapped at sugarcane sites, and all the restored sites, except forests. This shrew has a wide habitat tolerance and is commonly found at sites close to water with sufficient ground cover (Dippenaar & Baxter, 2013). Similarly, shrew species including *C. flavescens* were captured near a large pond at the sugarcane sites.

Only one *S. infinitesimus* was captured at one 2012 restored site, hence the high shrew species richness of this site. However, this species occurs in a wide range of habitats and is regarded as fairly common in KwaZulu-Natal (Taylor, 1998). The low presence of this species suggests that historical, environmental or biotic processes prevented their establishment at restored sites (Dippenaar & Baxter, 2013). When environmental conditions are not favourable *S. infinitesimus* reduces its cost of metabolism by using abandoned termitaria where microclimates are stable, and in some cases enters a state of torpor (Dippenaar & Baxter, 2013). Further, *S. infinitesimus* rarely enters traps even in cases where traps are situated alongside termitaria (Avenant, 2011). These behavioural traits may explain the low trap success of this species at Buffelsdraai.

Although the main difference in species richness across sites can be attributed to the capture of a single *S. infinitesimus*, differences in shrew richness among sites could be due to differences in habitat features. Shrew species richness is strongly correlated to vegetation features such as tree height and grass height because these characteristics provide protection against predators (Monadjem & Perrin, 2003). Additionally, low leaf litter depth can negatively impact the abundance of shrew species (Greenberg *et al.*, 2007), however leaf litter depth was not measured. Additionally, earthworms make up an important component of many shrew species' diet, however I did not find earthworms at any of the study sites. Earthworm diversity may be low at Buffelsdraai because earthworms are sensitive to land use changes including agricultural practices (Tondoh *et al.*, 2007; de Vries *et al.*, 2013; Dewi & Senge, 2015). An environmental assessment performed in 2011 recorded one shrew species at Buffelsdraai Landfill Site: a single *Suncus lixus* individual which was caught in the forest. This suggests that shrew abundance and species richness has increased at the Buffelsdraai Landfill Site.

Both rodent and shrew abundance was higher in winter (dry season) than summer (wet season). This is surprising given that food supply and plant cover is usually higher in the wet season (Mortelliti & Boitani, 2009; Lima *et al.*, 2001). In support, Habtamu & Bekele (2008), Lamani (2014), Workeneh *et al.*, (2012), Hurst *et al.*, (2013) and Rautenbach *et al.*, (2014), found that small mammal diversity was higher during the wet summer months. Indeed, seasonal variation in rainfall influences the breeding season of small mammals (Monadjem, 1998; Makundi *et al.*, 2007). On the other hand, previous studies in southern Africa also found higher small mammal diversity during the dry winter months (Cheeseman & Delany, 1979; Fuller & Perrin, 2001; Monadjem & Perrin, 2003; Schradin & Pillay, 2006, Habtamu & Bekele, 2013). One reason may be the delayed response in the temporal

availability of resources (Hernandez *et al.,* 2005). Alternatively, high food availability during the wet season may have rendered the bait in the traps less attractive to rodents than during the dry season when food abundance is low (Monadjem, 1999). Additionally, rodent species richness and abundance may decrease when productivity is high because strong competitors may exclude other species when resources are limiting (Perrin & Bodbijl, 2001).

4.3. Stable isotope composition of small mammals in response to restoration

Regardless of season, the overall isotopic niche occupied by rodent species was greatest at the 2010 restored sites. Stable isotope composition of *D. melanotis*, *G. dolichurus*, *L. rosalia*, *M. natalensis*, *M. minutoides*, *O. auratus* and *S. pratensis* aggregated within the stable isotope composition of the vegetation and insects of the 2010 restored sites. This suggests that these species' diets were most similar to the plants and invertebrates present at the 2010 restored sites. Except, *A. ineptus* and *G. murinus* were strongly associated with forest sites where they were captured. These results are consistent with evidence that small mammals utilised restored sites more than reference sites (Converse *et al.,* 2006).

Rodents conformed to their presumed diets (Hanney, 1965; Rowe-Rowe, 1986; Ellison, 1990; Wirminghaus & Perrin, 1992; Leirs *et al.,* 1994; Miller, 1994; Monadjem 1997; Monadjem, 1999). Further, diets of rodents exhibited little variation between sites and seasons. Except *M. natalensis* captured at 2010 and 2012 restored sites had diets that comprised largely of tree material and grass leave, seeds and stems, whereas individuals captured at sugarcane sites fed mainly on forbs and grass seeds and stems. Further, *M. natalensis* captured at 2010 restored sites consumed a higher percentage of grasses during the dry months compared to the wet months, and individuals captured at the sugarcane sites consumed mainly grasses during the wet months, and green plant material during the dry months. *Mastomys natalensis* is a highly opportunistic generalist, whose diet reflects what its habitat provides (Caro, 2001).

Nonetheless, rodents exhibited some plasticity in their diets. Rodents consumed both C_3 and C_4 plants, yet carbon composition of rodent hairs were more enriched during the dry season because rodents consumed primarily abundant C_4 plants (Symes *et al.,* 2013). Nitrogen isotopic composition were more enriched during the dry season. An enrichment of nitrogen isotopes in animal tissues is generally associated with aridity (Popa-Lisseanu *et al.,* 2015), when an animal is fasting or resources are limiting (Hobson *et al.,* 1993), and increase in the consumption of seeds because vegetation is less abundant (Nakagawa, 2007).

My results show that irrespective of site captured and tissue type analysed, *M. natalensis* exhibited diet most similar to the carbon and nitrogen isotopic composition of vegetation at the 2010 restored sites. The stable isotopes of bone, hair, liver and RBC tissues collected from *M. natalensis* represented different periods of feeding, because each tissue has a different metabolic turnover (Tiezen *et al.*, 1983). Specifically, red blood cells - two weeks (Russel & Bernstein, 1966); liver - one month (MacAvoy *et al.*, 2005); hair - four to six months (Kurle, 2009); and bone - a year (DeNiro & Epstein, 1981).The isotope values of the different tissues reflected a consistent pattern: diets of *M. natalensis* remained most similar to the trophic resources available at the 2010 restored sites. This indicates that consistently, for up to a year *M. natalensis* individuals, had diets comprising vegetation most similar to the isotopic composition of vegetation at the 2010 restored sites, irrespective of the site of capture.

Conversely, isotopic composition of shrew hairs were most similar to the site at which individuals were captured. Additionally, there were no seasonal differences in carbon and nitrogen isotopic composition of shrew hairs. Shrews consumed invertebrates exclusively. There were no differences in *C. cyanea*, *C. flavescens* and *S. infinitesimus* diets among sites and between seasons. To the best of my knowledge, this is the first study to investigate carbon and nitrogen isotopic composition of shrew hairs at restoration sites.

4.4. Caveats

The main caveats of this study are as follows. Cryptic rodent or shrew taxa may have been overlooked. In southern Africa, there are probably a number of cryptic species complexes in small mammal lineages such as *Aethomys* (Linzey *et al.*, 2003), *Grammomys* (Monadjem *et al.*, 2015) and *Mastomys* (Venturi *et al.*, 2004). Future studies should include DNA analyses of specimens captured in the field. Additionally, future studies should consider the influence of body condition on dietary niches.

Future studies should analyse the substrates of different sites, and determine if substrate per se plays a role in community structure and diversity at restored sites. Additionally, more detailed analyses of vegetation structure and diversity should be included in future studies.

Fire events play important roles in small mammal community dynamics. Small mammal populations have been recorded to decline post-fire (Sutherland & Dickman, 1999; Fuller & Perrin, 2001; Letnic & Dickman, 2005). Such observations are linked to changes in vegetation structure (Monadjem & Perrin, 2003), with reduced vegetation cover there is increased predation risk (Sutherland & Dickman, 1999). Fire also influences the availability of food (Yarnell *et al.,* 2007). Therefore this should be considered in future analyses.

A limited number of sites were sampled using only sherman-like traps. Although species richness indicators suggest that inventories were fairly complete, small mammal diversity, particularly shrew diversity at Buffelsdraai may be an underestimate. Specifically, pitfall traps may be more effective than sherman traps to sample shrews (Rautenbach *et al.,* 2014). Future studies should incorporate additional sites and use different trapping methods to verify the small mammal diversity reported in this study.

Additionally I sampled small mammal communities for 1 year only. Small mammal assemblage dynamics often show marked changes among seasons and across years (Monadjem & Perrin, 2003, but see Avenant, 2005, 2011; Avenant & Cavallini 2007; Avenant *et al.,* 2008 for contrasting results. Long term studies are necessary to consider seasonal and yearly variation in rodent and shrew population levels (Pearce & Venier, 2005), therefore future studies should increase sampling intensity so that fine-grained dietary patterns can be analysed.

Three processes can potentially complicate the reconstruction of diets from stable isotopes (Gannes *et al.,* 1997): dietary components may be integrated at different efficiencies; isotopic fractionation changes isotopic values in tissue relative to the source; and metabolic routing which will disproportionally distribute the source element among different tissues. All three approaches are based on the basic principle of tissue specific isotopic turnover. Because I analysed different tissues with different turnover rates, the results reflect the average diet of individuals (Tiezen *et al.,* 1983). Furthermore, species-specific diet-tissue fractionation factors should be determined under laboratory conditions for southern African rodents (Arneson & MacAvoy, 2005; Miller *et al.,* 2008; MacAvoy *et* al., 2012).

4.5 Management implications

The results of this study have important implications for the design and management of forest restoration projects in agricultural and urban landscapes. First, forests cannot be restored in a short period of time (Kanowski *et al.,* 2003). For example, only after 18 years

did ant assemblages in restored sites in KwaZulu-Natal begin to resemble ant assemblages in reference forests (Majer & de Kock, 1992). Nonetheless, the 2010 restored sites did appear to provide trophic resources that most resident rodents preferred, hence there is evidence that there has been progressive succession in the scarp forest after 10 years.

Second, rodents may be better bioindicators of restoration success than shrews. Herbivores and granivores may be better bioindicators than insectivores because they have direct trophic links with the restored vegetation, whereas insectivores are indirectly related via the invertebrates that they feed on (Keesing, 2000; Goheen *et al.*, 2004; Hurst *et al.*, 2014). However, the results for shrews may simply be an artefact of sampling methods, given that shrews were sampled with less effective methods than the rodents.

4.6 Conclusions

To assess restoration success I took a multi-pronged approach, investigating three ecological attributes that are key indicators (Ruiz-Jaen & Aide, 2005). My results suggest that the reforestation effort at Buffelsdraai Landfill site has been successful: vegetation structure increased significantly in complexity and cover from sugarcane to 2010 restored sites; small mammal abundance increased at the restored sites with the highest abundance recorded at the 2010 restored sites; and trophic resources found at the 2010 restored sites were preferred by most rodents.

This study is the first to assess restoration success using these three ecological attributes, and therefore provides baseline data to assess the restoration success in other human-impacted landscapes. This study highlights the value of focussing on the smaller, less conspicuous small mammal species and taking a holistic research approach to restore biodiversity in human-impacted landscapes, with a view to achieve goals within the broader conservation agenda (Entwistle & Dunstone, 2000).

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APPENDICES

Appendix 1: Collection dates and number of invertebrate specimens collected of each order present at each study site of the Buffelsdraai Landfill Site between November 2015 and July 2016.

Date collected	Study Site	Order	No. of specimens collected
06 - 11 - 2015	Sugarcane	Araneae	5
		Hymenoptera	32
		Orthoptera	12
	2014 restored	Araneae	9
		Coleoptera	5
		Hemiptera	6
		Hymenoptera	20
		Orthoptera	13
13 - 11 - 2015	2012 restored	Araneae	4
		Diplopoda	3
		Hymenoptera	21
		Lepidotera	2
		Orthoptera	7
	2010 restored	Araneae	8
		Hemiptera	12
		Hymenoptera	29
		Lepidotera	11
		Orthoptera	18
20 - 11 - 2015	Forest	Araneae	14
		Coleoptera	8
		Hemiptera	21
		Hymenoptera	35
		Orthoptera	13
13 – 05 - 2016	Sugarcane	Araneae	1
		Hemiptera	10
		Hymenoptera	23
		Orthoptera	15
	2014 restored	Araneae	5
		Coleoptera	7
		Hymenoptera	33
		Lepidotera	12
		Orthoptera	25
20 - 05 - 2016	2012 restored	Araneae	2
		Coleoptera	10
		Hemiptera	11
		Hymenoptera	23
		Lepidotera	17
		Orthoptera	20
	2010 restored	Araneae	3
		Hemiptera	4
		Hymenoptera	13
		Orthoptera	18
21 - 05 - 2016	Forest	Araneae	8
		Coleoptera	12

	Hymenoptera	24
	Lepidotera	9
	Orthoptera	15

Appendix 2: Collection dates and the part of each plant collected at each study site of the Buffelsdraai Landfill Site between November 2015 and July 2016.

Date collected	**Study Site**	**Species**	**Tree/ Forb/ Grass/ Sugarcane**	**Leaves/Stem/Fruit/Seeds**
05 - 11 - 2015	Sugarcane	*Acacia caffra*	Tree	Leaves & stem
		Acacia sieberiana	Tree	Leaves & stem
		Erythrina lysistemon	Tree	Leaves & stem
		Melia azedarach	Tree	Leaves & stem
		Trichilia dregeana	Tree	Leaves & stem & fruit
		Forb spp. 1	Forb	Leaves & stem
		Forb spp. 2	Forb	Leaves & stem
		Forb spp. 3	Forb	Leaves & stem
		Forb spp. 4	Forb	Leaves & stem
		Forb spp. 5	Forb	Leaves & stem
		Forb spp. 6	Forb	Leaves & stem
		Aristida spp.	Grass	Leaves & stem
		Eragrostis curvula	Grass	Leaves & stem & seeds
		Panicum natalense	Grass	Leaves & stem & seeds
		Saccharum officinarum	Sugarcane	Leaves & stem
06 – 11 - 2015	2014 restored	*Acacia caffra*	Tree	Leaves & stem
		Acacia sieberiana	Tree	Leaves & stem
		Brachylaena discolor	Tree	Leaves & stem
		Bridelia micrantha	Tree	Leaves & stem & fruit
		Erythrina lysistemon	Tree	Leaves & stem
		Millettia grandis	Tree	Leaves & stem
		Strelitzia nicolai	Tree	Leaves & stem
		Syzigium cordatum	Tree	Leaves & stem
		Trichilia dregeana	Tree	Leaves & stem
		Ziziphus mucronata	Tree	Leaves & stem
		Forb spp. 1	Forb	Leaves & stem
		Forb spp. 2	Forb	Leaves & stem
		Forb spp. 3	Forb	Leaves & stem
		Forb spp. 4	Forb	Leaves & stem
		Forb spp. 5	Forb	Leaves & stem
		Forb spp. 6	Forb	Leaves & stem
		Forb spp. 7	Forb	Leaves & stem
		Forb spp. 8	Forb	Leaves & stem
		Forb spp. 9	Forb	Leaves & stem
		Eragrostis curvula	Grass	Leaves & stem & seeds
		Melinis repens	Grass	Leaves & stem & seeds
		Panicum maximum	Grass	Leaves & stem & seeds
		Panicum natalense	Grass	Leaves & stem & seeds
		Themeda trianda	Grass	Leaves & stem & seeds
12 - 11 - 2015	2012 restored	*Acacia caffra*	Tree	Leaves & stem
		Acacia natalitia	Tree	Leaves & stem
		Acacia sieberiana	Tree	Leaves & stem

		Brachylaena discolour	Tree	Leaves & stem
		Bridelia micrantha	Tree	Leaves & stem & fruit
		Clerodendrum glabrum	Tree	Leaves & stem
		Erythrina lysistemon	Tree	Leaves & stem
		Ficus glumosa	Tree	Leaves & stem
		Millettia grandis	Tree	Leaves & stem
		Strelitzia nicolai	Tree	Leaves & stem
		Syzigium cordatum	Tree	Leaves & stem
		Trichilia dregeana	Tree	Leaves & stem & fruit
		Ziziphus mucronata	Tree	Leaves & stem
		Forb spp. 1	Forb	Leaves & stem
		Forb spp. 2	Forb	Leaves & stem
		Forb spp. 4	Forb	Leaves & stem
		Forb spp. 6	Forb	Leaves & stem
		Forb spp. 10	Forb	Leaves & stem
		Eragrostis curvula	Grass	Leaves & stem & seeds
		Melinis repens	Grass	Leaves & stem & seeds
		Panicum maximum	Grass	Leaves & stem & seeds
		Panicum natalense	Grass	Leaves & stem & seeds
		Themeda trianda	Grass	Leaves & stem & seeds
13 - 11 - 2015	2010 restored	*Acacia caffra*	Tree	Leaves & stem
		Acacia natalitia	Tree	Leaves & stem
		Acacia sieberiana	Tree	Leaves & stem
		Albizia adianthifolia	Tree	Leaves & stem
		Brachylaena discolour	Tree	Leaves & stem
		Bridelia micrantha	Tree	Leaves & stem & fruit
		Clerodendrum glabrum	Tree	Leaves & stem
		Dombeya rotundifolia	Tree	Leaves & stem
		Erythrina lysistemon	Tree	Leaves & stem
		Ficus glumosa	Tree	Leaves & stem
		Ficus sur	Tree	Leaves & stem
		Millettia grandis	Tree	Leaves & stem
		Schotia brachypetala	Tree	Leaves & stem
		Strelitzia nicolai	Tree	Leaves & stem
		Syzigium cordatum	Tree	Leaves & stem
		Trichilia dregeana	Tree	Leaves & stem
		Ziziphus mucronata	Tree	Leaves & stem
		Forb spp. 1	Forb	Leaves & stem
		Forb spp. 2	Forb	Leaves & stem
		Forb spp. 4	Forb	Leaves & stem
		Forb spp. 7	Forb	Leaves & stem
		Forb spp. 10	Forb	Leaves & stem
		Forb spp. 11	Forb	Leaves & stem
		Eragrostis curvula	Grass	Leaves & stem & seeds
		Melinis repens	Grass	Leaves & stem & seeds
		Panicum maximum	Grass	Leaves & stem & seeds
		Panicum natalense	Grass	Leaves & stem & seeds
20 - 11 - 2015	Forest	*Albizia adianthifolia*	Tree	Leaves & stem
		Combretum edwardsii	Tree	Leaves & stem
		Dalbergia armata	Tree	Leaves & stem
		Dalbergia obovata	Tree	Leaves & stem
		Dichrostachys cinerea	Tree	Leaves & stem
		Dombeya rotundifolia	Tree	Leaves & stem

		Ficus burtt-davyi	Tree	Leaves & stem
		Ficus glumosa	Tree	Leaves & stem
		Heteropyxis natalensis	Tree	Leaves & stem
		Schotia brachypetala	Tree	Leaves & stem
		Scolopia zeyheri	Tree	Leaves & stem
		Searsi chirindensis	Tree	Leaves & stem
		Tabernaemontana ventricosa	Tree	Leaves & stem
		Trichilia dregeana	Tree	Leaves & stem & fruit
		Forb spp. 3	Forb	Leaves & stem
		Forb spp. 6	Forb	Leaves & stem
		Forb spp. 12	Forb	Leaves & stem
		Forb spp. 13	Forb	Leaves & stem
		Aristida spp.	Grass	Leaves & stem
		Oplismenus hirtellus	Grass	Leaves & stem
12 - 05 - 2016	Sugarcane	*Acacia caffra*	Tree	Leaves & stem
		Acacia sieberiana	Tree	Leaves & stem
		Erythrina lysistemon	Tree	Leaves & stem
		Melia azedarach	Tree	Leaves & stem
		Trichilia dregeana	Tree	Leaves & stem
		Forb spp. 1	Forb	Leaves & stem
		Forb spp. 2	Forb	Leaves & stem
		Forb spp. 3	Forb	Leaves & stem
		Forb spp. 5	Forb	Leaves & stem
		Aristida spp.	Grass	Leaves & stem
		Eragrostis curvula	Grass	Leaves & stem & seeds
		Saccharum officinarum	Sugarcane	Leaves & stem
03 – 05 - 2016	2014 restored	*Acacia caffra*	Tree	Leaves & stem
		Acacia sieberiana	Tree	Leaves & stem
		Brachylaena discolour	Tree	Leaves & stem
		Bridelia micrantha	Tree	Leaves & stem
		Erythrina lysistemon	Tree	Leaves & stem
		Millettia grandis	Tree	Leaves & stem
		Strelitzia nicolai	Tree	Leaves & stem
		Syzigium cordatum	Tree	Leaves & stem
		Trichilia dregeana	Tree	Leaves & stem
		Ziziphus mucronata	Tree	Leaves & stem
		Forb spp. 1	Forb	Leaves & stem
		Forb spp. 2	Forb	Leaves & stem
		Forb spp. 4	Forb	Leaves & stem
		Forb spp. 5	Forb	Leaves & stem
		Forb spp. 6	Forb	Leaves & stem
		Forb spp. 9	Forb	Leaves & stem
		Eragrostis curvula	Grass	Leaves & stem & seeds
		Melinis repens	Grass	Leaves & stem & seeds
		Panicum maximum	Grass	Leaves & stem & seeds
		Panicum natalense	Grass	Leaves & stem & seeds
		Themeda trianda	Grass	Leaves & stem & seeds
19 - 05 - 2015	2012 restored	*Acacia caffra*	Tree	Leaves & stem
		Acacia natalitia	Tree	Leaves & stem
		Acacia sieberiana	Tree	Leaves & stem
		Brachylaena discolour	Tree	Leaves & stem
		Bridelia micrantha	Tree	Leaves & stem

		Clerodendrum glabrum	Tree	Leaves & stem
		Erythrina lysistemon	Tree	Leaves & stem
		Ficus glumosa	Tree	Leaves & stem
		Millettia grandis	Tree	Leaves & stem
		Strelitzia nicolai	Tree	Leaves & stem
		Syzigium cordatum	Tree	Leaves & stem
		Trichilia dregeana	Tree	Leaves & stem
		Ziziphus mucronata	Tree	Leaves & stem
		Forb spp. 1	Forb	Leaves & stem
		Forb spp. 4	Forb	Leaves & stem
		Forb spp. 6	Forb	Leaves & stem
		Forb spp. 10	Forb	Leaves & stem
		Panicum maximum	Grass	Leaves & stem & seeds
		Panicum natalense	Grass	Leaves & stem & seeds
		Themeda trianda	Grass	Leaves & stem & seeds
20 - 05 - 2015	2010 restored	*Acacia caffra*	Tree	Leaves & stem
		Acacia natalitia	Tree	Leaves & stem
		Acacia sieberiana	Tree	Leaves & stem
		Albizia adianthifolia	Tree	Leaves & stem
		Brachylaena discolour	Tree	Leaves & stem
		Bridelia micrantha	Tree	Leaves & stem
		Clerodendrum glabrum	Tree	Leaves & stem
		Dombeya rotundifolia	Tree	Leaves & stem
		Erythrina lysistemon	Tree	Leaves & stem
		Ficus glumosa	Tree	Leaves & stem
		Ficus sur	Tree	Leaves & stem
		Millettia grandis	Tree	Leaves & stem
		Schotia brachypetala	Tree	Leaves & stem
		Strelitzia nicolai	Tree	Leaves & stem
		Syzigium cordatum	Tree	Leaves & stem
		Trichilia dregeana	Tree	Leaves & stem
		Ziziphus mucronata	Tree	Leaves & stem
		Forb spp. 1	Forb	Leaves & stem
		Forb spp. 4	Forb	Leaves & stem
		Forb spp. 7	Forb	Leaves & stem
		Forb spp. 11	Forb	Leaves & stem
		Eragrostis curvula	Grass	Leaves & stem & seeds
		Melinis repens	Grass	Leaves & stem & seeds
		Panicum natalense	Grass	Leaves & stem & seeds
27 - 05 - 2015	Forest	*Albizia adianthifolia*	Tree	Leaves & stem
		Combretum edwardsii	Tree	Leaves & stem
		Dalbergia armata	Tree	Leaves & stem
		Dalbergia obovata	Tree	Leaves & stem
		Dichrostachys cinerea	Tree	Leaves & stem
		Dombeya rotundifolia	Tree	Leaves & stem
		Ficus burtt-davyi	Tree	Leaves & stem
		Ficus glumosa	Tree	Leaves & stem
		Heteropyxis natalensis	Tree	Leaves & stem
		Schotia brachypetala	Tree	Leaves & stem
		Scolopia zeyheri	Tree	Leaves & stem
		Searsi chirindensis	Tree	Leaves & stem
		Tabernaemontana ventricosa	Tree	Leaves & stem

	Trichilia dregeana	Tree	Leaves & stem
	Forb spp. 3	Forb	Leaves & stem
	Forb spp. 12	Forb	Leaves & stem
	Forb spp. 13	Forb	Leaves & stem
	Aristida spp.	Grass	Leaves & stem
	Oplismenus hirtellus	Grass	Leaves & stem