# The role of seasonality, environmental correlates and edge effects on the diversity and abundance of small mammals in Afromontane forest patches, Eastern Cape, South Africa

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**By Kyle Junkuhn**

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# DEDICATION

I dedicate this thesis to my parents, Armin and Helen. Without your love, support and all the opportunities you have provided me, I wouldn’t be the person I am today. I thank you unconditionally.

# ABSTRACT

The Eastern Cape contains the majority of the remaining forested areas in South Africa (95.8% Coastal forests and 47.6% Afromontane forests), however these occur in fragments. Due to the rapid rise in the human population and its needs, conversion of forests to agricultural land and the extraction of timber increases, natural vegetation are lost and this therefore leads to forests becoming fragmented into small forest patches.

One of the main consequences of forest fragmentation is loss of contiguous habitat, which is the dominant threat to species globally as it negatively affects both species richness and genetic diversity. This research investigates the effects that forest fragmentation has on small mammal diversity and abundance. The study sampled small mammals in Southern Mistbelt Afromontane forest patches in the Eastern Cape.

The first aim of this study was to identify which environmental variable, or combinations of variables, affect the diversity and abundance of small mammals in fragmented forest patches in the Eastern Cape during the austral summer. The second aim was to compare seasonal changes in small mammal diversity and abundance at a forest edge compared to the forest core at Beggars Bush, an Afromontane forest in the Eastern Cape, South Africa.

To identify the environmental impacts that different forests have on small mammals, nine different forests were chosen in the Eastern Cape and sampled during the summer in 2013. These nine forests were: Fort Fordyce Forest, Thaba Ndoda Forest, Hogsback Forest, Dassie Kraans Forest, Langeni Forest, Burchell's Reserve Forest, Maiden Dam Forest and Kagaberg Forest. At each site, three transects of 30 Sherman Traps each were used. Trapping periods within each forest patch lasted between three to five consecutive nights and traps were inspected once a day in the morning. Nine environmental variables were initially identified and after being tested for normality and colinearity, five variables were chosen. These variables were Altitude, Mean Annual Temperature, Gradient, Patch Size and Mean annual potential evaporation. The number of individuals captures and species richness were then incorporated as dependent variables for best subset multiple regression model selection using the Akaike Information Criterion (AIC).

For the second aim, the same trapping methods were used when comparing seasonal changes to diversity and abundance within the forest core. However at the forest edge, due to the small size of the forest, only 20 Sherman traps were used in each transect with five traps (25%) placed in the grassland and 15 (75%) placed in the forest along each transect.

This study was one of few to survey small mammal diversity and abundance in fragmented forests in the Eastern Cape. Gradient, patch size and mean annual potential evaporation were the variables that best predicted the individual number of small mammal captures while only gradient best explained species richness. It does however, need to be noted that capture rates and species richness were very low and this would therefore effect the analysis of environmental variables. Future studies should have a larger sample size of forest patches and include more microhabitat environmental variables to determine their effects on small mammal diversity and abundance. However, it should be noted that through climatic extinction filtering, forest mammals are resilient generalists that can tolerate fragmentation effects.

Furthermore, it was found that forest edges appear to play a significant role in small mammal diversity and abundance in the Beggars Bush Afromontane forest. Some species were habitat specialists such as *Rhabdomys pumilio* preferring the grassland habitat and *Graphiurus murinus* and *Aethomys namaquensis* the forest habitat, while *Myosorex varius* was the only species that was not habitat dependant. It was found that there was a greater diversity and abundance at the forest edge compared to within the forest core throughout most of the seasons. One possible flaw was that the sampling methods were different at the edge and within the forest and therefore future studies should ensure that the method stays uniform throughout the study.

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# Chapter 1

# General Introduction

## 1.1 Forests and Forest patches

South African indigenous forest is a multi-layered vegetation unit dominated by trees with overlapping crowns where grass and herbaceous vegetation is infrequent (Mucina & Rutherford 2006). Forests support a high proportion of South Africa's biodiversity yet they form the smallest biome in southern Africa (Lawes *et al.* 2000; Hayward *et al* 2005). These forests are broadly divided into two main categories: Afrotemporate and Coastal forests. Afrotemporate forests are more persistent and ancient while coastal forests are younger having expanded southwards from Mozambique (Lawes *et al.* 2007). Afrotemperate forests occur at higher altitudes and experience colder temperatures as well as considerable seasonal variation in rainfall when compared to coastal forests (Lawes *et al.* 2007). The forested areas of South Africa are mainly small isolated islands surrounded by other vegetation such as grassland, Albany thicket, fynbos and savanna (Mucina & Rutherford 2006). These forest patches are scarce, and while only a few large recognised forests are left in South Africa, the many small and isolated patches are still significant in maintaining biodiversity. The Eastern Cape Province of South Africa contains the majority of the remaining forested areas in the country (95.8% of all Coastal forests and 47.6% of all Afromontane forests), however these only occur in fragments (Low & Rebelo 1996), with the majority of these forests not larger than 1 km$^2$ (Rutherford & Westfall 1986).

The fundamental unit within a landscape is a patch and hence landscapes can be defined as patch mosaics (Hanski & Gilpin 1991). A patch can be defined as an area that differs in nature or appearance from its surroundings (Kotliar & Wiens 1990). Patches come in various shapes, types, sizes and are differentiated by physical gaps, biotic and abiotic gradients, and may have resulted from human land-use activities and/or changes over time (Bowers & Matter 1997).

Historically, it is believed that the general patchy distribution of South African forests has two major probable agents (excluding man) that caused this spatial pattern – climate and fire

(Mucina & Rutherford 2006). The refugia theory suggests that forests were repeatedly fragmented during cold glacial periods and expanded during warm wet interglacials and yet during these unfavourable climatic periods, fauna persisted in fragmented forests (Levinsky *et al.* 2013). Along with substrate and rainfall environmental variables, fire determines the distribution and shape of forest patches, particularly in mountainous areas (Geldenhuys 1994). This is because wind fans fires thereby causing fire patterns and the interaction with natural boundaries such as mountain ranges determines a fires' spread (Geldenhuys 1994). The wind is channelled through valleys and its direction is determined by physical barriers. Hence forests generally exist in wind shadowed areas (Geldenhuys 1994). Forests also occur in South Africa on wetter south-facing slopes while grasslands occur on drier north-facing slopes (Holland & Steyn 1975). However, it is also believed that extreme changes in vegetation and climate, including the last glacial maximum, were responsible for the distribution and patchiness of South Africa's forests (Lawes *et al.* 2007).

Despite the role of climate and fire in shaping the distribution of forests in South Africa, the influence of humans cannot be ignored. Iron Age farmers are believed to have cut down portions of forest (Mucina & Rutherford 2006) and this practice has essentially continued to the present day with forests being cleared for agriculture, and exploited for timber, food and medicines (Feely 1980; Castley & Kerley 1996; Lawes *et al.* 2004). It is this human exploitation that has led to the continued fragmentation of forests in South Africa.

## 1.2 Forest Fragmentation and Small Mammals

The composition of the faunal assemblages of South African forests has been shaped by large scale historical and regional processes (Lawes *et al.* 2007). These include palaeoclimatic change, speciation, extinction and geographic dispersion which have dominated ecological processes (Lawes *et al.* 2000). Lawes *et al.* (2007) believe that climatic extinction filtering and species recolonization after the last glacial maximum (LGM) have influenced the distribution and structure of South African forest fauna. At a local scale, the extent of fragmentation effects on small mammals depends on the magnitude and type of human activity surrounding the forest patches and the plant and animal species affected by forest destruction (Schmid-Holmes & Drickamer 2001; Castley & Kerley 1996).

Numerous forest patch characteristics have been identified which are related to small mammal abundance and diversity in temperate forests in southern Illinois, USA (Schmid-Holmes & Drickamer 2001). These include habitat patch size, patch shape and related edge effects, patch isolation, inter-patch distance and matrix character, type of anthropogenic activity, vegetation characteristics and microclimate. However, the effects of patch characteristics on small mammals differ with the type of habitat patch and the small mammal species investigated. For example, Buckner & Shure (1985) found that the abundances of two sympatric *Peromyscus* species were inversely related to patch size within clear-cut patches in a forested landscape. However, Bowers & Matter (1997) reviewed 12 publications assessing the effects of patch size on small mammal diversity and found a positive relationship (increase in size increases diversity) in four species, a negative for seven and 21 species for which it had no effect.

## 1.3 Small mammals

South African indigenous forests contain 14% of threatened vertebrates and are therefore of high conservation value. However, little is known about the fauna inhabiting these forests (Hayward *et al.* 2005). One particular group of forest dwelling vertebrates are small mammals. Small mammals are defined as mammals weighing less than 300g and are non-volant terrestrial mammal species (Skinner and Chimimba 2005). In southern Africa, this group consists of three mammalian orders, namely Rodentia (rodents), Eulipotypha (shrews) and Macroscelidea (elephant shrews).

Rodents are the largest order of mammals, comprising 32 extant families and approximately 2 277 species worldwide (Carleton & Musser 2005). Rodents are vital components in ecosystems as they function in habitat modification, nutrient cycling, and the dispersal of seeds and provide a link between primary and secondary consumers (Ecke *et al.* 2001; Avenant & Cavallini 2007). Rodent community structure and species richness have been related to habitat complexity, structure, productivity, area, surrounding landscape and distances between similar habitats (Armstrong & van Hensbergen 1996; Avenant 2000). Small mammals are thus good indicators of overall mammalian diversity and are often used in ecological surveys (Skinner & Chimimba 2005). Rodents can be used as ecological indicators where disturbances in their habitats are associated with decreases in rodent species richness or the presence or absence of indicator species (Avenant & Cavallini 2007).

## 1.4 Aims

Some research has been conducted on the effects of forest fragmentation on biodiversity (Castley & Kerley 2006; Fahrig 2003; Lawes *et al.* 2000; Lawes *et al.* 2007) while other research has been done on small mammals and their effect on ecosystem function (Armstrong & van Henbergen 1996; Avenant & Cavallini 2007; Kok *et al.* 2012). However, little research has been done on the effects of forest fragmentation on small mammal diversity in South Africa. Due to the severe threat of deforestation, it is imperative that trapping surveys take place to determine the distribution and abundance of small mammals in South Africa's fragmented forests (Stephenson 1994). This study investigates the effects that forest fragmentation has had on small mammal diversity and abundance.

There were three broad aims for this study:

1. To, firstly, document small mammal diversity and abundance in Afromontane forest patches in the Eastern Cape, South Africa;

2. To determine which environmental variable, or combinations of variables, affect the diversity and abundance of small mammals in fragmented forest patches in the Eastern Cape during the austral summer; and

3. To investigate the seasonal diversity and abundance of small mammals at the edge and within in an Afromontane forest patch in the Eastern Cape, South Africa.

# Chapter 2

# The environmental correlates of Small Mammal diversity and abundance in Afromontane forest patches in the Eastern Cape, South Africa

## 2.1 Introduction

### 2.1.1 Fragmentation

Fragmentation is a landscape-level occurrence where species survive in habitat remnants and are confronted with a modified environment of increased isolation, reduced natural area, new ecological boundaries and a loss of original habitat (Bentley *et al*. 2000). The forest biome in South Africa is an extremely fragmented landscape (Lawes *et al.* 2000), highly susceptible to both climate changes and fire and is under severe threat from human activities (de Villiers & White 2002; Mucina & Rutherford 2006). Anthropogenic habitat modification is the largest cause of habitat loss in the landscape (Fahrig 2003; Ewers & Didham 2006). Due to the rapid rise in the human population and its resource needs (Mucina & Rutherford 2006), conversion of forests to agricultural land and the extraction of timber increases results in the loss of natural vegetation and this therefore leads to forests becoming fragmented into small forest patches (Schmid-Holmes & Drickamer 2001; Fischer & Lindenmayer 2007).

One of the main consequences of forest fragmentation is the loss of contiguous habitat, which is the dominant threat to both fauna and flora species globally as it negatively affects both species richness and genetic diversity (Sala *et al.* 2000; Fahrig 2003; Asquith & Mejia-Chang 2005; Fischer & Lindenmayer 2007). Landscape modification, such as fragmentation, classifies threatening processes associated with it into two broad groups; exogenous or endogenous (Fischer & Lindenmayer 2007). Exogenous threatening processes are independent of species biology and include threats that often coincide with habitat loss such as habitat degradation, habitat sub-division and habitat isolation (Liu *et al.* 2001). An example would be habitat loss of natural vegetation which is important to many species through agricultural expansion (Kerr & Deguise 2004). Endogenous threatening processes are part of the species biology and include changes in behaviour, biology and interactions with other species (Fischer & Lindenmayer 2007). An example of this could be that landscape modification leads to changes in home-ranges and this may affect the predation and competition of a species (Pope *et al.* 2004).

Fragmentation effects are grouped under five categories: fragment area, fragment isolation, edge effects (more on this in Chapter 3), and fragment shape and matrix structure (Ewers & Didham 2006). Firstly, in terms of fragment area, the reduced habitat area threatens biodiversity because habitat remnants support smaller populations of species and these populations are therefore at a greater risk of extinction through processes such as reduced genetic variability and demographic stochasticity (Andren 1994; Asquith & Mejia-Chang 2005; Ewers & Didham 2006; Wilson *et al.* 2010).

Secondly, in terms of fragment isolation, the distance between remnants is then increased due to reduced habitats, which impedes inter-patch dispersal of individuals which further decreases the fragmented landscape's ability to maintain biodiversity (Andren 1994; Laurance *et al.* 2002; Ewers & Didham 2006; Wilson *et al.* 2010). Small mammals may be more restricted compared to species such as birds or larger mammals which are more mobile (Schmid-Holmes & Drickamer 2001).

Fragment shape is the core habitat remaining in a habitat fragment and is determined by the interaction between fragment area and the perimeter (Ewers & Didham 2006). Fragments with complex shapes have a greater proportion of edge than core habitat, leading to greater edge effects of the habitat (Ewers & Didham 2006). An irregular shape can also lead to division of core habitats, separated by edge-affected habitat (Ewers 2004). One of the most pertinent characteristics about the shape of habitat fragments is that complex fragments, which are irregular in shape, are colonised by mobile species more frequently than compact patches, which are more regular in shape (Collinge & Palmer 2002; Cumming 2002). This is because the high shape complexity has a greater amount of edge and therefore has an increased likelihood that the patch will be encountered by a moving individual (Collinge & Palmer 2002).

Lastly, matrix structure and quality are crucial in determining the abundance and composition of species within fragments (Cook *et al.* 2002; Ewers & Didham 2006). Invasion of both fauna and flora from outside a fragment changes the habitat characteristics within the fragment, and the exterior matrix might be hospitable to fragment-dwelling fauna (Gustafson & Gardener 1996; Lawes *et al* 2007). There is often species ‘spill-over’ or ‘species overlap’ between fragment and matrix where an overlap occurs along the ecotones of large patches and in small patches that are dominated by processes common to ecotones (Cook *et al.* 2002).

As human-dominated landscapes become more expansive, it is important to understand the effects that this has on fragmented natural habitats (Ewers & Didham 2006). Fragmentation causes species to survive in habitat fragments with a modified environment of a reduced area, new ecological boundaries and increased isolation (Ewers & Didham 2006). Due to the destruction of these natural forests through increased human activities, a large number of remnant forest patches are now surrounded by non-indigenous vegetation (Wethered & Lawes 2003). The matrix that surrounds these forest remnants is presently the most connected and extensive and therefore has the greatest effect on landscape functioning (Forman 1997). Therefore, to understand the effects of forest fragmentation on forest fauna, fragmented forest patches must be examined as a part of a dynamic landscape mosaic (Lindenmayer *et al.* 2002).

### 2.1.2 Environmental Variables and Small Mammals

There are many environmental variables that may explain the observed patterns in small mammal abundance and diversity (Andrews & O’Brien 2000). These can include climatic variables such as temperature and precipitation, microhabitat variables such as herbaceous cover and plant species richness or physical factors such as altitude, aspect and gradient (Andrews & O’Brien 2000). Many studies have focused on the effects of individual environmental variables such as the relationship between habitat area and number of species in forest fragments (Bower & Matter 1997; Didham *et al.* 1998; Goodman & Rakotondrvony 2000; Laurance *et al.* 2002; Wethered & Lawes 2003; Pardini *et al.* 2005). Some studies investigated plant richness, potential evaporation and temperature (Andrews & O’Brien 2000) while only a few (Schmid-Holmes & Drickamer 2001; Lawes *et al.* 2000; Lawes *et al.* 2005; Kotze & Lawes 2007) have taken into account a combination of environmental gradients to determine which one or combination of variables has the greatest effect on abundance and diversity. Rowe (2009) also found that a combination of environmental variables and not just one are responsible for small mammal variability.

Climate is believed to have an important role in determining the floristic diversity of a habitat and ultimately affects small mammal diversity (Andrews & O'Brien 2000). However, the extent of climate-related effects on small mammals is not clear because mammals are endothermic and are mobile and can therefore tolerate changes in climate. It is also assumed that mammal diversity is related to plant diversity (Avery 1993) and climate has an effect on vegetation. Therefore indirectly, climate may have an effect on mammal diversity. Temperature and rainfall play important roles in maintaining faunal diversity (Yarnell *et al.* 2007) while individual species distributions are set by temperature and rainfall increases primary production which leads to increased species diversity (Rowe 2009). In terms of altitude, the hypothesis for the effects of altitude on small mammals states that diversity will peak at mid altitudes (Rowe 2009; Kok *et al.* 2012). This theory is explained by the mid-domain effect which is defined as the overlap of species ranges towards the centre of a geographic domain, constrained by geometric boundaries (Colwell & Less 2000). The result is that there is increased species richness towards the centre of that domain.

The aim of this chapter was to identify which environmental variable or combinations of variables influence the diversity and abundance of small mammals in fragmented forest patches in the Eastern Cape during the austral summer. The variables assessed were patch size, elevation, gradient, mean annual rainfall, mean annual temperature, mean annual potential evaporation, heat units annually, herbaceous diversity and herbaceous cover.

## 2.2 Study Sites

South Africa has numerous small forest patches scattered around the eastern and southern escarpments of the country (Figure 2.1). Nine different forest patches in the Eastern Cape, South Africa were used as study sites for this part of the study (Figure 2.2). They are all Afromotane forests patches classified as Southern Mistbelt Forest (Mucina & Rutherford 2006). One of the reasons the Eastern Cape was selected as a study area was because it holds the majority of forested areas remaining in South Africa yet they occur mainly as fragments (Low & Rebelo 1996; Hayward *et al.* 2005).

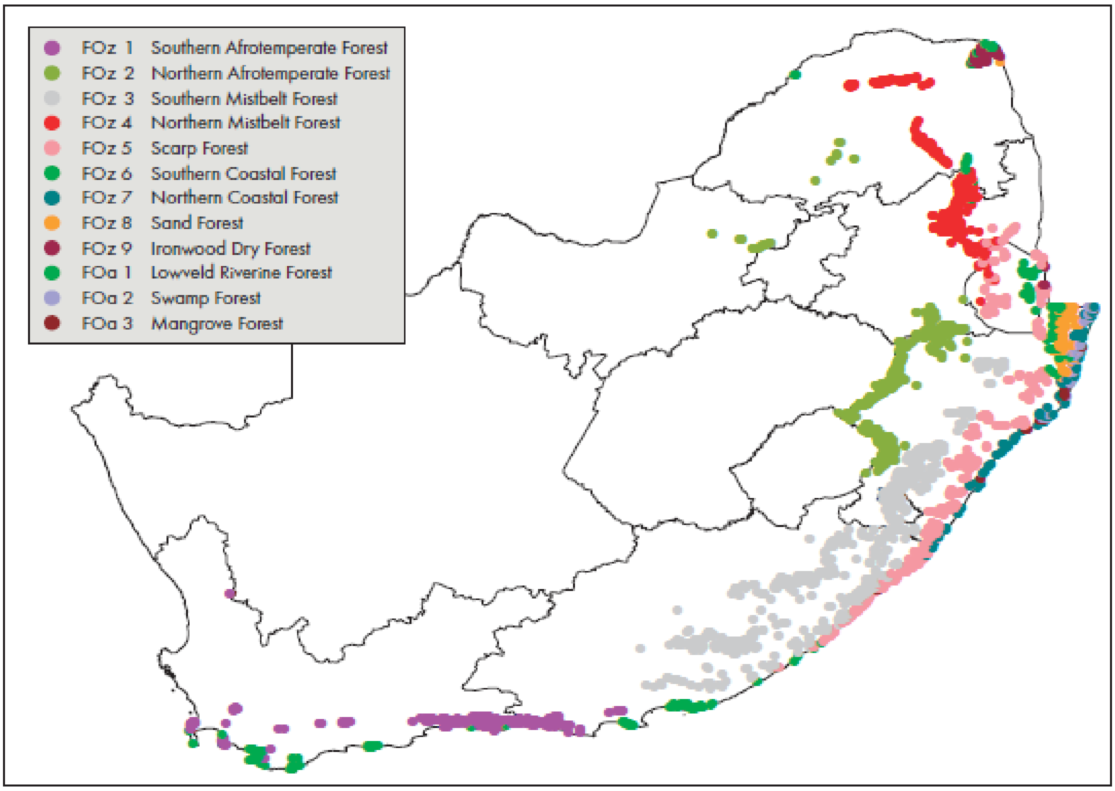

Figure 2.1: Distribution of mapped indigenous forest units in South Africa and Lesotho (Mucina & Rutherford 2006)

Figure 2.2: A Google Earth Map of all nine Afromontane forest study sites in the Eastern Cape, South Africa.

### 2.2.1 Southern Mistbelt Forests

Southern Mistbelt Forests occur on south and south-east facing slopes in fire-shadow habitats along the Great Escarpment in the KwaZulu-Natal and Eastern Cape Provinces of South Africa (Mucina & Rutherford 2006) (Figure 2.1). Forest patches are found in various sizes and are found at altitudes between 850 – 1600 m. However, the majority of forests are between 1000 – 1400 m. At high altitudes, these forests are generally tall (between 15 – 20 meters) and multi-layered, meaning that they have two layers of trees, a well-developed herb layer and a dense shrubby understorey (Mucina & Rutherford 2006). At lower altitudes, southern Mistbelt Forests are generally lower with fewer tree layers (Mucina & Rutherford 2006).

In the Eastern Cape, Southern Mistbelt forests are classified as Amotole Mistbelt forests. They are located on the Transkei escarpment and are characterised by emergent trees of *Afrocarpus falcatus* and deciduous and semi-deciduous species such as *Zanthoxylum davyi, Vepris lanceolata, Celtis africana and Calodendrum capense.* Some of the endemic herbaceous taxa include: *Eugenia zuluensis, Plectranthus elegantulus, P. rehmannii, Pyrrosia africana, Streptocarpus bolusii, S. candidus, S. fanniniae, S. silvaticus.*

The majority of soils found under Southern Mistbelt Forests have a high nutrient status, are deep, loamy and are developed on weathered dolerite intrusions or sandstones, shales and mudstones of the Karoo Supergroup (Mucina & Rutherford 2006). Low-lying forests have shallower soils and are developed on quartzitic Wittenberg Sandstones or sandstones of the Karoo Supergroup.

Rainfall is higher during the summer in Mistbelt Forests with a Mean Annual Precipitation (MAP) of 988mm (Mucina & Rutherford 2006). In addition, precipitation tends to increase with altitude and increases from west to east across the study area (Mucina & Rutherford 2006). For example, Beggars Bush (33° 17' 16.39" S, 26° 41' 02.24" E) situated 15km East of Grahamstown has an elevation of 684 m with annual precipitation of 495 mm, while Langeni Forest (31° 24' 40.95" S, 28° 32' 15.93" E) near Umtata has an elevation of 1092 meters and an annual precipitation of 1157 mm (Schultze 1997). The Mean Annual Temperature (MAT) in Southern Mistbelt Forests is 15.7 ° C. However, temperature decreases with both altitude and latitude (Mucina & Rutherford 2006).

Table 2.1: The physical variables of each of the nine Afromontane forests in the Eastern Cape, South Africa

| Study Site | Longitude | Latitude | Altitude (m) | Patch Size (km²) | Gradient | Aspect | Herbaceous Diversity | Herbaceous Cover (%) |
|---|---|---|---|---|---|---|---|---|
| Kaggaberg | 32° 38' 49.33" S | 26° 04' 06.74" E | 942 | 7.9 | 0.31 | S | 76 | 25 |
| Burchells Reserve | 33° 13' 46.75" S | 26° 08' 14.22" E | 846 | 1.05 | 0.3 | S | 73 | 22 |
| Dassie Kraans | 33° 19' 43.59" S | 26° 30' 01.93" E | 741 | 0.1 | 0.23 | SW | 24 | 11 |
| Fort Fordyce | 32° 40' 46.33" S | 26° 28' 17.43" E | 1014 | 6.93 | 0.27 | SW | 33 | 10 |
| Hogsback | 32° 36' 05.08" S | 26° 55' 44.52" E | 1164 | 7.34 | 0.31 | SE | 80 | 21 |
| Langeni | 31° 24' 40.95" S | 28° 32' 15.93" E | 1092 | 1.75 | 0.18 | SE | 133 | 62 |
| Maiden Dam | 32° 44' 06.83" S | 27° 17' 44.78" E | 551 | 10.1 | 0.04 | SE | 104 | 15 |
| Thaba Ndoda | 32° 48' 16.00" S | 27° 07' 44.31" E | 867 | 1.66 | 0.07 | S | 100 | 40 |
| Beggars Bush | 33° 17' 16.39" S | 26° 41' 02.24" E | 684 | 0.67 | 0.36 | S | 32 | 10 |

Table 2.2: The climate variables of each of the nine Afromontane forests in the Eastern Cape, South Africa

| Study Site | Mean Annual Rainfall (mm) | Mean Annual Potential Evaporation (mm) | Heat Units Annually (°days) | Mean Annual Temperature (°C) |
|---|---|---|---|---|
| Kaggaberg | 537 | 1723.3 | 108.9 | 16.7 |
| Burchells Reserve | 412 | 1884.8 | 111.3 | 14.5 |
| Dassie Kraans | 576 | 1761.6 | 114.7 | 13.1 |
| Fort Fordyce | 701 | 1771.7 | 111.8 | 12.4 |
| Hogsback | 1371 | 1675.8 | 103.6 | 11.9 |
| Langeni | 1157 | 1578.8 | 101.3 | 13.7 |
| Maiden Dam | 898 | 1719.5 | 119 | 12.4 |
| Thaba Ndoda | 653 | 1723.7 | 118.4 | 13.3 |
| Beggars Bush | 495 | 1734.8 | 118.6 | 13 |

## 2.3 Methods

### 2.3.1 Experimental Design

To identify the impacts that different forests patches may have on small mammals, nine different forests were chosen in the Eastern Cape Province, South Africa. These nine forests were selected as they were known to be accessible, span the range of the climatic gradient and varied in size, altitude and latitude. These nine forests were: Fort Fordyce Forest, Thaba Ndoda Forest, Hogsback Forest, Dassie Kraans Forest, Langeni Forest, Burchell's Reserve Forest, Maiden Dam Forest and Kagaberg Forest (Figure 2.2). All the forests sampled were Southern Mistbelt Forests and were all found on Southern-facing slopes. The nine forest patches were sampled for small mammals during the Austral Summer in 2013 (between January and the beginning of March). The summer season was chosen as this is when small mammals are most active due to the warmer temperatures (Skinner and Chimimba 2005). At each site, three transects of 30 Sherman Traps (229 x 76 x 89 mm) (Pearson & Ruggiero 2003; Kok *et al.* 2012) were laid out perpendicular to the slope, with transects located along contours at the top, middle and bottom of the slope. In each transect, traps were spaced ~10 meters apart (Pearson & Ruggiero 2003; Keller & Schradin 2008), while transects were spaced between 50 to 100 meters apart from one another depending on forest size. A study in South Africa on coastal dune forests showed that transects spaced 50 m apart or more showed sufficient sampling independence for millipedes, birds and rodents (Weiermans & van Aarde 2003). Therefore, transects were assumed to be statistically independent.

### 2.3.2 Trapping Protocol

Live-trapping allows the captured species to be identified, sexed, weighed and marked for possible recapture (Jones *et al.* 1996). Sherman Traps are effective at capturing small to medium sized rodents (20 – 50 g) and have been used successful in a number of studies on small mammal diversity (Maddock 1992; Madikiza *et al.* 2010; Wilson *et al.* 2010; Kok *et al.* 2012). The thermoregulatory demands of small mammal's increases due to stress caused by captivity or due to capture during colder periods (especially winter) and needs to be minimised (Sikes *et al.* 2011). This was done by ensuring a sufficient supply of food and nesting material within the live trap (Sikes *et al.* 2011). The bait used to capture small mammals was a mixture of oats with peanut butter as this was found to be the most successful bait for trapping small mammals (Schmid-Holmes & Drickamer 2001; Kok *et al.* 2013). A ball of cotton wool was used as nesting material and this was sufficient to separate the small mammal from the conducting surface of the metal trap and keep them warm during periods of cold temperatures (Sikes *et al.* 2011). Traps were placed next to prominent microhabitat features such as rocky terrain, logs or tree trunks to increase the likelihood of small mammal capture (Stephenson 1994; Schmid-Holmes & Drickamer 2001).

Trapping periods within each forest patch lasted between three to five consecutive nights and these varied due to time constraints (Caro et al. 2001; Schmid-Holmes & Drickamer 2001; Yarnell *et al.* 2007). Traps were inspected once a day in the morning (Stephenson 1994; Van Aarde *et al.* 1996; Schmid-Holmes & Drickamer 2001). If a species was caught in a trap, it would be transferred to a ziplock bag, where it could be photographed, identified and marked so as to identify if the individual had been caught previously. If an individual was caught with no markings, it was toe or hair-clipped (Van Aarde *et al.* 1996). The species, location on transect and if it was a unique or a recaptured individual were marked on a recording sheet. Initially hair-clipping was done to identify recaptures, but the possibility of doing a genetic study as an adjunct project was considered, and toe-clipping was then used. The genetic material of one species collected (*Graphiurus murinus*) has since been used in a small student project.

Individuals that were toe-clipped had the toe tissue stored in 100% ethanol for DNA studies on genetic diversity (Sikes *et al.* 2011). Toe clipping is a widely recognised practise and serves a dual role: it is a permanent marking method, and can hence facilitate the calculation of the number of individuals caught per species using mark-recapture methods (Southwood & Henderson 2000), and it is also used as a non-lethal method to acquire DNA material for genetic studies (Gannon *et al.* 2007). Toe clipping was performed using sharp sterilized nail clippers (Braude & Cizek 1998; Gannon *et al.* 2007). Nail clippers were sterilized between samples by washing them in 97% ethanol followed by flaming (Braude & Cizek 1998) and toes were clipped at the first digit (Pavone & Boonstra 1985). Previous studies have conclusively demonstrated that toe-clipping does not significantly affect the survival (Pavone & Boonstra 1985) or weight loss, a surrogate for stress, (Korn 1987; Wood & Slade 1990). Ethical clearance for this study was obtained prior to the commencement of the fieldwork from the Rhodes University Ethical Standards Committee (clearance number: ZOOL-07-2012).

### 2.3.3 Data Analysis

Nine environmental variables likely to influence small mammal diversity and abundance were selected to be included in the analysis, namely: patch size ($km^2$), patch elevation (m), gradient, mean annual temperature (°C), mean annual rainfall (mm), mean annual potential evaporation (mm), heat units annually (° days), herbaceous diversity and herbaceous cover (%). The co-ordinates (Longitude and Latitude), elevations (meters) and gradients (elevation/distance) for each forest were identified or calculated using Google Earth. The gradient of the entire forest was calculated on Google Earth using the elevation and distance from highest point to lowest point within the forest. The elevation was calculated by taking the average elevation of the forest. The data for mean annual rainfall, mean annual precipitation, mean annual potential evaporation and heat units annually were found using SA Atlas on Agrohydrology and Climatology (Schultze 1997) on ARCview version 10 (ESRI, Redlands, California). Forest patch sizes were calculated using Google Earth and the freeware earthpoint.us which used boundary co-ordinates of the forest from Google Earth to calculate the areas (earthpoint.us). Herbaceous diversity was the total number of herbaceous species found in several quadrats within the forest while herbaceous cover was calculated by taking the average estimate of herbaceous cover (%) at several sites within each forest. These latter data were obtained from an unpublished MSc study conducted by Ms. A. Matiwane.

In order to determine the completeness of sampling, rarefaction curves were constructed for species richness at each site. This was done using the Estimate S v8.2 where individuals are set as samples and were then calculated using the ICE and Chao 2 estimators compared to the observed richness (Colwell 2005). To determine which environmental variable or combination of variables were the most significant on small mammal diversity and abundance, a multi-model selection procedure was performed (Burnham & Anderson 2002; Archibald *et al* 2005).The number of individuals and the species richness of small mammals caught each day in each forest were incorporated as dependent variables for a best subset multiple regression model selection using the Akaike Information Criterion (AIC) in the R Studio statistics program (Akaike 1983; Burnham & Anderson 2002; Archibald *et al* 2005). Firstly, Generalized Linear Models (GLMs) were set up for the number of captures per day and the species richness per day in each forest with the most complex models initially run. Secondly, the independent variables were tested for co-linearity and normality (Burnham &

Anderson 2002) using variance inflation factors (VIFs), calculated using the ‘car’ package in R (Fox & Weisberg 2011) where no values were >2 (indicating no problematic multi-collinearity (Menard 1995)). The model was run with all the variables and the variable with the highest value removed and the model rerun and variables removed until the model was left with all the variables having a value of less than 2 (Painting & Holwell 2014). The analysis found that Heat units, Mean annual rainfall, herbaceous cover and herbaceous diversity had values greater than 2 and were therefore excluded from the analysis.

Multiple regression models were then constructed for each dependant variable and every combination of variables giving a combination of models for each diversity measurement. The second order $AIC_c$ scores were used instead of normal AIC scores due to the small sample size (n = 9 sites) relative to the number of variables selected (Burnham & Anderson 2002; Rowe 2009; Symonds & Moussalli 2011). . Finally, a GLM was performed on the best model from the AIC data which is the model at the top of the AIC results. All statistical analyses AIC model building and GLMs were calculated in R Studio (RStudio 2014) and the R script used can be found in Appendix A.

### 2.3.4 Overall trapping results

The trapping results were summarised by seven parameters. These were the number of captures, the number of unique individuals caught, number of recaptures, traps success, species richness and Shannon diversity. A trap night was defined as the number of traps active per 24 hours (Rowe-Rowe & Meester 1982). Trap success was calculated by taking the total number of small mammal captures divided by the number of trap nights and multiplied by 100 (Rowe-Rowe & Meester 1982). Species richness was the number of observed small mammal species, which according to Magurran (2004) is a count of the number of different species observed from sampling at a site. The other classification of species richness is true species richness which is the total number of species present at a site during sampling (Magurran 2004). This, however, is almost impossible as not all species are likely to be sampled by means of physical sampling during the three to five trap nights at sites (Magurran 2004). Therefore, the species richness represented throughout the analyses will be observed species richness as this can be used to calculate other diversity measures (Magurran 2004). The Shannon diversity index was used to analyse heterogeneity (Shannon 1984) using the following equation:

$$H = -\sum_{i=1}^{S} p_i \ln p_i$$

*H'* is diversity, s is the number of species, $p_i$ is the proportion of species found in the *i*th species (Rowe-Rowe & Meester 1982; Magurran 2004).

## 2.4 Results

Table 2.3: Summary of trapping data for nine different forests sampled during the summer season of 2013 in the Eastern Cape, South Africa.

| Study Site | No. of Captures | No. of Individuals | No. of Recaptures | No. of Trap Nights | Trap Success (%) | Species Richness | Shannon Index |
|---|---|---|---|---|---|---|---|
| Kagaberg | 3 | 3 | 0 | 450 | 0.67 | 2 | 1.89 |
| Burchells Reserve | 12 | 11 | 1 | 270 | 4.44 | 2 | 1.84 |
| DassieKraans | 3 | 3 | 0 | 270 | 1.11 | 2 | 1.89 |
| Fort Fordyce | 2 | 2 | 0 | 450 | 0.44 | 2 | 2 |
| Hogsback | 8 | 6 | 2 | 450 | 1.78 | 2 | 1.89 |
| Langeni | 16 | 13 | 3 | 270 | 5.93 | 2 | 1.33 |
| Maiden Dam | 3 | 2 | 1 | 270 | 1.11 | 2 | 1.89 |
| ThabaNdoda | 3 | 2 | 1 | 450 | 0.67 | 2 | 2 |
| Beggars Bush | 28 | 23 | 5 | 450 | 6.22 | 3 | 1.68 |
| **Total** | 78 | 65 | 13 | 3330 | | 4 | |

There was a total of 78 captures of which 65 were individuals and 13 recaptures over 3330 trap nights across the nine forest patches in the Eastern Cape (Table 2.3). The overall trap success was very low at 2.34 %. At eight of the nine sites only two species were caught, however the two species varied at each site while three species were found at Beggars Bush. Photographs of the nine forest can be found in Appendix B, Figures 5 to 7.

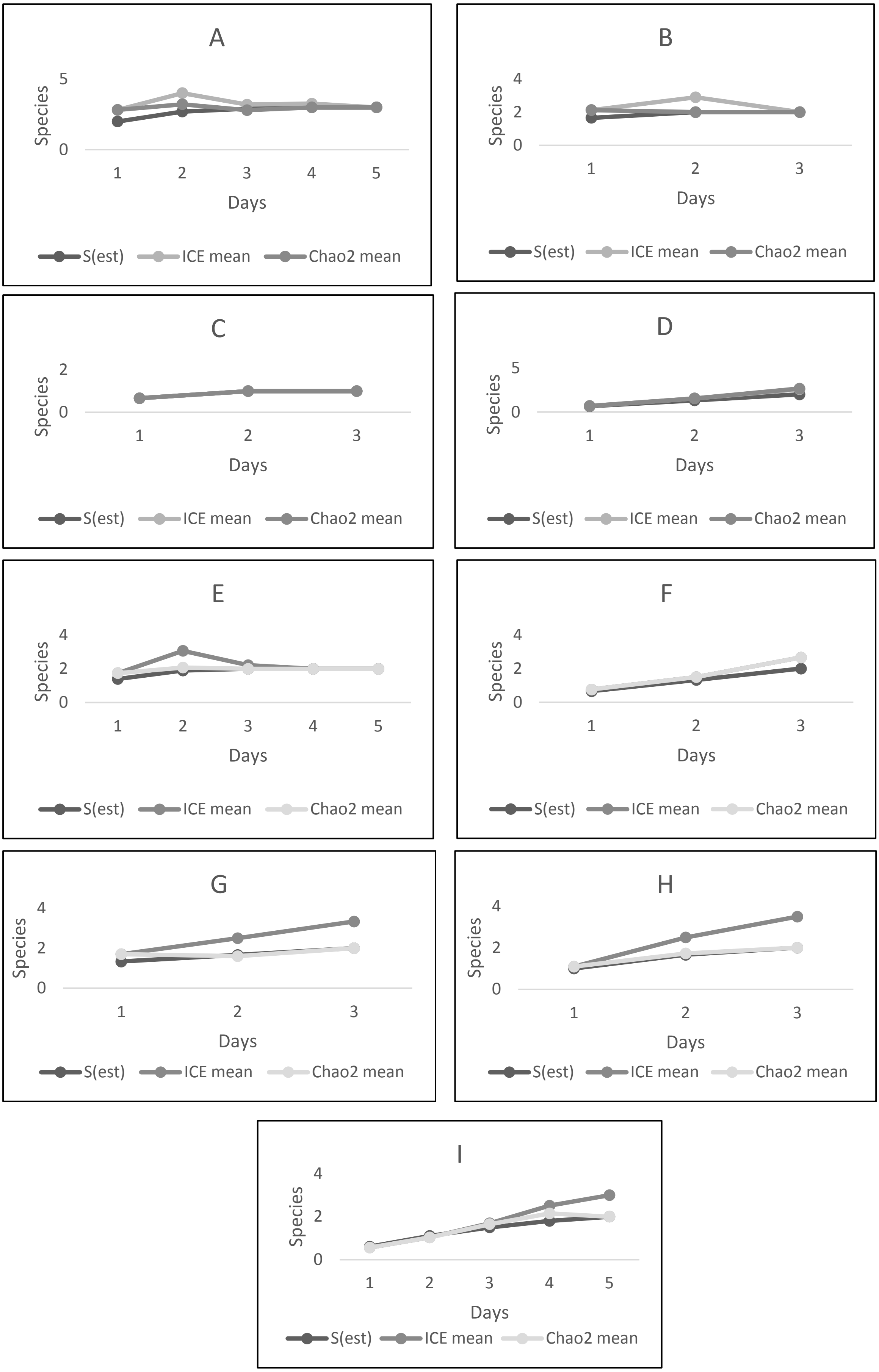

Figure 2.3: Rarefaction curves of species sampled in each of the nine forests. A – Beggars Bush, B – Burchells Reserve, C – Dassie Kraans, D – Fort Fordyce, E – Hogsback, F – Kaggaberg, G – Langeni, H – Maiden Dam & I – Pirie.

Figure 2.3 shows that sampling saturation was achieved in the majority of the forest sites sampled. This means that the observed species richness was equal to the two estimators (ICE mean and Chao 2 mean) at Beggars Bush, Burchells Reserve, Dassie Kraans, Fort Fordyce, Hogsback and Kaggaberg forests. While at Langeni, Maiden Dam and Pirie forests the observed richness was slightly greater.

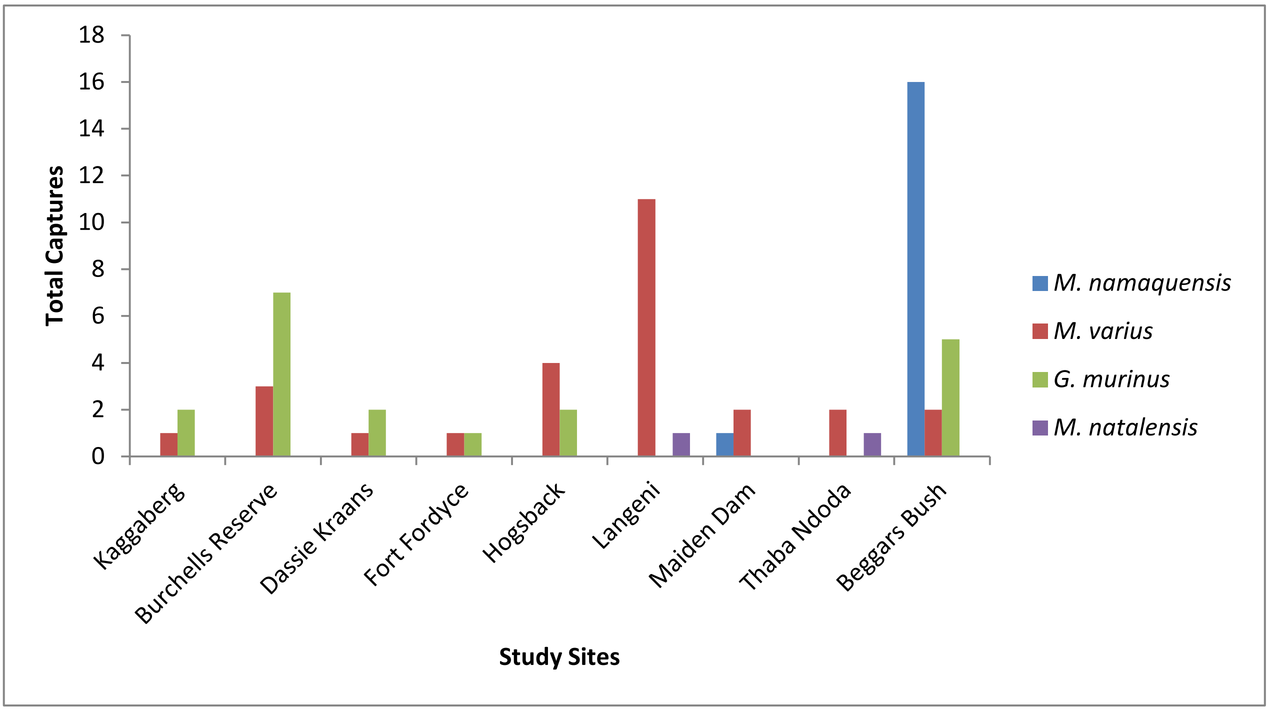

Figure 2.4: Total number of small mammal captures at the nine forest sites in the Eastern Cape, South Africa.

Overall, four species of small mammals were caught; *Aethomys namaquensis* (Namaqua Rock Mouse), *Myosorex varius* (Forest Shrew) *Graphiurus murinus* (Woodland Dormouse) and *Mastomys natalensis* (Multimammate Mouse) (Figure 2.4). *Mastomys namaquensis* had the highest number of captures but was only found in Beggars Bush and one caught at Maiden Dam. *M. varius* was the next most captured species in Langeni and it was the only species to be caught at each of the nine forests. While *G. murinus* was also caught at each of the nine sites. The photographs of each of the species caught and the forests can be found in Appendix B, Figure 1. The raw data for this figure can be found in Appendix C, Table 1.

Table 2.4: GLM table of coefficients of the best AIC models for Individual Captures and Species Richness

| Individual Captures | | Estimate | Std Error | z value | Pr(>\|z\|) |
|---|---|---|---|---|---|
| | Intercept | 7.821 | 2.693 | 2.905 | 0.003 |
| | Gradient | 4.989 | 1.356 | 3.68 | 0.002 |
| | MAPE | -0.005 | 0.002 | -2.78 | 0.005 |
| | Patch Size | -0.179 | 0.041 | 0.041 | 0.007 |
| Species Richness | | | | | |
| | Intercept | -0.568 | 0.479 | -1.186 | 0.236 |
| | Gradient | 2.79 | 1.708 | 1.634 | 0.102 |

Table 2.4 shows the best model from the AIC analysis for both individual captures and species richness. From the table it can be seen that for individual captures, as gradient increase so does the individual captures while when Mean Annual Potential Evaporation (MAPE) and patch size increase, individual captures decreases.

## 2.4 Discussion

In this study the average trap success of the nine forests was 2.34 %. In a similar study, in different sized Brazilian Atlantic forest fragments trap success was similar at 2.39 % (da Fonseca & Robinson 1990). However, a study by Schmid-Holmes & Drickamer (2001) in the temperate woodlands of southern Illinois, USA, showed that trapping success varied in different years, having a success rate of around 50 % in 1996 and around 14 % in 1997. In a fragmented forest in Queensland Australia, trap success was 2.9% (Laurance 1994). These show that trap success does vary; however, the study in Brazil which is closest in methodology does have a similar trap success.

Across all nine forests only four small mammal species were found. *M. varius* was the only species to be found in all nine study sites while the next most common species was *G. murinus* which was found at six sites (Figure 2.4). *Graphiurus murinus* has a large distribution over South Africa and is found in a wide range of habitats from grasslands to rocky areas, however, within the Eastern Cape they are only found in forests (Mzilikazi *et al.* 2012) and in particular have been found in Afromontane forests. This could be a possible reason for finding this species in six of the nine forests sampled. *Mastomys natalensis*is often found in forest clearings but rarely in the forest itself (Coetzee 1975) and is often the first coloniser in areas recovering from disturbance before it gets replaced by specialist species through succession (Perrin *et al.* 2001). This possibly explains why it was caught so few times in the forests sampled and possibly why it was caught in Langeni as the natural forest in surrounded by pine plantations and forest clearings.

Apart from Beggars Bush, there were a relatively low number of total captures, number of individuals and species richness which affected the Shannon Index values due to these low numbers. Interestingly, the two sites that had the most captures (Langeni - 16 and Beggars Bush - 28) had the most varied environmental variables compared to each other. For every variable, apart from mean annual temperature, the one site would have a high value and the other a low.

The graphs in Figure 2.3 show that sampling saturation was achieved in six of the nine sites. This means that the sampling effort was sufficient in those six sites, showing a true representation of the small mammal community at each site. The three that did have the richness estimators greater than the observed were however, never greater than one. This means that overall, the methods used to sample the small mammal richness at these sites was sufficient for this study. Other studies also found that sampling saturation was achieved between 4 – 5 days (Caro et al. 2001; Schmid-Holmes & Drickamer 2001; Yarnell *et al.* 2007; Avenant 2011).

The analysis of environmental variables on captures and species richness (Table 2.4) shows that only gradient affected both dependant variables. The GLM analysis found that when gradient increases, captures and species richness also increase while when patch size (-0.179) and MAPE (-0.005) increase, captures decreases. It would therefore seem that variables measured at both larger spatial scales (MAPE) and local spatial scales (gradient and patch size) both play roles affecting small mammal community structure. These results should, however, be analysed with the number of captures and species richness recorded at each site. Both are very low and because of these low captures and number of species, this effects the results and is difficult for true conclusions to be drawn from this.

Not many articles have been published about the effect of gradient on small mammals, however, many papers have been published on elevational gradients (altitude) and it has been found that there is a clear pattern of mid-elevational peaks in species richness (McCain 2005; Rowe 2009; Kok *et al.* 2012).Factors such as food and shelter are crucial to small mammal's survival to reproduce and survive (Hansson 1978, Hansson 1997; Andrews & O'Brien 2000) and therefore food sources and vegetation structure should affect the diversity and abundance of small mammals. However, my results found herbaceous cover and diversity to be too highly correlated to the other variables and were therefore not included. Other studies found different results; Ecke *et al.* (2001) found the cover of logs (coarse woody debris) to be of the greatest importance to explain small mammal diversity and abundance in birch and coniferous forests in northern Sweden. It is generally believed that cover from vegetation and logs as well as the scrub layer are three crucial variables to provide small mammals with food and protection from predation (Ecke *et al.* 2001). Hansson (1978) demonstrated that rocky outcrops and dense herbaceous cover were the two variables in forests that showed greater small mammal abundance. In Beggars Bush, there were many rocky outcrops throughout the forest and it was near these physical features where the majority of captures occurred. It is believed that these rocky outcrops are ideal for small mammals to escape predation by being in close proximity to them (Hansson 1978). Therefore, measuring the rocky outcrops in a forest could have been another variable to include as other factors such as rainfall and temperature have no effect on them..

Schmid-Holmes & Drickamer (2001) undertook a similar study on the effects of forest patch characteristics (patch size, shape, isolation, vegetation characteristics and microclimate) on small mammal communities in the temperate woodlands of southern Illinois, USA. Their study revealed that different small mammal species respond differently to environmental gradients, possibly due to ecological requirements. Their study found that temperature conditions did have an impact on small mammal abundances while spatial characteristics such a patch size and shape had little impact. Andrews and O'Brien (2000) found that mean annual temperature has little relation to mammal diversity and it is only large mammals (45-90kg) that are correlated with annual temperature, similar to results obtained here.

This study showed that patch size does have an impact on abundance, which is in agreement with Bower & Matter (1997) but is in contrast to other studies which show that patch size has a minimal impact of small mammal abundance (Schmid-Holmes & Drickamer 2001). Conversely to my study, other studies found that small patch sizes (>1km$^2$) have fewer species and that species are less abundant with decreasing patch size (Didham *et al.* 1998; Goodman & Rakotondrvony 2000; Nupp & Swihart 2000; Laurance *et al.* 2002; Pardini *et al.* 2005). However, Goodman & Rakotondrvony (2000) show that the sampling effort decreased with the decreasing size of the forest and this could have led to some bias in the analysis. In my study, patch size was inversely related to captures. However, a possible reason for this could be that the forest patches in my study did not represent a sufficient range of sizes to determine environmental gradients.

It must be noted that different environment gradients also affect small mammal abundances at different temporal scales. For example, temperature conditions will affect abundances at a short temporal scale while gradients such as forest structure will affect abundance at a longer temporal scale. Small mammals are well known to have inter-annual population fluctuations and this may obscure the impact of some environmental gradients on small mammal abundances in different sampling years (Schmid-Holmes & Drickamer 2001). Therefore, in order to get a true representation of the impacts of temporally different environmental variables, a study site would have to be observed over a longer period of time.

In conclusion, gradient, patch size and mean annual potential evaporation are the combination of variables that have a high probability of affecting the abundance of small mammals in Eastern Cape forests while gradient alone effects the species richness. However, the study had low capture rates and found the species richness to be low even though sampling saturation was achieved. Lawes *et al*. (2000) and Lawes *et al.* (2007) found that Afrotemperate fauna have a low species richness, fewer forest specialists and are vulnerable to invasion by generalist species from surrounding habitats. This therefore shows that my findings of low species richness and the universal presence of generalists such as *G. murinus* are similar to other South African Forest studies.

# Chapter 3

# The role of seasonality on the diversity and abundance of small mammals at the edge of an Afromontane forest patch, Eastern Cape, South Africa

## 3.1 Introduction

### 3.1.1 Climate and Fragmentation

Climate change and destruction of natural habitats are two of the greatest threats to biodiversity (Travis 2003). This is because specialist species with low colonization ability and distribution are  prone to extinction from both climate change and habitat destruction (Travis 2003). The susceptibility of a species to climate change may increase due to habitat fragmentation and therefore reducing the ability of the species to survive (Travis 2003). Changes in temperature and precipitation influence the frequency, duration and intensity of natural disturbances such as fire, drought, windstorms or introduced species (Baker 1995). This chapter looks at the effects of short term seasonal changes rather than the long term effects of climate change.

Due to isolation through fragmentation, species become vulnerable to extreme weather and seasonal changes because range expansion is restricted (McLaughlin *et al.* 2002; Opdam & Wascher 2004; Ewers & Didham 2006). An example of this was found by McLaughlin *et al.* (2002) where changes in precipitation amplified population fluctuations in the checkerspot butterfly (*Euphydrase dithabayensis*) and this led to the local extinction of two isolated populations. McLaughlin *et al.* (2002) considered that throughout its history, the checkerspot butterfly must have persisted through much larger variations in climate but would have only become susceptible to local extinctions through reduced habitat.

Species responses to seasonal changes, depends on a number of factors, such as the biology of the species, degree of resource specialisation, intensity of fluctuations and the presence/absence of predators (Leigh 1982). Therefore different species may respond differently to seasonal changes within the same community. Hence, seasonal changes from hot summers to cold winters could affect small mammal behaviour. Within disturbed habitats, such as fragmented forest patches, where their isolation, patch size and edge effects already play possible roles in changing the behaviour of small mammals, seasonal changes in climate could affect their behaviour due to these fragmentation characteristics.

Small mammals may change their behaviour due to changes in climate in various ways. One way is through seasonal hypothermia, which is the lowering of body temperature during colder months to conserve energy, while many species also breed seasonally (Eisenberg & Gould 1970; Stephenson 1994). It could therefore be expected that the abundance and diversity of small mammals would differ during different seasons of the year. The seasonal effects on small mammals can also be related to climate changes throughout the year. Andrews & O'Brien (2000) found that the effects of climate on mammal diversity are indirect and that climate affects the growth of vegetation which is a source of food and shelter for mammals and therefore has an indirect effect on mammal diversity.

### 3.1.2 Habitat Edges and Edge Effects

Habitat edges and their effect on vertebrate species are extensively studied by ecologists as their responses are crucial in understanding landscape structure and quality of habitat (Ries & Sisk 2004). Edges are defined as the border between patches of differing qualities as perceived by a specific individual of species and this alters their behaviour (Lidicker 1999; Strayer *et al.* 2003). Edges therefore separate habitat types, are species specific and edge widths vary with habitat types (Lidicker 1999). These habitat edges have important effects on the abundance and diversity of fauna and flora (Harris 1988; Fischer & Lindenmayer 2007). This is because habitat fragmentation diminishes habitat quality by increasing the proportion of habitat edge exposed to the surrounding matrix and some species may utilize these habitat edges differently compared to core habitats (Schmid-Holmes & Drickamer 2001; Wilson *et al.* 2010). The extent to which exterior environmental conditions penetrate a habitat fragment is determined by the conditions in the matrix surrounding a habitat fragment (Ewers & Didham 2006). These reduce the unaffected portions called core habitats and therefore increase the risk of extinction for species that inhabit the fragment (Ewers & Didham 2006; Wilson *et al.* 2010).

The outermost section of a forest, adjacent to the edge, becomes a zone of transition where changes in microclimate, species composition and vegetation structure occur (Stevens &

Husband 1998). With the consequences of habitat fragmentation, edges can have both positive, negative or have no effects on biodiversity depending on the specific edge or type encountered; these are known as edge effects (Ries & Sisk 2004). For example, a positive effect is that edges often result in a more diverse and greater vegetative complexity due to the mixture of two or more habitat types and therefore providing a combination of food and shelter (Heske 1995). Initially, when edge effects were first described by Leopold (1933), it was thought that all edges had positive effects on diversity and were encouraged by wildlife managers (Lidicker 1999; Harper *et al.* 2005). However, more recent findings suggest that the nesting success of songbirds decreases and predation and parasitism increase at forest edges (Gates & Gysel 1978). Increasing the habitat edge will also increase the fragmentation and isolation of patches, therefore decreasing population sizes and the potential movement between patches (Lidicker 1999).

Edge effects are changes in biological and physical conditions at an ecosystem boundary and can therefore be abiotic and biotic (Murcia 1995). Abiotic edge effects are changes in the physical conditions such as temperature, humidity and soil nutrients (Chen *et al.* 1990; Matlack 1993). Biotic edge effects are changes in biological variables such as species composition and predation (Malcolm 1994; Valladares *et al.* 2006). However, edge effects can also be caused by ecological flows across edges (Candenasso *et al*. 2003), changes in interspecific interactions (Fagan *et al.* 1999) or a combination of some of these and other factors. Ultimately the magnitude of these effects depends on the type and extent of human activity as well as the species under consideration (Schmid-Holmes & Drickamer 2001; Ewers & Didham 2006). Nupp & Swihart (2000) suggest that habitat fragmentation may affect behaviourally inflexible species and habitat specialists more than habitat generalists and behaviourally flexible species.

Three mechanisms have been identified as the most common mechanisms to explain increased abundance near edges: spill over, edges as enhanced habitats and complementary resource distribution (Ries & Sisk 2004). Spill over occurs when individuals cross the boundary into habitats that are not their preferred habitat due to proximity and the edge may be enhanced by its adjacency to a higher quality habitat (Shmida & Wilson 1985). For example, forests near open edges will be hotter, drier and have more light compared to forest interiors (Chen *et al.* 1999). By contrast, open habitats near forest edges will have lower temperatures and higher humidity resulting from increased shade from trees (Cadenasso *et al*. 1997). The second mechanism that enhances edge effects is by containing resources rare or absent from adjoining patches (Ries & Sisk 2004). Complimentary resource distribution is the third mechanism, which occurs when the edge allows easiest access to different resources found in bordering patches (McCollin 1998; Fagan *et al*. 1999).

The forest edge has a strong influence on the microclimate of a forest fragment (Turner 1996). A tall and relatively continuous forest canopy produces a cool, dark and humid microclimate (Turner 1996) in the forest interior, while at the edges there is an increase in temperature, up to 2° C (Nichol 1994), and a decrease in humidity. These effects on microclimate have been recorded up to 40 meters into the interior (Kapos 1989) while others have found the extent to be much less, around 7 – 12 meters (MacDougall & Kellman 1992).

The depth of edge effects varies widely from a few meters to a couple of kilometres depending on the ecological boundary type and the specific edge effect (Fischer & Lindenmayer 2007). The surrounding matrix is the most extensive landscape element and therefore has the dominant impact on landscape functioning. In South African Afromontane forests, the fragmented remnants are dominated by grassland matrices and this has had an effect on how forest species evolved (Wethered & Lawes 2003).

The aim of this study was to compare the seasonal changes in small mammal diversity and abundance at a forest edge in an Afromontane forest in the Eastern Cape, South Africa.

## 3.2 Methods

To assess the influence of seasonality on small mammal community structure, a small forest patch (0.67km$^2$) located in the Beggars Bush Nature Reserve near Grahamstown, was selected. This particular forest was chosen due to its close proximity to Rhodes University and had the right characteristics; a small and isolated Afromontane forest patch (See Chapter 2). This site was visited twice during each season: One sampling event comprised five consecutive trap nights within the core forest habitat (Stephenson 1994) and the second four consecutive trap nights at the forest edge (Wilson *et al.* 2010). This was done because captures were compared between the forest edge and within the forest core.

Within the forest, three transects each of 30 Sherman Traps (229 x 76 x 89 mm) were laid out perpendicular to the slope, with transect lines located along the top, middle and bottom of the slope (Pearson & Ruggiero 2003). In each transect, traps were spaced ~10 meters apart (Pearson & Ruggiero 200; Keller & Schradin 2008), and transects were spaced ~80 meters apart instead of 100 meters apart (See Chapter 2) due to the small forest size (Figure 3.1). The first transect line was spaced only 50 meters away from the forest edge, again this was due to the small forest patch size.

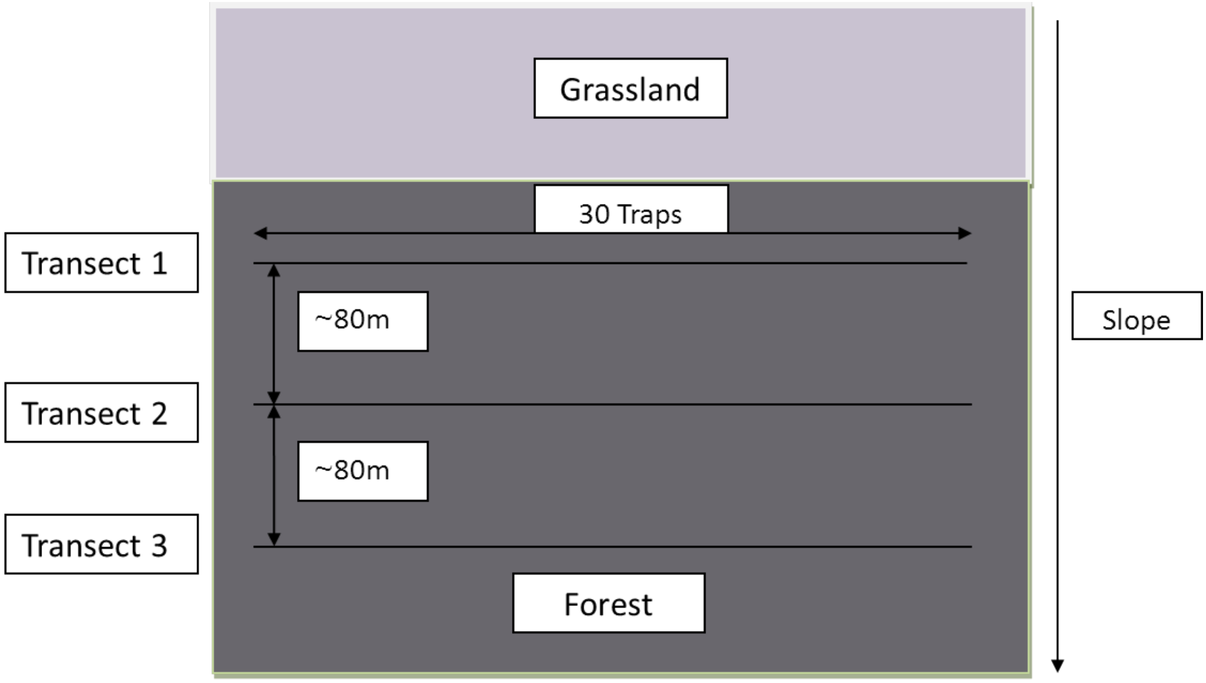

Figure 3.1: Diagram of the trapping layout for the seasonal forest study in the Beggars Bush Forest, Eastern Cape, South Africa

At the forest edge, three transect lines spaced 100 meters apart were laid out perpendicular to the forest border (Wilson *et al.* 2010). Along the transect, 75% of the traps were within the forest and 25% were in the neighbouring grassland (Wilson *et al.* 2010). At Beggars bush, the forest edge is situated on the top of a South-facing slope, adjacent to a grassland matrix (Figure 4 in Appendix B). Three transects lines were used that ran from the grassland at the top and down into the forest core (Figure 3.2). Due to the small size of the forest, only 20 Sherman traps were used in each transect with five traps (25%) placed in the grassland and 15 (75%) placed in the forest along each transect (Wilson *et al.* 2010). Captured small mammals were hair-clipped or toe-clipped to identify if the individual had been recaptured during the four trap nights. If an individual was recaptured, it would be marked on the recording sheet. Due to its small size, captured *M. varius* individuals were not toe or hair clipped.

The reason for the two different sampling methods is because literature cited different methods for sampling small mammals within the forest (Pearson & Ruggiero 2003) and compared to sampling at the forest edge (Wilson *et al.* 2010). Also, only after the sampling was it decided that the edge and forest interior would be compared seasonally as opposed to analysed separately against seasonal differences.

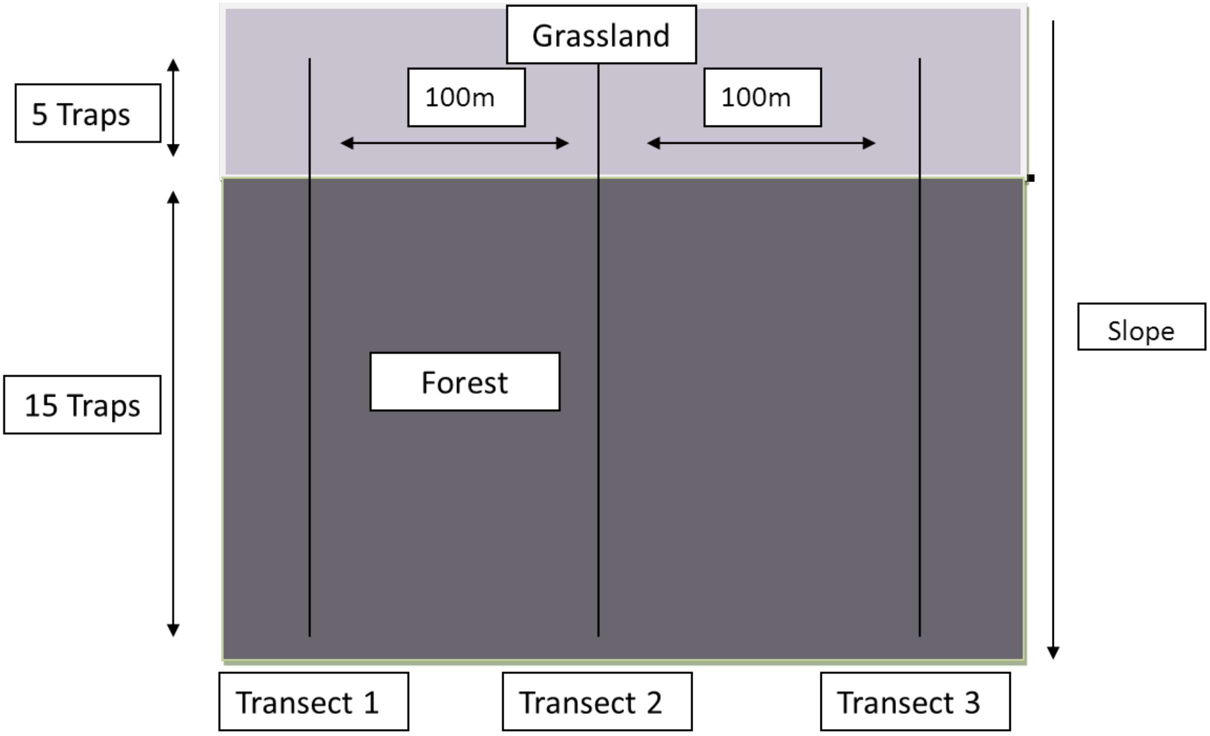

Figure 3.2: Diagram of the trapping layout for the seasonal edge effects in the Beggars Bush Forest, Eastern Cape, South Africa.

Photographs of the transects located in Beggars Bush can be found in Appendix A, Figure 2 and Figure 3.

## 3.3 Results

### 3.3.1 Total and Individual Small Mammal Captures

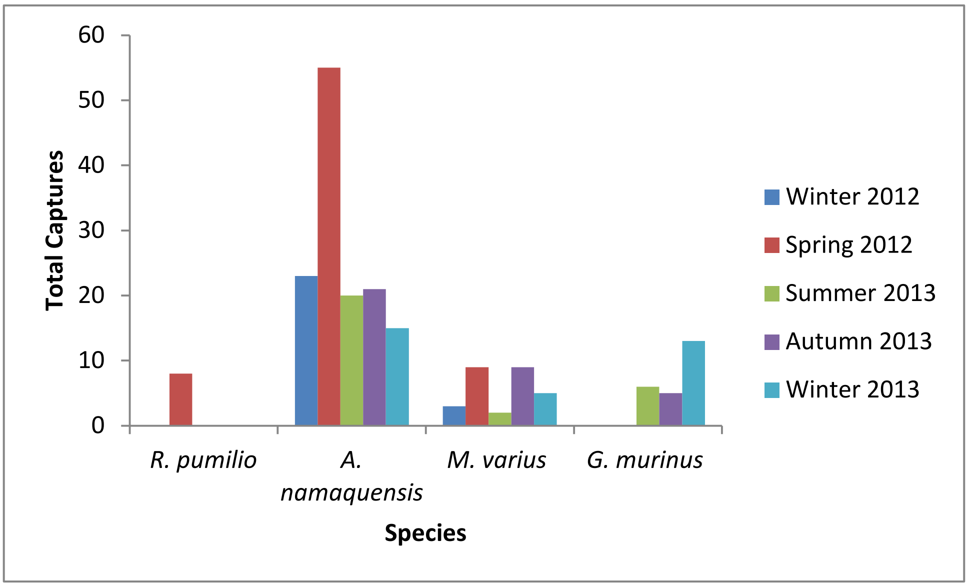

Figure 3.3: Total small mammal captures per season in Beggars Bush Forest Interior, Eastern Cape, South Africa. (*R. pumilio* – Striped Field Mouse, *A. namaquensis* – Namaqua Rock Mouse, *M. varius* – Forest Shrew & *G. murinus* – Woodlands Dormouse.

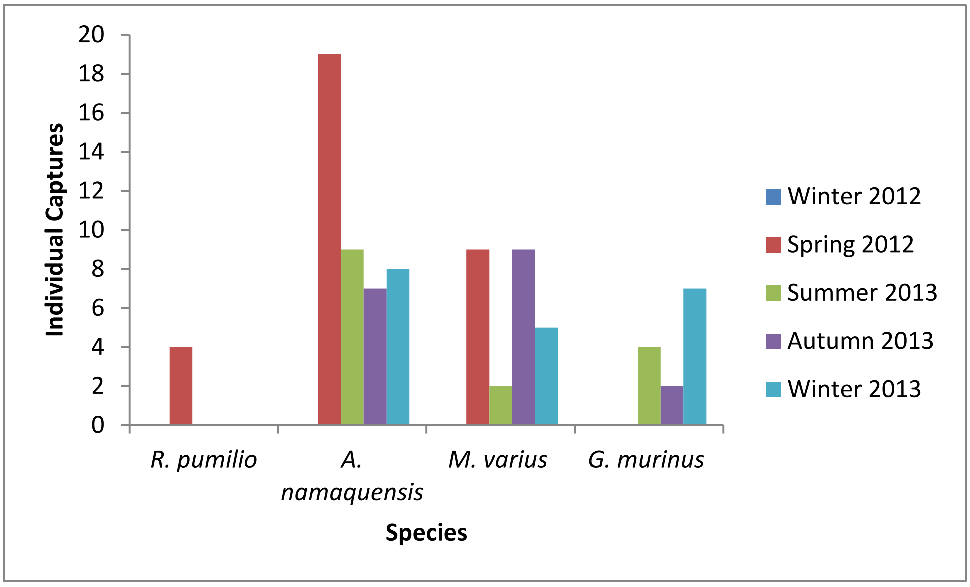

Figure 3.4: Unique individual small mammal captures per season in Beggars Bush Forest Interior, Eastern Cape, South Africa. Note: In the first Winter collection, no fur or toes were cut to identify individuals.

Within the forest, over the five sampling periods (Winter and Spring 2012; Summer, Autumn and Winter 2013) four species of small mammal were caught (Figure 3.3 and 3.4). The species, total number of captures and number of individual captures (in brackets) for each species were: *Aethomys namaquensis* (Namaqua Rock Mouse) n = 134 (43), *Myosorex varius* (Forest Shrew) n = 28 (25), *Graphiurus murinus* (Woodlands Dormouse) n = 24 (13) and lastly the *Rhabdomys pumilio* (Striped Field Mouse) n = 8 (4).

Seasonally, within the forest, there were changes in abundance and diversity for both total captures and individual captures (Figure 3.1 and Figure 3.3). *Aethomys namaquensis* and *M. varius* were the only two species caught in all five sampling sessions while *R. pumilio* was only caught during the Spring sampling trip. *Graphiurus murinus*was only caught in Summer, Autumn and in the Winter 2013 sampling session but not in 2012. There were no individual captures in the 2012 Winter period as this was a trial sampling session and individuals were not toe or hair clipped to identify individual species (Figure 3.3). Due to its small size, captured *M. varius* individuals were not toe or hair clipped within and at the edge of the forest and therefore recaptured individuals could not be identified.

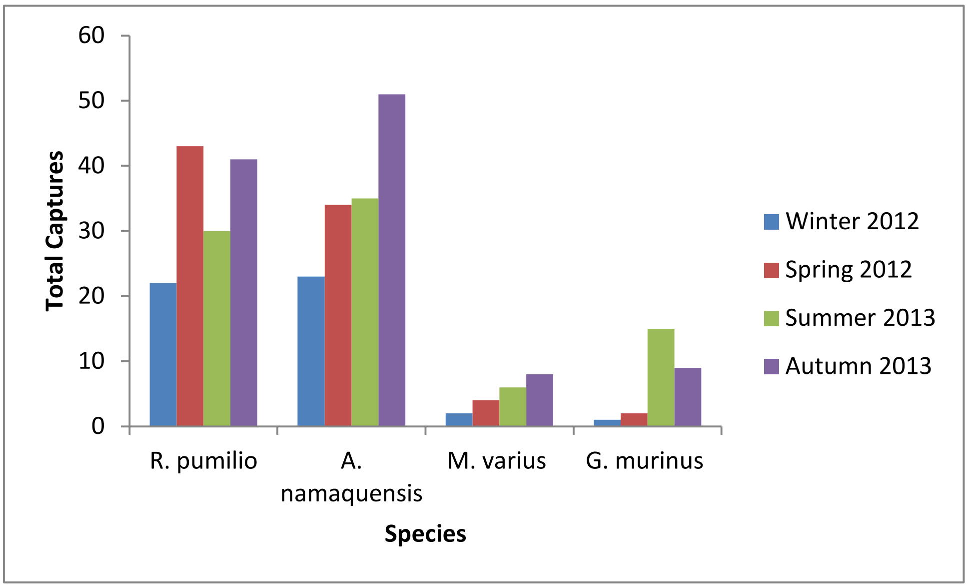

Figure 3.5: Total Small mammal captures per season at the Beggars Bush Forest Edge, Eastern Cape, South Africa

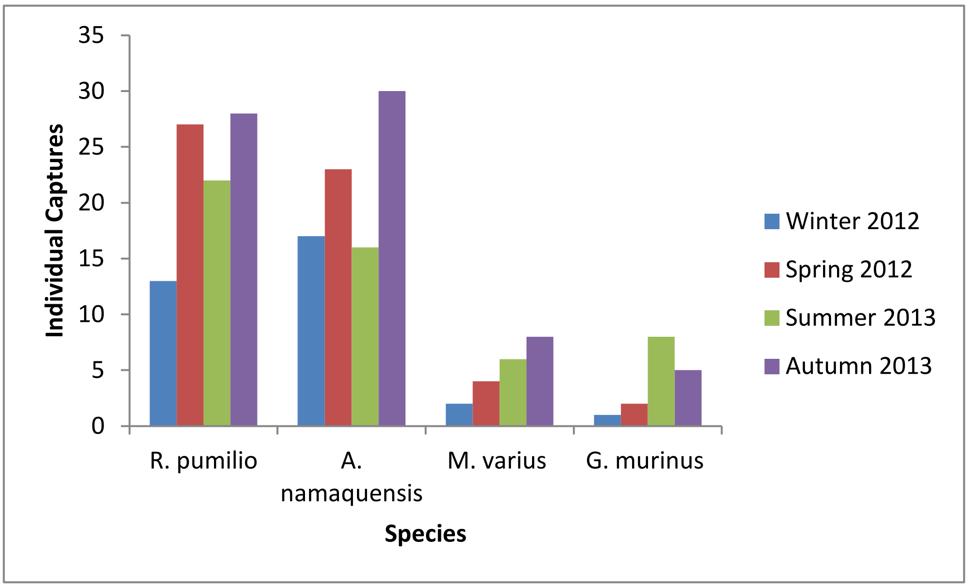

Figure 3.6: Individual Small mammal captures per season at the Beggars Bush Forest Edge, Eastern Cape, South Africa.

At the edge of the forest, over four consecutive seasons (Winter and Spring 2012 and Summer and Autumn 2013), a total of four small mammal species were caught and they were the same four species that were caught within the forest. The species, total number of captures and number individual captures (in brackets) for each species were (Figures 3.5 and 3.6). *A. namaquensis* n = 143 (86), *M. varius* n = 20 (20), *G. murinus* n = 27 (16) and *R. pumilio* n = 136 (90).

Unlike within the forest core, all four species were caught in every season at the edge. *Aethomys namaquensis* was caught the most in every season except in Spring, where *R. pumilio* was caught more frequently.

*Aethomys namaquensis*, *G. murinus* and *R. pumilio* had more individual and total captures at the edge than within the forest (Figures 3.3 to 3.6). Only *M. varius* was caught within the forest more frequently than it was caught at the forest edge (Figure 3.4 and 3.6). Seasonally, *R. pumilio* was caught more in each season at the edge than within the forest. *Aethomys namaquensis* was only caught more times within the forest than at the edge during Spring while *M. varius* was only caught more times at the edge during the Summer (Figures 3.3 to 3.6). Lastly, *G. murinus* was caught more times at the edge for every season than within the forest, except in the winter of 2013 where it was caught more than the previous winter within the forest. In terms of the total number of each species caught; *A. namaquensis, G. murinus* and *M. varius* had similar captures at the edge compared to within the forest (Figure 3.3 and 3.5). The only difference was that at the edge, of the total *R. pumilio* captures, 94% (136) were caught at the edge while there were only eight total captures within the forest.

### 3.3.2 Trapping Summary

Table 3.1: Summary of trapping data during five consecutive seasons, within the Interior of Beggars Bush Forest, Eastern Cape, South Africa

| | Total Captures | Number of Individuals | Number of Recaptures | Number of Trap Nights | Trap Success (%) | Species Richness | Shannon Index |
|---|---|---|---|---|---|---|---|
| Winter | 26 | 26 | 0 | 450 | 5.78 | 2 | 1.43 |
| Spring | 72 | 32 | 40 | 450 | 16 | 3 | 1.66 |
| Summer | 28 | 15 | 13 | 450 | 6.22 | 3 | 1.68 |
| Autumn | 35 | 18 | 17 | 450 | 7.78 | 3 | 1.87 |
| Winter 2 | 33 | 20 | 13 | 450 | 7.33 | 3 | 1.97 |
| Total | 194 | 111 | 83 | 2250 | 8.62 | 4 | |

Within the forest, a total of 194 captures occurred over the 2250 trap nights with a trap success of 8.62% over the whole sampling period (Table 3.1). A total of 111 were individual captures while 83 were recaptures. Spring had the most total captures (72) and individual captures (32) while Summer had the least total captures (28) and individual captures (15). In terms of diversity indexes, the highest Shannon index value was in Winter 2013 (1.97) while the lowest was Winter 2012 (1.43) with the other values very similar. The raw data can be seen in Appendix C, Table 2.

Table 3.2: Summary of trapping data during four consecutive seasons at the edge of Beggars Bush Forest, Eastern Cape, South Africa.

| | Total Captures | Number of Individuals | Number of Recaptures | Number of Trap Nights | Trap Success (%) | Species Richness | Shannon Index |
|---|---|---|---|---|---|---|---|
| Winter | 48 | 32 | 15 | 240 | 20 | 4 | 1.24 |
| Spring | 83 | 55 | 27 | 240 | 34.58 | 4 | 1.27 |
| Summer | 86 | 52 | 34 | 240 | 35.83 | 4 | 1.63 |
| Autumn | 109 | 71 | 38 | 240 | 45.42 | 4 | 1.49 |
| Total | 327 | 210 | 114 | 960 | 33.96 | 4 | |

At the forest edge, 327 captures occurred over a total of 960 trap nights resulting in an overall trap success of 34% (Table 3.2). A total of 210 individuals were caught while 114 were recaptures. Autumn had the most total captures (109) and individual captures (71), Spring and Summer had a similar amount of total captures (83 and 86 respectively) and individuals captures (55 and 52 respectively) while Winter had the lowest total (48) and individual (32) captures. Summer had the highest Shannon index value (1.63) while winter had the lowest (1.24). The raw data can be seen in Appendix C, Table 3.

The number of trap nights at the edge and within the forest varied due to differing methodology (See methods). The forest had 450 trap nights (3 transects x 30 traps x 5 nights) while the edge had 240 trap nights (3 transects x 20 traps x 4 nights). Yet, although the forest interior had more trap nights, the forest edge had more total captures (326), more individuals were caught (210), had more recaptures (114), a higher trap success (33.96%), and in each season, all four species found in Beggars Bush were caught. In each season, the forest edge also had the most total and individual captures than the forest interior.

### 3.3.3 The effect of the forest edge

Table 3.3: Total abundances of small mammals caught at the forest edge between the two habitats (grassland and forest) and the dividing edge at Beggars Bush Forest, Eastern Cape, South Africa. (T1 –T20 refers to the trap number along the transect)

| | Grassland (T1-T4) | Edge (T5-T6) | Forest (T7-T20) | Total |
|---|---|---|---|---|
| *R. pumilio* | 108 | 27 | 2 | 137 |
| *A. namaquensis* | 0 | 0 | 143 | 143 |
| *M. varius* | 5 | 2 | 13 | 20 |
| *G. murinus* | 0 | 3 | 24 | 27 |
| Total | 113 | 32 | 182 | 327 |

Different species preferred different habitats (Table 3.3). *Rhabdomys pumilio* (137 captures) and *A. namaquensis* (143 captures) make up the majority of captures (86%) in all three transects over all four seasons. Seventy nine percent of *R. pumilio* captures were in the grassland and 20% on the edge while 100% of *A. namaquensis* captures were in the forest core. *Myosorex varius* was the only species that truly caught in all three transects across ecotones although it was mainly caught in the forest core (65%). Lastly, *G. murinus* was caught 89% of the time in the forest core and the rest of the captures were along the forest edge. All species were caught in the forest core while only two were caught in the grassland (*R. pumilio* and *M. varius*) and all were caught along the edge except *A. namaquensis* (Table 3.3). The forest core had the majority of captures (56%), but it did contain the majority of traps. Therefore, the trap success of each habitat was the grassland at 59%, the edge at 33% and the forest at 27%.

At the forest edge, all four species were caught in every season but in varying abundances and habitat (Figure 3.2). *Aethomys namaquensis* (143 captures) was the most abundant in each season and was only caught in the forest core followed by *R. pumilio* (108 captures) in the grassland matrix. Spring (83) and Summer (86) had very similar total captures of all species combined while Autumn had the most total captures (109) particularly due to the high capture rate of *A. namaquensis* (51) within the forest core. *Myosorex varius* varied the most seasonally over the three ecotones while *R. pumilio* was relatively the most constant each season between grassland and edge captures.

## 3.4 Discussion

Within the forest interior at Beggars Bush, there was an overall trapping success rate of 8.6% (Table 3.1) while the forest edge had a trap success of 33.96%. A study on the effects of season on small mammals in two Madagascar forests showed trap success at 3% and 8% (Stephenson 1994). A study by O'Connell (1989) in a premontane forest in northern Venezuela had a trap success rate of 6% in the forest patch studied throughout the seasons. They highlighted that inter-annual fluctuations are common in small mammal studies and that unusual temperatures were possible reasons. These studies show that although small mammal capture does vary seasonally and within different forests, an 8.6% capture rate is similar to other studies.

At the forest edge, the trap success was 33.96%, which is higher than the forest interior success but similar to a study by Wilson *et al.* (2010) where the same method was used on a forest edge in the Blyde River Canyon, Mpumalanga, South Africa. The trap success rate of their forest-grassland edge study was 24.38%. The trap success was possibly higher at the edge than within the forest because more species were found at the edge due to the mixing of vegetation and specifically to my study, *R. pumilio* was the most frequently caught species (136) while only eight of these captures occurred within the forest.

In terms of diversity over the seasons, three species were caught in each season except only two in the first inter trapping period in the forest interior, whereas all four species were caught in each season at the forest edge. The diversity in the forest interior differs from that reported by Stephenson (1994) who also assessed the seasonality effects of trapping in forests in Madagascar and found that more species were caught over the austral wet summer season than in the Winter season. My results showed a constant diversity throughout the seasons within the forest (3 species) and at the edge (4 species). Therefore, not only did the edge have a higher trap success, small mammal diversity was also greater at the edge than within the forest. However, it must be noted that there was very low species richness for this study.

*Aethomys namaquensis* was the most frequently caught species (134 captures of 34 individuals) within the forest while it was caught 143 times with 86 individuals at the edge. Possible reasons for this are that Beggars Bush has many rocky outcrops and scattered rocks within the forest which are the ideal nesting and living sites of this species (de Graaff 1981). They are also partly arboreal and therefore the Beggars Bush habitat of both rocks and trees suits its natural habitat (Skinner & Chimimba 2005). *Aethomys namaquensis* is also a communal species and this could be a reason for the many captures as well as capturing different individuals in the same trap during a five night trapping session. Wilson (1975) also found this in Wankie National Park (now Hwange National Park), Zimbabwe, where seven individuals were taken from under one rock in a period of four days.

*Rhabdomys pumilio* was only caught eight times within the forest and this only occurred during spring. According to de Graaff (1981), *R. pumilio* prefers grassland but have been known to be captured along forest edges. This correlates with the findings that this species was caught 136 times (90 individuals) at the forest edge. *Rhabdomys pumilio* breeds in the summer, from September to April/May and this is the reason for higher number of captures from spring to autumn than in winter (Skinner & Chimimba 2005).

*Graphiurus murinus* is presently the smallest known African mainland hibernator and these bouts of torpor become progressively longer during winter (Mzilikazi *et al.* 2012). A study by Madikiza (2010) on *G. murinus* in a forest in the Great Fish River Reserve found the species to be active all year round while a study in the same forest by Mzilikazi *et al.* (2012) did not have any captures during their winter collection. This is similar to my results within Beggars Bush forest where I did not catch *G. murinus* in the winter of 2012 but I did in 2013. Interestingly, in another study by Madikiza *et al.* (2010) in the Great Fish River Reserve found no ground captures of *G. murinus*. Furthermore, *G. murinus* diet comprises of ground-dwelling tenebrionid beetles and millipedes so it would be expected to forage on the ground (Mzilikazi *et al.* 2012). However, similarly to my study where the species was caught on the ground, Qwede (2003) in the Afromontane forest in Hogsback (140km from Beggars Bush) and Kaplan (1995) in a forest remnant in Thomas Baines Nature Reserve (<10km from Beggars Bush) also caught *G. murinus* on the ground.

*Graphiurus murinus* showed a seasonal difference, having not been captured in winter or spring of 2012 within the forest, yet it was caught in the summer, autumn and a peak in the winter of 2013. However, during the winter and spring of 2012 at the forest edge, the species was caught only a few times. The findings within the forest differs slightly to Madikiza *et al.* (2010) who found that abundance increases in Spring, peaks in Summer and Autumn and decreases in Winter. Interestingly, this pattern was found to be exactly the same when assessing the edge effects at Beggars Bush, which is unusual because the edge effects trapping session took place only a few days after the interior trapping session. Reasons for this variation are likely to be that young are born in the Spring/Summer and can be trapped after three-four weeks (Madikiza *et al.* 2010), while the decrease in Winter is explained by decreased mobility in the cold (hibernation) and possible mortalities (Bertolino *et al.* 2001). A study by Wirminghaus & Perrin (1993) found higher abundances of small mammals in a Kwa-Zulu Natal forest during the summer than the winter. They believe that it could be due to poor food quality over the colder months of the year and hence less breeding. A possible reason for the differences between the interior and edge trapping variations in my study could be due to trap placing, as it has already been mentioned that *G. murinus* are arboreal could thus not have the exact recordings of this species.

The results from both the forest edge and the forest core studies showed that the different species preferred different habitats. *Rhabdomys pumilio* was found mainly in the grassland matrix while *G. murinus* was found mainly in the forest matrix. These findings are similar to Wilson *et al.* (2010) who also performed a study on edge effects between a forest core, its edge and a grassland matrix in an Afromontane forest in South Africa. Their results also showed that *R. pumilio* was mainly found in grassland areas and that *G. murinus* was found mainly in covered habitats in the forest core with only a few captures along the edge. The reason for *R. pumilio* findings is that it predominately prefers grassland habitats as it eats grass seeds but can also be found in a variety of habitats (Perrin *et al.* 2001). As stated in Chapter 2, a possible reason for *G. murinus* findings is that although its species has a wide range of habitats from grasslands to rocky areas, within the Eastern Cape they are only found in forests (Mzilikazi *et al.* 2012) and in particular have been found in Afromontane forests. *Myosorex varius* was found in both the forest and grassland as their habitat around South Africa ranges from dense grasslands to forests (Skinner & Chimimba 2005).

The small mammal community around the Beggars Bush forest edge comprised of a selection of forest and grassland species rather than edge associated specialists. Other studies had similar findings (Pimm 1998; Anderson & Burgin 2002; Laurance *et al.* 2007; Wilson *et al.* 2010) while Fox *et al.* (1997) and Villard (1998) also showed a greater diversity at habitat edges. Wilson *et al.* (2010) found that species diversity increased from the grassland matrix (2 species) through the edge (3 species) and into the forest core (4 species). However, Stevens & Husband (1998) found that both the diversity and abundance of small mammals caught in Brazilian Atlantic forests increased with increasing distance from the edge.

Similarly to this study, Bider (1968) found a higher diversity of mammals along a forest-field edge than in either habitat in Montreal, Canada. In Heske’s (1995) study, transects were set along forest edges and within the forest interior, and found no significant difference between the edge and interior. This study was similar to my study in that both study sites had “hard” edges, meaning there was an abrupt transition from forest core to surrounding matrix. Both of the edges were narrow zones (less than five meters) and could therefore possibly not provide the amount or type of vegetation sufficient to affect population communities significantly (Heske 1995).

In conclusion, both small mammal diversity and abundance were greater at the edge than within the forest interior. Over the seasons, small mammal diversity was the same at the edge and within the forest but captures varied seasonally in the forest and at the edge. Lastly, the findings show that the species captured comprised both habitat specialists and habitat generalists.

# Chapter 4

# General Discussion

This study is one of the most recent to survey small mammal diversity and abundances in fragmented forests in the Eastern Cape. It therefore provides a species list of small mammals found within these forests (Photographs in Appendix B, Figure 1). This information can therefore be used to compare to other forest types in South Africa or for future studies on diversity in fragmented forests.

Small mammal diversity and abundance in Eastern Cape forests seems to be low compared to other studies in Africa and around the world. A possible reason for the low diversity could be due to the fragmented nature of these forests and their destruction is leading to a decline in many species (Ewers & Didham 2006). However, this cannot be proven because no studies were done before the forests were fragmented. Another possible reason could be that inter-annual fluctuations are common in small mammal studies and this could have been a year when capture rates were low (Schmid-Holmes & Drickamer 2001). A study by Lawes *et al.* (2007) supported the hypothesis that Afrotemperate faunas have low species richness and contain less forest specialists than coastal forests. The authors believes that palaeoclimatic extinction filtering is the cause of this lowered species richness, and suggest that Afrotemperate communities are vulnerable to invasion by generalist species. This could be the reason for finding species such a *G. murinus* within these forest patches.

This study has shown that gradient, patch size and mean annual potential evaporation are a combination of variables that affect the individual captures while only gradient effects species richness. Therefore, these variables can be used to explain small mammal abundance and diversity in these Eastern Cape forests. Both gradient and mean annual potential evaporation are interesting variables, as there is little literature on how these affects mammal abundances. While patch size is a variable found in many papers explaining small mammal communities (Didham *et al.* 1998; Goodman & Rakotondrvony 2000; Nupp & Swihart 2000; Laurance *et al.* 2002; Pardini *et al.* 2005), they all found an increasing patch size to correlate to increased abundance, yet my study found the opposite. It does however, need to be noted that in my study capture rates and species richness were very low and this would therefore affect the analysis of environmental variables.

Future studies should have a larger sample size and the study should be carried out more than just once at each forest in order to get accurate trapping results from a whole year. More microhabitat environmental variables could also be used as they are better predictors of small mammal abundance and diversity (Schmid-Holmes & Drickamer 2001). Variables such as herbaceous vegetation height and rock cover should be determined as these are crucial for shelter from predation.

In conclusion, it needs to be noted that ecologically sensitive fauna are likely to be removed by climatic filtering (Balmford 1996) resulting in species-poor yet ecologically robust and persistent communities (Lawes *et al.* 2000). Therefore, variation in forest history to climatic, ecological and anthropogenic challenges would have filtered out assemblages of species that are vulnerable to current threats such as habitat fragmentation (Lawes *et al* 2007). Therefore, although fragmentation can affect small mammals, these forest mammals can tolerate these changes because they resilient generalists (Lawes *et al.* 2007).

From the results in Beggars Bush, it can be concluded that forest edges did play a significant role in small mammal diversity and abundance in forests (Murcia 1995; Sisk *et al.* 1997; Wilson *et al*. 2010). Ries & Sisk (2004) conclude that many species respond variably to edges and suggests that mechanisms are complex with very few general patterns. However, these responses are predictable when based on patterns of resource use by a species and its distribution.

The results from Beggars Bush also show that species richness is greater along the edge and decreases into the fragment interior (as found by Ewers & Didham 2006; Kotze & Lawes 2007). The mixing of fragment and matrix fauna at the habitat edge is the most common explanation for the greater species richness. Species composition is also affected by edge effects with a similar decreasing with distance from edge to interior (Ewers & Didham 2006). Forest fragment edges are also affected by climate as edges become warmer, less humid and better lit (Fetcher *et al*. 1985; Asquith & Mejia-Chang, 2005). Therefore, both diversity and abundance are greater at the forest edge than within the forest core.

Seasonally, the edge had more captures and more species were caught. However, the high number of *R. pumilio* caught at the edge does increase the total captures significantly because the other species were more or less evenly caught in each season. The responses of small mammals to habitat fragmentation depend on individual species traits (Ewers & Didham 2006). Table 3.3 shows that the species caught were habitat specialists with *R. pumilio* preferring the grassland habitat and *G. murinus* and *A. namaquensis* the forest habitat while *M. varius* was the only species that was not habitat dependant. The fact that *G. murinus* and *A. namaquensis* were not caught in the grassland could also mean that these two species have now become isolated within this small forest patch and migration to another forest patch is not possible due to the surrounding grassland matrix.

To conclude, the diversity and abundance of small mammals in Beggars Bush forest are affected seasonally within and at the forest edge and it is the edge that has the greater diversity and abundance.

Additional research on fragmented forests will give us a better understanding of how fragmentation has affected the faunal and floral diversity within these forest patches. Also by looking at the effects of more variables on small mammals, a better indication of small mammal diversity in fragmented forests will be obtained (Schmid-Holmes & Drickamer 2001). This, along with comparisons to other forest types around South Africa will lead to better understanding of factors that influence and drive small mammal diversity. However, it is also argued that these forest communities have evolved naturally in these forest patches, and although they are small, are still able to hold sustainable populations (Eeley *et al.* 1999 & Lawes *et al.* 2007). Other questions that need to be answered include: what are the long term effects of forest fragmentation and can these effects be mitigated in the future? This information can also help with conservation of small mammals and essentially the conservation of these fragmented patches across South Africa.

# References

AKAIKE, H. 1983. Information sharing and model selection. *Bulletin of the International Statistical Institute,* **50**, 277-290.

ANDERSON, L. & BURGIN, S. 2002. Influence of woodland remnant edges on small skinks (Richmond, New South Wales). *Austral Ecology*, **27**, 630-637.

ANDREN, H. 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat. *Oikos*, **71** (3), 355-366.

ANDREWS, P. & O'BRIEN, E. M. 2000.Climate, vegetation and predictable gradients in mammal species richness in southern Africa. *Journal of Zoology, London*, **251**, 205-231.

ASQUITH, N. M. & KEMJIA-CHANG, M. 2005. Mammals, edge effects, and the loss of tropical forest diversity. *Ecology*, **86** (2), 379-390.

ARCHIBALD, S., BOND, W. J., STOCK, W. D. & FAIRBANKS, D. H. K. 2005. Shaping the landscape: fire-grazer interactions in an African savanna. *Ecological Applications*, **15**, 96-109.

ARMSTRONG, A. J. & VAN HENSBERGEN, H. J. 1996. Small mammals in afforestable montane grasslands of the northern Eastern Cape Province, South Africa. *South African Journal of Wildlife research,* **26**, 11-18.

AVENANT, N. L. 2000. Small Mammal community characteristics as indicators of ecological disturbance in the Willem Pretorius Nature Reserve, Free State, South Africa. *South African Journal of Wildlife Research,* **30**, 26-33.

AVENANT, N. L. & CAVALLINI, P. 2007. Correlating rodent community structure with ecological integrity, Tussen-die-Riviere Nature Reserve, Free State province, South Africa. *Integrative Zoology*, **2**, 212-219.

AVENANT, N. L. 2011. The potential utility of rodents and other small mammals as indicators of ecosystem integrity of South African grasslands. *Wildlife Research*, **38**, 626-639.

AVERY, D. M. 1993. Last interglacial and Holocene altithermal in South Africa and Namibia: micromammalian evidence. *Palaeogeogrpahy, Palaeoclimatology & Palaeoecology*, **101**, 221-228.

BAKER, W. 1995. Long-term response of disturbance landscapes to human intervention and global climate change. *Landscape Ecology*, **10**, 143-159.

BALMFORD, A. 1996. Extinction filters and current resilience: the significance of past selection pressures for conservation biology. *Trends in Ecology & Evolution*, **11**, 193-196.

BATEMAN, B. L., KUTT, A. S., VANDERDUYS, E. P. & KEMP, J. E. 2010. Diversity patterns of small mammals in Zamales Mts., Luzon, Philippines. *Mammalian Biology*, **74**, 456-466.

BENTELY, J. M., CATTERALL, C. P. & SMITH, G. C. 2000. Effects of fragmentation of Araucarian Vine Forest on Small mammal communities. *Conservation Biology*, **14** (4), 1075-1087.

BERTOLINO, S., VIANO, C., & CURRADO, I. 2001. Population dynamics, breeding patterns and spatial utilization of the garden dormouse (*Eliomysquercinus*) in an Alpine habitat. *Journal of Zoology London*, **253**, 513-521.

BIDER, J. R. 1968. Animal activity in uncontrolled terrestrial communities as determined by a sand transect technique. *Ecological Monographs*, **38**, 269-308.

BOWERS, M. A. & MATTER, S. F. 1997. Landscape ecology of mammals: relationships between density and patch size. *Journal of Mammalogy*, **78**, 999-1013.

BRAUDE, S. B & CIZEK, D. 1988. Survival of the naked mole-rats marked by implantable transponders and toe-clippping. *Journal of Mammalogy*, **79**, 360-363.

BROWN, C. R., HUNTER, E. M. & BAXTER, R. M. 1997. Metabolism and Thermoregulation in the Forest Shrew *Myosorexvarius* (Soricidae: Crocidurinae). *Comparative Biochemistry and Physiology*, **118** (4), 1285-1290.

BUCKNER, C. A. & SHURE, D. J. 1985. The response of *Peromyscus* to forest opening size in the southern Appalachian mountains. *Journal of Mammalogy*, **66** (2), 299-307.

BURNHAM, K. P. & ANDERSON, D. R. 2002. *Model selection and inference: a practical information-theoretical approach*, 2ndedn, Springer-Verlag, New York, USA.

CADENASSO, M. L., TRAYNOER, M. M. & PICKETT, S. T. A. 1997. Functional location of forest edges: gradients of multiple physical factors. *Canadian Journal of Forest Research*, **27**, 774-782.

CANDENASSO, M. L., PICKETT, S. T. A, WEATHERS, K. C. & JONES, C. G. 2003. A framework for the theory of ecological boundaries. *BioScience*, **53** (8), 750-758.

CARLETON, M. D. & MUSSER, G. G. 2005. Order Rodentia. In *Mammal species of the world: A taxonomic and geographic reference*, 3rded, vol 2, ed. Wilson, D. E. & Reeder, D. M., 745-752. Baltimore: Johns Hopkins University Press.

CARO, T. M. 2001. Species richness and abundance of small mammals inside and outside an African national park. *Biological Conservation*, **98**, 251-257.

CASTELY, J. G. & KERLEY, G. I. H. 1996. The paradox of forest conservation in South Africa. *Forest Ecology*, **85**, 35-46.

CHEN, J., FRANKLIN, J. F. & SPIES, T. A. 1990.Microclimate pattern and basic biological responses at the clearcut edges of old-growth Douglas firstands. *Northwest Environmental Jounral*, **6**, 424-425.

CHEN, J. Q., SAUNDERS, S. C., CROW, T. R., NAIMAN, R. J., BROSOFKE, K. D. MROZ, G. D., BROOKSHIRE, B. L. & FRANKLIN, J. F. 1999. Microclimate in forest ecosystems and landscape ecology. *BioScience*, **49**, 288-297.

COLWELL, R. K. & LEES, D. C. 2000. The mid-domain effect: geometric constraints on the geography of species richness. *Trends in Ecology and Evolution*, **15**, 70-76.

COOK, W. M., LANE, K. T., FOSTER, B. L. & HOLT, R. D. 2002. Island theory, matrix effects and species richness patterns in habitat fragments. *Ecology Letters*, **5**, 619-623.

COLLINGE, S. K. & PALMER, T. M. 2002. The influences of patch shape and boundary contrast on insect responses to fragmentation in California grasslands. *Landscape Ecology*, **17**, 647-656.

COWELL, R. 2005. Estimates: *Statistical estimation of species richness and shared species from samples*. Version 8.2 user's guide and application. http://purl.oclc.org/estimates

CUMMING, G. S. 2002. Habitat shape, species invasions, and reserve design: insights from simple models. *Conservation Ecology,* **6** (1), 3.

CURRIE, D. J. 1991. Energy and large-scale patterns of animal and plant species richness. *The American Naturalist*, **137** (1), 27-49.

DA FONSECA, G. A. B. & ROBINSON, J. G. 1990. Forest size and structure: Competitive and predatory effects of small mammal communities. *Biological Conservation*, **53,** 265-294.

DE GRAAFF, G. 1981. *The Rodents of Southern Africa*. Butterworths, South Africa

DE VILLIERS, D. J. & WHITE, R. M. 2002. *Are current conservation practices conserving indigenous forest fauna?* In: Seydack, A. H. W., Vorster, T., Vermeulen, W. J. & van der Merwe, I. J. (eds), Multiple use management of natural forest woodlands: policy refinements and scientific progress. Natural Forests and Woodlands symposium III (pp. 110-121). Department of Water Affairs and Forestry, Pretoria.

DICKMAN, C. R., MAHON, P. S., MASTERS, P. & GIBSON, 1999. Long-term dynamics of rodent populations in arid Austrailia: the influence of rainfall. *Wildlife Research*, **26**, 389-403.

DIDHAM, R. K., HAMMOND, P. M., LAWTON, J. H., EGGLETON, P. & STORK, N. E. 1998. Beetle species responses to tropical forest fragmentation. *Ecological Monographs*, **68**, 295-323.

EELEY, H. A. C., LAWES, M. J. & PIPER, S. E. P. 1999. The influence of climate change on the distribution of indigenous forest in KwaZulu-Natal, South Africa. *Journal of Biogeography*, **26**, 595-617.

ECKE, F., LOFGREN, O., HORNFELDT, B., EKLUND, U. ERICSSON, P. & SORLIN, D. 2001. Abundance and diversity of small mammals in relation to structural habitat factors. *Ecological Bulletins*, **49**, 165-171.

EISENBERG, J. F. & GOULD, E. 1970.The tenrecs, a study in mammalian behavior and evolution. *Smithsonian Contributions to Zoology*, **27**, 1-127.

EWERS, R. M. 2004.The extent of forest fragmentation in New Zealand and its effects on arthropod biodiversity. *Ph.D. Thesis, University of Canterbury, Christchurch.*

EWERS, R. M. & DIDHAM, R. K. 2006. Confounding factors in the detection of species responses to habitat fragmentation. *Biological Reviews*, **81**, 117-142.

FAGAN, W. E., CANTRELL, R. S. & COSNER, C. 1999. How habitat edges change species interactions. *American Naturalist*, **153**, 165-182.

FAHRIG, L. 2003. Effects of habitat fragmentation on biodiversity. *Annual Review of Ecology, Evolution, and Systematics*, **34**, 487-515.

FEELY, J. M. 1980. Did iron age man have a role in the history of Zululand’s wilderness landscapes? *South African Journal of Science*, **76**, 150-152.

FETCHER, N., OBERDAUCER, S. F. & STRAIN, B. R. 1985. Vegetation effects on microclimate in lowland tropical forest in Costa Rica. *International Journal of Biometerology*, **29**, 145-155.

FISCHER, J. & LINDENMAYER, D. B. 2007. Landscape modification and habitat fragmentation: a synthesis. *Global Ecology and Biogeography*, **16**, 265-280.

FORMAN, R. T. T. 1997. *Land mosaics: The ecology of Landscapes and Regions*. Cambridge University Press, Cambridge.

FOX, B. J., TAYLOR, J. E., FOX, M. D. & WILLIAMS, C. 1997. Vegetation changes across edges of rainforest remnants. *Biodiversity Conservation*, **82**, 1-13

FOX, J. & WEISBERG, S. 2011. An R Companion to Applied Regression, Second edition, Vienna, Austria, Sage Publications.

GATES, J. E. & GYSEL, I. W. 1978. Avian nest dispersion and fledging success in field-forest ecotones. *Ecology*, **59**, 871-833.

GANNON, W. L., SIKES, R. S. & the animal care and use committee of the American Society of Mammalogists. 2007. Guidelines of the American Society of Mammalogists for the use of wild mammals in research. *Journal of Mammalogy,* **88**, 809-823.

GELDENHUYS, C. J. 1994. Bergwind fires and the location pattern of forest patches in the southern Cape landscape. *South African Journal of Biogeography*, **21**, 49-62.

GUSTAFSON, E. J. & GARDENER, R. H. 1996.The effect of landscape heterogeneity on the probability of patch colonization. *Ecology*, **77**, 94-107.

GOODMAN, S. M. & RAKOTONDRAVONY, D. 2000.The effects of forest fragmentation and isolation on insectivorous small mammals (Lipotyphla) on the Central High Plateau of Madagascar. *Journal of Zoology (London)*, **250**, 193-200.

HANSKI, I. & GILPIN, 1991. Metapopulation dynamics: brief history and conceptual domain. *Biological Journal of the Linnean Society*, **42**, 3-16.

HANSSON, L. 1978. Small mammal abundance in relation to environmental variables in three Swedish forest patches. *Studia Forestalia Suecica*¸ **147**, 1-39.

HANSSON, L. 1997. Population growth and habitat distribution in cyclic small rodents: to expand or to change. *Oecologia*, **112**, 345-350.

HARPER, K. A., MACDONALD, E., BURTON, P. J., CHEN, J., BROSOFSKE, K. D., SAUNDERS, S. C., EUSKIRCHEN, E. S., ROBERTS, D., JAITEH, M. S. & ESSEEN, P. 2005. Edge influence on forest structure and composition in fragmented landscapes. *Conservation Biology*, **19** (3), 768-782.

HARRIS, L. D. 1988. Edge effects and Conservation of Biotic Diversity. *Conservation Biology*, **2** (4), 330-332.

HAYWARD, M. W., WHITE, R. M., MABANDLA, M. & BUKEYE, P. 2005. Mammalian fauna of indigenous forest in the Transkei region of South Africa: an overdue survey. *South African Journal of Wildlife Research*, **35** (2): 117-124.

HESKE, E. J., 1995. Mammalian abundances on forest-farm edges versus forest interiors in Southern Illinois: Is there an edge effects? *Journal of Mammalogy*, **76** (2), 562-268.

HOLLAND, P. G. & STEYN, D. G. 1975. Vegetation responses to latitudinal variations in slope angle and aspect. *Journal of Biogeography*, **2**(3), 179-183.

JONES, C., MCSHEA, W., CONROY, M. & KUNZ, T. 1996. *Capturing mammals*, In: Measuring and monitoring biological diversity: standard methods for mammals, (eds) D. Wilson, F. Cole, J. Nichols, R. Rudran & M. Foster, Smithsonian Institution Press, Washinton DC.

KAPLAN, T. E. 1995. The habitat and ecology of small mammals in an Eastern Cape forest remnant. B.Sc. Honours Thesis, Rhodes University, Grahamstown, South Africa.

KAPOS, V. 1989.Effects of isolation on the water status of forest patches in the Brazilian Amazon. *Journal of Tropical Ecology*, **5**, 173-185.

KELLER, C. & SCHARDIN, C. 2008. Plant and small mammal richness correlate positively in a biodiversity hotspot. *Biodiversity and Conservation,* **17** (4): 911-923.

KERR, J. T. & DEGUISE, I. 2004. Habitat loss and the limits to endangered species recovery. *Trends in Ecology and Evolution,* **7**, 1163-1169.

KOK, A. D., PARKER, D, M. & BARKER, N. P. 2012. Life on high: the diversity of small mammals at high altitude in South Africa. *Biodiversity Conservation*, **21** (11), 2823-2843.

KOK, A. D., PARKER, D. M. & BARKER, N. P. 2013. Rules of attraction: the role of bait in small mammal sampling at high altitude in South Africa. *African Ecology,* **48** (1), 84-95.

KORN, H. 1987. Effects of live-trapping and toe-clipping on body weight of European and African rodent species. *Oecologia*, **71**, 597-600.

KOTLIAR, N. B. & WIENS, J. A. 1990. Multiple scales of patchiness and patch structure: a hierarchical framework for the study of heterogeneity. *Oikos*, **59**, 253-607.

KOTZE, D. J. & LAWES, M. J. 2007. Viability of ecological processes in small Afromontane forest patches in South Africa. *Aust. Ecol.*, **32**, 294-304.

LAURANCE, W. F. 1994. Rainforest fragmentation and the structure of small mammal communities in tropical Queensland. *Biological Conservation*¸ **69**, 23-32.

LAURANCE, W. F. LAURANCE, S. G., Ferreira, L. V., RANKIN-DE MERONA, J. M., GASCON, C., LOVEJOY, T. E. 1997. Biomass collapse in Amazonian forest fragments. *Science*, **278**, 1117-1118.

LAURANCE, W. F., LOVEJOY, T. E., VASCONCELOS, H. L., BRUNA, H. L., DIDHAM, R. K., STOUFFER, P. C., GASCON, C., BIERREGAARD, R. O., LAURANCE, S. G. & SAMPAIO, E. 2002. Ecosystem decay of Amazonian forest fragments: a 22-year investigation. *Conservation Biology*, **16**, 605-618.

LAURANCE, W. F., NASCIMENTO, H. E. M., LAURANCE, S. G., ANDRADE, A., EWERS, R. M., HARMS, K. E., LUIZAO, R. C. C. & RIBEIRO, J. E. 2007. Habitat fragmentation, variable edge effects, and the landscape-divergence hypothesis. *PLoS ONE¸* **2**(10), 1-8.

LAWES, M. J., MEALIN, P. E. & PIPER, S. E. 2000. Patch occupancy and potential metapopulation dynamics of three forest mammals in fragmented Afromontane forests in South Africa. *Conservation Biology*, **14** (4), 1088-1098.

LAWES, M. J. MACFARLANE, D. M. & EELEY, H. A. C. 2004. Forest landscape pattern in the KwaZulu-Natal midlands, South Africa: 50 years of change or stasis? *Austral Ecology*, **29**, 613-623.

LAWES, M. J., KOTZE, D. J., BOURQUIN, S. L. & MORRIS, C. 2005. Epigaeic invertebrates as potential ecological indicators of Afromontane forest condition in South Africa. *Biotropica*, **37**, 109-118.

LAWES, M. J., EELEY, H. A. C., FINDLAY, N. J. & FORBES, D. 2007. Robust and persistent forest faunal communities in South Africa: a legacy of palaeoclimatic change and extinction filtering? *Journal of Biogeography*, **34**, 1246-1264.

LAWES, M. J., JOUBERT, R., GRIFFITHS, M. E., BOUDREAU, S. & CHAPMAN, C. A. 2007. The effect of spatial scale of recruitment on tree diversity in Afromontane forest fragments. *Biological Conservation*, **139**, 447-456.

LEIGH, E. G. Jr. 1982. *Introduction*, In: The ecology of a tropical forest, (eds) E. G. Leigh, Jr, A. S. Rand & D. M. Windsor, Smithsonian Institution Press, Washinton, D.C.

LEOPOLD, A. 1933. *Game management*. Chas. Scribner's Sons, New York.

LEVINSKY, I., ARAUJO, M. B., NOGUES-BRAVO, D., HAYWOOD, A. M., VALDES, P. J. & RAHBEK, C. 2013. Clomate envelope models suggest spatio-temporal co-occurance of refugia of African birds and mammals. *Global Ecology and Biogeography*, **22**, 351-363.

LINDENMAYER, D. B., CUNNIGHMAN, R. B., DONNELLY, C. F., NIX, H. & LINDENMAYER. B. D. 2002. Effects of forest fragmentation on bird assemblages in a novel landscape context. *Ecological Monographs*, **72**, 1-18.

LIDICKER, Jr., W. Z. 1999. Responses of mammals to habitat edges: an overview. *Landscape Ecology,* **14**, 333-343.

LOVEGROVE, B. G. 2003. The influence of climate on basal metabolic rate of small mammals: a slow-fast metabolic continuum. *Journal of Comparative Physiology B.*, **173**, 87-112.

LOW, A. B. & REBELO, T. G. 1996. *Vegetation of South Africa, Lesotho and Swaziland*. Department of Environmental Affairs and Tourism, Pretoria.

LIU, J. G., LINDERMAN, M., OUYANG, Z. Y., AN, L., YANG, J. & ZHANG, H. M. 2001. Ecological degradation in protected areas: the case of Wolong Nature Reserve for giant pandas. *Science*, **292**, 98-101.

MACDOUGALL, A. & KELLMAN, M. 1992. The understorey light regime and patterns of tree seedlings in tropical riparian forest patches. *Journal of Biogeography*, **19**, 667-675.

MACE, G. M. & HARVEY, P. H. 1983. Energetic constraints on home range size. *American Naturalist*, **121**, 120-132.

MCCOLLIN, D. 1998. Forest edges and habitat selection in birds: a functional approach. *Ecography*, **21**, 247-260.

MADDOCK, A. H. 1992. Comparison of two methods for trapping rodents and shrews. *Journal of Zoology,* **38**, 333-340.

MADIKIZA, Z. K. J. 2010. Population biology and aspects of social-spatial organization of the woodlands dormouse¸ *Graphiurus murinus* in the Great Fish River Reserve, South Africa. Msc dissertation, University of Fort Hare.

MADIKIZA, Z. K. J., BERTOLINO, S., BAXTER, R. M. & DO LINH SAN, E. 2010. Seasonal, sexual and age-related variations in the live-trapping success of Woodlands Dormice *Graphiurus murinus, Zoological Studies,* **49** (6), 797-805.

MAGURRAN, A. E. 2004. *Measuring Biological Diversity*. Blackwell Publishing, Oxford.

MALCOLM, J. R. 1994. Edge effects in central Amazonian forest fragments. *Ecology*, **75**, 2438-2445.

MATLACK, G. R. 1993. Microenvironment variation within and among forest edge sites in Eastern United States. *Biological Conservation*, **66**, 610-616.

McCAIN, C. M. 2005. Elevational gradients in diversity of small mammals. *Ecology*, **86**, 366-372.

MCLAUGHLIN, J. F., HELLMANN, J. J., BOGGS, C. L. & EHRLICH, P. R. 2002. Climate change hastens population extinctions. *Proceedings of the National Academy of Science*, **99**, 6070-6074.

MENARD, S. 1995. Applied logistic regression analysis. Thousand Oaks, CA, Sage Publication

MUCINA, L. & RUTHERFORD, M. C. 2006. *The vegetation of South Africa, Lesotho and Swaziland*. *Strelitzia*, *19*. South African National Biodiversity Institute, Pretoria.

MURCIA, C. 1995. Edge effects in fragmented forests – implications for conservation. *Ecological Evolution*, **10**, 58-62.

MZILIKAZI, N., MADIKIZA, Z., OELKRUG, R. & BAXTER, R. M. 2012. Chapter 4: Hibernation in Free-Ranging African Woodland Dormice, *Graphiurus murinus*. In: *Living in a Seasonal World*, (eds) T. Ruf*et al*, Springer-Verlag, Berlin.

NICHOL, J. E. 1994. An examination of tropical rainforest microclimate using GPS modelling. *Global Ecology and Biogeography Letters*, **4**, 69-78.

NUPP, T. E. & SWIHART, R. K. 2000. Landscape-level correlates small mammal assemblages in forest fragments of farmland. *Journal of Mammalogy*, **81**, 512-526.

O'CONNELL, M. A. 1989. Population dynamics of Neotropical small mammals in seasonal habitats. *Journal of Mammalogy,* **70** (3), 532-548.

OPDAM, P. & WASCHER, D. 2004. Climate change meets habitat fragmentation: linking landscape and biogeographcal scale levels in research and conservation. *Biological Conservation*, **117**, 285-297.

PAINTING, C. J. & HOLWELL, G. I. 2014. Flexible alternative mating tactics by New Zealand giraffe weevils. *Behavioural Ecology*, **25** (4), 1-8.

PARDINI, R. 2004. Effects of forest fragmentation on small mammals in an Atlantic forest landscape. *Biodiversity and Consevation*, **13**, 2567-2586.

PAVONE, L. V. & BOONSTRA, R. 1985.The effects of toe-clipping on the survival of the meadow vole (*Microtus pennysylvanicus*). *Canadian Journal of Zoology*, **63**, 499-501.

PEARSON, D. E. & RUGGIERO, L. F. 2003. Transect versus grid trapping arrangement for sampling small-mammal communities. *Wildlife Society Bulletin*, **31** (2): 454-459.

PERRIN, M. R., ERCOLI, C. & DEMPSTER, E. R. 2001. The role of agonistic behaviour in the population regulation of two synoptic African grassland rodents, the striped field mouse *Rhabdomyspumilio* (Sparrman 1784) and the multimammate mouse *Mastomys natalensis* (A. Smith 1834) (Mammalia Rodentia). *Tropoical Zoology*, **14** (1), 7-29.

PIMM, S. L. 1998. The forest fragment classic. *Nature*, **393**, 23-24.

POPE, M. L., LINDENMAYER, D. B. & CUNNINGHAM, R. B. 2004. Patch use by the Greater Glider in a fragmented forest ecosystem. *Wildlife Research*, **31**, 559-568.

QWEDE, K. 2003. Observations of two populations of the woodlands dormouse (*Graphiurus murinus*) in the Eastern Cape, South Africa. Unpublished B.Sc Thesis, University of Fort Hare, Alice, South Africa.

R STUDIO. 2014. RStudio: intergrated development environment for R. (Version 0.98.1078). [Computer Software] Boston, MA.

RIES, L. & SISK, T. D. 2004.A predictive model of edge effects. *Ecology*, **85** (11), 2917-2926.

ROSENBLATT, D. L., HESKE, E. J., NELSON, S. L., BARBER, D. M., MILLER, M. A. and MACALLISTER, B. 1999. Forerst fragments in east-central Illinois: islands or habitat patches for mammals? *American Midland Naturalist*, **141**, 115-123.

ROWE-ROWE, D.T. & MEESTER, J. 1992.Habitat preferences and abundance relations of small mammals in the Natal Drakensberg. *South African Journal of Zoology*, **17**, 202-209.

ROWE, J. R. 2009. Environmental and geometric drivers of small mammal diversity along elevational gradients in Utah. *Ecography*, **32**, 411-422.

RUTHERFORD, M. C. & WESTFALL, R. H. 1986. Biomes of southern Africa – an objective categorization. *Memoirs of the Botanical Survey of South Africa*, **54**, 1-642.

SALA, O. E., CHAPIN, F. S., ARMESTO, J. J., BERLOW, E., BLOOMFIELD, J., DIRZO, R., HUBER-SANWALD, E., HUENNEKE, L. F., JACKSON, R. B., KINZIG, A., LEEMANS, R., LODGE, D. M., MOONEY, H. A., OESTERHELD, M., POFF, N. L., SYKES, M. T., WALKER, B. H., WALKER, M. & WALL, D. H. 2000. Global diversity scenarios for the year 2100. *Science*, **287**, 1770-1774.

SCHMID-HOLMES, S. & DRICKAMER, L. C. 2001. Impact of forest patch characteristics on small mammal communities: a multivariate approach. *Biological Conservation*, **99**, 293-305.

SCHRADIN, C. 2005. When to live alone and when to live in groups: ecological determinants of sociality in the African striped mouse (*Rhabdomys pumilio*, Sparrman, 1784). *Belgian Journal of Zoology*, **135**, 77-82.

SCHRADIN, C. & PILLAY, N. 2005.Intraspecific variation in the spatial and social organisation of the African Striped Mouse. *Journal of Mammalogy*, **86** (1), 99-107.

SCHULZE, R. E. 1997. South African atlas of agrohydrology and climatology. Water Research Commission, Pretoria, Report, TT82/96, University of Natal, Pietermaritzburg

SHANNON, C. E. 1948. A mathematical theory of communication. *Bell system technical journal*, **27**, 379 – 423, 623 – 656.

SHMIDA, A. & WILSON, M. V. 1985.Biological determinants of species diversity. *Journal of Biogeography*, **12**, 1-20.

SIKES, R. S., GANNON, W. L. &the animal care and use committee of the American Society of Mammalogists. 2011. Guidelines of the American Society of Mammalogists for the use of wild mammals in research. *Journal of Mammalogy*, **92** (1), 235-253.

SISK, T. D., HADDAD, N. M. & EHRLICH, P. R. 1997. Bird assemblages in patch woodlands: modelling the effects of edge and matrix habitats. *Ecological Application*, **7**, 1170-1180.

SKINNER, J. D. & CHIMIMBA, C. T. 2005.*The mammals of the Southern African Subregion*. Cambridge University Press, Cape Town. p. 814.

STEPHENSON, P. J. 1994. Seasonality effects on small mammal trap success in Madagascar. *Journal of Tropical Ecology*, **10**, 439-444.

STEVENS, S. M. & HUSBAND, T. P. 1998. The influence of edge on small mammals: evidence from Brazilian Atlantic forest fragments. *Biological Conservation*¸ **85**, 1-8.

STRAYER, D. L., POWER, M. E. & FAGAN, W. F. 2003.A classification of ecological boundaries. *BioScience*, **53**, 723-729.

SOUTHWOOD, R. & HENDERSON, P. A. 2000. *Ecological Methods*, Blackwell Science, Oxford.

SULLIVAN, T. P., SULLIVAN, D. S., RANSOME, D. B. & LINDGREN, P. M. F. 2003. Impact of removal-trapping on abundance and diversity attributes in small-mammal communities. *Wildlife Society Bulletin*, **31**, 464-474.

SYMONDS, M. R. E. & MOUSSALLI, A. 2011. A brief guide to model selection,multimodel inference and model averaging in behavioural ecology using Akaike's information criterion. *Behavioural Ecological Sociobiology*, **65**, 13-21.

TRAVIS, J. M. J. 2003. Climate change and habitat destruction: a deadly anthropogenic cocktail. *Royal Society of London*, **270**, 467-473.

TURNER, I. M. 1996. Species loss in fragments of tropical rain forest: a review of the evidence. *Journal of Applied Ecology*, **33**, 200-209.

VAN AARDE, R. J., FERREIRA, S. M., KRITZINGER, J. J., VAN DYK, P. J., VOGT, M. & WASSENAAR, T. D. 1996. An evaluation of habitat rehabilitation on coastal dune forests in northern KwaZulu-Natal, South Africa. *Restoration Ecology*, **4** (4), 334-345.

VALLADARES, G., SALVO, A. & CAGNOLO, L. 2006. Habitat fragmentation effects on trophic processes of insect-plant food webs. *Conservation Biology*, **20**, 212-217.

VILLARD, M. A. 1998. On forest interior species, edge avoidance, area sensitivity, and dogmas in avian conservation. *Auk*, **115**, 801-805.

WEIERMANS, J. & VAN AARDE, R. J. 2003. Roads as ecological edges for rehabilitating coastal dune assemblages in northern KwaZulu-Natal, South Africa. *Restoration Ecology*, **11**, 43-49.

WETHERED, R. & LAWES, M. J. 2003.Matrix effects on bird assemblages in fragmented Afromontane forests in South Africa. *Biological Conservation*, **114**, 327-340.

WILSON, V. J. 1975. Mammals of the Wankie National Park, Rhodesia. *Memoirs of the national Museum of Rhodes,* **5**, 1-147.

WILSON, J. W., STIRNEMANN, R. L., SHAIK, Z. S. & SCANTLEBURY, M. 2010. The response of small mammals to natural and human-altered edges associated with Afromontane forests of South Africa. *Forest Ecology and Management*, **259**, 926-931.

WIRMINGHAUS, J. O. & PERRIN, M. R. 1993. Seasonal changes in density, demography and body composition of small mammals in a southern temperate forest. *Journal of Zoology, London*, **229**, 303-318.

WOOD, M. D. & SLADE, N. A. 1990. Comparison of ear-tagging and toe-clipping in prairie voles, *Microtusochrogaster*, *Journal of Mammalogy*, **71**, 252-255.

YARNELL, R. W., SCOTT, D. M., CHIMIMBA, C. T. & METCALFE, D. J. 2007. Untangling the roles of fire, grazing and rainfall on small mammal communities in grassland ecosystems. *Oecologia*, **154**, 387-402.

YATES, M. D., LOEB, S. C. & GUYNN Jr, D. C. 1997. The effect of habitat patch size on small mammal populations. In: *Proceedings of the annual conference of the Southeastern Association of Fish and Wildlife Agencies*, **51**, 501-510.

# Appendix A

## The R Script used for determining the AIC and GLM values

```
rm(list=ls())

setwd("C:\\Users\\Kyle\\Desktop\\Forest determinants")

mydata<-read.csv("determinants.csv", header=T)

attach(mydata)

mydata[1:10,]

require(MuMIn)

require(car)

fit1 <- glm(richness~eledens+yearsPA+yearsELE+rainfall+thicketDENS+height+complex, data = mydata, family=poisson)

vif(fit1)

models1 <- dredge(fit1)

Models2

#Check assumptions

par(mfrow=c(2,2))

plot(fit1)

termplot(fit1,se=T,partial.resid=T)

best.fit1 <- glm(richness~rainfall*yearsELE, data = mydata, family=poisson)

summary(best.fit1)
```

# Appendix B

# Photographs

Figure 1: Photographs of the five species caught in the nine forest, Eastern Cape, South Africa *Myosorex varius* (A), *Rhabdomys pumilio* (B), *Graphiurus murinus* (C), *Aethomys namaquensis* (D) and *Mastomys natalensis* (E))

Figure 2: Sherman traps and the vegetation along Transects 1 (A), Transect 2 (B) and Transect 3 (C) at Beggars Bush, Eastern Cape, South Africa.

Figure 3: Photographs of a trap in transect 2 during all four seasons in Beggars Bush Forest, Eastern Cape, South Africa. Starting top left clockwise – winter (A), spring (B), summer (C) & autumn (D).

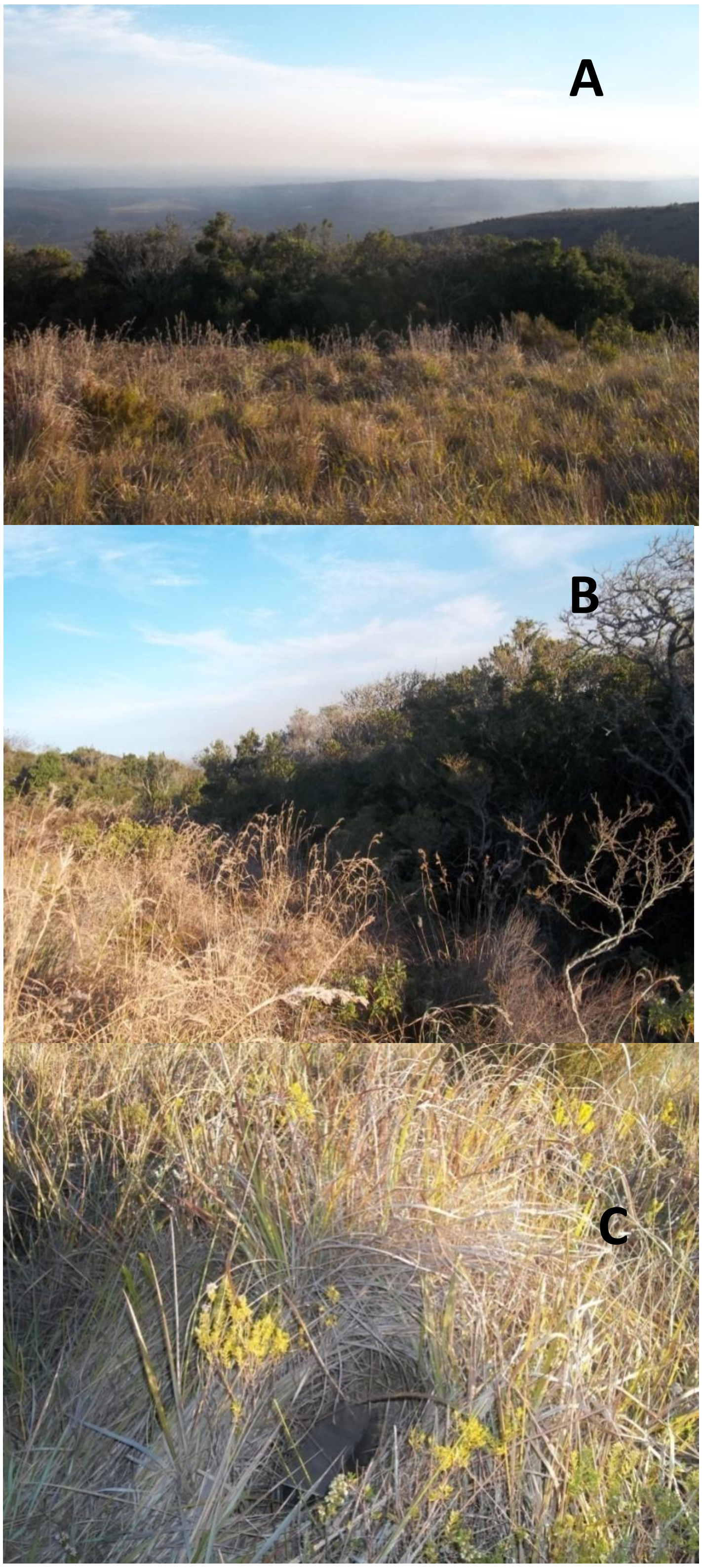

Figure 4: Photographs of the Forest-Grassland Edge at Beggars Bush, Eastern Cape, South Africa (A and B). A Sherman trap in the grassland matrix (C)

Figure 5: Photographs of the Bedford/Kagaberg Forest (1), Burchells Reserve Forest (2) and Dassie Kraans (3) in the Eastern Cape, South Africa and photographs of traps within the forest (1A, 2B, 3C).

Figure 6: Photographs of Fort Fordyce Forest (1), Hogsback Forest (2) and Langeni Forest (3) in the Eastern Cape, South Africa and photographs of traps within the forest (1A, 2B, 3C).

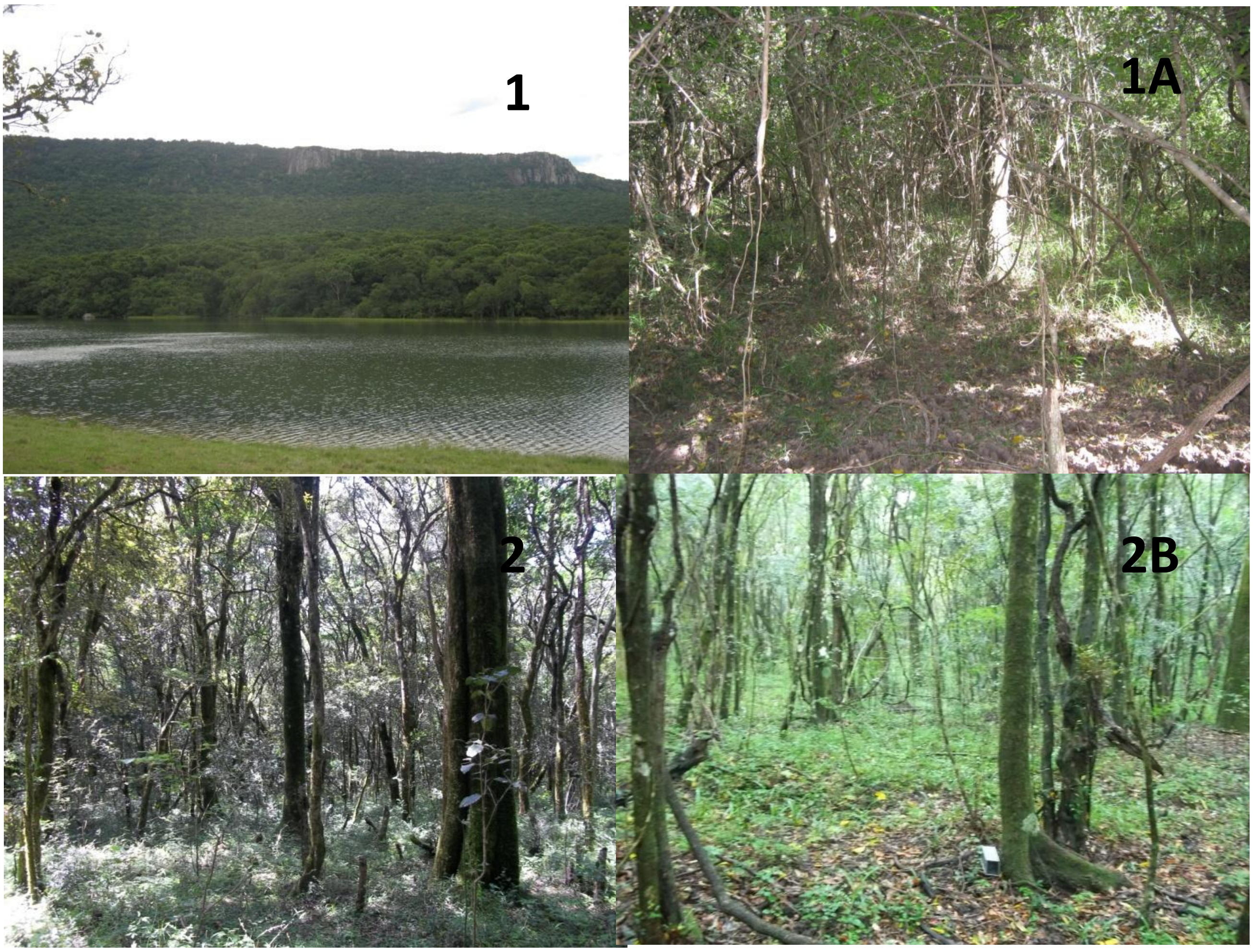

Figure 7: Photographs of Maiden Dam Forest (1) and Thaba Ndoda Forest (2) in the Eastern Cape, South Africa and photographs of traps within the forest (1A, 2B)

# Appendix C

# Data

Table 1: The raw data of the total number of species caught in each of the nine forests.

| | *M. namaquensis* | *M. varius* | *G. murinus* | *M. natalensis* | Total |
|---|---|---|---|---|---|
| Kaggaberg | | 1 | 2 | | 3 |
| Burchells Reserve | | 3 | 7 | | 10 |
| Dassie Kraans | | 1 | 2 | | 3 |
| Fort Fordyce | | 1 | 1 | | 2 |
| Hogsback | | 4 | 2 | | 6 |
| Langeni | | 11 | | 1 | 12 |
| Maiden Dam | 1 | 2 | | | 3 |
| Thaba Ndoda | | 2 | | 1 | 3 |
| Beggars Bush | 16 | 2 | 5 | | 23 |
| Total | 17 | 27 | 19 | 2 | 65 |

Table 2: The raw data of species caught in the Interior of Beggars Bush.

| **Season/Species** | **Transects** | | | **Totals** |
|---|---|---|---|---|
| | | | | |
| Winter | T1 | T2 | T3 | Total |
| *A. namaquensis* | 15 | 6 | 2 | 23 |
| *M. varius* | 3 | 0 | 0 | 3 |
| Total | 18 | 6 | 2 | **26** |
| | | | | |
| Spring | T1 | T2 | T3 | Total |
| *R. pumilio* | 8 | | | 8 |
| *A. namaquensis* | 20 | 12 | 23 | 55 |
| *M. varius* | 3 | 1 | 5 | 9 |
| Totals | 31 | 13 | 28 | **72** |
| | | | | |
| Summer | T1 | T2 | T3 | Total |
| *G. murinus* | 2 | 4 | | 6 |
| *A. namaquensis* | 7 | 5 | 8 | 20 |
| *M. varius* | 2 | | | 2 |
| Totals | 11 | 9 | 8 | **28** |
| | | | | |
| Autumn | T1 | T2 | T3 | Total |
| *G. Murinus* | 2 | 3 | | 5 |
| *A. namaquensis* | 3 | 11 | 7 | 21 |
| *M. varius* | 5 | 3 | 1 | 9 |
| Totals | 10 | 17 | 8 | **35** |
| | | | | |
| Winter 2 | T1 | T2 | T3 | Total |
| *G. murinus* | 10 | 3 | 0 | 13 |
| *A. namaquensis* | 3 | 8 | 4 | 15 |
| *M. varius* | 2 | 0 | 3 | 5 |
| Totals | 15 | 11 | 7 | **33** |

Table 3: The raw data of species caught at the edge of Beggars Bush.

| **Season/Species** | **Transects** | | | **Totals** |
|---|---|---|---|---|
| | | | | |
| Winter | T1 | T2 | T3 | Total |
| *R. pumilio* | 6 | 6 | 10 | 22 |
| *A. namaquensis* | 6 | 5 | 12 | 23 |
| *M. varius* | | 1 | 1 | 2 |
| *G. murinus* | | | 1 | 1 |
| Total | 12 | 12 | 24 | **48** |
| | | | | |
| Spring | T1 | T2 | T3 | Total |
| *R. pumilio* | 13 | 14 | 16 | 43 |
| *A. namaquensis* | 18 | 5 | 11 | 34 |
| *M. varius* | 3 | 1 | | 4 |
| *G. murinus* | | | 2 | 2 |
| Total | 34 | 20 | 29 | **83** |
| | | | | |
| | T1 | T2 | T3 | Total |
| Summer | 9 | 6 | 15 | 30 |
| *R. pumilio* | 14 | 9 | 12 | 35 |
| *M. varius* | 2 | 3 | 1 | 6 |
| *G. murinus* | 5 | 6 | 4 | 15 |
| Total | 30 | 24 | 32 | **86** |
| | | | | |
| Autumn | T1 | T2 | T3 | Total |
| *R. pumilio* | 9 | 14 | 18 | 41 |
| *A. namaquensis* | 24 | 20 | 7 | 51 |
| *M. varius* | 6 | 2 | 0 | 8 |
| *G. murinus* | 0 | 3 | 6 | 9 |
| Total | 39 | 39 | 31 | **109** |