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Ecological corridors in urban landscapes: identifying and assessing an ecological corridor in Johannesburg

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A dissertation submitted to the Faculty of Science, University of the Witwatersrand, Johannesburg in fulfilment of the requirements for the degree of Master of Science

18 February 2019 in Johannesburg

DECLARATION

I declare that this dissertation is my own unaided work. It is being submitted for the degree of Master of Science in the University of Witwatersrand, Johannesburg. It has not been submitted before for any degree or examination in any other university.



_____

Tyrone Brent Mckendry

18/02/2019

ABSTRACT

With the ever increasing rate of urbanisation, and the negative impacts that it has on ecosystems and biodiversity, urban ecologists must find effective and sustainable conservation strategies to implement in the future. Ecological corridors have been suggested as one possible management strategy for the conservation of urban wildlife in isolated habitat fragments. Increasing the level of connectivity between habitat patches by creating corridors is believed to allow for the integration of populations into a single demographic unit, thus improving the probability of survival. This study sought to assess the value of an ecological corridor (n=1) in Johannesburg. The corridor was identified using spatial mapping software and assessed against a number of landscape criteria essential for a functional corridor. Based on this assessment, one corridor was identified for further research. A focal mammal species was used to perform a least-cost path analysis on the selected corridor to determine a suitable route for wildlife movement along the corridor. Vegetation structure, cover and composition were assessed on site and all three sampling sites along the corridor were found to be comparable in both structure and species composition. A Generalised Linear Model showed no significant differences in vegetation structure or composition between sites, nor were there any significant differences between plant species richness and disturbance or vegetation structure and disturbance. Only one site had a lower percentage of exotic plant species which may be attributed to good land management.

Camera traps and live trapping were used to record the (<50 kg) small mammal species richness at each site. A total of 13 small mammal species were recorded in the corridor. Two of the study sites yielded higher small mammal species richness values than the third site which is a protected area but the result was not significant. This may indicate a lack of connectivity between this site and the rest of the corridor. This study suggests that an

urban ecological corridor that is being impacted by urbanisation is still able to provide habitat for small mammals and possibly allow for movement within the urban matrix. However, the presence and absence of species within the corridor does not take into consideration population demography which may be influenced by factors such as social behaviour. Therefore, the overall effectiveness of the urban ecological corridor as a management strategy will likely require careful consideration of not only the habitat characteristics of the corridor but also the genetic, behavioural and demographic parameters of a particular target species.

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List of acronyms and abbreviations

CBA- Critical Biodiversity Area

CoJ- City of Johannesburg

C-Plan- Gauteng Conservation Plan

DEA- Department of Environmental Affairs

EMF- Environmental Management Frameworks

ESA- Ecological Support Area

GCSA- Gauteng Conservancy Association and Stewardship Association

GDARD- Gauteng Department of Agriculture and Rural Development

GDP- Gross Domestic Product

GEMF- Gauteng Environmental Management Framework

NACSSA- National Association of Conservancies/Stewardship of South Africa

NEMA- National Environmental Management Act

RSDF- Regional Spatial Development Framework

Chapter 1.Introduction and literature review

The human population is growing at an exponential rate and is predicted to reach 9.3 billion people by 2050 and 10.1 billion by 2100 (Lee, 2011). According to the United Nations Population Fund (UNFPA, 2007) more than half of the world's population resided in urban areas in 2008. This trend in urbanisation is expected to increase over the coming decades, especially in developing countries, where it is expected that approximately 80% of the population will be living in urban communities by 2030 (UNFPA, 2007). Humans alter landscapes drastically for several reasons, but by doing so they also impact on species richness and diversity through fragmentation and degradation of natural environments (Kattel *et al*., 2013). It is therefore important to understand these patterns of degradation by looking at past land-use change to determine possible risks of those changes to the environment and coupled with past biodiversity changes, help make sustainable planning choices in the future (Tjallingii, 2000). However, it is difficult to understand the full impacts of land-use change on biodiversity because we do not fully understand ecosystem resilience, how much disturbance an ecosystem can withstand and still remain functional (Gunderson *et al.,* 2002).

One study on ecosystem resilience (Isbell *et al.,* 2015) suggested increased productivity of diverse grassland and plant communities during and after drought events compared with degraded communities but subsequent testing produced mixed results. Anthropogenic disturbances to communities appear to decrease ecosystem stability thereby reducing reliance (Isbell *et al.,* 2015).Cities "for example" are known to be major contributors to climate change through high levels of greenhouse gas (GHGs) emissions, the creation of 'heat island' effects, degrading ecosystem services and high levels of pollution

(Arnfield, 2003; Rozenweig *et al*., 2010). There is great concern that, unless these negative impacts are reduced or eliminated, there is a high risk of reaching a tipping point where the demands on environmental resources will exceed its supply or bio-capacity (Meikle and Elkadi, 2012).

Reducing negative impacts and achieving sustainability goals in cities is challenging and requires input from urban designers and planners but also an in-depth understanding of the urban ecosystem functions and 'green infrastructure' to meet the demands of the people for quality living space and access to public natural areas (Kattel *et al*., 2013). It is widely recognised that creating sustainable frameworks for development requires interdisciplinary collaboration by linking ecological and social systems (Zipperer *et al*., 2011). Many frameworks and models have been designed with these values in mind, but due to the complex nature of the urban landscape, many of these may not be able to address the problems in urban ecosystems and sustainability (Kattel *et al*., 2013).The urban landscape is highly dynamic and complex in nature, and within it open spaces or green areas are generally considered valuable to human well-being and wildlife (Pickett *et al*., 2004). However, urbanisation has directly and indirectly transformed natural areas across the globe, resulting in several irreversible negative impacts on ecological systems at various spatial scales (Luck and Wu, 2002). Since urbanisation is regarded as one of the major contributors to the degradation of the environment, there is a need for improved environmental protection plans in urban areas, including better nature conservation initiatives (Cilliers *et al*., 2004).

1.1 Impacts of urbanisation on biodiversity

Urban expansion is very well studied and documented worldwide. In the early 1900's urban development was based on concentric circles of land-use with the highest density development occurring at the centre and lower densities further away from the centre (Burgess, 1924). Over the last century high density development has doubled or even tripled in parts of America and Europe (Theobald, 2001). Urban environments are subjected to rapid expansion over a long period of time which threatens many ecosystems and places pressure on the wildlife populations that continue to exist in the remaining habitat fragments (George and Crooks, 2006). These habitat fragments rarely contain the levels of biodiversity that existed there prior to the fragmentation (Laurence *et al*., 2002) as the damage to natural environments often results in drastic declines in local biodiversity (Beardsley *et al*., 2009; Ortega-Alveraz and MacGregor-Fors, 2009). Fragments of native vegetation are dispersed within an agricultural or urban matrix (Gehring and Swihart, 2002) which creates areas of discontinuous habitat and may form barriers for many faunal species, limiting their movement between favourable habitats (Savidge, 1973; Oxley *et al*., 1974; Barnett *et al*., 1978; Cambell, 1981; Mader, 1984; Brás *et al.,* 2013; Samways and Pryke, 2016).

Some aspects of urbanisation may actually lead to an increase in biodiversity, usually through the introduction of exotic species at a more rapid rate than the rate at which native species are being lost (McKinney, 2006). Many native birds are attracted to urban forests in people's gardens, regardless of whether the trees are exotic species or not (Chace and Walsh, 2006). Other native animal species (birds, mammals and reptiles) use exotic vegetation for shelter or nesting sites. Australian researchers found an increase in species richness in Adelaide from 1836 to 2002 due to the introduction of exotic plant and animal species at a higher rate than that at which native species were lost. However, the mammals in this study

experienced a net loss in species richness (Tait *et al*., 2005). Vertebrate species richness may thus not increase to the same extent as plant species richness. One reason for this is because many vertebrates need larger areas to sustain a healthy population as opposed to plants which can have a healthy population in a relatively small area i.e. the species-area effect (McKinney, 2008). The level of impact on an environment by urbanisation is often described as a gradient from rural to urban areas, with rural areas being least impacted and urban areas being the most impacted (McKinney, 2008).Extreme urbanisation (i.e. City centres and industrial zones) often leads to a reduction in species richness for plants, vertebrates and invertebrates (McKinney, 2008); whereas areas that are affected by moderate urbanisation (i.e. suburban areas) or development gradients have species richness values which can be highly variable (McKinney, 2008).

One of most significant causes of habitat fragmentation is the development of extensive road networks which is central to the expansion of urban areas (Watts *et al*., 2007). While many species have been shown to avoid roads with high traffic levels, roadkill continues to be a large contributor to wildlife mortalities (Malo *et al.*, 2004). Studies have shown that roads can have major negative impacts on the population size of many different species (Fahrig and Rytwinski, 2009) through increased collision-related mortality (Shuttleworth, 2001), home range fragmentation, habitat loss, alteration of natural foraging and reproductive behaviours (Siemers and Schlaub, 2011). Fragmentation by roads can lead to population isolation for some species, decreasing genetic diversity and overall population health (Rico *et al*., 2009). The actual conservation value of roadside vegetation is dependent on the surrounding landscape, and in areas with sufficient natural habitat and high levels of biodiversity, these roadside refuges do not provide the suitable habitat for local threatened species (O'Farrell and Milton, 2006). In human-dominated landscapes where natural habitat

is less abundant, these refuges could be critical to maintaining biodiversity and ecosystem function (Burel, 1996).

McKinney (2008) summarised 105 studies that reported the effects of urbanisation on species richness along urban-rural gradients. From low to moderate levels of urbanisation, plants showed an increase in species richness or no change at all, while non-avian vertebrates tended to show a decrease is species richness. When the changes in species richness from moderate to high urbanisation for the different groups were analysed, it was found that plant species richness decreased in about 50% of the studies and increased in the other 50%, while vertebrates appeared to be far more affected with all 21 vertebrate-related studies showing a decrease in species richness which suggests a negative impact on native species. Thus it appears that out of the groups tested, vertebrates were the best indicators of the impacts of urbanisation along a rural-urban gradient.

Habitat quality is often used as a way to describe the ability of an environment to support organisms (Johnson, 2007), hence for the purposes of this study I used habitat quality as a measure of the species richness and composition of plants within a habitat, and habitat type to describe the percentage of vegetation cover and its structure (as per Kent and Coker, 1994 and Ives *et al*., 2011). For this study I used land-use as an indicator to differentiate between the types of disturbance that the environment had been exposed to (as per Kong, 2010).

1.2 The value of urban biodiversity

Biodiversity is seen as the basis of functioning ecosystems which provide a variety of ecosystem services (Bolund and Hunmar, 1999; Miller, 2006). There is a concern that a loss

of biodiversity may negatively affect these ecosystem services which we depend on for clean air, water, food and shelter as well as many other social values (Reid *et al*., 2005). Some studies have also provided scientific evidence to highlight the role of biodiversity in building resilience to climatic change (Elmqvist *et al*., 2003; Jansson and Polasky, 2010). Therefore, the natural environment should be considered to be as important as all other land-uses (ICUN, 2005; in Borgström *et al*., 2013). Nature is thought to be linked to human health and well-being and several studies have been done in an attempt to understand the exact reasons for this, but these relationships are likely very complex and difficult to understand (Dallimer *et al*., 2012). People from different backgrounds have different ideas about biodiversity for various reasons (Millennium Ecosystem Assessment, 2005).The benefits gained from urban nature are experienced by a wide variety of people in many kinds of green spaces such as domestic gardens, urban parks and semi-natural habitat patches (Irvine *et al*., 2010).However, recreational activities often conflict with biodiversity conservation, so the pressure on urban green spaces to service biodiversity and human recreation is higher than ever (Qiu *et al*., 2013).

One example of conflict between conservation and urbanisation is human-wildlife conflict (HWC) which is defined as any adverse effect occurring between people and animals (Conover, 2002). Some species, such as large carnivores, can be a problem in urban environments. In America, species such as the Grizzly Bear (*Ursus arctos*) frequently occur in urban areas where they pose a threat to both humans and domestic animals (Primm and Clark, 1996). As urbanisation continues to expand and replace natural habitat, it is likely that these human-wildlife conflicts will increase (Hansen and Rotella, 2002). The cause of these conflicts is well documented (Balmford *et al.,* 2002; Young *et al.,* 2010) but the solutions are not as clear and will likely require area-specific assessments. The attitudes of people living in

close proximity to wildlife will be essential in developing wildlife conservation plans and minimising HWC (Decker *et al.,* 2012). The primary causes for people's attitudes to living with wildlife and particularly those living adjacent to protected areas is usually dependant on the costs and benefits associated with the particular species being dealt with (Linnell *et al.,* 2010). This is much clearer in rural environments where conflicts between predators and livestock can have severe economic implications, thus creating negative attitudes towards wildlife (Kansky and Knight, 2014).

1.3 Protected Areas as a conservation strategy

While nature reserves play an important role in wildlife conservation, not all protected areas have the same conservation value. An example of this can be seen in America where some people refer to some National Parks as "rock and ice" because the focus on conservation here is on their unique geological characteristics and they occur in deserts and mountains (Weber, 2004).

Nature conservation in urban areas is considered to be important but is also problematic because urban ecology has unique biotic and abiotic characteristics (Ramalho and Hobbs, 2012). Urban areas tend to have more generalist species and less habitat specialists (Elmqvist *et al*., 2008) which may make it difficult to use general red data species lists as a support for nature conservation strategies (Borgström *et al*., 2013). Another problem is that these areas of 'natural' habitat are selected for protection based on certain threatened species (Borgström, 2009).The urban matrix is considered to be dangerous to many species and thus the remaining species that exist in the protected area may be difficult to support

long-term within an urban landscape, unless these areas are connected by an extensive structure of refuges or corridors (Andersson and Boden, 2008).

There has been a significant increase in urbanisation in South Africa with over 62% of the population residing in urban areas and approximately 50% of the urban population living below the poverty line (Stats SA, 2017), resulting in significant competition for space (Cilliers *et al*., 2014). As a result, many nature conservation issues are side-lined in favour of goals to improve human well-being, which focuses primarily on poverty alleviation (Hindson, 1994 in Cilliers *et al*., 2014). Another problem associated with urbanisation and poverty is the increase in the number of informal settlements which contribute to the process of habitat fragmentation and degradation of natural ecosystems as large open spaces are cleared of their natural vegetation to support lateral growth (Cilliers *et al.*, 2014).

In South Africa approximately 22% of terrestrial ecosystems are well protected, but around 78% remain unprotected. One of the most successful conservation strategies in the last decade has been the establishment of biodiversity stewardship programmes in several provinces in South Africa. These programmes allow conservation authorities to enter into agreement with landowners and managing these areas in a sustainable way (NBA, 2011). While these strategies appear to be effective for large portions of land, this same strategy becomes less effective and more difficult to manage in urban areas where parcels of land are much smaller and consequently there are many more landowners in a larger area which need to agree to the stewardship programme (Borgström, 2012). Thus, a more viable conservation strategy in urban areas may be protecting and prioritising the naturally occurring ‘corridors’ that are unsuitable for development.

1.4 Urban green spaces and ecological corridors

'Urban green space' is a broad term used to describe a range of urban land uses that provide ecological functions such as supporting biodiversity, sequestering carbon, filtering water, facilitating storm water infiltration and acting as a buffer for climatic fluctuations (Anderson *et al*., 2014). Urban green spaces are regarded as vital to the survival of urban biodiversity and the provision of ecosystem services in urban areas (Bolund and Hunhammar, 1999). These green spaces are highly variable in nature as they are exposed to different types of disturbance, but most of them to some degree or another still provide habitat for organisms. However, these green spaces are of limited value if they are isolated from one another (Borgström, 2012).

Problems frequently associated with low connectivity between populations are a loss of genetic diversity and greater competition between species in the same patches due to limited space or resources (Luque *et al*., 2012). In order to counter these negative effects, the establishment and protection of ecological corridors to link habitat fragments has been suggested as a conservation strategy (Brás *et al*., 2013).

Greenways or ecological corridors are created to join fragmented areas of habitat with the aim of providing natural areas for an array of species to survive, reproduce and potentially use as a means of dispersal between the patches (i.e. fragments of habitat that have become isolated through disturbance); local fauna are thought to benefit from this continuous habitat (Carlson *et al*., 1989). It is well known that linear strips of habitat are used by many types of fauna for refuge, foraging and possibly movement; these strips can include riparian areas, hedges, plantations and roadsides (Lewis, 1969; Polard and Relton, 1970; Roberts, 1994). Urban green spaces are becoming increasingly important for biodiversity (Cornelis and Hermy, 2004) as these areas act as refuges for many species existing in the urban

environment (Chace and Walsh, 2006; McKinney, 2008). If the purpose of an urban environmental management plan is to ensure the long-term survival of native wildlife populations, then linking fragments of remnant vegetation with landscape corridors could counter the negative effects associated with population isolation (Bolger *et al*., 2001).

The ecological value of these corridors has been recognised and some conservation plans include them as 'bio-highways' to connect natural areas within urban and peri-urban landscapes (Bryant, 2006; Parker *et al*., 2008). Understanding the biodiversity of these ecological corridors is complicated because there are a range of different interactions between the organisms and their environment at varying scales. Ecological corridors are often considered to be linear units of habitat when in reality the organisms utilising these spaces may move through them in a non-linear fashion, rather depending on them for refuge during inactive times of the day or night or using them for dispersal between habitat patches. It is therefore challenging to quantify biodiversity because it can be perceived in different ways and can change greatly over space and time (Magurran, 2004). This complexity means that conservation management of ecological corridors is difficult and focusing on one single characteristic or function is unlikely to encompass the full range of ecological values they provide (Hess and Fischer, 2001). Conservation assessment techniques that have been used previously focus mainly on corridor width (Spackman and Hughes, 1995), but there are various other environmental factors that may influence biodiversity such as the surrounding land use (Moffatt *et al*., 2004), soil properties (Désilets and Houle, 2005) and the level of habitat connectivity between patches of remnant vegetation (Tabacchi *et al*., 1998; Ward *et al*., 2002).

There are several examples of corridors being used for conservation around the world. In Australia, the Department of Sustainability, Environment, Water, Population and

Communities has implemented a National Wildlife Corridors Plan (2012) to address problems associated with urbanisation such as landscape fragmentation and degradation, climate change and invasive species. The plan was initiated by the Australian Government and was designed to support habitat connectivity to allow natural ecological movement and functions in the landscape (National Wildlife Corridors Plan Advisory Group (Australia), 2012). Another example, is the Florida Wildlife Corridor which encompasses and area of 15.8 million acres (9.5 million acres are already protected) and provides habitat for 42 endangered species and 24 threatened species (FWC, 2015). A large corridor project like this one serves as an example of the scale of conservation that can be achieved with good planning and research.

Angold *et al*., (2006) suggested that the use of corridors as a means of dispersal may be more important to small and medium sized mammals than to plants and invertebrates, highlighting the importance of using a target species or group of species when assessing the functionality of corridors for dispersal rather than as habitat. The plant species composition along different ecological corridors is highly variable but generally is composed of species that are adapted to disturbance (Naiman *et al*., 1997; Joubert *et al*., 2016). However, species which are sensitive to disturbance may be unable to survive in areas which are being subjected to disturbance and thus only the most disturbance-resistant species will remain (Dornelas, 2010). Due to the lack of available data on the effects of corridors on factors such as population density and genetic diversity, a more realistic way for me to assess the value of corridors in this study was by looking at which species are likely to use them as habitat or for movement between patches rather than "functionality" as per Angold (2005) (Bolger *et al*., 2001).

1.5 Ridges and wetlands as corridors

Faunal and floral species richness and diversity are generally much higher in geomorphological heterogeneous habitats such as rocky ridges; due to variation in aspect, slope, altitude, soil and hydrological conditions. As much as 42% of Gauteng Province, South Africa's Red Data and 41% of the endemic plant species are confined to ridges in the province (Pfab, 2001). Landscapes exhibiting spatially heterogeneous abiotic conditions provide more ecological niches for plants and animals than homogenous landscapes (Pfab, 2001). Sites with high geomorphological heterogeneity have been found to have much higher floral species richness and diversity and it can be reasonably assumed that the associated faunal communities will also be more diverse in heterogeneous landscapes (Burnett *et al*., 1998).Ridges can have significantly different biotic communities between the higher and lower lying areas (Samways and Hatton, 2000). The protection of these ridges in Gauteng can therefore provide habitat for a diverse range of species allowing them to survive in a province that is undergoing rapid urbanisation (Pfab, 2001).

Wetlands are aquatic-terrestrial ecotones that have unique physical characteristics and species composition (Naiman *et al*., 1997; Wiens, 2002). Riparian corridors are strips of vegetation that occur along watercourses such as rivers, streams and drainage lines; they are common in urban areas that have been affected by human activities (Naiman *et al*., 1993; Lee *et al*., 2004). Riparian corridors are able to provide some very important ecological functions such as water purification through wetlands which improves water quality, and use by animals for movement between remnant patches of natural vegetation and as habitat (Fabos and Ahern, 1995). They are protected because of their environmental benefits to the adjacent stream ecosystems and for their value as remnant vegetation in degraded urban environments, but in the past they were avoided by developers because of their unfavourable topography for

development and/or their susceptibility to flooding (Osborne and Kovacic, 1993; Tabacchi *et al*., 1998). These riparian corridors are sometimes referred to as riparian buffers, setbacks or greenways (Ives *et al*., 2011). The ecological value of these corridors has been recognised and conservation plans include them as 'bio-highways' to connect natural areas within urban and peri-urban landscapes (Bryant, 2006; Parker *et al*., 2008).

1.6 Current state of Johannesburg's ecosystems

Gauteng is a relatively small province compared with the other provinces in South Africa, covering just 1.4% of the total land area of the country (World Atlas, 2018). It is considered to be highly urbanised and is recognised as the financial capital of Africa (Stats SA, 2018). It is also one of the most densely populated provinces with over12 million people and this number increases every year (World Atlas, 2018). Despite its small size, Gauteng contributes more than 33% of national income to the country's economy and 10% to the GDP of the entire African continent. Johannesburg is the largest city in South Africa and is the capital of Gauteng Province; it is linked to other major cities by large highway interchanges. The urban area extends in all directions essentially uninterrupted through a number of towns. North of Johannesburg is the city of Pretoria which is the capital of South Africa. Pretoria's southern suburbs are expanding further south and beginning to merge with Johannesburg's urban sprawl.

The National Biodiversity Assessment (NBA) was designed to assess South Africa's biodiversity and ecosystems (NBA, 2011). The South African National Biodiversity Institute (SANBI) led the project in partnership with various organisations, including the Department of Water Affairs, Council for Scientific and Industrial Research (CSIR) and South African

National Parks (SANParks). The report found that approximately 40% of terrestrial ecosystems in South Africa are threatened (9% critically endangered, 11% endangered and 19% vulnerable). Grassland was highlighted as one of the most threatened biomes in South Africa. One of the most significant findings from the NBA is that the rate of loss of natural habitat is so high in some parts of the country such as Gauteng, KwaZulu-Natal and North West Province that if this rate continues, there will be no natural habitat left outside of the protected areas by the year 2050 (NBA, 2011*et al*., 2012).

High levels of development and urban sprawl have impacted the natural environment in large portions of the city of Johannesburg (Compaan, 2011). The area that was once dominated by vast open grasslands has been fragmented and transformed to such an extent that today many grassland types (such as Egoli Granite Grassland) are considered threatened and can only be found in a few remnant patches scattered mostly on the outer edges of the city (Compaan, 2011).

1.7 Gauteng conservation strategy

The Gauteng Department of Agriculture and Rural Development (GDARD) is the governmental department responsible for agricultural and rural development in Gauteng. Their vision is to provide integrated management for sustainable agriculture, rural development and the use of natural resources in order to improve the quality of life for all in Gauteng. They had a five year strategic plan (2010-2014) that was an outcome-based approach to achieve their goals with a focus on sustainable rural communities (outcome 7) and protecting and enhancing environmental assets and natural resources (outcome 10) (GDARD, 2014). GDARD's conservation plan (C-Plan) highlights the priority areas for

biodiversity conservation in the Gauteng Province. As part of the systematic conservation planning process, the C-plan produced a map that delineated the biodiversity priority areas which are Protected Areas, Critical Biodiversity Areas, and Ecological Support Areas, the map is called the Critical Biodiversity Areas Map (Compaan, 2011). While the C-plan provides a strong basic dataset on species data and a good quality land cover map, it was identified in the report that certain refinements would improve the quality and strength of the product (Compaan, 2011).

The C-Plan shows Critical Biodiversity Areas (CBAs) within Gauteng which are considered to be irreplaceable, important and protected areas (Compaan, 2011). The C-Plan 3.3 showed that 26% of Gauteng is comprised of CBAs and only 2.4% of them are formally protected (Compaan, 2011). In addition, Ecological Support Areas (ESAs) are an important part of the C-Plan to provide long-term sustainability (Compaan, 2011). ESAs are an integral part of the hierarchy of biodiversity but it is not possible to include every feature of biodiversity. These ESAs include dolomite outcrops, rivers, wetlands, pans, corridors for climate change and species migration and ridges (Compaan, 2011).

In addition to the nature reserves managed by GDARD and the City of Johannesburg, there are also several conservancies which have been established by non-governmental organisations (NGOs), which fall under the Gauteng Conservancy Association (GCA handbook, 2009). The GCA represents rural and urban conservancies in Gauteng and they promote conservation on private property to counter the loss of green-belt areas. By definition “a conservancy is the voluntary co-operative environmental management of an area, by its community and user groups, and in respect of which registration has been granted by the relevant provincial nature conservation authorities.” (GCA Handbook, 2009, pg. 4).

According to GDARD's C-Plan (Compaan, 2011), ecological corridors are highlighted as important features creating a network of habitat which may facilitate wildlife dispersal and species migration between biodiversity priority areas. According to the report, GDARD also acknowledges the importance of these corridors in maintaining ecological processes. A standard buffer zone of 600m was applied to the identified corridors and these areas are labelled as Ecological Support Areas (ESA's). According to GDARD's C-Plan technical report (2014) future iterations of the C-Plan will aim to tightly prioritise biodiversity features and identified areas to increase the spatial efficiency of the conservation plan which can be considered as "land hungry" for South Africa's major development hub; the plan should also aim to identify the most efficient possible network of priority areas to maintain biodiversity within Gauteng (GDARD, 2014).

This type of 'network' is exactly what corridor conservation planning is about. In order to implement a corridor plan for Gauteng, the corridors need to be assessed on their current ecological state to determine whether or not they are able to sustain biodiversity within the city. Currently there are little to no data to support the protection of these corridors within the province and thus research of this type could be imperative to the future survival of wildlife within the region. This research could also provide data that may be used to help develop corridor conservation models in other parts of South Africa, thus aiding in large scale protection for biodiversity. If however the corridors are found to be too degraded to support or sustain biodiversity, then conservation planners and organisations can focus their efforts elsewhere to develop more effective nature conservation strategies for Gauteng and South Africa.

1.8 Aim and objectives

My study aimed to assess the condition of an ecological corridor in Johannesburg and its potential for species movement and habitat provision. Due to the scale of the data collection for this research project, only one corridor was assessed. Objective 1 was to locate urban green spaces in Johannesburg and identify one ecological corridor that was connected to protected areas or nature reserves. Objective 2 was to determine the most efficient path for wildlife movement. Objective 3 was to assess the ecological condition of the corridor based on vegetation characteristics, disturbance levels, human activity and potential movement barriers. Objective 4 was to assess the small mammal diversity along the corridor to determine its usefulness for small mammals.

1.9 Layout of the dissertation

My dissertation comprises of an introductory chapter (Chapter1), three experimental chapters (Chapter 2-4) and a general discussion (chapter 5). Chapter1 provides a detailed literature review related to urban ecology and corridor science, as well as the aim and objectives of my study. Chapter 2 discusses the findings of spatial analysis to identify corridors in Johannesburg. Chapter 3 provides a habitat assessment of a selected corridor using vegetation data and Chapter 4provides an assessment of the value of the corridor to small mammals. The final discussion evaluates at the usefulness of the urban ecological corridor in the study in providing connectivity and habitat for wildlife and revisits key concepts in order to explain the processes driving the system. Pages are sequential for the entire dissertation but figures and tables are in sequence for each chapter. There is a single reference section provided at the end of the dissertation.

Chapter 2.Identifying and assessing the landscape features of an ecological corridor in Johannesburg

2.1 Introduction

Anthropogenic activities are known to negatively impact on the natural environment by altering landscapes to suite human needs (Pierik *et al*., 2016). For example, urbanisation has been shown to cause habitat fragmentation and degradation which often result in a loss of biodiversity (Wu, 2008; Beninde *et al*., 2015). Rapid urbanisation may result in conflicting demands for open space and land for development that require careful spatial planning to ensure that human needs are met without compromising biodiversity conservation objectives (Penteado, 2013). The urban environment is typically a stark contrast to the natural environment and increasing development often results in developed areas which have strips or fragments of ecologically important habitat nearby (Marzluff and Ewing, 2001; Kong *et al*., 2010). The impacts of urbanisation are mostly irreversible and as such it is essential to implement the best spatial planning techniques to ensure sustainable use of the landscape and to mitigate any negative effects on the natural system (Kong *et al*., 2010).

Ecological corridors aim to counter habitat fragmentation by connecting protected areas and natural areas with high natural value, both structurally and functionally, throughout the urban landscape (Samways and Pryke, 2016). Recent research suggests that the management of connectivity could be useful in achieving sustainable development in urban areas (Kong *et al*., 2010). Several researchers have suggested that connectivity should only be considered as important once the size and quality of the protected areas have been improved because the value of larger protected areas to species conservation is more certain than the

value of having a high level of connectivity (Hodgson *et al*., 2009; Kong *et al*., 2010). However, it can be argued that the range that certain species may need cannot be provided by expanding the size of the protected areas (Krosby *et al*., 2010). One of the primary motivations for increasing connectivity is to mitigate the negative impacts of fragmentation in an urban environment which could allow for species movement and thereby reduce the likelihood of local extinction (Krosby *et al*., 2010). Thus the management of connectivity can be seen as a method for promoting animal dispersal within the urban landscape, allowing them to utilise natural, semi-natural and potentially urban spaces through the creation of ecological corridors and habitat patches that encourage matrix permeability (Krosby *et al*., 2010).

The management of connectivity involves ecological restoration which is the process of changing disturbed or damaged ecosystems to represent a 'natural' state that provides structure, diversity and healthy functioning of the specific ecosystem (Lewis, 1969). It has been suggested that there is not enough time to gather sufficient baseline data to find a solution to the current biodiversity crisis and researchers must rely on current fundamental conservation knowledge to produce an intuitive solution (Meir *et al*., 2004; Lindenmayer *et al*., 2013; Samways and Pryke, 2016). One of the challenges of connectivity between protected areas is the variability in ecological requirements for different species and improving connectivity for one species may create a barrier for another species (Mimet *et al*., 2013).

Ecological corridors may play a significant role in urban resilience to climate change (Isbell *et al.*, 2015). They can increase the value of protected areas by providing additional space and spatial options for times when climatic conditions are poor, such as during droughts or flooding events (Morecroft *et al.*, 2012). It is likely that organisms have, through

natural selection, developed survival strategies for climatic changes by adjusting their movement patterns to find suitable living conditions. Ecological corridors support whole communities and ecosystem processes which are considered to be more important than a single species approach to conservation (Samways and Pryke, 2016).

2.1.1 Choosing the correct scale for spatial analysis

Landscape ecologists generally focus on the spatial patterns created by the interactions between different elements which form landscape mosaics (Wiens *et al*., 1993). Consequently, many research projects aim to distinguish between "suitable" habitat and the "unsuitable" matrix. This allows for a quick visual assessment of the extent of habitat which has been cleared, identification of isolated patches, and to generate models which simulate possible movement patterns of wildlife between habitat fragments. Using this method of landscape interpretation does not accurately allow for an assessment of habitat heterogeneity because it considers the patch as the smallest unit of investigation and does not consider within-patch dynamics (Fahrig and Merriam, 1994). Wiens (1997) showed that habitats are heterogeneous at virtually any level of scale. Additionally, it was shown that organisms may 'perceive' the level of landscape heterogeneity differently (Wiens and Milne, 1989) and therefore accurate analysis of a corridors functionality should aim to assess landscape elements and an organisms response within the same domain (Wiens, 1989).

The discipline of wildlife corridor research often uses a simplistic approach assuming that connected habitat patches are homogenous (Haddad *et al*., 2003). This means that ecological corridors are frequently viewed as "suitable" habitat surrounded by "unsuitable" matrix and therefore represents the smallest scale for assessment (Peng *et al.*, 2017). When

the habitat patch is considered as the smallest sampling unit then the life-history of any organisms being investigated within that patch must be considered (Haddad *et al*., 2003). Corridor research is often done in fragmented landscapes which actually represent two spatial scales: the suitable habitat within the matrix; and resources within the habitat patches which can be defined as non-continuous (Hanski, 1994).

2.1.2 Urbanisation in South Africa

Many nature conservation issues are ignored in favour of goals to improve human well-being, which is based primarily on poverty and the various issues surrounding poverty (Hindson, 1994; Cilliers *et al*., 2004). In South Africa approximately 22% of terrestrial ecosystem types are well protected but around 35% remain unprotected (Compaan, 2011). One of the most successful conservation strategies in the last decade has been the establishment of biodiversity stewardship programmes in several provinces in South Africa. These programmes allow conservation authorities to enter into agreement with landowners to manage these areas in a sustainable way (NBA, 2011). While these strategies appear to be effective for large portions of land, this same strategy becomes less effective and more difficult to manage in urban areas where pieces of land are much smaller and consequently there are many more landowners per unit of land which need to agree to the stewardship programme. Thus a more viable conservation strategy in urban areas could possibly be achieved by protecting and prioritising the naturally occurring 'corridors' that are unsuitable for development or are still undeveloped.

There are numerous nature reserves throughout Gauteng which are managed and protected by the Gauteng Department of Agriculture and Rural Development (GDARD). In

GDARD's Strategic Plan 2010-2014 they have adopted an outcomes-based approach to achieve their goals. These outcomes aim to provide vibrant, equitable and sustainable rural communities with food security for all and to protect and enhance our environmental assets and natural resources by reducing the impacts of climate change, applying sustainable natural resource management and protecting biodiversity (GDARD, 2009). In GDARD's latest Strategic Plan (2010-2014), protecting biodiversity is listed as the focus of one of their strategic objectives for Outcome 10 which includes the following points: Gauteng Protected Areas Expansion Strategy, four priority habitats mapped and protected per annum, and four action plans developed to restore, maintain or reduce the decline of population of species of selected taxonomic groups (GDARD, 2009).

According to GDARD's Gauteng Conservation Plan Version 3.3 (Compaan, 2011), corridors are highlighted as important to facilitating wildlife dispersal and species migration between biodiversity priority areas, which is particularly important in response to climate change. According to the report they also acknowledge the importance of these corridors in maintaining ecological processes. In order to establish these ecological corridors as an urban conservation strategy in Johannesburg, GDARD and the City of Johannesburg require empirical evidence to determine if the existing corridors are functional. The first step in determining if the corridor is functional is by first identifying one within the city and assessing its physical features by using spatial mapping techniques. Therefore the aim of this study was to identify a potential and viable ecological corridor in and around Johannesburg using digital land-use maps, aerial photographs and criteria identified by corridor scientists to be of importance for a viable corridor (from an ecological perspective). The objectives of the study were (1) to locate urban green spaces in Johannesburg and identify one ecological corridor that was connected to protected areas or nature reserves. Objective 2 was to

determine the most efficient path for wildlife movement and (2) to determine the most efficient path along the corridor for wildlife movement.

2.2 Materials and Methods

The study area formed part of the Johannesburg Municipality, Gauteng Province, South Africa (26° 11' S and 28° 2' E). Google Earth Pro (2017) and ESRI's ArcGIS 10.3.1 (ArcGIS Desktop: Release 10. Redlands, CA: Environmental Systems Research Institute. 2015) were used to create a map showing the Protected Areas in Johannesburg (Appendix A) which were used as a starting point to identify any potential ecological corridors that might occur in the region.

2.2.1 Selecting a corridor

Using the protected areas in Appendix A and green space layers provided by the Gauteng Department of Agriculture and Rural Development, I identified several possible corridors in the city using ArcGIS and narrowed this selection down to one based on its length of continuous connectivity. Only one corridor was tested for this study due to time and resource constraints. The corridor was assessed on corridor width, length, habitat connectivity (suitable stretches of continuous habitat) and site accessibility. I selected one corridor which looked suitable for the purpose of this research based on presence of a protected area, the presence of sufficient open space/natural (minimum 50 m as per Cotter *et al.*,2014) land-use next to or near to the protected area and proximity (land-use adjacent to the corridor) to different urban land-use types which may impact the corridor (roads, developed areas, fences, walls etc.), as well as corridor width, length, and potential barriers to movement. Corridor

width was a key component in selecting the corridor but this value can be highly variable and dependent on specific ecosystems and the wildlife utilising them, for example Bond (2003) suggested a minimum size of approximately 300m for a wildlife corridor for larger mammals while Cotter *et al.*, (2014) suggested a corridor of 40-50 m to be sufficient for amphibian movement. In this study I used a minimum width of 50 m as a way to eliminate unsuitable corridors and because corridors within Johannesburg rarely appear to be wider than 100m due to high levels of development. The level of accessibility of each corridor was determined during site visits and was based on my ability to gain access (on foot) to multiple sites along the corridor to conduct my biodiversity surveys. Several sections of the corridor were on private property restricting access for surveys.

2.2.2 Least-cost path analysis

Spatial characteristics were used to select the corridor and then least-cost path analysis in ESRI's ArcMap 10.3.1 was used to confirm that the corridor provided a suitable path for wildlife movement. The least-cost path function is a raster-based optimisation algorithm in the Grid module that was initially designed to determine a path between two sites by using a theoretical cumulative-cost surface from the source outwards towards the destination (Walker and Craig head, 1997).

Before the least-cost path could be assessed, the core habitat areas which act as sources or destinations within the corridor had to be identified (as per Kong *et al*., 2010). These core habitats were used to determine the most optimal sample sites, where possible at even intervals.

Each land-use type (e.g. residential areas, roads, agricultural lands etc.) was assigned a weight (impedance value) which was assumed to represent the degree of anthropogenic disturbance that would have an impact on the dispersal of wildlife along the ecological corridor (as per Kong *et al.*, 2010). The impedance values are theoretical variables used to estimate the resistance to movement that an animal may experience in each land-use type (Knaapen *et al.*, 1992). It is important to note that the impedance value will have different values for different species; for example a water body may be a barrier for most terrestrial mammals but not necessarily for birds. Some construction areas will include residential green space which could function as habitat or corridors but if they have been impacted by human disturbance and isolation from other patches they will be assigned high impedance values (Kong *et al.*, 2010). For the purposes of this study, I selected a focal small mammal species which was known to occur within the study area (Compaan, 2011) which was used to assign impedances values to the corridor. The species that I selected to represent the impedance value was the black-backed jackal (*Canis mesomelas*) which is a habitat generalist which we do not expect to be able to climb over walls and fences, making it particularly susceptible to anthropogenic barriers and an ideal candidate to represent corridor impedance. The impedance values that were used to create a weighted land-use map, which was then used to generate a cost surface in the least-cost path analysis, are shown in Table 1(as per Kong *et al.*, 2010). The least-cost path analysis modelling software in ESRI's ArcMap 10.3.1 required a one to ten gradient rating system for each type of impedance. In this cost surface, man-made physical structures such as buildings and roads were given a high rating of 9-10. Natural barriers such as water bodies were also assigned a high value of 8 because the focal mammal species is terrestrial and unlikely to be able to cross large areas of water. Residential areas generally include some green space depending on the density of the development but a

frequently associated with high levels of disturbance which gave them impedance rating of 5-7. Golf courses and cultivated land were given an impedance rating of 4. Bare open land was given an impedance rating of 2. All natural green spaces including parks were given an impedance value of 1.

Table 2, 1.Land-use type and impedance values for ecological corridor usage adapted from Kong (2010).

Category	Description	Impedance value (1-10)
Buildings	Man-made structure inhibits movement	10
Industrial	High impact land-use	10
Mining	Medium to high impact land-use	9
Roads	Dangerous barrier to cross	9
Water	Dams and large rivers	8
Sports/Recreation	High human disturbance	7
Residential-cluster	High density development	7
Residential	Medium density development	6
Small holdings	Low density development	5
Golf Courses	Large open space but human disturbance	4
Cultivated	Low intensity use	4
Bare	No habitat but no barrier	2
Natural	Green spaces including parks	1

2.3 Results

2.3.1 Spatial characteristics of the urban ecological corridor

The selected corridor was located in the northern part of Johannesburg and included Rietfontein Nature Reserve, the only formally protected area in this part of the city (Appendix B). The nature reserve is located relatively close (approximately 4km) to the Greater Kyalami Conservancy (GEKCO; Appendix B) which promotes open space and connectivity within its borders. The habitat consists predominantly of grasslands and rocky

ridges with sections of riparian vegetation adjacent to the Jukskei River that dissects the area, running from east to west. The altitude ranges from approximately 1460m to 1521ma.s.l.

Overall the study area was characterised as residential but included small holding developments with scattered portions of built-up and commercial areas, sport sites including golfcourses and schools. Site 3 was largely characterised by small holdings but was also nearby to residential, industrial and commercial developments. Site 2 had the highest percentage of open space and the nearest land-use was for sports and recreation. Site 1 was Rietfontein nature reserve and was characterised by green space but is surrounded by high density residential development as well as a highway (Appendix B).

The regional climate in the study area consists of summer rainfall from October to March and dry winters from May to August (Mucina and Rutherford, 2006). The mean annual rainfall for the region is 680mm but can range from 620mm to 680mm. The mean annual temperature for the region is 16.0°C but can range from about 35°C in summer to below freezing 0°C in winter (Mucina and Rutherford, 2006).

For this study I selected three similarly sized habitat patches within the study area to be used as sample sites (Appendix C). Site 1 was Rietfontein Nature Reserve, the only protected area in the corridor. Sites 2 and 3 were the largest habitat patches that were accessible and evenly spaced from one another. Rudd *et al.,* (2002) suggested that the larger a habitat patch is, the greater its ecological value is likely to be because it can support greater species diversity and large, persistent populations as wildlife abundance is related to patch size.

The spatial characteristics of the corridor determined using mapping software are shown in Table 2.1. The total area of the corridor was 13, 4 km^2comprising of 5 core habitats

making up an area of 3.96 km^2 (29, 5% of the total area) (Table 2.2). The mean corridor width was 650 m wide and the mean corridor length was 2, 4 km long (Table 2.2). The corridor was located in a transitional zone between the high density residential and commercial development of the northern suburbs (e.g. Bryanston, Rivonia and Morningside) and the lower density small holdings on the outskirts of the city (e.g. Kyalami and Midrand). The study area was characterised by mixed land-use types including residential areas, commercial properties, industrial activities, agriculture and natural patches occurring within it. According to the GDARD Conservation plan (Compaan, 2011); the corridor included one of the five major ridge systems in Gauteng (Appendix C; Pfab, 2001). Rietfontein Nature Reserve is classified as a class 3 ridge (35-65% transformed) and there is a small portion of class 1 ridge (0-5% transformed) in the north-east part of the study area which was not included in the study. Site 2 is located at the edge of the class 2 ridge but is comprised predominantly of riparian/wetland habitat. The wetlands in the corridor form part of the Jukskei River catchment which drains from east to west and forms part of the larger Limpopo primary catchment region which is drained by the Jukskei and Klein Jukskei River towards the Crocodile River. The Jukskei River is the only major river in the study area and bisects it flowing from east to west. There are five small dams that occurred along wetland lines and several hillslope seeps that all flow into the Jukskei river. Site 2 was the only site which occurs along the wetland.

Table 2.2 Spatial characteristics of the corridor determined using spatial mapping software

Factors	GEKCO-Rietfontein
Total area (km^2)	13.4
Core habitats (> 50m)	5
Core habitat area (km^2)	3.96
Mean Corridor width (km)	0.65
Mean Corridor length (km)	2.4

Connectivity (1-5)	4
Accessibility	4

2.3.2 Corridor for movement based on least-cost path analysis

The least-cost path analysis tool in ESRi's ArcMap was used to show the most optimal path for movement of small mammals in the study area (Figure 2.4). The route generated by impedance followed the most optimal route for the focal species to move between study sites or habitat patches. The corridor corresponds to the relative impedance values of different land-use types that were generated (as per Kong *et al.,* 2010) to determine a theoretical route that an animal would use to move between the selected habitat patches in the corridor. The path that is generated can be used to show which parts of the corridor should be prioritised for management to increase continuity and connectivity. Roads and buildings were considered as serious barriers for wildlife movement by increasing habitat isolation but were not major factors in this corridor (figure 2.3). Residential property represented a large percentage of the surrounding land-use particularly in the southern section of the corridor (figure 2.3). Reducing impedance and increasing connectivity is the best way to improve an ecological corridor (Kong *et al.*, 2010) and the least-cost path analysis showed the areas of greatest impedance along this corridor (figure 2.4) which can be used to improve it in the future.

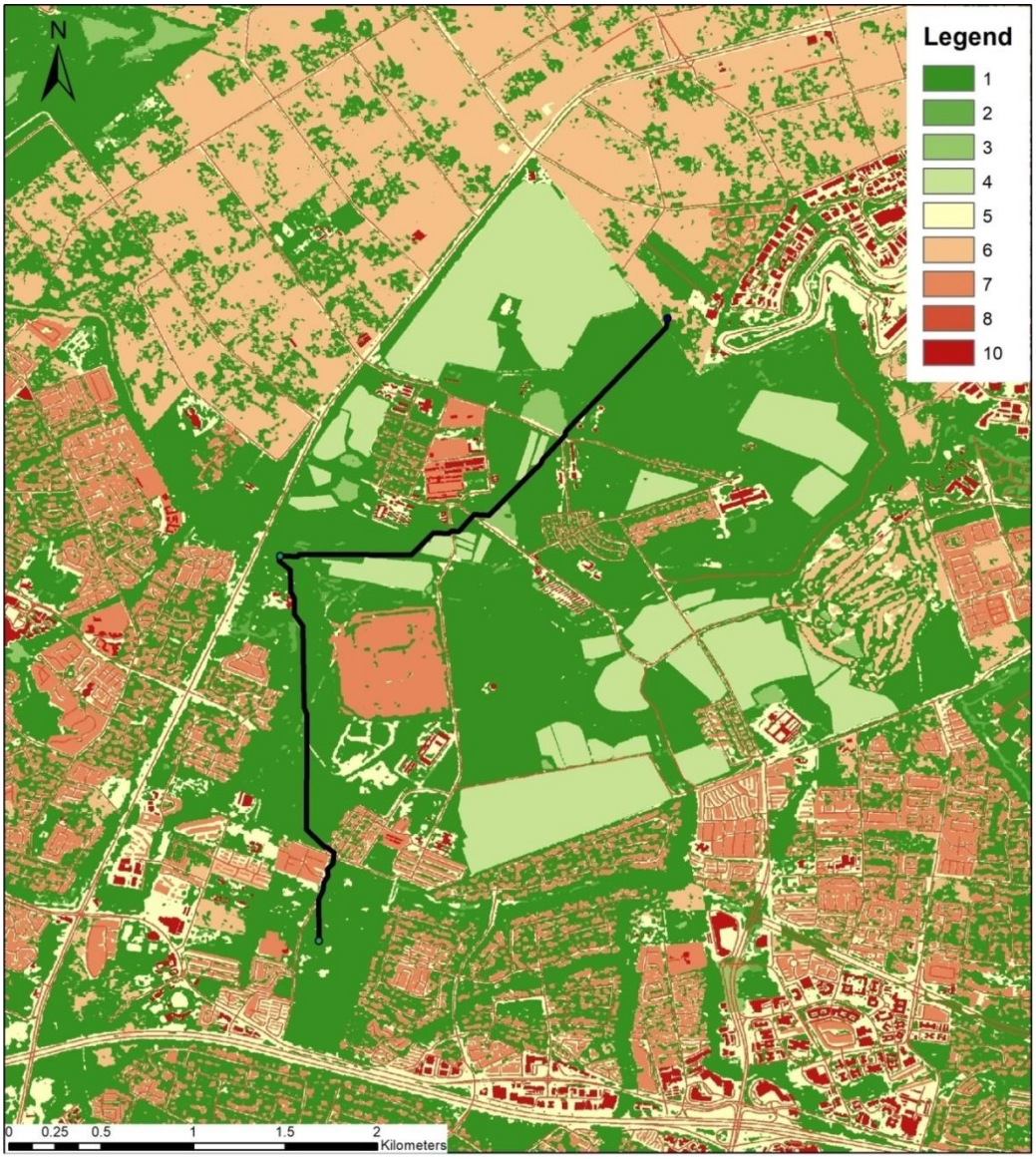


Figure 2, 1: ecological corridor generated by least-cost path analysis. Colour scale represents the theoretical relative impedance per land-use type. (Created in ESRI ArcMap 10.3.1).

2.4 Discussion

Using landscape features and spatial characteristics, I was able to identify an ecological corridor in Johannesburg and assess it using least-cost path analyses (Kong *et al.*, 2010). The use of impedance values to determine the optimal movement patterns for small mammals within the urban matrix allowed for more accurate assessment of the landscape

between links (Kong *et al.*, 2010). The spatial characteristics of the corridor (refer to Table 2) showed that it has been impacted by urbanisation which was evident from the number of core habitats that were found within the relatively small area, suggesting fragmentation. However, this corridor had a mean width of 650m suggested there may still have been sufficient habitat within the corridor to support biodiversity and allow for wildlife movement (Bond, 2003). It was evident from the land-use shapefile that Rietfontein Nature Reserve was surrounded by residential development and roads which may reduce the level of connectivity that is had with the rest of the corridor. Spatial analyses did not test for isolation but flora and fauna were used as an indicator for connectivity in Chapters 3 and 4 of this dissertation.

The urban matrix is generally considered to be hostile to many native species and thus preserving them within the habitat fragments is probably difficult unless the fragments are connected by linear strips of habitat (Borgström *et al.*, 2013). The data from this study do not provide evidence for the value of the corridor to biodiversity conservation but do suggest that structural landscape connectivity does exist. Previous studies on the effectiveness of corridors in urban environments have suggested that they may not be useful to some groups of organisms for dispersal but they still provide valuable habitat (Angold *et al.*, 2006) which was tested in Chapters 3 and 4 of this thesis.

A study done by Walker and Craighead (1997) used the least-cost path analysis to simulate movement corridors, using specific species habitat preferences to calculate cumulative-cost. It was assumed that wildlife would choose a route that offered the most favourable habitat where possible and as such wildlife movement would be facilitated by these routes. Incorporating ecological corridors into future green space planning in Johannesburg would be a major step forward in achieving higher levels of biodiversity conservation and overall sustainability. However, there are many challenges that need to be

addressed before this can become a reality. Broad landscape planning is useful for creating general ecological corridors or green spaces but species-specific analysis should be done to determine the connectivity of specific populations (Hoctor, 2003). Different species may experience different levels of connectivity (Kindlmann and Burel, 2008) in varying landscapes which must be taken into account in planning. Ultimately the ecological corridors form part of a much bigger network of open space extending out of the city and province, so development should be at multiple scales to take into account the connectivity of the region. Establishing connected green spaces should be both functional and structural connectivity. Samways and Pryke (2016) suggest using protected areas as a means of comparison for corridors by assessing species composition and interactions to determine if the ecosystem is 'functional'. Management of the corridors would also pose some challenges as natural disturbances, such as fires and grazing, would need to be mimicked to ensure correct ecosystem function (Samways and Pryke, 2016).

Ecological corridors should be built upon existing green spaces (Li *et al.,* 2015). This would reinforce current green spaces while allowing future development to improve the overall ecological corridor and spatial development planning efficiency (Li *et al.,* 2015). Preserving the habitat patches will act as "stepping stones" between the current nature reserves and open space (Kong *et al.,* 2010). Protected habitat patches may benefit surrounding corridors and improve resilience to further habitat fragmentation (McKinney, 2006). However, this study showed that a protected habitat patch such as site 3, Rietfontein Nature Reserve, may become isolated if the surrounding open space is not excluded from development planning, ultimately reducing overall ecological value, thus spatial planning must be improved.. Links to habitat patches outside the study area may play an important role in the functionality of an ecological corridor. Higher levels of connectivity between urban

and rural habitat patches are likely to improve wildlife dispersal and migration (Brás *et al*., 2013). However, making these links may be challenging due to increasing fragmentation resulting from urban development.

This analysis was based purely on theoretical impedance values for a small mammal spp. movement and the overall functionality of the corridor may differ if considering an alternative species. The development of urban ecological corridors will not be able to address all the biodiversity conservation problems that exist but may offer sustainable and cost-effective ways to compliment current conservation strategies (Linehan *et al*., 1995). A significant amount of research is required to fully understand the benefits of ecological corridors in urban environments but several studies have found evidence to support their effectiveness for conservation (Samways and Pryke, 2016).

Chapter 3.Assessing the habitat condition of an urban corridor – a case study

3.1 Introduction

Urban ecology and development processes are extremely complex, forcing researchers to apply diverse theoretical approaches in order to understand them (Ahern, 2013). Urbanisation results in damage to natural environments by changing landscapes, causing drastic declines in local biodiversity (Beardsley *et al*., 2009). Human competition for urban resources, primarily land, leads to landscapes becoming divided into patches or fragments of 'natural areas' which creates ecological niches that experience similar pressures and exhibit similar characteristics within an agricultural or urban matrix (Gehring and Swihart, 2002). Previous studies have suggested that in landscapes where habitats have been separated by urbanisation, connectivity may be reduced resulting in barriers for wildlife movement(Savidge, 1973; Oxley *et al*., 1974; Barnett *et al*., 1978; Cambell, 1981; Mader, 1984). Early research efforts by various ecologists (Odum, 1963; Boyden *et al*., 1981) helped establish conceptual models of cities as urban ecosystems. Although considerable progress has been made in the field of urban ecology, one challenge is that most of this research has been done in isolation within several disciplines, creating oversimplified models in favour of either ecological or social dimensions (Ahern, 2013). Most ecological researchers remain sceptical of integrating social and ecological processes to avoid inaccurate representations of each field. Yet urban landscape patterns are developed through micro-scale interactions between ecological and human processes and ecosystem functions are impacted by human and ecological patterns (Alberti, 2008). Urban processes occur at multiple spatial and temporal scales and vary from city to city. These complex urban ecosystems are constantly

changing and evolving according to the various drivers influencing them such as land-use change and biophysical agents (geomorphology, climate and natural disturbances). These changes will have different impacts on the condition of urban ecosystems, both directly (within the city) and indirectly through emissions, land conversion and use of resources (Alberti, 2008).

Bryant (2006) listed four major effects of urbanisation as increasing temperatures, water runoff, reduced native species diversity and increased levels of exotic species, and greater production of carbon dioxide. There is often a simplification of the vegetation structure as shrubs and deadwood are removed and replaced by grasses and herbs (Marzluff and Ewing, 2001). These changes tend to have a negative impact on the diversity of birds and other animals whose diversity is generally associated with complex plant communities (Savard *et al*., 2000). Vertebrates for example need large areas to sustain a healthy population as opposed to plants which can have a healthy population in a relatively small area (McKinney, 2008). Extreme urbanisation often leads to a reduction in indigenous species richness in plants, mammals, reptiles, amphibians and invertebrates which can most likely be explained by a species-area effect, where suitable habitat is lost and degraded by pollution, traffic and other disturbances. However, areas that are affected by moderate urbanisation (i.e. suburban areas) vary considerably among groups and show less consistent tendency to reduce species richness (McKinney, 2008).

Biodiversity is seen as the basis of functioning ecosystems which provide a variety of ecosystem services (Walter Reid *et al*., 2005). Ecosystem services can be defined as "conditions and processes through which natural ecosystems and the species that make them up, sustain and fulfil human life" (Daily, 1997, page 3). A loss of biodiversity may negatively impact these ecosystem services which humans are dependent on for clean air, water, food

and shelter as well as many other social values (Walter Reid *et al*., 2005). Some studies have also provided scientific evidence to support the role of biodiversity in building resilience to climatic changes (Elmqvist *et al*., 2003; Jansson and Polasky, 2010). Changes to the ecological systems in place may compromise the quality of these systems by introducing exotic plants, weeds and grasses which may displace native species, increase soil erosion and reduce water quality (Harris, 1984).

The aim of this case study was to assess the current state of a selected urban corridor in Johannesburg, South Africa to determine its potential as an urban corridor. Specifically, I set out (a) to assess the plant species composition and vegetation structure, and (b) to determine the degree of disturbance, pollution and frequency of physical barriers along it such as fences. I predicted that the corridor would be impacted heavily by urbanisation, resulting in multiple physical barriers as well as degraded habitat throughout the study area.

3.2 Materials and Methods

3.2.1 Study area

The study area is located on the outskirts of the major suburbs north of Johannesburg City (Appendix A) which lies in the southern half of Gauteng Province, South Africa (26° 11' S and 28° 2' E). It consists mainly of large agricultural holdings with portions of medium to high density residential land-use, and open areas are characterised by grassland and rocky ridge mixed vegetation (Compaan, 2011).

ESRi's ArcMap 10.3.1(ArcGIS Desktop: Release 10. Redlands, CA: Environmental Systems Research Institute. 2015) was used to identify an undeveloped corridor between Rietfontein Nature Reserve and the Greater Kyalami Conservancy which was selected prior

as a potential corridor for further research (Chapter 2), based on the fact that it included multiple habitat patches or nodes, areas that were large enough to be considered as ecologically sustainable, portions that had been impacted by urbanisation at varying levels, accessibility and safety, and the potential for the corridor to connect open areas and habitat patches (Figure 2,3, Chapter 2). GDARD's (Gauteng Department of Agriculture and Rural Development) C-Plan (Conservation Plan 3.3) was used to identify Ecological Support Areas (ESA) and Critical Biodiversity Areas (CBA) within the area.

Three sizable study sites were selected along the corridor (Figure 2.2, Chapter 2). The first site was part of Rietfontein Nature Reserve (26° 1'49.67"S; 28° 2'29.57"E) which is a formally protected area of approximately 22ha that is surrounded by suburban residential housing. Site 1 was characterised primarily by rocky ridge habitat. The altitude ranged from a low of approximately 1460m in the north to 1521m a.s.l.at the beacon which is located near the centre of the reserve. The second study site was part of a commercial operation south of Kyalami and was located in the centre of the study area (26° 0'40.18"S; 28° 2'20.11"E) with an area of 25,5ha and had the lowest elevation (1400ma.s.l.) along the Jukskei River (Figure 2,3, Chapter 2). The third site was located within the Greater Kyalami Conservancy (GEKCO) at the far north end of the corridor (26°59'57.88"S; 28° 3'29.32"E) on private property which is 9.67ha and has similar characteristics to the Rietfontein site (elevation: 1527ma.s.l.; Figure2,3,Chapter 2).



Figure 2, 2: Satellite image of the Rietfontein-GEKCO corridor showing the quadrats within each study site. (Google Earth 2017).

3.2.2 Vegetation surveys

The vegetation surveys were conducted between December 2015 and March 2016. The assessment identified dominant plant species, topography, rocky outcrops, ground cover, slope, aspect, plant species composition and level of disturbance for each sample plot (Figure 5) with each study site. I assessed plant species richness, and vegetation structure at all three study sites along the corridor. Quadrats are traditionally used to establish a standard area for sampling vegetation at each site (as per Kent and Coker, 1992), so I set up five 3x3m quadrats spaced evenly across each of the three sites, taking care to minimise slope and aspect variations and ensuring that quadrats were at least 3m apart (as per Anderson *et al*., 2014). In order to select the location of each 3 x 3 m quadrat, I employed a semi-random "over-shoulder-toss" method using a marker and then measured out the quadrat using the marker as the centre (as per Bullock, 1997; Hill *et al*., 2005). Sampling was done during the summer months (December 2015; January, February and March 2016) when the vegetation was flowering and seeding to assist with species identification.

Structural attributes were assessed for the five random quadrats at each of the three sites along the corridor. Ground cover and percentage of bare ground present were determined for each quadrat using the Braun-Blanquet cover-abundance scale (see Kent and Coker, 1994, Ives *et al*., 2011; Table 3,1). Plant structure was recorded as an estimated percentage of each structure type occurring within each quadrat (structure types included trees; long grass < 60 cm; medium grass 20 – 60 cm; short grass > 20 cm; medium herbaceous vegetation > 20 cm and bare ground) for the classification and description of plant communities. An overall percentage vegetation ground cover and structure was then determined for each of the sites by averaging across five quadrats.

Table 3.1.Braun-Blanquet cover-abundance scale used in present study.

Index Value	Description	Percentage Cover
5	Plants cover 3/4 reference area	>75
4	Plants cover 1/2 to 3/4 reference area	50-75
3	Plants cover 1/4 to 1/2 reference area	26-50
2	Plants 1/20 to 1/4 reference area	5 to 25
1	Plants cover less than 1/20 reference area	<5
+	None	0

To determine species richness, a species list of all the trees, shrubs, herbaceous plants and grasses identified was compiled for each of the study sites. Species were identified using field guides where possible and any unidentified species were photographed for later identification at the University of the Witwatersrand's Herbarium. Additionally, I recorded any anthropogenic disturbances at each quadrat for all three study sites. The types of disturbance at the sites included: pollution (plastic, glass, tins and rubble), low-to-medium impact human activities (e.g. walking, cycling, and horse riding), high impact human activities (e.g. snares), and physical barriers (e.g. walls, fences and roads) within each 3 x 3 m quadrat. Each category of anthropogenic disturbance was given a score of 1 to 5 (1 = minimum impact: 5 = maximum impact) based on the visible impact it had made to each quadrat and averaged across 5 quadrats. Each site was subsequently given a theoretical disturbance rating out of five based on the estimated impact that the disturbance would have on the habitat and fauna and flora at each site. A rating of 0 indicated no visible disturbance; 1very low evidence of disturbance; 2 disturbance present but infrequent or dispersed; 3 medium disturbance level; 4 a high level of disturbance; and 5 indicates the highest level of disturbance.

3.2.3 Data analyses

Univariate and multivariate statistical techniques were used to identify the factors influencing biodiversity within the corridor. Plant species richness and structural composition were the two aspects of biodiversity measured in this study. Plant species richness was determined for all observed species and for native species only. This allowed for assessment of any effects that exotic species might have on native plant assemblages. The vegetation data were arcsine transformed and a General Linear Model (GLM) with a Fisher's Exact post hoc test was used to identify if there were any significant differences in the vegetation cover and structure between and within sites. A multiple-site similarity measure was done to determine any significant differences in plant species richness between the three sites (as per Diserud and Ødegaard, 2007).The number of exotic plant species was expressed as a percentage of the total plant species richness to act as an additional explanatory variable for any observed differences between sites.. For all statistical analyses, the significance level was set at $p<0.05$.

All statistical analyses were done using Statistica Version 13 (Statsoft, 2016 Tulsa, OK.).

3.3 Results

3.3.1 Vegetation cover, structure and composition

Vegetation attributes at sites 1 and 3 were the compositionally similar, being represented by rocky ridge habitat, while the central site, Site 2, consisted primarily of riparian habitat. Structurally, all three sites offered similar percentages of ground cover (see Figure 3, 2).The vast majority of the corridor was characterised by grasslands with scattered

patches of woodland. A GLM showed that there was no significant difference in vegetation cover or structure between the sites (F_3=0.087 p=0.96, Figure 3, 2).

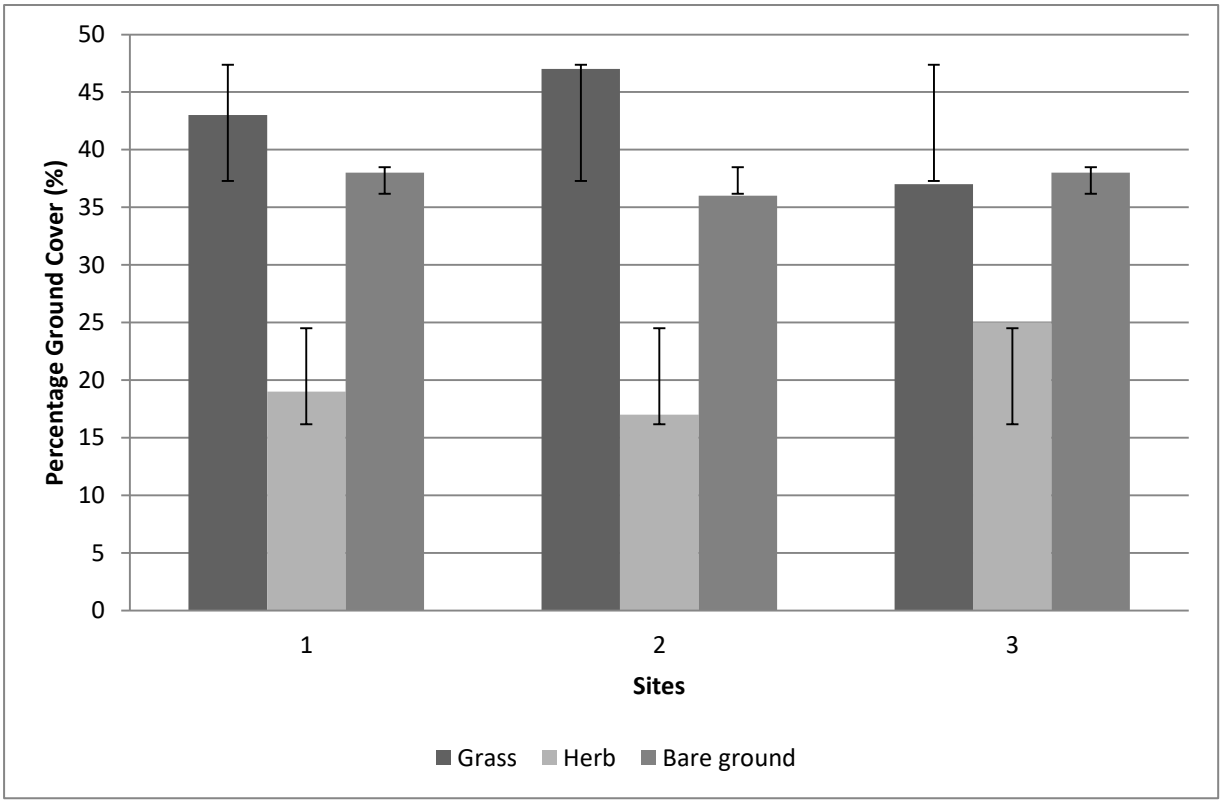


Figure 3, 2: .Mean (+/-SD) percentage vegetation cover and structure for each site in the present study.

A total of 123 plant species were recorded along the corridor. Of these, 98 species were indigenous and 25 species were exotic (Table 3, 2). The plant species for each of the sites were recorded for comparison (Appendix A). The species composition was found to be similar between the sites according to the multiple-site similarity index (C_s= 0.455) (Ricotta and Paving, 2015). Rietfontein Nature Reserve (Site 1) was found to have the highest number of species with 84 plant species, while sites 2 and 3 had relatively similar totals for species richness with 40 and 55 respectively (Table3,2). At site 1, 19.3% of the plant species recorded were exotic species, while site 2's plant species consisted of 22% exotic species.

Site 3 had the lowest percentage of exotic plants at only 7.4% (Figure 16).The most widespread exotic plant species recorded in the corridor were Mexican clover (*Richardia brasiliensis*), Jerusalem cherry *(Solanum pseudocapsicum),* tick-berry *(Lantana camara),* syringaberry *(Melia azedarach),* southern cone marigold *(Tagetes minuta),*pompom weed *(Campuloclinium macrocephalum),* and Spanish needles (*Biddens bipinnata)*. Site 1 had the highest density of Kikuyu grass (*Pennisetum clandestinum),* queen of the night *(Cereus jamacaru)* and Brazilian nightshade *(Solanum seaforthianum).* Site 2 had sizable patches of blue morning glory (*indica),* common morning glory (*purpurea),* and high density stands of silver poplar (*Populus alba)* (refer to Appendix E).

Table 3, 2. Overall plant species richness and cover percentage along the corridor

Vegetation type	**Vegetation cover percentage**			**Species Richness**
	Site 1	**Site 2**	**Site 3**	
Trees and Shrubs	42.8	22.5	36.4	41
Forbs	34.5	40	43.6	52
Grasses	22.6	37.5	20	30
Exotics	19	22.5	7.2	25
Total				**123**

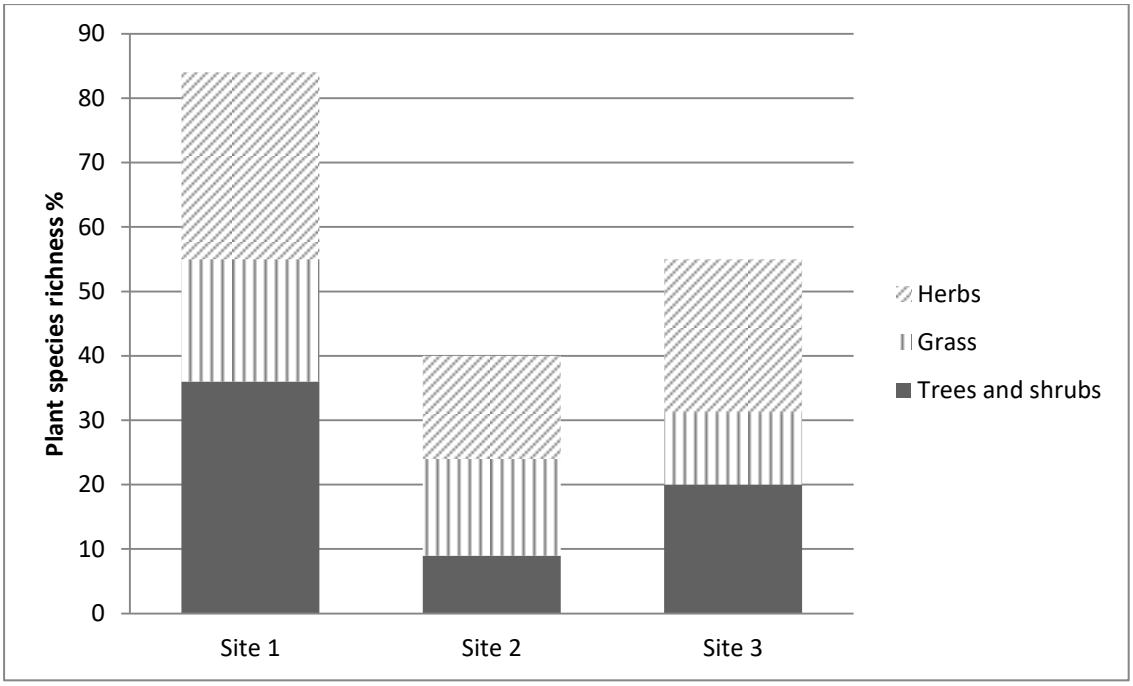


Figure 3, 3.Plant species richness at each site in the present study.

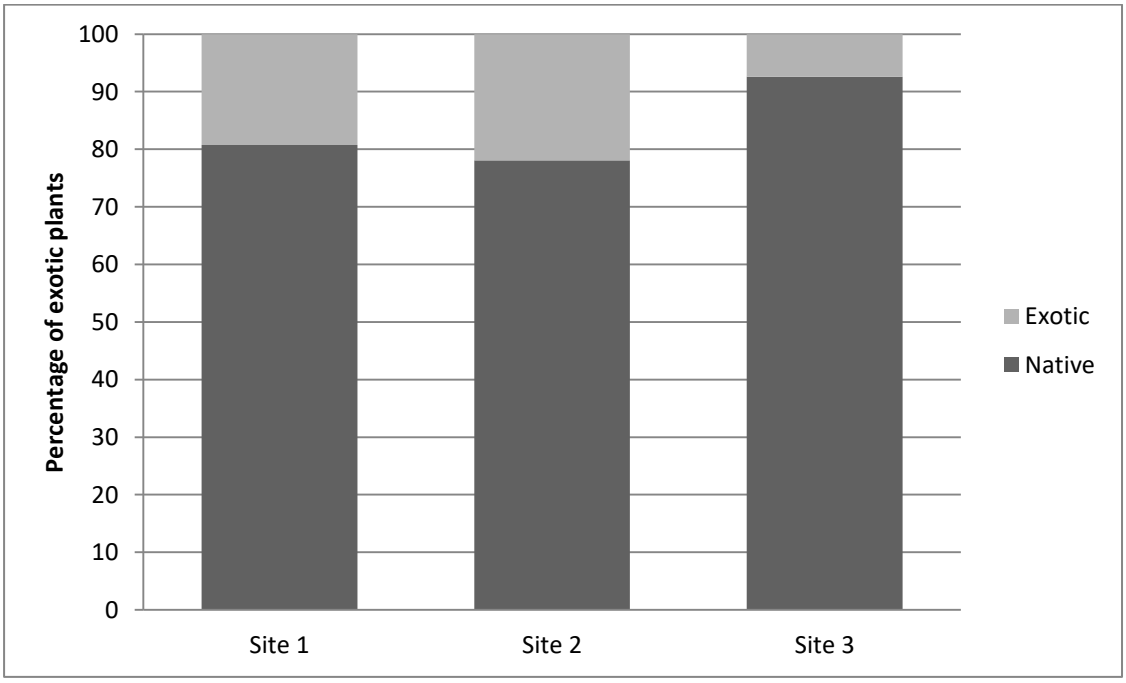


Figure 3.4.Percentage of native and exotic plant species at each site in the present study.

3.3.2 Anthropogenic disturbance

Anthropogenic disturbance ratings between sites were relatively similar (Figure3, 4); with site 3 having the lowest disturbance rating of 0.3 and sites two and three having a rating of 1.1 and 1.3 respectively. Plant species richness was also not significantly influenced by the disturbance variables. A generalised linear model (GLM) showed that there was no significant relationship between plant species richness and disturbance between sites (Wilks ƛ =0.19 $F_{(12)}$=1.363 p=0.264).

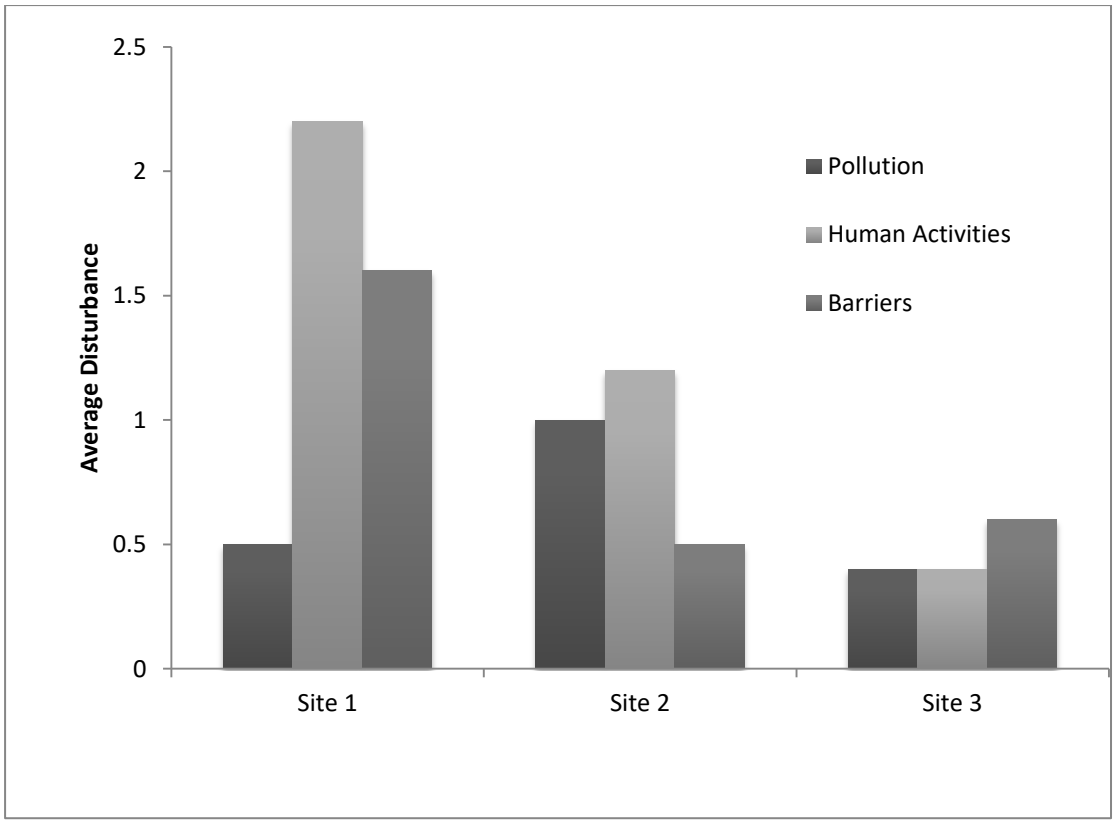


Figure 3.5 Anthropogenic disturbance averages per study site in present study.

3.4 Discussion

The aim of this study was to assess the current state of a selected urban corridor in Johannesburg, South Africa and to determine its potential as an urban ecological corridor. We

assessed the plant species composition, vegetation structure and recorded the degree of anthropogenic disturbance.

In this study, 123 plant species were recorded across all three study sites within the corridor. This diverse range of vegetation contributed to a high species richness despite the relatively small study area (Figure 3.3). As a Class 3 ridge, the Rietfontein-GEKCO corridor no longer represents its natural state (Compaan, 2011) but still maintains patches of remnant vegetation, separated by novel vegetation units or anthropogenic structures. The presence of exotic vegetation suggests that anthropogenic disturbances may have changed the composition of plant communities, especially in low lying areas which are typically represented by grassland habitat.

The comparative assessment of physical structure and plant species composition between the corridor and connected habitat patches indicated that the vegetation within the corridor had low structural or compositional variation. Impacts associated with urbanisation may alter the vegetation composition of an area but it may also increase the structural diversity of the vegetation (Chace and Walsh, 2006). However in this study the percentage of vegetation cover and structure was not significantly different between different sites along the corridor, indicating continuity of vegetation type along the corridor. While the sites did not show significant variation as a whole, differences in each sample quadrat may indicate influences by factors such as edge effects and exotic species. This continuity is expected due to the sites forming part of the same ridge system (see Appendix B). Site 1 was the only site with differences in vegetation that could present a barrier for the movement of wildlife. The presence of anthropogenic habitat types at the site such as lawns may not provide sufficient cover for species to move through safely and thus acting as a barrier. Despite this however, Site 1 still contained the second highest percentage of grass cover in the study (Figure 3, 3).

The plant species richness between sites was not significantly different. Plant species richness may be a useful indication of the habitat quality (Angold *et al*., 2006). However there are many factors that may influence the plant species richness other than disturbance and thus these results may not be directly linked to the impacts of urbanisation (Lite *et al*., 2005). Corridor width is generally considered to be an important factor in corridor research due to the higher levels of species diversity frequently found in wider corridors, and lower species diversity in isolated habitat patches surrounded by anthropogenic land uses (Pryke and Samways, 2001). However in this study the corridor width at each site was not significantly different (See Chapter 2, Table 2.2). Previous studies have shown that the number of plant species in an area increases in response to urbanisation, as exotic species are added to the native species, giving a high overall species richness, after which it decreases at the levels of disturbance increase (McKinney, 2008). Rietfontein Nature Reserve (Site 1) had very distinct differences between the anthropogenic vegetation and 'natural' vegetation communities. This is because the nature reserve has an area near the offices, buildings and parking areas which have been highly transformed to cater for recreational purposes. These transformed habitats include species which do not occur in the area naturally but have been planted in the gardens and lawns. This habitat was represented by species such as paperbark thorn (*Vachelia sieberiana),* white stinkwood *(Celtis africana),* monkey thorn *(Vachelia galpinii),* cheesewood (*Pittosporum viridiflorum),* common coral tree *(Erythrina lysistimon),* Cape honeysuckle *(Tecoma capensis),* couch grass *(Cynodon dactylon) and* Kikuyu grass. In areas that are becoming urbanised, native plant communities are frequently replaced by introduced plant communities, creating novel environments and influencing the ecological and environmental relationships in the ecosystem (Chace and Walsh, 2006; Kowarik, 2011, Joubert, 2016). According to the Rietfontein Nature Reserve Ecological Evaluation Draft

Report which was completed in 2014, 293 plant species have been identified at the site. The relatively high number of plant species recorded at the Rietfontein Nature Reserve site suggests that it has been impacted by urbanisation because a large number of exotic species were present at the site, similar to the other two sites. However, this site, being a rocky ridge habitat, also had a large number of native species. While the plant species richness may be a good indicator of the level of disturbance at a site, the apparent high species richness at Rietfontein Nature Reserve could be due to habitat heterogeneity, increasing the number of niches and plant communities. However I have no baseline data from the area before it was impacted by development, so I cannot suggest whether or not the plant species have declined. For this study we only did a sample of the plant species richness for comparison between sites rather than a complete flora assessment.

3.4.1 The impact of anthropogenic disturbance on vegetation

Vegetation percentage cover, structure and species composition are known to be impacted by urbanisation as thus have regularly been used as indicators of disturbance (McKinney, 2008; Kowarik, 2011). In this study it was determined that the average disturbance rating between sites was not significantly different. This suggests that the entire corridor has been subjected to similar levels of disturbance generally associated with urbanisation. Figure 3,4 shows the different categories of disturbance that were assessed with respective values and one that stands out is the barriers for site 1 was much higher than the other two sites. During the study period a fence was erected around the site which appeared to be significant barrier causing damage to the vegetation on both sides of the entire

perimeter and likely reducing connectivity. Future studies will have to be done to determine what effects this fence might have on this portion of the corridor.

Chapter 4.Small mammals movement and habitat use in an urban corridor

4.1 Introduction

Identifying the factors that influence the distribution patterns of specific species is one of the most important aspects of ecological research (Gehring and Swihart, 2003). Species distribution patterns can be complex to understand but previous research has identified climate, topography, competition, land-use change, altitude and predation (Brown 1984; Caughley *et al.*, 1987; Msuha *et al*., 2012) as factors which may determine species distribution. Mammal species richness may serve as an effective indicator for biodiversity conservation strategies and environmental decision making and is thought to be driven by habitat condition under the influence of local environmental variation, which may be affected by varying levels of disturbance (Ramesh *et al.*, 2016). The geographic distribution of individual species is frequently associated with specific climatic conditions. Species distributions may also be limited by temperature and water availability (Currie, 1991; O'Hara and Tittensor, 2010). Further, vegetation productivity and landscape structure may also drive species richness at multiple levels (Currie, 1991; Yates *et al*., 2010).

Habitat degradation and fragmentation have negatively impacted on biodiversity in recent centuries by reducing suitable habitat availability and increasing levels of isolation of different populations (Fahrig, 2003). Land-use changes alter natural habitats which usually results in a decline in local species richness along an urban development gradient (Fahrig, 2003; McKinney, 2008; Ramesh *et al*., 2016). These land-use changes fragment the natural landscape into smaller and more isolated patches over time which reduces habitat connectivity, creates barriers for wildlife movement and impacts ecological processes (Crookes *et al*., 2013).The distance between these habitat patches may determine the chances

of survival for specific species and thus increased changes to these habitats may accelerate species extinction (Morelli, 2013). Therefore, it is important to assess the effects of landscape structure on mammal species richness which may help to determine their distribution patterns and how habitat fragmentation and degradation affect them in an urban environment (Ramesh *et al*., 2016).

It has been suggested that at smaller spatial scales, the use of species richness models may benefit local conservation actions (Ramesh *et al*., 2016). However, there are many possible determinants, especially in urban landscapes, for species richness, including food availability, predator pressure, poaching and hunting for food or medicine, and the presence of domestic animals such as dogs and cats (Hawkins and Porter, 2003; Greve *et al*., 2012; Kiffner *et al*., 2015). There is often a simplification of the structure of the vegetation as shrubs and deadwood are removed and replaced with grasses and herbs (Marzluff and Ewing, 2001). These changes tend to have a negative impact on the species richness of birds and other animals whose diversity is generally associated with complex plant communities (Savard *et al*., 2000).

4.1 Wildlife conservation in urban areas

Nature reserves are considered to be one of the most important ways to halt the decline of biodiversity worldwide (Borgström *et al*., 2012). Nature conservation in urban areas is important but challenging due the unique biotic and abiotic characteristics that occur in urban landscapes (Ramalho and Hobbs, 2012). Urban environments are typically dominated by generalist species and less habitat specialists (Elmqvist *et al*., 2008) which may make the use of red data species lists as support for nature conservation strategies flawed

(Borgström *et al*., 2013). The problem is that the areas of remaining habitat are frequently selected based on their function to protect certain threatened species (Borgström, 2009). However, the urban matrix is considered to be hazardous to many species and the ones that exist in these protected areas may be difficult to sustain long-term, unless these areas are connected by an extensive structure of refuges (Andersson and Boden, 2008). Hence there is a need to assess the value of the space between habitat fragments as ecological corridors in urban landscapes, particularly for small mammals which play an important role in ecosystem functioning (Redon *et al*., 2015). Mammals may be placed at considerable risk when trying to move through a hostile urban matrix between habitat patches or fragments which could reduce their potential or willingness to successfully traverse the landscape (Dornelas, 2010).

Landscape connectivity can be assessed from two perspectives, 'structural connectivity' and 'functional connectivity.' Structural connectivity can be described as "the physical relationship between elements" while functional connectivity refers to "the degree to which landscapes actually facilitate or impede the movement of organisms and processes" (Meiklejohn *et al*., 2009, pg. 1). Assessing connectivity is challenging because researchers must take into account that the corridors will provide different levels of connectivity for each species. This would require monitoring the movement of each species which could prove costly and potentially challenging to accomplish. Selecting the correct scale for the correct landscape analysis is crucial because of the variation in patterns and processes that function at different scales (Blazquez-Cabrera *et al*., 2014). In the past, ecological corridors have been described as linear or narrow stretches of habitat but more recent definitions acknowledge the fact that corridors facilitate movement which occurs at different scales, in different directions and patterns depending on the species, and thus corridors must be defined from the perspective of a specific species (Meiklejohn, *et al*., 2009). However, different species are

capable of changing their movement patterns at different scales according to their requirements, e.g. migratory movements versus daily foraging movements (Blums *et al*., 2003).

According to Lidicker and Stenseth (1992), the dispersal process involves three stages: emigration, travel and immigration, all of which may be negatively impacted by encountering hostile land-uses, which is frequently associated with fragmented landscapes. Emigration is generally used to describe the movement of individuals away from their home territory. If individuals are motivated to disperse but are unable to because of inadequate resources within the surrounding matrix habitat, dispersal may become frustrated (Stenseth and Lidicker, 1992), resulting in increased population densities and pressure on local resources (Cockburn, 1992). During the process of emigration, the individual is placed at greater risk in areas of unsuitable habitat due to predation, food shortage and lack of suitable shelter, thereby decreasing the individual's chances of survival. If the individual successfully reaches the recipient population, it must still enter the breeding population and produce viable offspring for this to be considered a successful dispersal (Barton, 1992). Ecological corridors could enhance connectivity and thereby increase the potential for mammal movement and decrease encounters with hostile habitat (Tigas *et al*., 2002). Movement of an individual away from its natal home range (Stenseth and Lidicker, 1992) can improve the survivability of the animal by changing its habitat (Cockburn, 1992). This movement may also improve both source and recipient populations by changing sex ratios, age structures, social dynamics, and genetic variation (Stenseth and Lidicker, 1992).

The functionality of a corridor can be approached from a direct or generational movement perspective (Stenseth and Lidicker, 1992). Generational movement is required to determine connectivity but the potential usefulness of the corridor will be dependent on the

ability of an individual to complete its entire life cycle within the corridor (Harrison, 1992). This means that a corridor must be able to provide sufficient resources and suitable conditions to support the organism through all stages of its lifecycle (Harrison, 1992) which may be considerably different to the resources required to support temporary movement along the corridor (Beyer and Leo, 1992). Bennet (1999) suggested that obtaining information on the increased resources necessary to sustain generational movement is important in landscapes that have been degraded and fragmented, possibly limiting an organism's ability to disperse relative to the level of patchiness. An ecological corridor that provides sufficient connectivity for movement but lacks adequate resource availability may in fact reduce the fitness of species within the connected patches (Soulé and Gilpin, 1991).

The aim of this study was to determine the value of an urban corridor for small mammal conservation by allowing for movement between habitat fragments and providing suitable habitat. The primary objectives of the study were (1) to assess the mammal species richness and diversity at regular intervals along the corridor and (2) to identify common mammal species occurring within the corridor which could be used as indicators to assess the usefulness of the corridor.

4.2 Materials and Methods

4.2.1 Study area

The study area was located on the outskirts of the major suburbs in the north of Johannesburg City which lies in the southern half of the Gauteng Province, South Africa (26° 11' S and 28° 2' E). A natural corridor between Rietfontein Nature Reserve and the Greater

Kyalami Conservancy was selected within the greater Johannesburg area (Appendix A). Three study sites were selected along the GEKCO-Rietfontein corridor. The first site was part of Rietfontein Nature Reserve (26.0300° S; 28.0397° E) which is a formally protected area of approximately 22 ha that is surrounded by suburban residential housing. The site is primarily a rocky ridge habitat with areas towards the bottom of the ridge that have been transformed by anthropogenic activities. The second study site was part of a commercial operation south of Kyalami and was located in the centre of the study area at the lowest altitude of the corridor along the Jukskei River. The third site was located within the Greater Kyalami Conservancy (GEKCO) at the far north of the corridor on private property which has similar characteristics to the Rietfontein site with less anthropogenic activity (pers. obs., Chapter 3).

4.2.2 Small mammal species richness and activity

At each study site two unbaited camera traps (Bushnell Trophy Cam HD) were placed in an area most likely to be used by animals, such as wildlife trails, ridges and riparian crossings. The camera traps were set up at least 50m apart at each site approximately 30 cm off the ground to allow for capture of rodents and other small mammals. Camera traps were set up at the three study sites for 12 weeks (84 trap nights) (as per Goad *et al*., 2014). Camera traps were checked on a weekly basis for maintenance (e.g. replacing batteries and clearing memory cards).

The site identity, date and time were recorded for every mammal captured on a camera trap. During each site visit, any mammal sightings or evidence of their presence (e.g. spoor and scat) were recorded. All three sites had varying levels of human activity (especially on weekends) and as a result there was considerable risk that the camera traps would be

tampered with or stolen. Therefore the camera traps were placed in well-hidden locations (where possible) to reduce the likelihood of them being stolen, but this meant that the cameras were not always necessarily set up in the most ideal places to capture mammal activity. The camera traps were set up in different habitats within each site to maximise the potential of capturing a range of different mammals including the cryptic or less active species. Camera traps were moved to a new location within each site once per week to ensure as many possible habitats were assessed. Existing species lists provided by Johannesburg City Parks and The Greater Kyalami Conservancy were used to assess if all likely species had been recorded during the study. An index of relative activity (RA) was calculated from the number of photographs of a species on a single camera trap divided by the total number of nights that the camera was active (as per Goad *et al.,* 2014).

Small mammal trapping was done using standard steel Sherman live traps (26 x 8 x 8m), which were set up 10m apart (n = 16 per transect) in a 8x2 parallel line arrangement at a suitable location with sufficient vegetation cover for small mammals (as per Steele *et al.,* 1984). The traps were set up for 6 nights per trapping session at each site in summer (December 2015 and January 2016), ensuring adequate sampling for small mammal species richness (see Jones *et al.,* 1997; Pearson and Ruggiero, 2003; Hurst *et al*., 2014).

During the trapping period, grass from the surrounding area and wooden boards were placed on top and on the sides of the Sherman traps to protect the animals from the heat of the day (if shade was available it was used), cold of the night and possible rain. The Sherman traps were baited with a mixture of sunflower seeds, peanut butter, salt and sunflower oil (Lancaster and Pillay, 2010). Checking traps and setting them up with fresh bait over a large area is time consuming; so traps were only able to be checked every morning and every afternoon. If a capture was recorded, the site and trap number were noted. Each individual

was carefully removed from the trap and identified. A small section of fur was always cut from the hind leg area to identify re-captured individuals. Morphometric measurements were not recorded during this study. Photographs were taken of individuals for identification later on. The re-capture status of the individual was recorded before the animal was released, and the trap was then re-set and rebaited. During the daily checking of the Sherman traps, additional sightings or signs of small mammals were recorded. The traps were thoroughly washed between trap sessions to remove any odour from previous rodent captures.

4.2.3 Data analyses

To determine differences in small mammal species richness between sites, presence and absence data for small mammal species recorded on the camera traps and Sherman live traps were used. It was not possible to differentiate between individuals of each mammal species, so absolute densities could not be calculated, but the index provided useful insight to the frequency that each species was being captured at each site (George and Crooks, 2006). If multiple individuals of a species were recorded in a single photograph, each individual was recorded as a separate sighting.

A General Linear Model (GLM) with a Fisher's Exact post hoc test was used to determine whether site or vegetation structure (refer to Chapter 3) were predictors for small mammal species richness in the corridor. For all statistical analyses, the significance level was set at $p<0.05$.

All statistical analyses were done using Statistica Version13 (Statsoft, 2016 Tulsa, OK).

4.3 Results

4.3.1 Small mammal species richness and relative activity

A total of fourteen small mammal species were recorded along the corridor during this study (Table 4.1). Twelve weeks of camera trapping at 6 trapping sites with 84 trap-nights yielded 455 independent photograph captures of 11 species of indigenous mammals, 1 introduced mammal species and 2 species of domestic mammals (Table 4, 1).

Table 4.1 Indigenous mammal species recorded in the study area during the study period

	Species	Site 1	Site 2	Site 3
	Atelerix frontalis (South African hedgehog)	x	x	x
	Galerella sanguinea (slender mongoose)	x	x	x
	Genetta tigrina (large-spotted genet)	x	x	x
	Lepus saxatilus (scrub hare)	x	x	x
	Mastomys natalensis (multimammate mouse)	x	x	x
	Sylvicapra grimmia (common duiker)	x	x	x
	Canis mesomelas (black-backed jackal)		**x**	**x**
	Hystrix africaeaustralis (Cape porcupine)		**x**	**x**
	Otomys angoniensis (Angoni vlei rat)		**x**	**x**
	Cynictis penicillata (yellow mongoose)			**x**
	Atilax paludinosus (water mongoose)		**x**	
	Procavia capensis (rock hyrax)		**x**	
	Damaliscus pygargus phillipsi (blesbok)	**x**		
Total	**13**	**7**	**11**	**10**

Live trapping resulted in the capture of two species with a total of 23 individuals over 45 nights. Multimammate mouse (*Mastomys natalensis)* was captured 35 times with 17 individuals recorded and Angoni vlei rat *(Otomys angoniensis)* was captured 6 times with 6 individuals. Trap availability was not a limiting factor in the number of captures with the majority of the trap remaining unsprung on most nights (pers.obs.).

Site 1 had the lowest small mammal species richness with seven native species recorded. Sites 2 and 3 had eleven and ten native species respectively with nine species occurring at both sites. Six native species were common to all three sites, while three species were recorded at two of the sites and four species occurred at only a single site (Table 4, 2). Introduced fallow deer (*Dama dama)* were recorded at sites 2 and 3. Domestic dogs (*Canis lupus familiaris)* were recorded at Site 2 and 3 while cats (*Felis catus*) were only recorded at Site 3. No additional species were identified by scat or spoor during this study.

Common duiker (*Sylvicapra grimmia*) was the most frequently recorded species and had the highest mean RA (0.85) across all sites (Table 4.2). However, most of the images were captured at site 1 (0.833), whereas the other two sites only showed a relative activity measure of 0.008.Rock hyrax *(Procavia capensis)* was only recorded at one site but was captured in 50 images, giving it a high mean RA (0.417) (Table 4.2). South African hedgehog *(Atelerix frontalis*) was captured on 62 images and occurred at all three sites, giving it the third highest RA for this study (0.517) (Table 4.2). Vegetation structure and disturbance were not found to be predictors of small mammal distribution (F_3=0.068 p=0.87).

Table 4, 2.Species captured by camera traps, the number of sites at which species were observed, total number of images of each species, and the weighted mean relative activity rates for entire trapping session

Species	Number of sites	Number of images (total)	Weighted mean relative activity			
			Total	Site 1	Site 2	Site 3
Sylvicapra grimmia	3	**102**	***0.85***	**0.833**	0.008	0.008
Genetta tigrina	3	17	0.142	0.05	0.058	0.033
Lepus saxatilus	3	20	0.167	0	0	0.125
Galerella sanguinea	3	40	0.333	0.025	0.025	0.283
Atelerix frontalis	3	**62**	**0.517**	0.425	0.025	0.067

Canis mesomelas	2	**74**	***0.617***	0	0.2	0.417
Hystrix africaeaustralis	2	12	0.1	0	0.058	0.042
*Dama dama**	2	**61**	**0.508**	0	**0.383**	0.125
*Canis lupus familiaris**	2	7	0.058	0	0.008	0.05
Damaliscus pygargus phillipsi	1	1	0.017	0.008	0	0
*Felis catus**	1	2	0.017	0	0	0.017
Procavia capensis	1	**50**	**0.417**	0	**0.417**	0
Atilax paludinosus	1	5	0.042	0	0.042	0
Cynictis penicillata	1	2	0.017	0	0	0.017

* denotes an exotic species

4.4 Discussion

The results of this study showed that species richness was similar along the majority of the corridor with many of the same species occurring at all three sites, but unexpectedly species richness was lower in the Nature Reserve (Site 1) than at other points in the corridor. The estimated richness was highest outside of the protected area, which may suggest better habitat continuity or higher quality habitat (McKinney, 2008). Mammal species richness is often associated with specific climates (O'Hara and Tittensor, 2010), however at smaller spatial scales where all study sites experience similar if not the same climatic conditions, species richness is likely determined by other factors (Ramesh *et al.*, 2016). These factors may include landscape features such as topography, competition, land-use change, altitude and predation (Brown 1984; Caughley *et al.*, 1987; Msuha *et al*., 2012). Domestic animals were recorded at all three sites which may have had a negative impact on the native wildlife populations (Veitch, 2002; Loss *et al.*, 2013).

4.4.1 Mammals as indicators for corridor usefulness

Many of the mammals recorded in my study were habitat generalists (Skinner and Chimimba, 2005) which indicated that they do not have specific habitat requirements and may be more successful in modified landscapes than habitat specialists. The results from these studies (McKinney, 2002; Angold *et al*., 2006) suggest that generalist species frequently benefit from urbanisation because they were able to use a diverse range of habitats and are usually mobile, allowing them to move relatively easily between habitat fragments (Gehring and Swihart, 2003).

Species richness is considered to be a good measure of biodiversity which can be a useful indication of the impacts on a native habitat by disturbance caused by urbanisation (McKinney, 2008). Different groups of organisms, both plants and animals, are often used as indicators of impacts along gradients of urbanisation (McKinney, 2002). Mammalian responses to habitat fragmentation have been well documented (See Naiman *et al.*, 1997; Angold *et al*., 2006; Goad *et al*., 2014).

4.4.2 Using mammals to assess corridor connectivity

In a corridor that has excellent connectivity one would expect that the relative activity (RA) levels of each species would be similar across each of the study sites (Goad *et al*., 2014). This appeared to be the case for most of the species that occurred at all three sites with the exception of common duiker (*Sylvicapra grimmia*) which had a much higher RA at Site 1 (Rietfontein Nature Reserve) than at the other two sites respectively. One possible

explanation for this result is that the common duiker within the reserve are unable to move out of it and thus they move through the same area repeatedly with a much higher frequency than the individuals outside of the reserve. However, it must be noted that the common duiker within the reserve appear to be habituated to human activity and frequent visits to the study site resulted in regular sightings of the species which may make them easier to capture on a camera trap and skew the results, while the animal was not seen at the other two sites. The high RA of rock hyrax (*Procavia capensis*) at only one site (0.417) was indicative of their dependence on a particular habitat type, a rocky outcrop adjacent to the Jukskei River, that only occurred at that this site and this species did not appear to move to other parts of the corridor.

Despite being the only red data species in this study, South African hedgehog was captured on 62 images and occurred along the whole corridor but did have a much higher RA (0.425) at Rietfontein than the other two sites (0.025 and 0.067), which was similar to the pattern observed with the common duiker. It is possible that this species is also isolated within the reserve which leads to a higher frequency of recordings at the site. However, small mammals like hedgehogs can potentially fit through small gaps in the walls and fences adjacent to the corridor, allowing for lateral movement and access to additional resources. Trapping could not be done along the entire length of the corridor. However, the vast majority of the corridor consisted of vegetation types that were similar to the trapping sites and were shown to be suitable for small mammals and thus I assumed that the length of the corridor would likely have this same suitability for this species.

The black-backed jackal (*Canis mesomelas*) was another species with a relatively high total RA (0.617) and yet it was absent from Rietfontein Nature Reserve which is another likely indication that the Reserve is isolated from the rest of the corridor. Fallow deer (*Dama*

dama) had a very high total RA (0.517) and were also absent from Rietfontein Nature Reserve which is further evidence that the reserve appears isolated from the rest of the corridor. Fallow deer were introduced into Leeukop Prison which is located between sites two and three where this species was recorded. For the purposes of this study fallow deer and other exotic animals were not used to assess the quality of the corridor but may serve as good indicators of connectivity. Corridors may facilitate the spread of exotic species as well as indigenous species (Resasco, *et al*., 2014).

Habitats that have higher heterogeneous vegetation attributes are expected to support higher mammal diversity but in this study vegetation units across the corridor were not significantly different between sites (Chapter 3). Vegetation composition is thus unlikely to be an explanatory variable for the lower number of native small mammal species recorded at Site 1. One possible explanation for this finding is the higher density of urbanisation that occurs around Site 1. Mammal species richness generally decreases with an increase in urbanisation through habitat loss and other associated disturbances (McKinney, 2002; Pardini *et al.*, 2004; McKinney, 2008; Dornelas, 2010). Lower small mammal species richness suggested a higher impact from urbanisation and the hostile higher density development surrounding Site 1 could present a barrier to movement and could potentially benefit from the inclusion of ecological corridor features in the landscape configuration.

The presence of blesbok (*Damaliscus pygargus phillipsi)* within this corridor was unexpected and not likely to occur naturally in this corridor system. It was only recorded at Site 1 and was likely introduced into the Nature Reserve. Its presence in this study may serve to provide additional evidence suggesting that site 1 has become isolated from the rest of the corridor for larger mammals but further research on this would need to be done.

The aim of the study was to assess the corridor for connectivity and usefulness for mammals but these values will vary for each species. Direct movement by a species only requires that the corridor is able to supply sufficient resources to the individual during that time of travel (Merriam and Saunders, 1993). The presence of South African hedgehog, slender mongoose *(Galerella sanguinea),* large-spotted genet (*Genetta tigrina),* scrub hare *(Lepus saxatilus),* multimammate mouse (*Mastomys natalensis)* and common duiker (*Sylvica pragrimmia)* along the entire corridor may indicate that the corridor is able to supply sufficient resources to support them. Daily travel to search for resources is the most likely mode of movement in this corridor but generational connectivity was not assessed in this study. The results of this study may, however, not reflect a true picture of the activities on site. The cameras were not always set up in the best spots to record species because of the high level of human activity at some of the sites which may have prevented the detection of some species. For this study, sampling was done in summer in 2016 when it was very hot and Johannesburg was experiencing drought conditions. Due to time constraints sampling was not done in winter but future studies should aim to sample in multiple seasons if possible.

The vegetation type along the corridor offers relatively high levels of aerial or canopy cover, providing ways for certain species to cross barriers such as fences and walls. Therefore, species which are proficient climbers would be poor indicators of habitat connectivity for this study. Large-spotted genet occurred at all three sites and its excellent climbing ability would likely increase its ability to cross barriers along the corridor and thus increase its chances of survival within an urban environment. A recent study done by Widdows (2015) on large-spotted genets in urban environments in KwaZulu-Natal, South Africa, demonstrated how successful and resilient this species is to anthropogenic disturbances.

The present study demonstrates that an ecological corridor in Johannesburg may provide sufficient resources to sustain several native small mammal species and possibly allow for movement between habitat patches. Areas with higher development pressures can negatively impact certain species and reduce the value of any corridors. Previous research on the mammalian response to urbanisation supports the theory that connecting habitat in fragmented landscapes can reduce the negative impacts associated with development (Maisonneuve and Rioux, 2001; Pardini *et al.*, 2004; Goad *et al.*, 2014; Hurst *et al*., 2014; Ramesh *et al*., 2016). Results from this study are largely inconclusive but provide evidence to suggest that urban ecological corridors may have conservation value but further research needs to be done to develop urban conservation strategies. Although this study only includes one season of data, the use of occupancy data and non-invasive techniques may form part of a model for future research.

Chapter 5. Discussion and Conclusion

Assessing ecological corridors for usefulness requires a clear understanding of the differences between “use” and “function”. Use simply refers to utilisation while function implies the completion of a particular task (Bennett, 1990). Many studies on ecological corridors aim to assess the functionality of the corridor but are in fact only assessing use (Mimet *et al*., 2013). Both terms are interchangeable if the aim of the research is to provide additional habitat. However, if connectivity is being assessed by studying the genetic or demographic interactions of populations between habitat fragments, then functionality has different implications (Horskins, 2005).The aim of this study was to analyse an ecological corridor in Johannesburg and assess its usefulness for biodiversity conservation. The data obtained in this study have given valuable insights into the spatial characteristics of an urban corridor, the condition of the flora and its potential value to facilitate small mammal movement and provision of suitable habitat.

Identification of potential ecological corridors can assist with determining which core habitats within the urban landscape offer the highest ecological value for biodiversity conservation. Corridors may also preserve ecological components, increasing an urban landscape’s functionality (Sandstrom *et al*., 2006). Landscape connectivity should be a priority in development planning and least-cost path analysis provides an effective way of determining the spaces critical to connectivity (Chapter 2). Maps generated from these methods showed where the corridors pass through degraded or fragmented habitats that would hinder the function of the corridor (Appendix B, C and D). In the study done by Yu *et al*., (2005) it was noted that Chinese cities frequently failed to take into consideration the size, location and species composition of the green spaces being developed. Development of

ecological corridors must prioritize these factors and using site-specific assessments would likely yield higher levels of diversity (Kong *et al*., 2010).

For this research project, demography and genetics were not assessed; therefore any reference to functionality of the corridor refers to the provision of habitat for wildlife. The data collected in this study gave valuable insights into the possible links between fragmented habitat patches due to 1) similar vegetation structure between sites (Chapter 3), and 2) the presence of several small mammal species occurring along the corridor (Chapter 4). A lower small mammal species richness value at the only site representing a protected area in the form of a Nature Reserve could be viewed as an indication of a potential barrier to movement to the rest of the corridor. Least-cost path analysis done in Chapter 2 may act as a guiding framework for planning but it does not take into account specific site conditions. A corridor that enables true connectivity would likely require individual animals that are capable of reproducing to live within and move along the corridor, depending on the species specific dispersal patterns (Horskins, 2005). Without these genetic or demographic data, the only assumption that can be made from the observations in this study is that this ecological corridor has the potential to allow for true connectivity but additional data would need to be obtained to show it.

5.1 Vegetation cover and structure

In areas that are becoming urbanised, native plant communities are frequently replaced by introduced plant communities, creating novel environments which influence the ecological and environmental relationships in the ecosystem (Chace and Walsh, 2006; Kowarik, 2011). The vegetation percentage cover, structure and species composition are

known to be impacted by urbanisation as thus have regularly been used as indicators of disturbance (McKinney, 2008; Kowarik, 2011). Urbanisation may alter the vegetation composition of an area but it may also increase the structural diversity of the vegetation (Chace and Walsh, 2006). However in this study the percentage of vegetation cover and structure along the corridor were not significantly different. Plant species richness, which may be a useful indication of the habitat quality (Angold *et al*., 2006), appeared to be similar between sites (see chapter 3) which may indicate that all the sites have been impacted by similar levels of disturbance.

Previous studies have shown that the number of plant species in an area increases in response to urbanisation, as exotic species are added to the native species, giving a high overall species richness but then decreases as the levels of disturbance increase (McKinney, 2008). However, there are many factors that may influence the plant species richness other than disturbance and thus these results may not be directly linked to the impacts of urbanisation (Lite *et al*., 2005). The relatively high number of plant species recorded at the all the sites suggests that they have been impacted by urbanisation but the total percentage of exotic species present at each site was found to be relatively low. So while the corridor showed evidence of disturbance by urbanisation, it appears to have retained a relatively high level of native biodiversity.

5.2 Corridor use by small mammals

The distribution of mammals across the ecological corridor being assessed in this study was unexpected (Chapter 4). Only one of the three study sites, Rietfontein Nature Reserve, was listed as a Protected Area and my initial expectation was that it would likely

have the highest mammal species richness value which would likely indicate a higher ecological value. The relatively low small mammal species richness at the Rietfontein Nature Reserve may suggest that one or more of the apparent physical barriers present (i.e. a fence surrounding the site, close proximity to a road and surrounding residential land-use) may provide an explanation for the differences in small mammal species richness between sites.

The results of Chapter 2 of this study suggest that choosing a focal species, to assess the usefulness of the corridor would likely be an effective method to test connectivity and allow for more conclusive evidence to support its value for connectivity between habitat fragments. Although mammals serve as good indicators to assess wildlife corridors due to their propensity for movement and high conservation interest, their complex social structuring may lead to generalisations of corridor usefulness which may be inaccurate (Horskins, 2005). The results from Chapter 4 of this study suggest that the corridor may be playing an important role in providing habitat for a variety of mammal species and possibly allowing for movement over a large portion of it, but in order to truly measure its effectiveness, further genetic research would need to be done and social behaviours for each selected species would need to be considered.

5.3 Ecological corridors in conservation

It has long been suggested that ecological corridors should be considered as conservation strategies in decision-making processes (Harris and Scheck, 1991; Harrison, 1992; Lindenmayer and Nix, 1993). These studies have been focussed primarily on physical attributes such as corridor width and vegetation structure and place emphasis on assessing habitat use for wildlife and less so on connectivity (Ahern, 1995). In order to maximise the

effectiveness of a corridor in providing connectivity, several additional factors may also be worth including in the planning stage.

Genetic diversity is known to be reduced in populations which have been impacted by habitat fragmentation (Luque *et al*., 2012). Genetic data were not used in this study but are worth consideration for future research. The implementation of a wildlife corridor may be considered as a management strategy to conserve endangered species, particularly in patches where a healthy population exists within one patch and not within an isolated nearby patch. In this situation, connecting the isolated habitat fragments may increase the available habitat by creating an ecological corridor. However, as mentioned in previous sections, the population must have the capacity to able to colonise the new habitat. Establishing a new corridor implies the creation of new habitat with high potential to create connectivity due to the absence of exiting residents in those areas. Thus, creating new corridors will likely result in an influx of new individuals into an area and establish new populations (Horskins, 2005).Connectivity will only persist if the social organisation of each species allows for it and may vary depending on the time needed for each species to become established in the new habitat(Horskins, 2005). This would suggest that any studies to assess the effectiveness of a new constructed corridor would need to take place over several time intervals after the construction.

5.4 Limitations to the study

This was a baseline study, so there are no previous data to compare my results with. The study was only done over a few months and mammal distributions may shift over time, which can result in some species being absent from a study site, reducing the species richness

and thus skewing the data. Future studies may consider trapping over multiple seasons to account for population changes over time. There may be other factors influencing the small mammal species richness that we did not identify or assess such as poaching, presence of domestic animals or recent events such as extended drought conditions (2016). The value of the ecological corridors as a means of actual mammal movement between habitat fragments was not assessed and could form part of a future study. One study group does not truly reflect the value of the corridor to biodiversity- mammals may respond in an entirely different way compared with another group of organisms such as birds which, some studies have shown, can be sensitive to the vegetation quality and may have stronger dependence on native habitat (Bolger *et al.*, 2001).

It has been suggested that connectivity requires more than just ecological corridors (Merriam, 1995). What this means is that connectivity requires more than just movement and residency to be considered as functional because true dispersal between habitat fragments requires integration of new individuals into the recipient population (Horskins, 2005). Trapping does not provide insight into the reproductive success of individuals and therefore cannot provide accurate assessments of connectivity. However, if sufficient trapping is able to show frequent movement between populations, it could suggest that the probability of population integration would be high (Horskins, 2005). The data from this study demonstrated how difficult it can be, using live traps, to obtain sufficient captures to draw any conclusions on movement patterns. Due to risk of theft and tampering, traps could not necessarily be placed in the areas that are most optimal to record mammals and therefore may skew results. The high number of traps used each night and the low number of new individuals captured may be as a result of urbanisation but there are many factors that could be influencing this outcome. It is unlikely that sufficient data will be collected from trapping

studies, within ecological corridors, at this relatively small scale, to give accurate comment on the degree of true connectivity. Alternative methods such as the use of DNA may provide greater insights to determine connectivity. This study did not quantify the specific impacts of surrounding land uses such as the value of gardens for the provision of resources for small mammals, which may invite certain species into the corridor.

5.5 Conclusion

Urbanisation is still happening at a rapid rate in many parts of Gauteng, including the Kyalami and Midrand area where the study corridor is located. There are several proposed developments such as major roads, large residential estates and industrial properties within the area which are likely to have negative impacts on the remaining habitat fragments and the local biodiversity in the future. This research has provided some evidence to support the idea that suggests that these small mammals may be using this corridor for resources and potentially movement between habitat patches. The small mammal species richness along the corridor suggests that it has value as habitat provision for the native mammals.

References

Ahern, J. 1995. Greenways as a planning strategy. *Landscape and Urban Planning* 33, 131–155.

Ahern, J. 2013. Urban landscape sustainability and resilience: the promise and challenges of integrating ecology with urban planning and design. *Landscape Ecology* 28, 1203–1212.

Alberti, M. 2008. *Advances in Urban Ecology: Integrating Humans and Ecological Processes in Urban Ecosystems*. Springer, Seattle, Washington, USA.

Andersson, E. and Boden, Ö. 2008. Testing network based models of habitat fragmentation in complex landscapes – implications for management. *Ecography* 32, 123–132.

Anderson, P.M.L., Avlonitis, G., Ernstson, H., 2014. Ecological outcomes of civic and expert-led urban greening projects using indigenous plant species in Cape Town, South Africa. *Landscape and Urban Planning* 127, 104–113.

Angold, P.G., Sadler, J.P., Hill, M.O., Pullin, A., Rushton, S., Austin, K., Small, E., Wood, B., Wadsworth, R., Sanderson, R., Thompson, K., 2006. Biodiversity in urban habitat patches. *Science of the Total Environment* 360, 196–204.

Arnfield, A. J. 2003. Two decades of urban climate research: A review of turbulence, exchanges or energy and water and the urban heat island. *International Journal of Climatology*, 23(1), 1–26.

Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R. E., *et al*. 2002. Economic reasons for conserving wild nature. *Science*, 297, 950–953.

Beier, P. and Loe, S. 1992. A checklist for evaluating impacts to wildlife movement corridors. *Wildlife Society Bulletin* 20, 434–440.

Barnett J.L., How, R. A., Humphreys, W. F. 1978.The use of habitat components by small mammals in eastern Australia. *Australian Journal of Ecology*, 3, 277–285.

Barton, H. N. 1992. *The genetic consequences of dispersal.* In*: Animal dispersal: small mammals as a model* (Eds. Stenseth, N. C. and Lidicker Jr., W.Z) 37-60. Chapman and Hall, London.

Beardsley, K., Thorne, J.H., Roth, N.E., Gao, S., McCoy, M.C., 2009. Assessing the influence of rapid urban growth and regional policies on biological resources. *Landscape and Urban Planning* 93, 172–183.

Beier, P, Loe, S. 1992, A checklist for evaluating impacts to wildlife movement corridors. *Wildlife Society Bulletin* 20, 434–440,

Beninde, J., Veith, M., Hochkirch, A., 2015. Biodiversity in cities needs space: a meta-analysis of factors determining intra-urban biodiversity variation. *Ecology Letters* 18, 581–592.

Bennett, A.F., 1990. Habitat corridors and the conservation of small mammals in a fragmented forest environment. *Landscape Ecology* 4, 109–122.

Blazquez-Cabrera, S., Bodin, Ö.,Saura, S., 2014. Indicators of the impacts of habitat loss on connectivity and related conservation priorities: Do they change when habitat patches are defined at different scales? *Ecological Indicators* 45, 704–716.

Blums, P. Nichols, D. J. Lindberg, S. M. Hines, E.J. and Mednis, A. 2003.Factors affecting breeding dispersal of European ducks on Engure Marsh, Latvia. *Journal of Animal Ecology* 72, 292–307.

Bolger, D. T., Scott, T. A., Rotenberry, J. T. 2001. Use of corridor-like landscape structures by bird and small mammal species. *Biological Conservation*, 102,213–224.

Bolund P, Hunhammar S. 1999. Ecosystem services in urban areas. *Ecological Economics* 29,293–301

Bond, M., 2003. *Principles of Wildlife Corridor Design. Center for Biological Diversity.*

Borgström, S. 2009. Patterns and challenges of urban nature conservation – A study of southern Sweden. *Environment and Planning A*, 41(11), 2671–2685.

Borgström, S., Cousins, S.A.O., Lindborg, R., 2012. Outside the boundary – Land use changes in the surroundings of urban nature reserves. *Applied Geography* 32, 350–359.

Borgström, S., Lindborg, R., Elmqvist, T. 2013.Nature conservation for what? Analyses of urban and rural nature reserves in southern Sweden. *Landscape Urban Planning*1909-2006.

Boyden, S., Millar, S., Newcombe, K., and O'Neill, B. 1981.*The Ecology of a City and Its People. The Case of Hong Kong*. Canberra: Australian National University Press.117,66–80.

Brás, R., Cerdeira, J.O., Alagador, D., Araújo, M.B., 2013. Linking habitats for multiple species. *Environmental Modelling and Software* 40, 336–339.

Brown J.H. 1984. On the relationship between abundance and distribution of species. *American Society of Naturalists* ;124, 255–279.

Bryant, M.M., 2006. Urban landscape conservation and the role of ecological greenways at local and metropolitan scales. *Landscape and Urban Planning* 76, 23–44.

Bullock, J. 1997. *Plants. In: Sutherland, W.J. (Ed.), Ecological Census Techniques: A Handbook.* Cambridge University, Cambridge, pp. 186–213.

Bullock, W. L., and Samways, M. J. 2005. Conservation of flower-arthropod association in remnant African grassland corridors in an afforested pine mosaic. *Biodiversity and Conservation* 14, 3093–3103.

Burel, F. 1996. Hedgerows and their role in agricultural landscapes, Critical Review in *Plant Sciences*, 15, pp. 169–190.

Burgess, E.W. 1924 *a.* "The Growth of the City: An Introduction to a Research Project." In *American Sociological Society, Eighteenth Annual Meeting, 1923, Papers and Proceedings*, vol. 18. Chicago: University of Chicago Press, 85–97.

Burnett RT, Cakmak S, Brook JR.1998. The effect of the urban ambient air pollution mix on daily mortality rates in 11 Canadian cities. *Canadian Journal of Public Health*. 89, 152–156.

Cambell, B. H. 1981. An aqueduct as a potential barrier to the movements of small mammals. The *Southwestern Naturalist*, 26, 84–85.

Carlson C, Canty D, Steiner F, Mack N. 1989. A path for the Palouse: An example of conservation and recreation planning. *Landscape and Urban Planning*. 17,1–19.

Caughley, G., Short, J., Grigg, G. and Nix, H. 1987. Kangaroos and climate: an analysis of distribution. *Journal of Animal Ecology*, 56, 751–61.

Cilliers, S., du Toit, M., Cilliers, J., Drewes, E., Retief, F., 2014. Sustainable urban landscapes: South African perspectives on transdisciplinary possibilities. *Landscape and Urban Planning* 125, 260–270.

Cilliers, S.S., Müller, N., Drewes, E., 2004. Overview on urban nature conservation: situation in the western-grassland biome of South Africa. *Urban Forestry and Urban Greening* 3, 49–62.

Chace, J. F. and Walsh, J. J. 2006. Urban effects on native avifauna: a review. *Landscape and Urban Planning*, 74, 46–69.

Cockburn, A. 1992.*Habitat heterogeneity and dispersal: environmental and genetic patchiness*. In: *Animal Dispersal: Small Mammals as a Model* (Eds. Stenseth, N. C. and Lidicker Jr, W.Z.) 65–85. Chapman and Hall, London.

Compaan, P., 2011, *Technical report: Gauteng conservation plan version 3.3 (C-Plan 3.3)*, unpublished report for the Gauteng Department of Agriculture and Rural Development, Johannesburg, South Africa, viewed 25 October 2017, from https://www.researchgate.net/publication/315771900_Gauteng_Conservation_Plan_Version_33_C-Plan_33.Conover M.R. 2002.*Resolving human- wildlife conflicts: the science of wildlife damage management*. 1st ed. Boca Raton: CRC Press.

Cornelis, J., Hermy, M. 2004. Biodiversity relationships in urban and suburban parks in Flanders. *Landscape and Urban Planning*, 69, 385–401.

Cotter, M., Berkhoff, K., Gibreel, T., Ghorbani, A., Golbon, R., Nuppenau, E.-A., Sauerborn, J., 2014. Designing a sustainable land use scenario based on a combination of ecological assessments and economic optimization. *Ecological Indicators*, 36, 779–787

Crookes, D.J., Blignaut, J.N., de Wit, M.P., Esler, K.J., Le Maitre, D.C., Milton, S.J., Mitchell, S.A., Cloete, J., de Abreu, P., Fourie, H., Gull, K., Marx, D., Mugido, W., Ndlovu, T., Nowell, M., Pauw, M. andRebelo, A. 2013. System dynamic modelling to assess economic viability and risk trade-offs for ecological restoration in South Africa. *Journal of Environmental Management*, 120, 138–147.

Currie, D. J. 1991. Energy and large-scale patterns of animal- and plant-species richness. *American Naturalist*, 137,, 27–49.

Daily, G. C., ed. 1997.*Nature's services: societal dependence on natural ecosystems*. Island Press, Washington, DC.

Dallimer M, Irvine K. N, Skinner MJ, Davies Z. G, Rouquette J. R, Maltby L. L, Warren P. H, Armsworth P. R, Gaston K. J. 2012. Biodiversity and the feel-good factor: understanding associations between self-reported human well-being and species richness. *Bioscience* 62, 47–55

Decker, D.J., Forstchen, A.B., Jacobson, C.A., Smith, C.A., Organ, J.F. and Hare, D. 2012. What does it mean to manage wildlife if public trust really matters? In: *Trans. 78th North American Wildlife and Natural Resources Conference,* 74, 47–54.

Department of Environmental Affairs and Tourism (DEAT),.1998.*Guideline Document: EIA Regulations–Implementation of Sections 21, 22 and 26 of the Environmental Conservation Act*. Department of Environmental Affairs and Tourism, Pretoria.

Department of Environmental Affairs (DEA). 2014. National Biodiversity Research Development & Evidence Strategy (2015-2025). Department of Environmental Affairs, Pretoria.39 Pages.

Désilets P, Houle G. 2005. Effects of resource availability and heterogeneity on the slope of the species-area curve along a floodplain-upland gradient. J. *Vegetation Science* 16, 487–496

Dornelas, M. 2010. Disturbance and change in biodiversity. *Philosophical Transactions of the Royal Society B,*365, 3719–3727.

Diserud, O.H., Odegaard, F., 2007. A multiple-site similarity measure. *Biology Letters* 3, 20–22.

NBA. 2011. National Biodiversity Assessment 2011: An assessment of South Africa's biodiversity and ecosystems. Synthesis Report. By Driver, A., Sink, K.J., Nel, J.N., Holness, S., Van Niekerk, L., Daniels, F., Jonas, Z., Majiedt, P.A., Harris, L. & Maze, K. South African National Biodiversity Institute and Department of Environmental Affairs, PretoriaElmqvist, T., Folke, C., Nyström, M., Peterson, G., Bengtsson, J., Walker, B., Norberg, J., 2003.Response diversity, ecosystem change and resilience. *Frontiers in Ecology and the Environment* 1, 488–494.

Elmqvist, T., Alfsén, C., Colding, J. 2008.Urban systems. In S.-E.Jörgensen, and B.Fath (Eds.), *Encyclopaedia of Ecology*. St. Louis, USA: Elsevier.

ESRI. 2014. ArcGIS Desktop 10.3. Environmental Systems Research Institute, Redlands.

Fabos, J., G, Ahern, J. 1995. *Greenways: The Beginning of an International Movement*. Elsevier, New York.

Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. *Annual review of Ecology and Systematics* 34,487–515

Fahrig, L. and Merriam, G. 1994. Conservation of fragmented populations. *Conservation Biology* 8(1), 50–59.

Fahrig, L., Rytwinski,T.. 2009. Effects of roads on animal abundance: an empirical review and synthesis. *Ecology and Society* 14(1), 21.

FWC (Florida Wildlife Corridor). 2018. About the wildlife corridor. http://floridawildlifecorridor.org/about/

Gauteng Conservancy Association (GCA).Conservancies Handbook. Retrieved from http://www.conservancies.org/Downloads/Condensed%20Conservancy%20Handbook.pdf.

Gauteng Provincial Government. 2014. Planning for 2014- making the future work today. 15 pages.

Gehring T. M, Swihart R. K. 2003. Body size, niche breadth, and ecologically scaled responses to habitat fragmentation: mammalian predators in an agricultural landscape. *Biological Conservation* 109, 283–295

George, S.L., Crooks, K.R., 2006. Recreation and large mammal activity in an urban nature reserve. *Biological Conservation* 133, 107–117.

Goad, E.H., Pejchar, L., Reed, S.E., Knight, R.L., 2014. Habitat use by mammals varies along an exurban development gradient in northern Colorado. *Biological Conservation* 176, 172–182.

Greve, M., Chown, S.L., van Rensburg, B.J., Dallimer, M. and Gaston, K.J. 2011. The ecological effectiveness of protected areas: a case study for South African birds. *Animal Conservation*, 14, 295–305

Gunderson, L. H.,Holling,C. S. **.** 2002. *Panarchy: understanding transformations in human and natural systems.* Island Press, Washington, D.C., USA.

Haddad, N. M., Bowne, D. R., Cunningham, A., Danielson, B. J., Levey, D. J., Sargent, S and Spira, T. 2003. Corridor use by diverse taxa. *Ecological Society of America* 84(3), 609–615.

Hanski, I. 1994. Spatial scale, patchiness and population dynamics on land. *Philosophical Transactions of the Royal* Society *A* B 343, 19–25.

Hansen, A. J., Rotella, J. J.2002. Biophysical factors, land use, and species viability in and around nature reserves. *Conservation Biology* 16(4), 1112–1122.

Harris, L.D., 1984. *The Fragmented Forest: Island Biogeography Theory and the Preservation of Biotic Diversity*. University of Chicago Press, Chicago, IL.

Harris, L. D. and Schleck, J. 1991. From implications to applications: the dispersal corridor principle applied to conservation on biological diversity. In: Nature Conservation (Eds. D.A. Saunders and R. J. Hobbs) 189–220. Surrey Beatty and Sons Pty Ltd, Sydney.

Harrison, R. L. 1992. Toward a theory of inter-refuge corridor design. *Conservation Biology* 6(2), 293–295.

Hawkins, B. A.,Porter, E. E.. 2003. Does herbivore diversity depend on plant diversity? The case of California butterflies. *American Naturalist* 161, 40–49.

Hess, G. R, Fischer, R. A. 2001. Communicating clearly about conservation corridors. *Landscape and Urban Planning* 55, 195–208

Hill, D, Tucker G, Fasham M, Shrewry, M, Shaw P.2005.*Handbook of Biodiversity Methods: Survey, Evaluation and Monitoring*. Cambridge University Press, Cambridge.

Hindson, D,Byerley, M., Morris, M. 1994. `From Violence to Reconstruction: The Making, Disintegration, and Remaking of an Apartheid City', Antipode 26(4), 323–350.

Hoctor, T. S. 2003. Regional landscape analysis and reserve design to conserve Florida's biodiversity. PhD Dissertation, University of Florida, Gainesville, FL.

Hodgson, J.A., Thomas, C.D., Wintle, B.A.,Moilanen, A. 2009 Climate change, connectivity and conservation decision making: back to basics. *Journal of Applied Ecology*, 46, 964–969.

Horskins, K. 2005. *The Effectiveness of Wildlife Corridors in Facilitating Connectivity: Assessment of a Model System from the Australian Wet Tropics.* PhD thesis, Queensland University of Technology.

Hurst, Z.M., McCleery, R.A., Collier, B.A., Silvy, N.J., Taylor, P.J., Monadjem, A., 2014. Linking changes in small mammal communities to ecosystem functions in an

agricultural landscape. *Mammalian Biology – Zeitschrift für Säugetierkunde* 79, 17–23.

Isbell, F., Craven, D., Connolly, J.,Loreau, M., Schmid, B., Beierkuhnlein, C., Bezemer, T.M., Bonin, C.,Bruelheide, H., de Luca, E., Ebeling, A., Griffon, J.N., Guo, Q., Hautier, Y., Hector, A., Jentsch, A.,Kreyling, J., Lanta, V., Manning, P., Meyer, S.T., Mori, A.S., Naeem, S., Niklaus, P.A., Polley, H.W., Reich, P.B., Roscher, C.,Seabloom, E.W., Smith, M.D., Thakur, M.P., Tilman, D., Tracy, B.F., van der Putten, W.H., van Ruijven, J.,Weigelt, A., Weisser, W.W., Wilsey, B.,Eisenhauer, N. 2015. Biodiversity increases the resistance of ecosystem productivity to climate extremes. *Nature* 526, 574–577.

IUCN. 2005. Benefits beyond boundaries. In *Proceedings of the Vth IUCN World Parks Congress* Cambridge.

Irvine, K. N., Fuller, R. A., Devine-Wright, P., Tratalos, J., Payne, S. R., Warren, P. H., Lomas, K. J., Gaston, K. J. 2010. *Ecological and psychological values of urban green space*. Pages 215–237 in Jenks M, Jones C, eds. Dimensions of the Sustainable City. Springer.

Ives, C.D., Hose, G.C., Nipperess, D.A., Taylor, M.P., 2011. Environmental and landscape factors influencing ant and plant diversity in suburban riparian corridors. *Landscape and Urban Planning* 103, 372–382.

IUCN, 2013.Documentation Standards and Consistency Checks for IUCN Red List Assessments and Species Accounts. IUCN Red List Committee and IUCN SSC Steering Committee.

Jansson, Å.,Polasky, S., 2010. Quantifying biodiversity for building resilience for food security in urban areas: getting down to business. *Ecology and Society* 15 (3), 20.

Johnson, M. D. 2007. Measuring habitat quality: A review. *The Condor*, 109(3), 489–504.

Jones, C.G., Lawton J. H., Shachak, M. 1997.Positive and negative effects of organisms as physical ecosystem engineers. *Ecological Society of America* 78, 1946–1957.

Joubert, L., Pryke, J.S., Samways, M.J., 2016. Past and present disturbances influence biodiversity value of subtropical grassland ecological networks. *Biodiversity and Conservation* 25, 725–737.

Kansky, R., Knight, A.T., 2014. Key factors driving attitudes towards large mammals in conflict with humans. *Biological Conservation* 179, 93–105.

Kattel, G.R., Elkadi, H., Meikle, H., 2013. Developing a complementary framework for urban ecology. *Urban Forestry and Urban Greening* 12, 498–508.

Kent, M., Coker, P. 1992. *Vegetation description and analysis. A practical approach*. John Eiley and Sons Ltd. England.

Kiffner, C., Albertini, M., Ede, A., Donnellan, B., Hahn, N., McGinnis, M.A., Nietlisbach, N.A., Tate, J., Kioko, J., 2015. Performance of individual species as indicators for large mammal species richness in Northern Tanzania. *Ecological Indicators.* 53, 70–77.

Kindleman, P., Burel, F., 2008. Connectivity measures: a review. *Landscape Ecology*, 23,879–890.

Knaapen, J. P., M. Scheffer, and B. Harms. 1992. Estimating habitat isolation in landscape planning. *Landscape and Urban Planning* 23,1–16.

Kong, F., Yin, H., Nakagoshi, N., Zong, Y., 2010.Urban green space network development for biodiversity conservation: Identification based on graph theory and gravity modelling. *Landscape and Urban Planning* 95, 16–27. https://doi.org/10.1016/j.landurbplan.2009.11.001

Kowarik, I., 2011. Novel urban ecosystems, biodiversity, and conservation. *Environmental Pollution* 159, 1974–1983.

Krosby, M., Tewksbury, J., Haddad, N. M., Hoekstra, J. 2010.Ecological connectivity for a changing climate. *Conservation Biology* 24 (6),1686–1689.

Lancaster, J., Pillay, N., Ganem, G., Meynard, C., Perigault, M., Edwards, S., Caminade, P., Watson, J. 2010.Environmental correlates and co-occurrence of three mitochondrial lineages of striped mice (*Rhabdomys*) in the Free State Province (South Africa).*Acta Oecologica*, 42, 30–40.

Laurence, F. W. 2001. Letter to The Independent (London) 02/022001.

Lee, P., Smyth, C., Boutin, S. 2004. Quantitative review of riparian buffer width guidelines from Canada and the United States. *Journal of Environmental Management*, 70, 165–180.

Lee, R., 2011. The outlook for population growth. *Science* 333, 569–573.

Lewis, T. 1969. The diversity of the insect fauna in a hedgerow and neighbouring fields. *Journal of Applied Ecology*, 6,453–458.

Li, H., Chen, W., He, W. 2015. Planning of green space ecological network in urban areas: an example of Nanchung, China. *International Journal of Environmental Research and Public Health12, 12889*–12904.

Lidicker Jr., W.Z.,Stenseth, N. C. 1992. To disperse or not to disperse: who does it and why? In: Animal dispersal: small mammals as a model (Eds. Stenseth, N. C. and Lidicker Jr., W. Z) 21-36. Chapman and Hall, London.

Lindenmayer, D.B., Laurance, W.F., Franklin, J.F. *et al*. 2013. New policies for old trees: averting a global crisis in a keystone ecological structure. *Conservation Letter*, 7, 61–69.

Lindenmayer ,D.B. and Nix, H.A.. 1993. Ecological principles for the design of wildlife corridors. *Conservation Biology*, 7, 627–630

Linehan, J., Gross, M., Finn, J., 1995. Greenway planning: developing a landscape ecological network approach. *Landscape and Urban Planning* 33, 179–193.

Linnell, J.D.C., Rondeau, D., Reed, D.H., Williams, R., Altwegg, R., Raxworthy, C.J. 2010. Confronting the costs and conflicts associated with biodiversity. *Animal Conservation*, 13,429–431

Lite, S.J., Bagstad, K.J., Stromberg, J.C. 2005.Riparian plant species richness along lateral and longitudinal gradients of water stress and flood disturbance, San Pedro River, Arizona, USA .*Journal of Arid Environments*,63, 785–813.

Loss, S.R., Will, T., Marra, P.P. 2013. The impact of free-ranging domestic cats on wildlife of the United States. *Nature Communications* 4, 1396.

Luck M. and Wu J. 2002. A gradient analysis of urban landscape pattern: A case study from the Phoenix Metropolitan region of USA. *Landscape Ecology* 17,327–329.

Luque, S., Suara, S., Fortin, M.J. 2012.Landscape connectivity analysis for conservation: insights from combining new methods with ecological and genetic data. *Landscape Ecology* 27, 153–157

Mader, H.J. 1984. Animal habitat isolation by roads and agricultural fields. *Biological Conservation*, 29,81–96.

Maisonneuve, C., Rioux, S., 2001.Importance of riparian habitats for small mammal and herpetofaunal communities in agricultural landscapes of southern Québec. *Agriculture, Ecosystems and Environment* 83, 165–175.

Magurran, A.E. 2004.*Measuring biological diversity*. Blackwell, Malden, MA.

Malo,J.E., Suárez, F., Díez, A. .2004. Can we mitigate animal-vehicle accidents using predictive models? *Journal of Applied Ecology*, 41, 701–710

Mansergh, I. M., D. J. Scotts. 1989. Habitat continuity and social organisation of the mountain pygmy-possum restored by tunnel. *Journal of Wildlife Management* 53,701–707.

Marzluff, J.M., Ewing, K. 2001. Restoration of fragmented landscapes for the conservation of birds: a general framework and recommendations for urbanizing landscapes. *Restoration Ecology.* 9, 280–292.

McKinney, ML. 2002. Urbanization, Biodiversity, and Conservation: The impacts of urbanization on native species are poorly studied, but educating a highly urbanized human population about these impacts can greatly improve species conservation in all ecosystems. *Biological Science* 52, 883–890.

McKinney, ML. 2006a. Urbanization as a major cause of biotic homogenization. Biological Conservation 127, 247–260

McKinney, M. L. 2008.Effects of urbanization on species richness- A review of plants and animals. *Urban Ecosystem*, 11, 161–176.

Meikle, H., Elkadi, H., 2012. Urban ecology and the future of cities. In: Healthy Cities Conference, Geelong, 2012.Available fromhttps://healthycities.com.au/BOP/PR.pdf

Meiklejohn K., Ament R., Tabor, G. 2009. Habitat Corridors and Landscape Connectivity: Clarifying the Terminology. Center for Large Landscape Conservation. *Landscape Ecology Practices in Planning:* Available from:www.climateconservation.org.

Meir, E., Andelman, S., Possingham, H. P. 2004. Does conservation planning matter in a dynamic and uncertain world? *Ecology Letters* 7, 615–622.

Merriam, G. 1995. Movement in spatially divided populations: responses to landscape structure. In: Landscape approaches in mammalian ecology and conservation (Eds. Lidicker Jr., W. Z.) 64–77. University of Minnesota Press, Minneapolis.

Millennium Ecosystem Assessment. 2005. *Ecosystems and Human Well-Being: Biodiversity Synthesis. A Report of the Millennium Ecosystem Assessment*. World Resources Institute, Washinton, DC, USA

Miller, J. R. 2006.Restoration, reconciliation, and reconnecting with nature nearby. *Biological Conservation* 127, 356–361

Mimet, A., Houet, T., Julliard, R., Simon, L., 2013.Assessing functional connectivity: a landscape approach for handling multiple ecological requirements. *Methods in Ecology and Evolution* 4, 453–463.

MaloJ.E., Suarez, F., Diez, A. 2004. Can we mitigate animal–vehicle accidents using predictive models? *Journal of Applied Ecology* 41 701–710

Moffatt, S.F., McLachlan, S.M., Kenkel, N.C. 2004.Impacts of land use on riparian forest along an urban - rural gradient in Southern Manitoba. *Plant Ecology*, 174, 119–135.

Morelli, F., Tryjanowski, P., 2014. Associations between species can influence the goodness of fit of species distribution models: the case of two passerine birds. *Ecological Complexity* 20, 208–2012.

Mori, M.D., Crick, H.Q.P., Duffield, S.J., Macgregor, N.A. 2015. Resilience to climate change: translating principles into practice. *Journal of Applied Ecology*49, 547–551.

Msuha, M.J., Carbone, C., Pettorelli, N. and Durant, S. M. 2012. Conserving biodiversity in a changing world: Land-use change and species richness in northern Tanzania. *Biological Conservation* 21, 2747–59.

Mucina, L. and Rutherford, M.C. (eds.). 2006. *The vegetation of South Africa, Lesotho and Swaziland*, South African National Biodiversity Institute, Pretoria.

Naiman, R.J., Decamps, H., Pollock, M. 1993. The Role of riparian corridors in maintaining regional biodiversity. *Ecological Applications,* 3,209–212.

Naiman, R.J., Decamps, H. 1997. The ecology of interfaces: riparian zones. *Annual Review of Ecology and Systematics* 28,621–658.

National Wildlife Corridors Plan Advisory Group (Australia), Australia, Department of Sustainability, E., Water, Population, and Communities, 2012. *National Wildlife Corridors Plan: a framework for landscape-scale conservation* 2012.

Odum, E. P. 1963. *Ecology*. Holt, Rinehart and Winston, Nueva York.

O'Farrell, P. J., Milton, S. J. 2006. Road verge and rangeland plant communities in the southern Karoo: Exploring what influences diversity, dominance and cover. *Biodiversity and Conservation* 15, 921–938.

O'Hara, T.D., Tittensor, D.P., 2010. Environmental drivers of ophiuroid species richness on seamounts. *Marine Ecology - An Evolutionary Perspective* 31, 26e38.

Ortega-Álvarez, R., I. Macgregor-Fors. 2009. Living in the big city: Effects of urban land-use on bird community structure, diversity, and composition. *Landscape and Urban Planning* 90, 189–195.

Osborne L.L, Kovacic D.A. 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. *Freshwater Biology* 29.243–258.

Oxley D.J, Fenton M.B, Carmody G.R. 1974. The effects of roads on populations of small mammals. *Journal of Applied Ecology* 11,51–59.

Pardini, R., de Souza, S.M., Brago-Neto, R., Metzger, J.P. 2004.The role of forest structure, fragment size and corridors in maintaining small mammal abundance and diversity in an Atlantic forest landscape. *Biological Conservation.*,124, 253–266.

Parker, K., Head, L., Chisholm, L.A., Feneley, N., 2008.A conceptual model of ecological connectivity in the Shellharbour Local Government Area, New South Wales, Australia. *Landscape and Urban Planning* 86, 47–59.

Peakall, R., Ruibal, L. Lindenmayer, D. B. 2003 Spatial autocorrelation analysis offers new insights into gene flow in the Australian bush rat, *Rattus fuscipes*. *Evolution* 57, 1182–1195.

Pearson, D. E., and L. F. Ruggiero. 2003. Transect versus grid trapping arrangements for sampling small mammal communities. *Wildlife Society Bulletin* 31(2), 454–459.

Peng, J., Zhau, H., Liu, Y. 2017. Urban ecological corridor construction: a review. *ActaEcologicaSinica*37, 23–30.

Penteado, H.M., 2013. Assessing the effects of applying landscape ecological spatial concepts on future habitat quantity and quality in an urbanizing landscape. *Landscape Ecology* 28, 1909–1921.

Pfab, M. 2001. Draft Development guidelines for ridges. Gauteng Department of Agriculture and Rural Development.Directorate of Nature Conservation. South Africa.

Pfab, M.F., 2002, 'An integrative approach for the conservation and management of South Africa's floristic diversity at the provincial level'. *Biodiversity and Conservation* 11, 1195–1204.

Pickett, S. T. A., Cadenasso M. L., Grove J. M. 2004. Resilient cities: meaning, models, and metaphor for integrating the ecological, socio-economic, and planning realms. *Landscape and Urban Planning* 69, 369–84.

Pierik, M.E., Dell'Acqua, M., Confalonieri, R., Bocchi, S., Gomarasca, S., 2016. Designing ecological corridors in a fragmented landscape: A fuzzy approach to circuit connectivity analysis. *Ecological Indicators* 67, 807–820.

Polard, E., Relton, J. Hedges.V.1970. A study of small mammals in hedges and cultivated fields. *Journal of Applied Ecology* 7,549–557.

Primm, S.A. and Clark, T.W. 1996.Making sense of the policy process for carnivore conservation. *Conservation Biology* 10, 1036–1045.

Pryke SR, Samways MJ. 2001. Width of grassland linkages for the conservation of butterflies in South African afforested areas. *Biological Conservation* 101,85–96.

Qiu, L., Lindberg, S., Nielsen, A.B. 2013. Is biodiversity attractive?- value in urban green space. *Landscape and Urban Planning* 119, 136–146.

Ramalho, C.E., Hobbs, R.J., 2012. Time for a change: dynamic urban ecology. *Trends in Ecology and Evolution* 27, 179–188.

Ramesh, T., Kalle, R., Downs, C.T., 2016. Predictors of mammal species richness in KwaZulu-Natal, South Africa. *Ecological Indicators* 60, 385–393.

Redon (de), L., Le Viol, I., Jiguet, F., Machon, N., Scher, O., Kerbiriou, C., 2015. Road network in an agrarian landscape: Potential habitat, corridor or barrier for small mammals? *Acta Oecologica* 62, 58–65.

Reid, K., Flowers, P. & Larkin, M. 2005.Exploring the lived experience. *The Psychologist*, 18, 20–23

Resasco, J., Haddad, N.M., Orrock, J.L., Shoemaker, D., Brudvig, L.A., Damschen, E.I., Tewksbury, J.J., Levey, D.J., 2014. Landscape corridors can increase invasion by an exotic species and reduce diversity of native species. *Ecology* 95, 2033–2039.

Rico, A., Kindlmann, P., Sedlacek, F., 2009. Can the barrier effect of highways cause genetic

subdivision in small mammals? *Acta Theriologica* 54, 297–310.

Ricotta, C., Pavoine, S. 2015, A multiple-site dissimilarity measure for species presence/absence data and its relationship with nestedness and turnover. *Ecological Indicators* 54, 203–206.

Roberts, D.C. 1994. The design of an urban open-space network for the city of Durban (South Africa). *Environmental Conservation,* 21, 11–17.

Rosenzweig, C., Solecki, W., Hammerm, S.A., Mehrotra, S., 2010. Cities lead the way in climate-change action. *Nature* 467, 909–911.

Rudd, H., Vala, J., Schaefer, V. 2002. Importance of backyard habitat in a comprehensive biodiversity conservation strategy: a connectivity analysis of urban green spaces. *Restoration Ecology* 10(2),368–375.

Samways, M. and Hatton, M.2000.Palmnut Post, Vol 3, No 2, 9–11

Samways, M.J., Pryke, J.S., 2016. Large-scale ecological networks do work in an ecologically complex biodiversity hotspot. *Ambio* 45, 161–172.

Sandström, Ulf G., Angelstam, P., Khakee, A., 2006. Urban comprehensive planning – identifying barriers for the maintenance of functional habitat networks. *Landscape and Urban Planning* 75, 43–57.

Savard, J.-P.L., Clergeau, P., Mennechez, G., 2000. Biodiversity concepts and urban ecosystems. *Landscape and Urban Planning* 48, 131–142.

Savidge, I.R. 1973. A stream as a barrier to homing in *Peromyscus leucopus*. *Journal of Mammalogy*, 54, 982–984.

Shuttleworth, C.M., 2001. Traffic related mortality in a red squirrel (Sciurus vulgaris) population receiving supplemental feeding. *Urban Ecosyst*ems 5, 109–118.

Siemers, B.M., Schaub, A., 2011. Hunting at the highway: traffic noise reduces foraging efficiency in acoustic predators. *Proceedings of the Royal Society B*1712, 1646–1652.

Skinner, J.D., and Chimimba, C.T. 2005. *The Mammals of the Southern African Subregion*3rd ed. Cambridge University Press.

Soulé, M.E. and Gilpin, M.E. 1991.The theory of wildlife corridor capability. In: *Nature Conservation 2: The Role of Corridors* (eds D.A. Saunders and R.J. Hobbs), pp. 3–8. Surrey Beatty and Sons, Chipping Norton.

Spackman, S.C., Hughes, J.W. 1995. Assessment of minimum stream corridor width for biological conservation: species richness and distribution along mid-order streams in Vermont, USA. *Biological Conservation* 71,325–332.

Stats SA (Statistics South Africa). 2017. Poverty trends in South Africa: an examination of absolute poverty between 2006 and 2015. Pretoria 002.

Stats SA (Statistics of South Africa). 2018. If South Africa's provinces were independent states. http://www.statssa.gov.za/?p=11092

Steele, B.B., Bayn, R.L., Val Grant, C., 1984. Environmental monitoring using populations of birds and small mammals: Analyses of sampling effort. *Biological Conservation* 30, 157–172.

Tabacchi, E., Correll, D. L., Hauer, R.,Pinay, G., Planty-Tabacchi, A. M., Wissmar, R. 1998.Development, maintenance and role of riparian vegetation in the river landscape. *Freshwater Biology* 40,497–516.

Tait, C.J., Daniels, C.B., Hills, R.S. 2005. Changes in species assemblages within the Adelaide Metropolitan Area, Australia, 1836-2002. *Ecological Applications* 15, 346–359.

Theobald, D. M., Miller, J. R., Thompson Hobbs, N. 1997. Estimating the cumulative

effects of development on wildlife habitat. *Landscape and Urban Planning* 39,25–36.

Tigas, L.A., VanVuren, D. H., Sauvajot, R. M. 2002.Behavioural responses of bobcats and coyotes to habitat fragmentation and corridors in an urban environment. *Biological Conservation* 108, 299– 306

Tjallingii, S.P. 2000. Ecology on the Edge. *Landscape and Urban Planning* 48,103–119.

UNFPA (United Nations Population Fund).2007.*State of World Population 2007: Unleashing the Potential of Urban Growth*.Available at:www.unfpa.org/sites/default/files/pub-pdf/695_filename_sowp2007_eng.pdf. Accessed on September 22, 2018.

Veitch, C. R. 2002. Eradication of Pacific rats (*Rattusexulans*) from TiritiriMatangi Island, Hauraki Gulf, New Zealand. In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*, pp. 360-364. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Walker, R., L. Craighead. 1997. Least-Cost-Path Corridor Analysis: Analyzing Wildlife Movement Corridors in Montana Using GIS, Proceedings of the 1997 ESRI User's conference.

Walter Reid *et al*., *Ecosystems and Human Well-Being- Synthesis, in Millennium Ecosystem Assessment*. 2005. (Washington, DC: Island Press, 2005).

Ward, J.V., Tockner, K., Arscott, D.B., Claret, C. 2002. Riverine landscape diversity. *Freshwater Biology* 47,517–539

Watts, R.D., Compton, R.W., McCammon, J.H., Rich, C.L., Wright, S.W., Owens, T., Ouren, D.S., 2007. Roadless space of the conterminous United States. *Science* 316,736–738.

Weber, M. 2004. Philosophy of Experimental Biology. Cambridge University Press, Cambridge, UK.

Widdows, C.D., Downs, C.T. 2015. A genet drive-through: are large spotted genets using urban areas for “fast food”? A dietary analysis. *Urban Ecosystems* 18, 907–920.

Wiens, J.A. 1989. Spatial scaling in ecology. *Functional Ecology* 3(4), 385–397.

Wiens, J. A. 1997a. The emerging role of patchiness in conservation biology. -In: Pickett, S. T. A., Ostfeld, R. S., Shachak, M.,Likens, G. E. (eds), *The ecological basis of conservation: heterogeneity, ecosystems, and biodiversity*. Chapman and Hall, New York, pp. 93-107.

Wiens, J. A. 1997b. Metapopulation dynamics and landscape ecology. In: Hanski, I. A. and Gilpin, M. E. (eds), Metapopulation biology. ecology, genetics, and evolution. Academic Press, San Diego, CA, pp. 43–62.

Wiens, J. A., Stenseth, N. C., Van Home, B. and Ims, R. A. 1993. *Ecological mechanisms and landscape ecology*. Oikos 66, 369–380.

Wiens, J.A. 2002. Riverine landscapes: taking landscape ecology into the water. *Freshwater Biology* 47, 501–515.

Wiens, J.A., Milne, B.T. 1989. Scaling of ‘landscapes’ in landscape ecology, or landscape ecology from a beetle’s perspective. *Landscape Ecol*ogy 3, 87–96.

WorldAtlas. 2018. The largest provinces of South Africa by size and population. https://www.worldatlas.com/articles/the-largest-provinces-of-south-africa-by-area-and-by-population.html

Wu, J., 2008. Making the Case for Landscape Ecology: An Effective Approach to Urban Sustainability. *Landscape Journal* 27, 41–50.

Yates, C. J., McNeill, A,.Elith, J.,Midgley, G. F.2010. Assessing the impacts of climate change and land transformation on Banksia in the South West Australian Floristic Region. *Diversity and Distributions* 16,187–20.

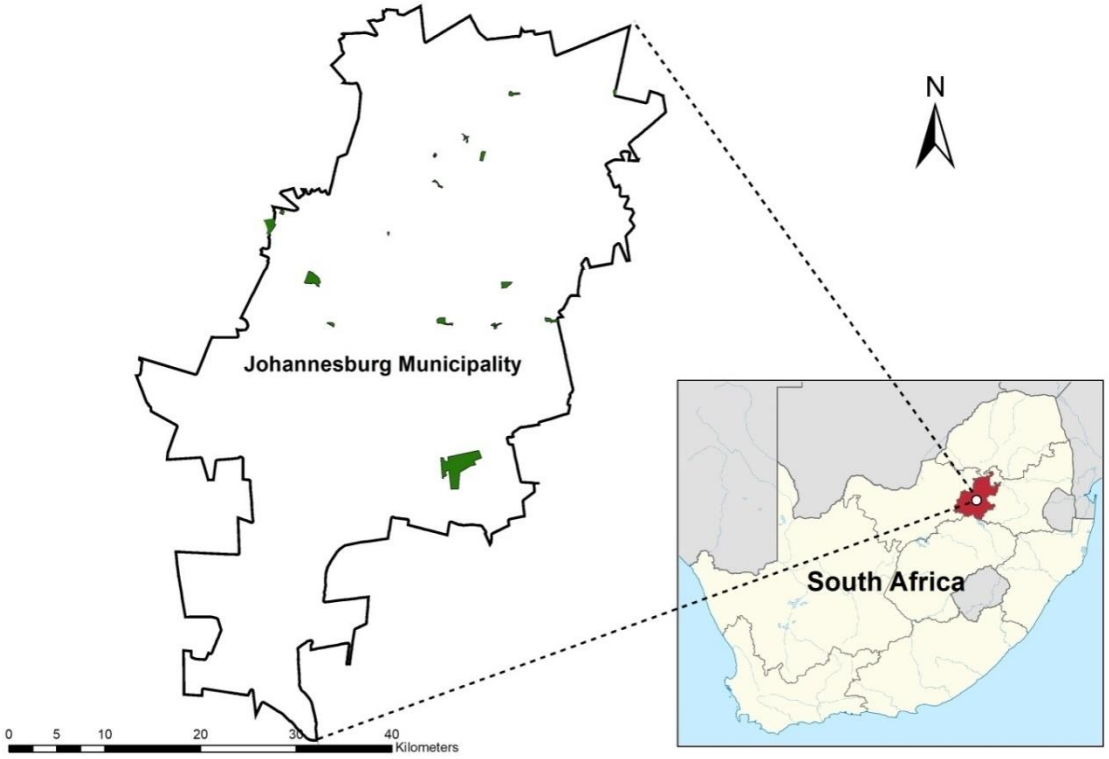
Young, J.C., Marzano, M., White, R.M., McCracken, D.I., Redpath, S.M., Carss, D.N. Quine, C., Watt. A.D. 2010. The emergence of biodiversity conflicts from biodiversity impacts: characteristics and management strategies. *BiodiversityConserv*ation 19 3973–3990

Yu, K. J., Li, W., Li, D. H., Li, C. B., Huang, G., Liu, H. L.2005. "Suitability analysis of heritage corridor in rapidly urbanizing region: a case study of Taizhou City", *Geographical Research*, Vol.24, No.1 (1).

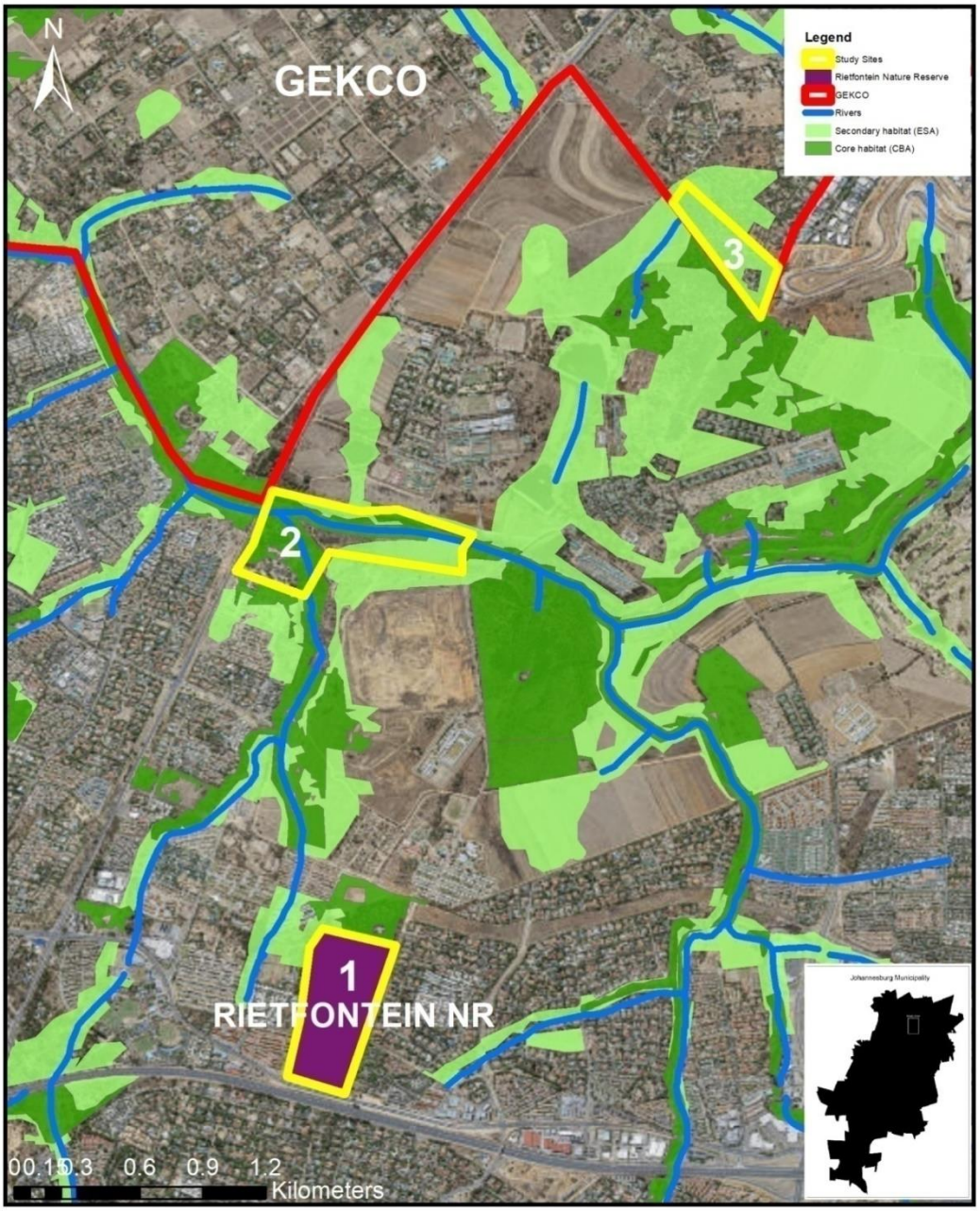
Zipperer, W.C., Morse, W.C., Gaither, J.G., 2011.Linking social and ecological systems. In: Niemelä, J., Breuste, J.H., Elmqvist, T., Guntenspergen, G., James, P., McIn-tyre, N.E. (Eds.), *Urban Ecology: Patterns, Processes, and Applications*. Oxford University Press, New York.

Appendices

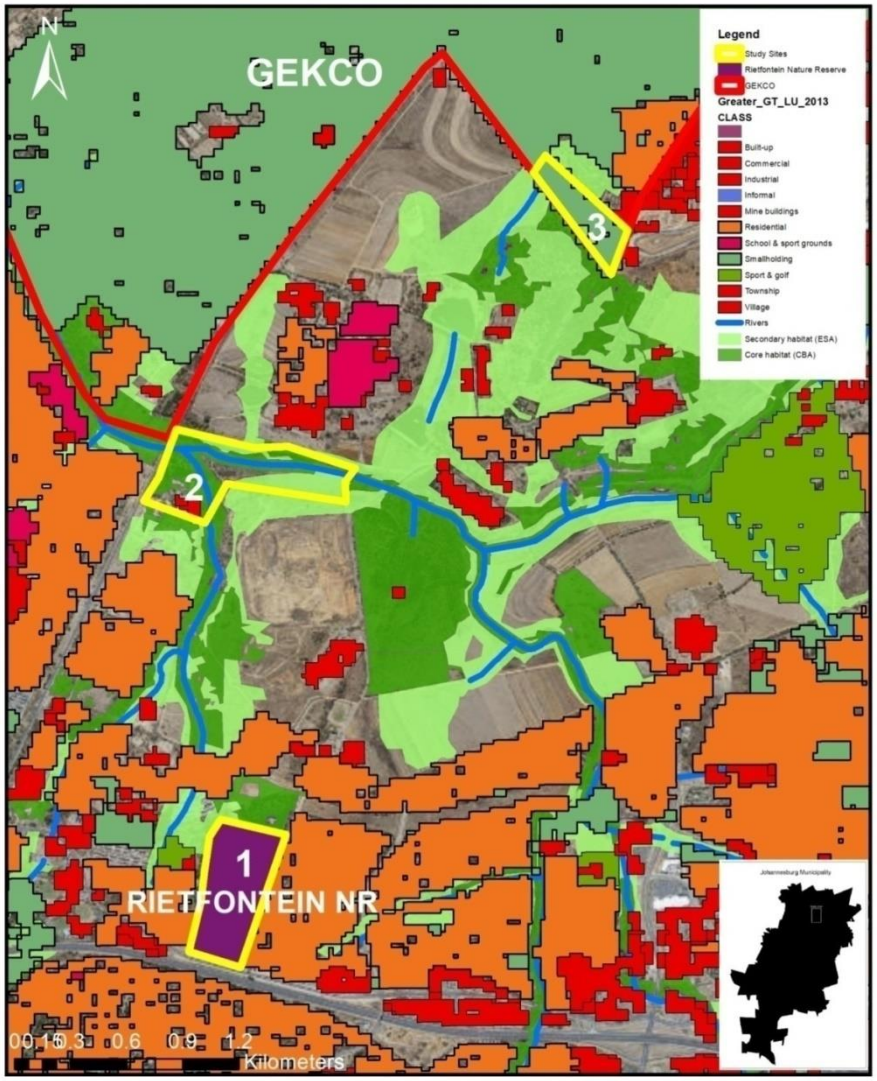
Appendix A: Map showing the Protected Areas within the Johannesburg Municipality, South Africa in 2017 (created in ESRI ArcMap 10.3.1)



Appendix B: Satellite image of the GEKCO-Rietfontein Nature Reserve and Johannesburg Bioregional layer showing the corridors(green) and study sites (yellow)(Created in ArcMap using C-Plan 3.3).



Appendix C: Satellite image showing corridors (green), study sites (yellow) and the surrounding land-use (Johannesburg Bioregion Plan, 2015; Greater GT Lu, 2013). (created in ESRI ArcMap 10.3.1)



Appendix D: Satellite images of the three study sites along the corridor in 2016



Fig A: Google Earth images showing study site 1, Rietfontein Nature Reserve in the north of Johannesburg, South Africa (Google Earth, 2015).



Fig B: Google Earth images showing study site 2, commercial operation in the north of Johannesburg, South Africa (Google Earth, 2015).



Fig C: Google Earth images showing study site 3, Private property in the north of Johannesburg, South Africa (Google Earth, 2015).

APPENDIX E: Complete plant species list

Trees and shrubs

Vacheliagalpinii
Vacheliasieberiana
Acokantheraoppositifolia
Afrocanthiumgilfillanii
Asparagus laricinus
Asparagus setaceus
Asparagus suaveolens
Brachylaenarotundata
Carissa bispinosa
Celtisafricana
*Cereus jamacaru**
Combretumerythrophyllum
*Cotoneaster franchetii**
Cussoniapaniculata
Cussoniaspicata
Dais continifolia
Diospyroslycoides
Dombeyarotundifolia
Ehritiarigida
Englerophytummagalismontanum
Erythrinalysistemon
Eucleacrispa
Gardenia sp.
Gymnosporiaheterophylla
Gymnosporiapolyacantha
*Melia azederach**
Munduleasericea

Mystroxylonaethiopicum subsp. burkeanum
Olea europaea
Pittosporum viridiflorum
Proteacaffra
Psydraxlivida
Searsialancea
Searsialeptodictya
Searsiapyroides
Tecoma capensis
Vanguariainfausta
Vangueriaparviflora
Xerophytaretinervis

Zanthoxylumcapense
Ziziphus mucronata

Forbs

*Agave americana**
*Agave sisalana**
*Ageratinaadenophora**
Aloe arborescens
Aloe greatheadii subsp. davyana
Aloe marlothii
Aloe zebrina
*Alternantherapungens**
Anthericumtransvaalense
Beciumobavatum
*Bidensbipinnata**
*Campulocliniummacrocephalum**
Cheilantheshirta
Chlorophytumcooperi
Commelinaafricana
Conyzapodocephala
Cyanotisspeciosa
*Datura stramonium**
Gnidiakraussiana
*Gomphrenacelosioides**
Helichrysumrugulosum
Hypoestusforskaolii
Hypoxishemerocallidea
Hypoxisiridifolia
Hypoxisrigidula
Indigoferaburkeana
*Ipomoea indica**

*Ipomoea purpurea**
*Lantana camara**
Ledebouriaovatifolia
Leucasmartinicensis
*Ligustrum sp.**
Lobiliaflaccida
Nidorellaanomala
Noletiararifolia
Oldenlandiaherbacia
Oxalis corniculata
Pellaeacalomelanos
*Podraneabrycei**
Polydoraposkeana

*Portulacaoleracea**
*Richardiabrasiliensis**
*Salvia tiliifolia**
Sansevieria sp.
Sarcostemmaviminale
Scadoxispuniceus
Schistostephiumheptalobum
*Solanum pseudocapsicum**
*Solanum seaforthianum**
*Tagetesminuta**
Viscumrotundifolium
*Zinnia peruviana**

Grasses

Aristidacongesta
Bewisiabiflora
Brachiariaserrata
Chloris pycnothrix
Chloris virgata
Cymbopogonexcavatus
Cymbopogonpospischilii
Cynodondactylon
Dactyloceteniumaegyptium
Digitariaeriantha
Diheteropogonamplectens
Eleusinecoracana
Eragrostiscurvula
Eragrostisracemosa
Eragrostissquarrosa
Hyparrheniahirta
Melinisnerviglumis
Melinisrepens
Panicummaximun
Paspalumnotatum
*Pennisetumclandestinum**
Pogonarthriasquarrosa
Schizachriumsanguineum
Setariapallide-fusca
Setariasphacelata
Sporobolusafricanus
Themedatriandra

Tragus berteronianus
Urochloamosambicensis
Urochloaoligotricha