Cape Peninsula University of Technology

MEDIUM AND LARGE MAMMAL COMMUNITY ASSEMBLAGES ACROSS CITY OF CAPE TOWN NATURE RESERVES

by

ANDREA KIM SCHNETLER

Thesis submitted in fulfilment of the requirements for the degree

Master of Conservation Science: Nature Conservation

in the Faculty of Applied Sciences

at the Cape Peninsula University of Technology

Supervisor: Dr F.G.T. Radloff
Co-supervisor: Prof M.J. O'Riain

Cape Town
November 2019

DECLARATION

I, Andrea Kim Schnetler, declare that the contents of this dissertation/thesis represent my own unaided work, and that the dissertation/thesis has not previously been submitted for academic examination towards any qualification. Furthermore, it represents my own opinions and not necessarily those of the Cape Peninsula University of Technology.

Signed

15 November 2019

Date

ABSTRACT

Urbanisation is associated with the loss and fragmentation of natural land, the disruption of ecosystem functioning and services, and the loss of biodiversity. Small remnants of natural land within cities not only serve as recreational green spaces that contribute to human wellbeing, but also as refugia for a variety of indigenous flora and fauna. While large mammal species, in particular those that pose a threat to humans and are rarely tolerated in urban reserves, small and medium mammals may persist and even thrive in human modified landscapes. Understanding which species survive best in urban protected areas and how reserve attributes such as size, shape and connectedness influence mammal assemblages and species richness is important for the conservation of urban ecosystems globally.

Cape Town is situated in the Cape Floristic Region (CFR) - a renowned biodiversity hotspot, with high rates of endemism. Cape Town is however one of the fastest growing cities in South Africa and both agricultural and housing demands are increasing pressure on remaining patches of natural land. Currently most of this land is conserved within 17 nature reserves that together comprise roughly 9% of the total surface area of the City of Cape Town (CCT) municipal area. Existing mammal species lists suggest that 22 mammal species still survive in these reserves but no formal, standardised surveys of the existing reserves have been conducted with a method that allows for comparisons between reserves and within reserves over time. The primary aim of this study was therefore to develop a standardised monitoring protocol for medium and large mammal species within the CCT reserves (range 30 - 8 400 ha). The secondary goal was to understand how reserve size, area to perimeter ratio, connectivity, vegetation heterogeneity and presence of permanent freshwater aquatic habitat might influence mammal community composition.

A standardised camera trap protocol was developed for the 12 CCT reserves larger than 30 ha and conducted from June 2017 to Feb 2019 with cameras positioned within every square kilometre of a reserve, with a minimum of five cameras per reserve irrespective of reserve size. Additional cameras were placed in unique habitat types not included or underrepresented in the standardised grid and a minimum of 1000 camera days of data were collected for each reserve. A total of 13 360 independent trigger events by medium and large mammals revealed 19 native species (11 carnivores, 7 herbivores, 1 omnivore), which was 86% of the 22 species listed in the databases (based on records of 2012 to 2017), and 49% of the 39 species believed to have been present historically. Species richness varied from 1 – 12 species (mean ± SD = 7±3.6) and Cape porcupine (*Hystrix africaeaustralis*), Cape grysbok (*Raphicerus melanotis*) and small grey mongoose (*Galerella pulverulenta*) were present in most reserves. The minimum survey effort required to effectively sample the

reserves varied from 210 to more than 1840 camera days and was affected by both reserve size and levels of connectivity. The use of camera traps with a placement protocol as used in this study together with the minimum camera day effort estimates presented for each reserve should allow for regular monitoring and provide comparable results.

Species richness was best explained by reserve area-perimeter ratio with richness lower in reserves with large perimeters relative to their total area. Large, better connected reserves also had higher species richness and included wide ranging large carnivores such as leopard (*Panthera pardus)*, while species with specialist habitat requirements such as otter (*Aonyx capensis*) were notably absent from reserves without the appropriate habitats. This study suggests that reductions in the size of existing CCT reserves and/or an increase in hard edges that reduce the core area may lower species richness and potentially drive more medium and large mammals to local extinctions. Extending existing reserves through the addition of core natural habitat and improved connectivity to tracts of natural land are both management interventions likely to maintain and improve the ability of urban reserves to sustain diverse, ecologically functional mammal assemblages.

ACKNOWLEDGEMENTS

I wish to thank:

- My supervisor, Dr Radloff, for the endless patience, knowledgeable guidance and insight, incredible support, trust and genuine interest for the duration of this study. I am extremely honoured and grateful for your time and effort spent assisting me with this project
- My co-supervisor, Prof O'Riain, for the time, guidance, knowledge and insight spent on this project, as well as the generous loan of equipment and workspace
- The City of Cape Town (CCT) Biodiversity Management Branch, for their permission and trust in allowing me to conduct research within the various CCT nature reserves and the incredibly generous and enthusiastic logistics support, use of equipment and invaluable field knowledge of all staff and students who assisted with this research
- Leighan Mossop, for organizing of equipment as well as the endless assistance, patience, trust and support provided throughout this project
- NCC Environmental Services, for their generous loan of cameras
- The Friends of Helderberg Nature Reserve, for their generous funding of batteries and other equipment that allowed me to start this project, and Owen Wittridge for facilitating this
- Zoë Woodgate, for the charger loans, fieldwork tips, data analysis recommendations and software support
- Susan Schnetler and Kyle Niddrie, for their incredible moral and financial support, confidence, understanding, consideration and encouragement that allowed me to devote time to this project

The financial assistance of the National Research Foundation towards this research is acknowledged. Opinions expressed in this thesis and the conclusions arrived at, are those of the author, and are not necessarily to be attributed to the National Research Foundation.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

APPENDICES

GLOSSARY

Acronyms/Abbreviations

AIC	Akaike’s Information Criterion
CCT	City of Cape Town
CFR	Cape Floristic Region
ha	Hectares
ICE	Incidence Coverage Estimator
km	Kilometres
NMDS	Non-metric Multidimensional Scaling
SD	Standard deviation
spp.	Species (plural)
VIF	Variance Inflation Factor

CHAPTER ONE:
GENERAL INTRODUCTION

1.1 Urbanisation and ecosystem functioning

Urban development is increasing as the human population and rural-urban migration increase globally (Van der Ree & McCarthy 2005, McDonald et al. 2008). Urban areas are expected to account for more than 1 000 000 km^2 globally by 2025 (McDonald et al. 2008) and more than 60% of the projected 8.1 billion humans are expected to be living within urban settlements by the year 2030 (McDonald et al. 2008, Pickett et al. 2011). Shochat et al. (2006) defines urban areas as those consisting of built structures at a density of more than 10 buildings per hectare, and indices of urbanisation (or the extent of urban development) are often characterised by human population density, building density, hard surfaces, road density or time since development (Pickett et al. 2011).

Urbanisation brings with it the transformation of natural land (Pickett et al. 2011) which may otherwise provide a multitude of ecosystem services for human populations (Anderson & O'Farrell 2012). More particularly, urbanisation results in highly fragmented landscapes, leaving isolated remnants of natural land in a matrix of human land uses (McKinney 2002, De Stefano & De Graaf 2003, Rebelo et al. 2011, Pickett et al. 2011, Ramesh et al. 2016). These natural land remnants not only serve as urban green spaces and contribute to human wellbeing (Anderson & O'Farrell 2012, O'Farrell et al. 2012, Cheesbrough et al. 2019), but may also serve as refugia for remaining wildlife (De Stefano & De Graaf 2003, Hobbs & Mooney 2008, Šálek et al. 2015). The biodiversity remaining in the urban matrix requires active and effective conservation efforts for some semblance of ecosystem function and services to remain and be sustainable, especially as an increase in urban development also has indirect impacts such as increases in pollution, natural resource use, human-wildlife contact and disease exposure (Ceballos & Ehrlich 2006, McDonald et al. 2008, Ordeñana et al. 2010, Visconti et al. 2011).

Because of the rate of urban development, it is important to understand the impact urbanisation has on remaining local biodiversity and the challenges it poses for conservation efforts (McDonald et al. 2008, Pickett et al. 2011, Anderson & O'Farrell 2012). It is of particular importance as human settlements tend to exist in areas with high biodiversity and endemism (Garden et al. 2006, McDonald et al. 2008). Research on the effects of urbanisation on species richness and ecosystem functioning and how to conserve these attributes is not relatively comprehensive (Van der Ree 2004, Garden et al. 2006, Anderson & O'Farrell 2012, Torres-Romero & Olalla-Tárraga 2015). The majority of existing urban ecology studies comprise of either species-specific investigations or focus on a particular management questions. The results tend to be descriptive rather than explorative of ecological or urban processes (Van der Ree 2004, Shochat et al. 2006, Pickett et al. 2011, Saito & Koike 2013). Despite these shortcomings, clear trends are already evident.

Pickett et al. (2011) found evidence for emerging ecological “urban syndromes” in which the effects of urbanisation on soil, water and wildlife dynamics are producing homogenised ecosystems. Disturbance, pollution and habitat transformation affect the soil, vegetation and water condition in fragments and alter ecological processes that may be supported within them. For indigenous fauna specifically, habitat fragmentation, disturbance and isolation because of urbanisation have been shown to have significant direct impacts on species richness and persistence (Ceballos et al. 2005, Van der Ree & McCarthy 2005, Ceballos & Ehrlich 2006, Garden et al. 2006, Visconti et al. 2011, Pickett et al. 2011, Pekin & Pijanowski 2012). Most faunal taxa show a negative relationship with increased urban densities (Garden et al. 2006, Pickett et al. 2011), with local extinction most prevalent in highly urbanised areas (Van der Ree & McCarthy 2005, Torres-Romero & Olalla-Tárraga 2015).

The fauna which seem to be most affected by urbanisation are most often endemic, habitat specialists and rare species (McDonald et al. 2008, Clavel et al. 2011, Pickett et al. 2011), or species reliant on successful dispersal (Pickett et al. 2011, Correa Ayram et al. 2016). Conversely species that are adaptable, generalists, have high reproductive rates, or are not reliant on large home ranges or natural habitat for movement, tend to thrive (McKinney 2002, De Stefano & De Graaf 2003, Garden et al. 2006, Baker & Harris 2007, Ordeñana et al. 2010, Pickett et al. 2011, Lowry et al. 2013, Šálek et al. 2015). For this reason, exotic, domestic and invasive species often become prevalent in urban land use zones (Pickett et al. 2011, Saito & Koike 2013). Either way, species richness generally tends to be reduced as homogenisation of community composition occurs (Clavel et al. 2011, Pickett et al. 2011, Torres-Romero & Olalla-Tárraga 2015).

Urbanisation also tends to affect trophic level dynamics within the urban environment (Pickett et al. 2011, Saito & Koike 2013). Human activity often enhances plant productivity and either deliberately (e.g. bird feeders) or indirectly (e.g. waste) supplements food sources (Pickett et al. 2011, Saito & Koike 2013). This may increase the abundance of certain species of arthropods, birds and small mammals, while at the same time other activities including persecution, pollution and poisons may firstly eliminate naturally occurring apex predators and subsequently replace them with human induced mortality (De Stefano & De Graaf 2003, Picket et al. 2011). Even detritivore composition can be different in urban environments, as leaf litter and detritus are often actively managed in gardens and public open spaces (Pickett et al. 2011). Fischer et al. (2012) describes a commonly found human-altered trophic structure resulting in what has been termed a “predation paradox”, where predator numbers increase with urbanisation but predation rates themselves decline. It is suggested that this is due to availability of anthropogenic food sources, which leads to less pressure from apex predators on prey species (Shochat et al. 2006, Faeth et al. 2005, Fischer et al. 2012, Saito & Koike 2013).

1.2 Urban mammal ecology

Traditional conservation efforts for mammal species have focused largely on establishing protected areas in which species are ostensibly protected from anthropogenic influences (McDonald et al. 2008). However, in urban areas, natural environments tend to be reduced to small, fragmented and often isolated pockets which, owing to edge effects, cannot remain unaffected by anthropogenic activity. Research shows how mammal species are becoming increasingly exposed to anthropogenic impacts due to urbanisation, often with detrimental consequences (De Stefano & De Graaf 2003, Ceballos et al. 2005, Van der Ree & McCarthy 2005, Pekin & Pijanowski 2012, Ceballos & Ehrlich 2006, Visconti et al. 2011).

Mammal species richness in urban fragments is known to be influenced by a number of physical characteristics, namely fragment size (Diamond 1975, De Stefano & De Graaf 2003, Kerley et al. 2003, Ceballos et al. 2005, Visconti et al. 2011, Matthies et al. 2017, Gonçalves et al. 2018), fragment shape (Diamond 1975), habitat heterogeneity (Ramesh et al. 2016, Matthies et al. 2017), connectivity to additional suitable habitat (Diamond 1975, Stevens et al. 2006, Correa Ayram et al. 2016), and surrounding land use and/or proximity to human activity (De Stefano & De Graaf 2003, Ceballos et al. 2005, McDonald et al. 2008, Visconti et al. 2011, Pekin & Pijanowski 2012, Mann et al. 2015, Torres-Romero & Olalla-Tárraga 2015, Gonçalves et al. 2018). In general the probability of a mammal species becoming endangered generally increases as the proportion of urban area within its distribution range grows, but as is the trend in other taxa, this exposure affects some species more negatively than others (Pickett et al. 2011, Pekin & Pijanowski 2012, Saito & Koike 2013).

The loss of specialist species in urban environments results in a change of community composition which has knock-on effects on trophic level interactions and ecosystem functioning (Saito & Koike 2013). Mammal species are important role players in ecosystem functioning and biodiversity maintenance (Kerley et al. 2003, De Stefano & De Graaf 2003, Ceballos et al. 2005, Visconti et al. 2011). For example, carnivorous mammals, particularly apex predators, control lower trophic level dynamics (Kerley et al. 2003, Ordeñana et al.

2010, Bateman & Fleming 2012), while herbivores influence a number of ecosystem functions through actions such as herbivory, trampling and seed dispersal (Kerley et al. 2003). Actions such as these affect plant and animal community dynamics and, ultimately, biodiversity (Augustine & McNaughton 1998, Kerley et al. 2003). Similarly, mammal species richness can be an indicator of overall ecosystem health and monitoring the effects urbanisation has on mammal species is thus important for the conservation and management of the mammal species as well as the ecosystems supporting them (Van der Ree 2004, Anderson & O'Farrell 2012, Fischer et al. 2012, Saito & Koike 2013).

1.3 Mammal conservation in the City of Cape Town

Within the Cape Floristic Region (CFR) in South Africa, conservation is met with a unique challenge. The CFR is a renowned biodiversity hotspot, hosting the Fynbos Biome and with it a wide variety of vegetation types and high rates of endemism (Kerley et al. 2003, Rebelo et al. 2011, Pressey et al. 2003). While scientific literature largely focuses on the CFR's floral diversity, the area also contains a significant diversity of fauna species (Boshoff et al. 2001, Kerley et al. 2003, Pressey et al. 2003). The CFR has been subject to variety of anthropogenic land use practices, with nearly 26 % of the CFR having been transformed for cultivated land alone (Rouget et al. 2003). According to Underwood et al. (2009), the CFR experiences the second highest population growth rate within the global Mediterranean biome (areas with cool, wet winters and dry, warm summers).

The City of Cape Town (CCT) municipal area covers 2 461 km^2 of the CFR (Rebelo et al. 2011) and currently has over 4 000 000 residents (Small 2017). The area's Mediterranean climate, natural fire regimes and high soil diversity support a high diversity of plant species, and in turn high faunal diversity, although sandy, nutrient-poor soils prevent high productivity (Rebelo et al. 2006). The extraordinary wealth of biodiversity in the CCT area was highlighted in April 2019 when citizen science aided in the recording of 4 157 individual fauna, flora and fungal species over four days to win the iNaturalist City Nature Challenge (iNaturalist Network 2019). However, as a coastal city with limited space for expansion to meet housing demands, development pressure is increasing on natural and agricultural land (Anderson & O'Farrell 2012, Holmes et al. 2012). Mountains to the west and east of the CCT area confine the urban settlement to the coasts and central lowlands (Anderson & O'Farrell 2012), which has resulted in a highly fragmented urban matrix (Figure 1.1). These factors accentuate the threat that increased urbanisation may have on the fragmentation and loss of natural land and with that the potential loss of mammal species.

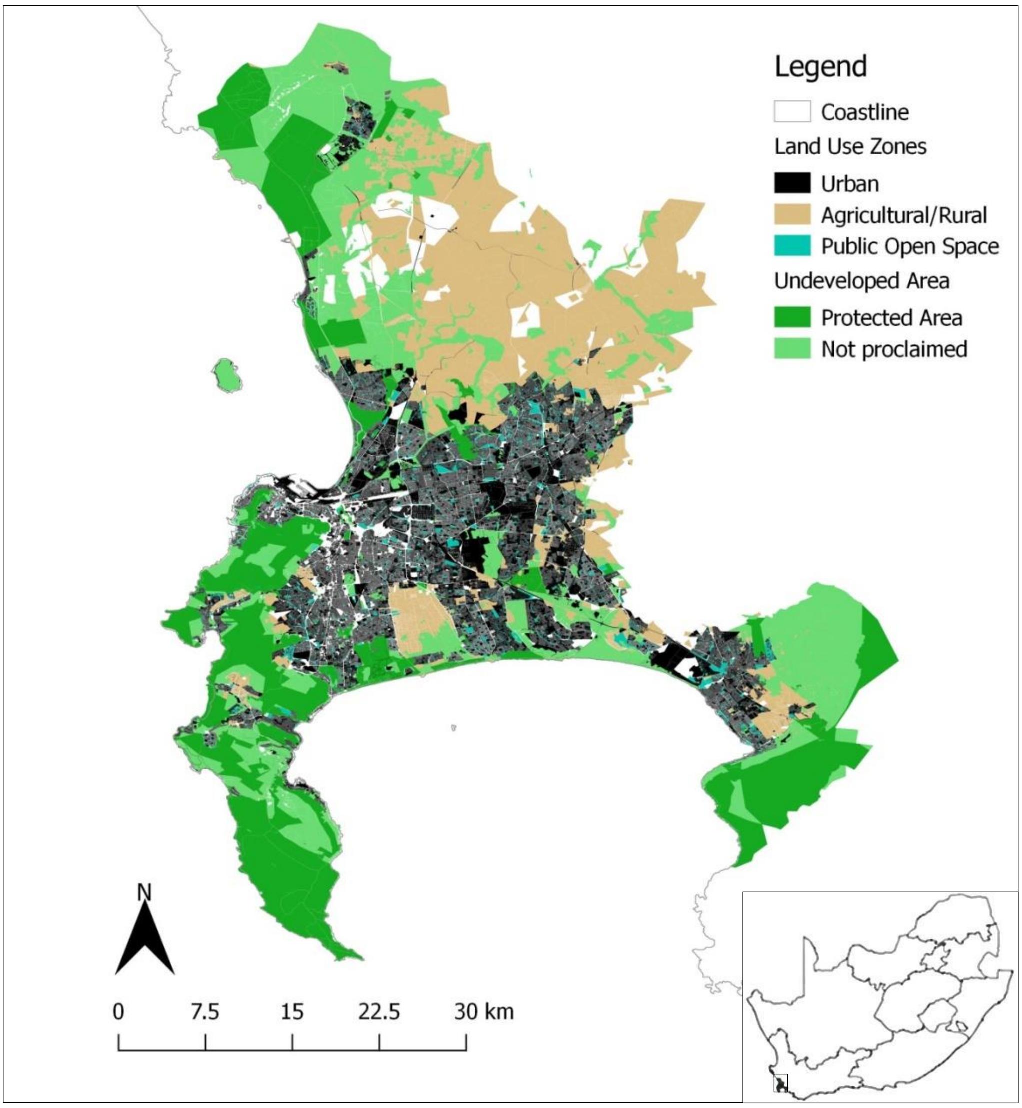


Figure 1.1: Land use zones and undeveloped areas within the City of Cape Town municipal area. "Protect areas" refers to formally protected conservation areas. Areas "not proclaimed" refer to open spaces, some of which may be managed as conservation areas but which are not protected by any formal legislation, and may include private property. (Adapted from City of Cape Town 2019a & 2019b)

Historically, 25 different vegetation types occurred within the boundaries of the CCT municipal area (Rebelo et al. 2006). Of these, 10 vegetation types are classified as Critically Endangered, four Endangered and four Vulnerable (Rebelo et al. 2011). All 25 vegetation types still occur within the CCT area, but in extremely fragmented and diminished ranges (Rebelo et al. 2011). Although a large area of land was initially transformed due to agriculture (Anderson & O'Farrell 2012), a rapid increase in urban development in the second half of 20th century left just under 40% of Cape Town's municipal area untransformed of which only 17.7 % is managed for conservation (Rebelo et al. 2011). Covering most of the peninsula, Table Mountain National Park makes up just over half of the protected land, and another quarter comprises of land within in the Kogelberg mountain range on the municipality's eastern boundary (Rebelo et al. 2011). However, very little of the remaining lowland area which comprises a number of highly vulnerable vegetation types (Rebelo et al. 2006) is conserved. Ensuring the integrity and persistence of the remaining vegetation types and their associated ecosystem processes will rely not only on conservation of flora but associated fauna as well (Rebelo et al. 2011). Many endemic plant species are reliant on very specific insect, bird and small mammals for pollination and seed dispersal (Rebelo et al. 2006, Biccard & Midgley 2009, Pauw & Hawkins 2011, Pauw & Louw 2012). Herbivores control bush encroachment and allow for nutrient cycling and seed dispersal, while predators control herbivore populations to prevent overgrazing and loss of vulnerable plant species (Rebelo et al. 2011).

Since the settlement of the Dutch in 1652, mammal species in what is now the Cape Town area have been directly impacted by human activity (Rebelo et al. 2011, Anderson & O'Farrell 2012). A total of 41 medium (> 0.5 kg) and large mammal species are thought to have occurred historically within the larger Cape Town area (Boshoff & Kerley 2001, Kerley et al. 2003), but many of the large carnivore and herbivore species such as lion (*Panthera leo*), black rhinoceros (*Diceros bicornis bicornis*) and eland (*Tragelaphus oryx*) were hunted to local extinction by the beginning of the 18th century (Rebelo 1992, Anderson & O'Farrell 2012). Wild animals and dangerous game in particular were seen as a threat to settler safety, and by 1656 large carnivores were actively exterminated in the area (Anderson & O'Farrell 2012). Hunting of large herbivores was also considered a pleasurable pastime and not only for sustenance (Rebelo et al. 2011, Anderson & O'Farrell 2012).

Urbanisation and fragmentation have further threatened mammal species with extinction (Rebelo 1992). In an effort to conserve the remaining biodiversity the City of Cape Town manages 17 protected areas within its boundaries. However, the majority of these conservation areas cannot support large mammals such as African elephant (*Loxodonta africanus*), black rhino and lion (Rebelo et al. 2011). Some existing conservation areas have attempted to reintroduce medium and large mammal species such as hippopotamus (*Hippopotamus amphibius*) that were reintroduced to the Rondevlei section of False Bay Nature Reserve in 1981 and still survive as a managed population in the reserve (Rebelo et al. 2011). Grey rhebuck (*Pelea capreolus*) were reintroduced to Tygerberg and Helderberg Nature Reserves, but there is uncertainty as to whether any of the released individuals remain. Eland and red hartebeest (*Alcelaphus buselaphus caama*) were successfully released into Blaauwberg Nature Reserve in 2016, while Cape grysbok (*Raphicerus melanotis*) was reintroduced to Kenilworth Racecourse Conservation Area where a viable population now thrives.

Direct impacts of urbanisation on the remaining fragments of natural land within the CCT metropole include fire regime changes (through absence of natural fires or increase in human-ignited fires), pollution, alien invasive vegetation and fauna, poaching (flora and fauna) and increased human presence within fragments (Rebelo et al. 2011). Additional factors such as degree of isolation/connectivity to other natural areas, area size and degrees of disturbance by invasive alien plants and human activity can also contribute to mammal species persistence risk within the reserves (Diamond 1975, De Stefano & De Graaf 2003, Garden et al. 2006, Stevens et al. 2006, Turgeon & Kramer 2012, Correa Ayram et al. 2016, Matthies et al. 2017). A Biodiversity Network of conservation land has been formed through the aggregation of protected areas, conservation management areas and undeveloped land earmarked for conservation (Holmes et al. 2012, City of Cape Town 2019b), with the intention of improving habitat availability and connectivity and thus biodiversity, ecosystem services and human wellbeing (Holmes et al. 2012, O'Farrell et al. 2012). Although this has allowed for flora conservation (Rebelo et al. 2011), the progressive efficacy of this in terms of mammal species conservation has not been determined, nor have the impacts of potential urban drivers been studied. As a result, it is not certain as to whether mammal species richness is being maintained by protected areas, whether any local extinctions are imminent, or even which species may actually be present across the area.

Garden et al. (2006) indicated that knowledge on urban fauna, that is necessary to inform conservation management, can be limited by a lack of multispecies studies across multiple ecological levels over time, and this is evidently applicable to the CCT area. Studies have been done on particular mammal species within the municipality, but have not been repeated regularly, if at all, and multispecies impact studies are generally lacking (De Stefano & De Graaf 2003, Kerley et al. 2003, Garden et al. 2006, Cilliers & Siebert 2012). Historical data on the general occurrence of species across the Cape Town area exist, but current species lists tend to be inconsistent or based largely on opportunistic sightings (Boshoff & Kerley 2001, Garden et al. 2006, O'Brien 2008). Species lists used by reserve management are based on the City of Cape Town Biodiversity Database, and are supplemented by iNaturalist (https://www.inaturalist.org) and iSpot (https://www.ispotnature.org) citizen science databases, as well as anecdotal records from reserve managers and staff, and so cannot be viewed as standardised sampling effort across reserves.

To effectively conserve the remaining wildlife and associated ecosystem processes in the conservation areas of the CCT, up-to-date species lists are needed and standardised monitoring protocols employed to allow for comparisons between reserves and within reserves over time. The need for an appraisal of all remaining wildlife species is great, but as discussed, even larger mammal species have not been accurately recorded. This study will focus on identifying the medium and large mammal species in CCT nature reserves with the use of camera traps.

1.4 Camera traps in mammal research

A number of medium and large mammal species surveying techniques are available to researchers and reserve managers. These include line transects, drive counts, night counts, sign surveys and live trapping (Munari et al. 2011). The use and efficacy of each of these techniques largely depends on the target species, survey aims and financial and/or time constraints (Munari et al. 2011). Methods can be biased and/or largely inaccurate if used inappropriately (Van der Ree & McCarthy 2005, Giman et al. 2007). Techniques that rely on observer skill such as line transects and night counts can be perceived as cost effective, but often lead to inaccurate estimates of species occurrence, in that density of vegetation, poor visibility, human presence and/or observer bias may skew results (Munari et al. 2011).

Remote-sensing camera traps are becoming increasingly popular as a survey tool, and have the potential to accurately record medium and large mammal species richness, diversity, abundance and behaviour (Kelly 2008, Rowcliffe et al. 2008, Tobler et al. 2008, Ordeñana et al. 2010, Rovero et al. 2010, Colyn et al. 2017). Being largely undetectable, camera traps allow for non-invasive surveying which increases the likelihood of recording evasive, nocturnal and/or rare species (Rowcliffe et al. 2008, Tobler et al. 2008, Ordeñana et al. 2010, Rovero et al. 2010, Si et al. 2014).

There is increasing evidence showing that when studying species richness in an area, the number of camera days sampled (number of cameras multiplied by survey period) are more important for accurate results than camera spacing or density (Kelly 2008, Tobler et al. 2008, O'Brien 2008, Si et al. 2014, Colyn et al. 2017). This means that surveys can be conducted over short periods of time, rather than the years required for accurate species list compilation through human observation only (Kelly 2008, O'Brien 2008). It seems that the majority of studies record 80-90% of estimated number of species occupying an area within 900 to 1 500 camera days (Giman et al. 2007, Tobler et al. 2008, Si et al. 2014), although this seems to vary with habitat type and number of rare/elusive species. For example, Tobler et al. (2008) found that they required 2 340 camera days to record 86% of species in Peruvian forest areas, but Trolle and Kery (2005) found that a section of the Pantanal wetland area in Brazil had been sufficiently surveyed within only 504 camera days.

A recent study conducted in the Fynbos shrubland of the Cape Peninsula indicated that more than 90% of species can be detected after approximately 1 000 days (Colyn et al. 2017). Si et al. (2014) suggests that for smaller areas, a higher number of cameras can be used over a shorter period of time, as they found that the optimal sampling period for an individual camera to detect an accurate diversity of species was approximately 40 days. This significantly lowers survey effort, which is important when studies need to be repeated regularly and cost-effectively (MacKenzie 2005), as would be the case in areas of rapid urbanisation.

Stratified grids of cameras tend to produce the most reliable data which then can also be used to inform distribution and/or occupancy models (O'Brien 2008). Cameras can be placed to target specific species, but biased estimates may be produced (Tobler et al. 2008). It is advised that for areas which cover more than one major habitat type, all habitat types should be included in the survey to account for habitat specialists (Tobler et al. 2008, O'Brien 2008). When surveying for medium to large mammals, Kelly (2008) suggests fixing cameras at a height of 20 – 30 cm from ground level in order to maximise detectability for the range in sizes.

Mammal species lists compiled from camera trap data can be assessed in conjunction with spatial data such as fragment size, connectivity, land use practices, disturbance and vegetation to identify associations between mammal species presence, abundance and diversity on an urban landscape scale (Ordeñana et al. 2010, Rovero et al. 2010). This information can then be used to better streamline conservation efforts on a site-specific and landscape-scale. For example, Cowling et al. (2003) suggest that the best way forward for the conservation of populations within the CFR at a landscape level is a network of connected protected areas, restorable habitat and habitat remnants. To do this one would be required to motivate for land acquisition and rezoning at an administrative level, which would require an understanding of the effects urbanisation has on mammal community assemblages and the best ways to mitigate these effects.

1.5 Research problem statement

The current status of medium and large mammal species within CCT reserves is unknown due to a lack of a reliable, standardised monitoring protocol. As discussed, the rapid growth of the human population and associated expansion of urban areas within the CCT area is a threat to the persistence of the remaining medium and large mammal species. To effectively conserve the mammal species and the ecosystems processes they associate with in the reserves, their current status needs to be determined and the drivers of species richness patterns understood.

1.6 Study objectives

The aim of this study was to determine the medium and large mammal species assemblages within the respective CCT reserves and to identify the potential drivers responsible for the expected differences in mammal community composition. The specific objectives were:

1. To determine which medium and large mammal species communities are still present in the City of Cape Town nature reserves that are larger than 30 ha and compare the results to historic and current species lists

2. To establish a camera trapping protocol that can effectively record and monitor medium and large mammal community composition in the CCT nature reserves with the minimum effort possible.

3. To identify potential drivers of species composition in the CCT nature reserves

1.7 Structure of the thesis

This thesis consists of five chapters. Chapters 3 and 4 are data chapters compiled as stand-alone manuscripts to facilitate publication in peer-reviewed journals.

Chapter 2 provides descriptions on the location, climate, topography and vegetation of the general City of Cape Town (CCT) municipal area and is followed by site descriptions of all the CCT nature reserves surveyed for this study.

In Chapter 3, historical and presumed current medium and large mammal species lists are compiled and compared with data obtained from the camera trap surveys conducted as part of this study in twelve CCT nature reserves larger than 30 ha. Minimum camera trap survey effort for the placement protocol used is determined with the help of species accumulation and species richness estimation curves.

Chapter 4 compares the species richness estimates of the respective reserves with the reserves' size, area-perimeter ratio, vegetation heterogeneity, presence of permanent freshwater aquatic habitat and connectivity to identify potential species richness drivers. Linear models are used to determine which covariates best describe species richness patterns.

Chapter 5 aims to consolidate the implications of the previous chapters for management action. A standardized monitoring protocol for medium and large mammals in the CCT reserves is discussed and suggestions made on how to best conserve the remaining medium and large mammal species. Aspects in need of further research are also identified.

1.8 Permits and ethical considerations

This study was conducted with the written permission of the City of Cape Town Biodiversity Management Branch and authorised under CapeNature permit number 0052-AAA041-00019. Data were collected using camera traps only, which were set up in consultation with reserve managers and staff and not placed in sensitive areas. The Bushnell infrared camera traps used emit only infrared light and are considered a minimally invasive survey technique and less detectable by animals than camera traps with white flash (Rovero et al. 2013, Caravaggi et al. 2017). No physical contact was made with fauna and plants were only trimmed, not removed, where necessary for camera placement.

1.9 References

Anderson, P.M.L. & O'Farrell, P.J.O. 2012. An ecological view of the history of the City of Cape Town. *Ecology and Society*, 17(3): 28.

Augustine, D.J. & McNaughton, S.J. 1998. Ungulate effects on the functional species composition of plant communities: herbivore selectivity and plant tolerance. *The Journal of Wildlife Management*, 62(4): 1165-1183.

Baker, P.J. & Harris, S. 2007. Urban mammals: What does the future hold? An analysis of the factors affecting patterns of use of residential gardens in Great Britain. *Mammal Review,* 37(4): 297–315.

Bateman, P.W. & Fleming, P.A. 2012. Big city life: Carnivores in urban environments. *Journal of Zoology*, 287(1): 1–23.

Biccard, A. & Midgley, J.J. 2009. Rodent pollination in *Protea nana*. *South African Journal of Botany*, 75(4): 720–725.

Boshoff, A.F. & Kerley, G.I.H. 2001. Potential distributions of the medium- to large-sized mammals in the Cape Floristic Region, based on historical accounts and habitat requirements. *African Zoology*, 36(2): 245–273.

Boshoff, A.F., Kerley, G.I.H. & Cowling, R.M. 2001. A Pragmatic Approach to Estimating the Distributions and Spatial Requirements of the Medium- to Large-Sized Mammals in the Cape Floristic Region, South Africa. *Diversity and Distributions*, 7(1): 29–43.

Caravaggi, A., Banks, P.B., Burton, A.C., Finlay, C.M.V., Haswell, P.M., Hayward, M.W., Rowcliffe, M.J. & Wood, M.D. 2017. A review of camera trapping for conservation behaviour research. *Remote Sensing in Ecology and Conservation*, 3(3): 109–122.

Ceballos, G. & Ehrlich, P.R. 2006. Global mammal distributions, biodiversity hotspots, and conservation. *Proceedings of the National Academy of Sciences*, 103(51): 19374–19379.

Ceballos, G., Ehrlich, P.R., Soberon, J., Salazar, I. & Fay, J.P. 2005. Global Mammal Conservation: What Must We Manage? *Science*, 309(5734): 603–607.

Cheesbrough, A.E., Garvin, T. & Nykiforuk, C.I.J. 2019. Everyday wild: Urban natural areas, health, and well-being. *Health and Place*, 56(January): 43–52.

Cilliers, S.S. & Siebert, S. 2012. Urban Ecology in Cape Town: South African Comparisons and Reflections. *Ecology and Society*, 17(3): 33.

City of Cape Town (CCT) 2019a. *Open Data Portal: Integrated Zoning Land Parcel*. https://web1.capetown.gov.za/web1/OpenDataPortal/DatasetDetail?DatasetName=Integrated%20zoning%20land%20parcel [4 April 2019]

City of Cape Town (CCT). 2019b. *Open Data Portal: Terrestrial Biodiversity Network*. http://odp.capetown.gov.za/datasets/4f2d7835518a4e6b8205ce12d77ff463_64 [4 April 2019]

Clavel, J., Julliard, R. & Devictor, V. 2011. Worldwide decline of specialist species: Toward a global functional homogenization? *Frontiers in Ecology and the Environment*, 9(4): 222–228.

Colyn, R, Radloff, F.G.T. & O'Riain, M.J. 2017. Camera trapping mammals in the scrubland's of the Cape Floristic Kingdom - the importance of effort, spacing and trap placement. *Biodiversity and Conservation*, 27(2): 503-520.

Correa Ayram, C.A., Mendoza, M.E., Etter, A. & Salicrup, D.R.P. 2016. Habitat connectivity in biodiversity conservation: A review of recent studies and applications. *Progress in Physical Geography*, 40(1): 7–37.

Cowling, R.M., Pressey, R.L., Rouget, M. & Lombard, A. T. 2003. A conservation plan for a global biodiversity hotspot - The Cape Floristic Region, South Africa. *Biological Conservation*, 112(1-2): 191–216.

De Stefano, S. & De Graaf, R.M. 2003. Exploring the ecology of suburban wildlife. *Frontiers in Ecology and the Environment*, 1(2): 95–101.

Diamond, J.M. 1975. The island dilemma: Lessons of modern biogeographic studies for the design of natural reserves. *Biological Conservation*, 7(2): 129–146.

Faeth, S.H., Warren, P.S., Shochat, E. & Marussich, W.A. 2005. Trophic dynamics in urban communities. *BioScience*, 55(5): 399-407

Fischer, J.D., Cleeton, S.H., Lyons, T.P., Miller, J.R., Fischer, J.D., Cleeton, S.H. & Timothy, P. 2012. Urbanization and the Predation Paradox: The Role of Trophic Dynamics in Structuring Vertebrate Communities. *BioScience*, 62(9): 809–818.

Garden, J., Mcalpine, C., Peterson, A., Jones, D. & Possingham, H. 2006. Review of the ecology of Australian urban fauna: A focus on spatially explicit processes. *Austral Ecology*, 31: 126–148.

Giman, B., Stuebing, R., Megum, N., Mcshea, W.J. & Stewart, C.M. 2007. A camera trapping inventory for mammals in a mixed use planted forest in Sarawak. *Raffles Bulletin of Zoology*, 55(1): 209–215.

Gonçalves, M.S.S., Gil-Delgado, J.A., Gosálvez, R.U., López-Iborra, G.M., Ponz, A. & Velasco, Á. 2018. Seasonal differences in drivers of species richness of waders in inland wetlands of La Mancha Húmeda Biosphere Reserve. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 28(6): 1414–1423.

Hobbs, R.J. & Mooney, H.A. 2008. Broadening the Extinction Debate: Population Deletions and Additions in California and Western Australia. *Conservation Biology*, 12(2): 271–283.

Holmes, P.M., Rebelo, A.G., Dorse, C. & Wood, J. 2012. Can Cape Town's unique biodiversity be saved? Balancing conservation imperatives and development needs. *Ecology and Society*, 17(2): 28.

iNaturalist Network. 2019. City Nature Challenge 2019. https://www.inaturalist.org/projects/city-nature-challenge-2019 [20 October 2019]

Kelly, M.J. 2008. Design, evaluate, refine: Camera trap studies for elusive species. *Animal Conservation*, 11(3): 182–184.

Kerley, G.I.H., Pressey, R.L., Cowling, R.M., Boshoff, A.F. & Sims-Castley, R. 2003. Options for the conservation of large and medium-sized mammals in the Cape Floristic Region hotspot, South Africa. *Biological Conservation*, 112(1-2): 169–190.

Lowry, H., Lill, A. & Wong, B.B.M. 2013. Behavioural responses of wildlife to urban environments. *Biological Reviews*, 88(3): 537–549.

MacKenzie, D.I. 2005. What Are the Issues with Presence-Absence Data for Wildlife Managers? *Journal of Wildlife Management*, 69(3): 849–860.

Mann, G.K.H., Lagesse, J. V., O'Riain, M.J. & Parker, D.M. 2015. Beefing Up Species Richness? The Effect of Land-Use on Mammal Diversity in an Arid Biodiversity Hotspot. *African Journal of Wildlife Research*, 45(3): 321–331.

Matthies, S.A., Rüter, S., Schaarschmidt, F. & Prasse, R. 2017. Determinants of species richness within and across taxonomic groups in urban green spaces. *Urban Ecosystems*, 20(4): 897–909.

McDonald, R.I., Kareiva, P. & Forman, R.T.T. 2008. The implications of current and future urbanization for global protected areas and biodiversity conservation. *Biological Conservation*, 141(6): 1695–1703.

McKinney, M.L. 2002. Urbanization, biodiversity and conservation. *BioScience*, 52(10): 883–890.

Munari, D.P., Keller, C. & Venticinque, E.M. 2011. An evaluation of field techniques for monitoring terrestrial mammal populations in Amazonia. *Mammalian Biology*, 76(4): 401–408.

O'Brien, T.G. 2008. On the use of automated cameras to estimate species richness for large- and medium-sized rainforest mammals. *Animal Conservation*, 11(3): 179–181.

O'Farrell, P.J., Anderson, P.M.L., Le Maitre, D.C. & Holmes, P.M. 2012. Insights and opportunities offered by a rapid ecosystem service assessment in promoting a conservation agenda in an urban biodiversity hotspot. *Ecology and Society*, 17(3): 27.

Ordeñana, M. A, Crooks, K.R., Boydston, E.E., Fisher, R.N., Lyren, L.M., Siudyla, S., Haas, C.D., Harris, S., Hathaway, S. A, Turschak, G.M., Miles, A.K. & Van Vuren, D.H. 2010. Effects of urbanization on carnivore species distribution and richness. *Journal of Mammalogy*, 91(6): 1322–1331.

Pauw, A. & Hawkins, J.A. 2011. Reconstruction of historical pollination rates reveals linked declines of pollinators and plants. *Oikos*, 120(3): 344–349.

Pauw, A. & Louw, K. 2012. Urbanization drives a reduction in functional diversity in a guild of nectar-feeding birds. *Ecology and Society*, 17(2): 27.

Pekin, B.K. & Pijanowski, B.C. 2012. Global land use intensity and the endangerment status of mammal species. *Diversity and Distributions*, 18(9): 909–918.

Pickett, S.T.A., Cadenasso, M.L., Grove, J.M., Boone, C.G., Groffman, P.M., Irwin, E., Kaushal, S.S., Marshall, V., McGrath, B.P., Nilon, C.H., Pouyat, R. V, Szlavecz, K., Troy, A. & Warren, P. 2011. Urban ecological systems: Scientific foundations and a decade of progress. *Journal of Environmental Management*, 92(3): 331–362.

Pressey, R.L., Cowling, R.M. & Rouget, M. 2003. Formulating conservation targets for biodiversity pattern and process in the Cape Floristic Region, South Africa. *Biological Conservation*, 112(1-2): 99–127.

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

Rebelo, A.G. 1992. Red Data Book species in the Cape Floristic Region: threats, priorities and target species. *Transactions Royal Society of South Africa,* 48(1): 55–86.

Rebelo, A.G., Boucher, C., Helme, N., Mucina, L. & Rutherford, M.C. 2006. Fynbos Biome. In: *The vegetation of South Africa, Lesotho and Swaziland*. Mucina, L. and Rutherford, M.C. (Ed.). Strelitzia 19, South African National Biodiversity Institute, Pretoria.

Rebelo, A.G., Holmes, P.M., Dorse, C. & Wood, J. 2011. Impacts of urbanization in a biodiversity hotspot: Conservation challenges in Metropolitan Cape Town. *South African Journal of Botany*, 77(1): 20–35.

Rouget, M., Richardson, D.M., Cowling, R.M., Lloyd, J.W. & Lombard, A.T. 2003. Current patterns of habitat transformation and future threats to biodiversity in terrestrial ecosystems of the Cape Floristic Region, South Africa. *Biological Conservation*, 112(1–2): 63–85.

Rovero, F., Tobler, M. & Sanderson, J. 2010. Camera trapping for inventorying terrestrial vertebrates. In: *Manual on field recording techniques and protocols for All Taxa Biodiversity Inventories and Monitoring*. Eymann, J., Degreef, J., Häuser, C., Monje, J.C., Samyn, Y., Van den Spiegel, D. (Ed.). Abc Taxa, 8(1): 100-128.

Rovero, F., Zimmermann, F., Berzi, D. & Meek, P. 2013. 'Which camera trap type and how many do I need?' A review of camera features and study designs for a range of wildlife research applications. *Hystrix*, 24(2): 148–156.

Rowcliffe, J.M., Field, J., Turvey, S.T. & Carbone, C. 2008. Estimating animal density using camera traps without the need for individual recognition. *Journal of Applied Ecology*, 45: 1228–1236.

Saito, M. & Koike, F. 2013. Distribution of Wild Mammal Assemblages along an Urban-Rural-Forest Landscape Gradient in Warm-Temperate East Asia. *PLoS ONE*, 8(5): e65464.

Šálek, M., Drahníková, L. & Tkadlec, E. 2015. Changes in home range sizes and population densities of carnivore species along the natural to urban habitat gradient. *Mammal Review*, 45(1): 1–14.

Shochat, E., Warren, P.S., Faeth, S.H., McIntyre, N.E. & Hope, D. 2006. From patterns to emerging processes in mechanistic urban ecology. *Trends in Ecology and Evolution*, 21(4): 186–191.

Si, X., Kays, R. & Ding, P. 2014. How long is enough to detect terrestrial animals? Estimating the minimum trapping effort on camera traps. *PeerJ*, 2: e374.

Small, K. 2017. 2016 *Community Survey: Cape Town Trends 1996 to 2016*. Research Branch, Organisational Policy and Planning Department, City of Cape Town

Stevens, V.M., Leboulengé, É., Wesselingh, R.A. & Baguette, M. 2006. Quantifying functional connectivity: Experimental assessment of boundary permeability for the natterjack toad (*Bufo calamita*). *Oecologia*, 150(1): 161–171.

Tobler, M.W., Carrillo-Percastegui, S.E., Leite Pitman, R., Mares, R. & Powell, G. 2008. An evaluation of camera traps for inventorying large- and medium-sized terrestrial rainforest mammals. *Animal Conservation*, 11(3): 169–178.

Torres-Romero, E.J. & Olalla-Tárraga, M. a. 2015. Untangling human and environmental effects on geographical gradients of mammal species richness: A global and regional evaluation. *Journal of Animal Ecology*, 84(3): 851–860.

Trolle, M. & Kery, M. 2005. Camera-trap study of ocelot and other secretive mammals in the northern Pantanal. *Mammalia*, 69(3-4): 405-412.

Turgeon, K. & Kramer, D.L. 2012. Compensatory immigration depends on adjacent population size and habitat quality but not on landscape connectivity. *Journal of Animal Ecology*, 81(6): 1161–1170.

Underwood, E.C., Viers, J.H., Klausmeyer, K.R., Cox, R.L. & Shaw, M.R., 2009. Threats and biodiversity in the Mediterranean biome. *Diversity and Distributions*, *15*(2): 188-197.

Van der Ree, R. & McCarthy, M.A. 2005. Inferring persistence of indigenous mammals in response to urbanisation. *Animal Conservation*, 8: 309–319.

Van der Ree, R. 2004. The impact of urbanisation on the mammals of Melbourne - do atlas records tell the whole story or just some of the chapters? In: *Urban Wildlife: more than meets the eye*. D. Lunney & S. Burgin, (Ed.). Mosman, NSW: Royal Zoological Society of New South Wales: 195–204.

Visconti, P., Pressey, R.L., Giorgini, D., Maiorano, L., Bakkenes, M., Boitani, L., Alkemade, R., Falcucci, A., Chiozza, F. & Rondinini, C. 2011. Future hotspots of terrestrial mammal loss. *Philosophical Transactions of the Royal Society of London B: Biological Sciences*, 366(1578): 2693–2702.

CHAPTER TWO:
STUDY SITE DESCRIPTIONS

2.1 Study Area

The City of Cape Town (CCT) municipality oversees 17 nature reserves within its boundaries (Fig 2.1). Thirteen of these reserves are larger than 30 ha and considered large enough to support viable populations of medium and large mammal species (Figure 2.1). Twelve of these were included in this study with Edith Stephens Nature Reserve (39 ha) being excluded because of repeated fires during the survey period.

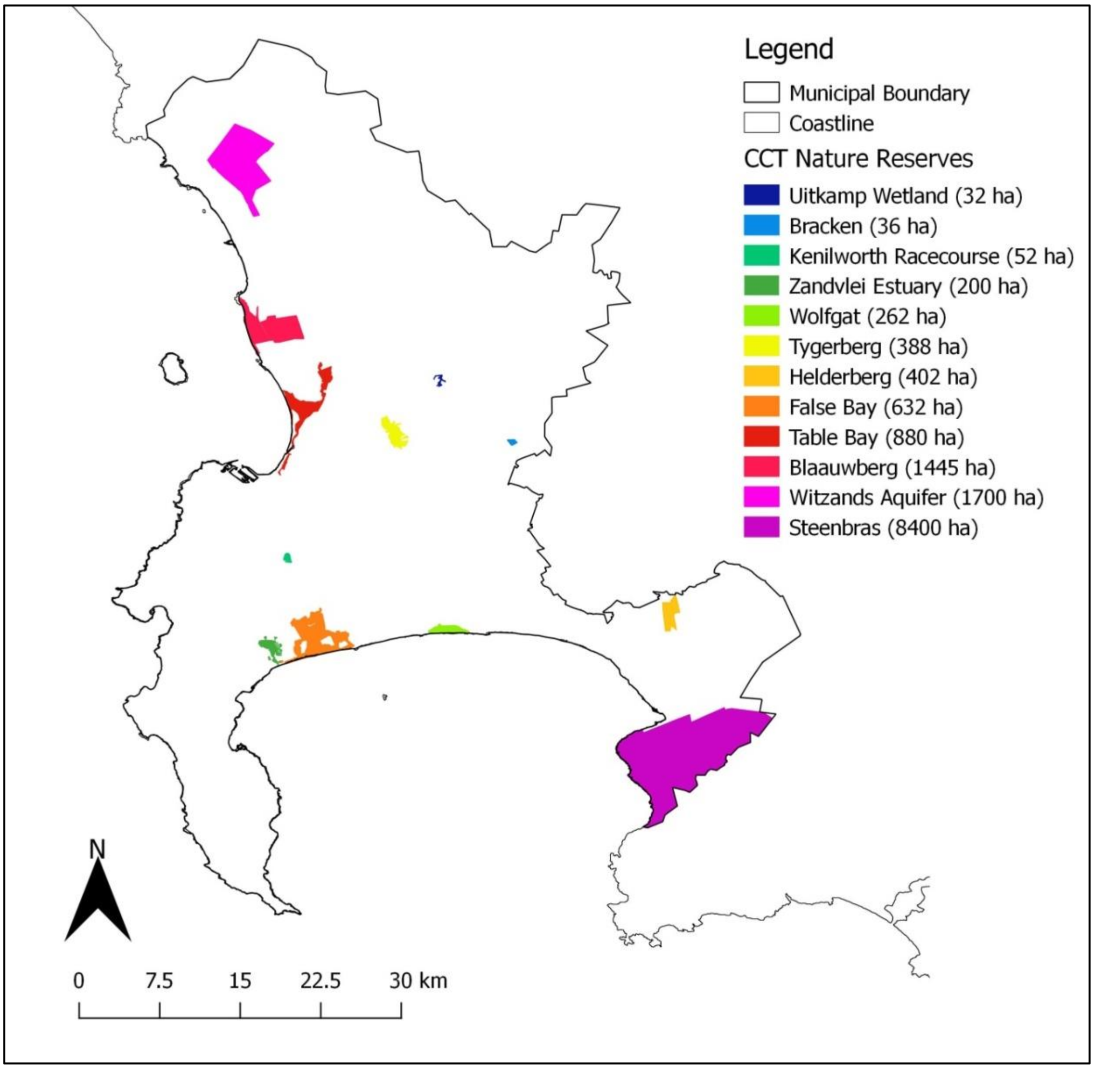


Figure 2.1: CCT nature reserve study sites (adapted from CCT 2019a & 2019c)

Climate

The CCT area (approximately 2 461 km^2) is situated within a Mediterranean climate region characterised by cool, wet winters and warm, dry and windy summers (Cowling et al. 1996, Rebelo et al. 2006). Mean annual rainfall varies with terrain and latitude from a low of 400 mm in the southern peninsula to 500-600 mm on the Cape Flats and 1300-2000 mm for the upper slopes of the northern peninsula (Cowling et al. 1996, Harris et al. 2010). Mean annual temperature for the general area varies between 16 and 22 °C (Cowling et al. 1996), with site specific variation from -0.2°C to 30.3°C (Rebelo et al. 2006). Prevailing winds average between 20 and 40 km/h and vary seasonally from north-westerlies in winter to south-easterlies in summer (Cowling et al. 1996, Rebelo et al. 2006).

Geology and topography

The study area is situated within the Cape Fold Belt formation (Rebelo et al. 2006), and incorporates the mountainous of the Cape Peninsula along the western boundary, extensive sand flats in the central and northern regions, and the Hottentots-Holland mountains on along the eastern boundary. Together these land formations are part of the Cape Supergroup (Rebelo et al. 2006) with mountainous areas comprised predominantly of weather-resistant sandstone and quartzite, with some exposed granite intrusions and narrow shale bands. The lower slopes are predominantly older Malmesbury shales (Cowling et al. 1996) with limestone cliffs forming part of the southern coastal boundary of the CCT area.

The diverse topography, parent rock material and rainfall have given rise to a wide variety of soils (Rebelo et al. 2006). The Cape flats are formed mainly of sandstone and quartzite deposits. The relatively high quartz content of the sandstone produces well-drained, nutrient-poor soil, as does that of the granite parent material. Soils on higher-lying sandstone flats are shallow and acidic (Cowling et al. 1996). Shale-derived soils on the lower mountain slopes are deeper and have a relatively higher nutrient content.

Vegetation

This soil diversity in the region contributes to diverse vegetation types within the CCT municipal area. These can be largely grouped into three major complexes, namely fynbos, renosterveld and strandveld (Rebelo et al. 2006). Typical fynbos is characterised by shrubland consisting of at least 5% Restionaceae species, with the presence of Ericaceae and Proteaceae shrubs in varying proportions and a low grass component (Rebelo et al. 2006). Fynbos systems are fire-prone and occur mainly in sandy, nutrient-poor soil. Renosterveld structure can vary between shrubland and grassland, consisting of small-leaved, evergreen *Asteraceae* shrubs, grasses and a large proportion of geophytes. Renosterveld typically excludes *Erica* and *Protea* species, and occurs on shale- and granite-derived clay soils. It is also fire-prone. Strandveld vegetation occurs along coastal areas but out of direct ocean spray (Rebelo et al. 2006). This vegetation has a medium to dense structure formed by sclerophyllous shrubs, and while restio species may be present, no

Protea and little to no *Erica* species occur. Strandveld relies on calcium-rich soils, and has a low fire frequency (Rebelo et al. 2006).

The three broad vegetation complexes are further subdivided into bioregions and within the confines of the CCT: the Southwest Fynbos, West Coast Renosterveld and West Strandveld are present (Figure 2.2). There are also some azonal areas, which include water-associated vegetation (e.g. wetland, riverine), and a total of twenty five distinct vegetation types. Approximately 60% of the CCT municipal area is has been transformed by urban rural and industrial developments (Figure 2.3), and only small remnants of natural vegetation persist (Rebelo et al. 2011).

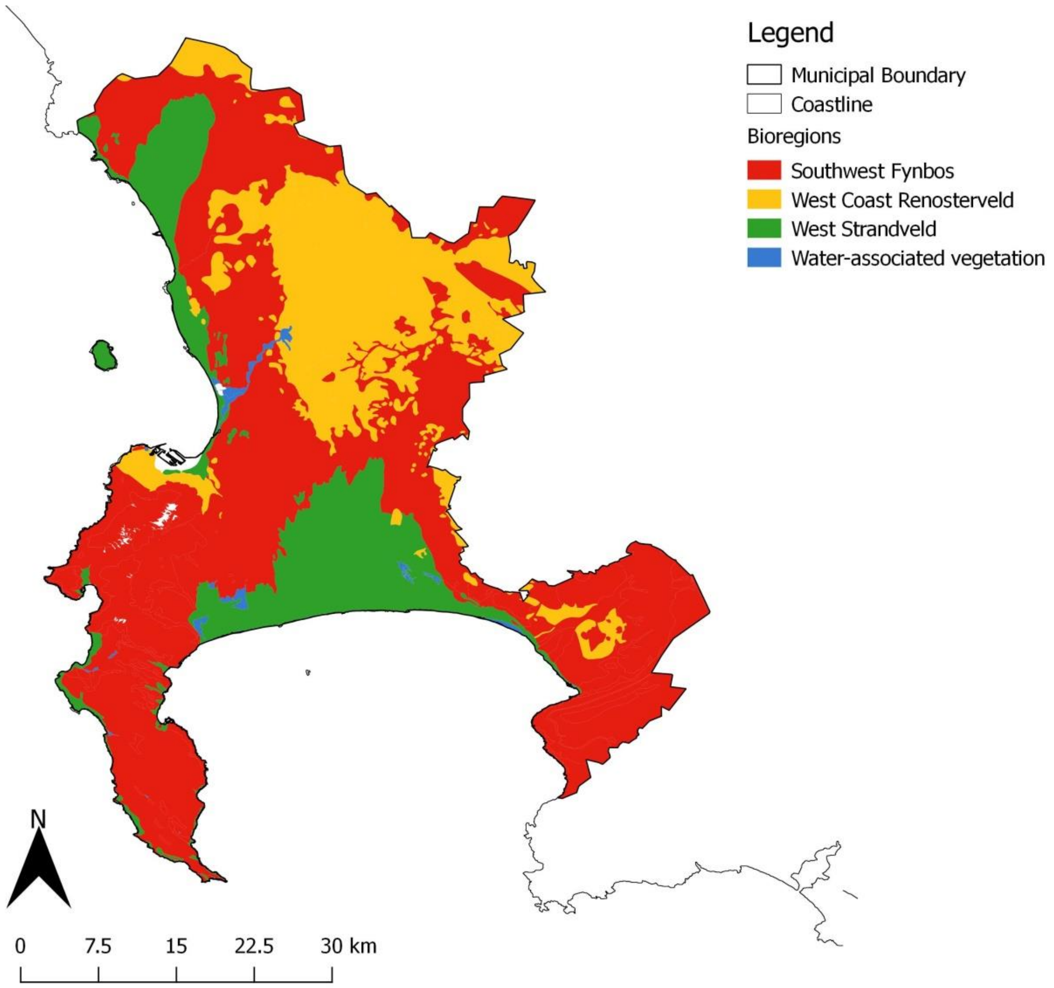


Figure 2.2: Historically-occurring bioregions within the CCT area (adapted from SANBI 2016 and CCT 2019a)

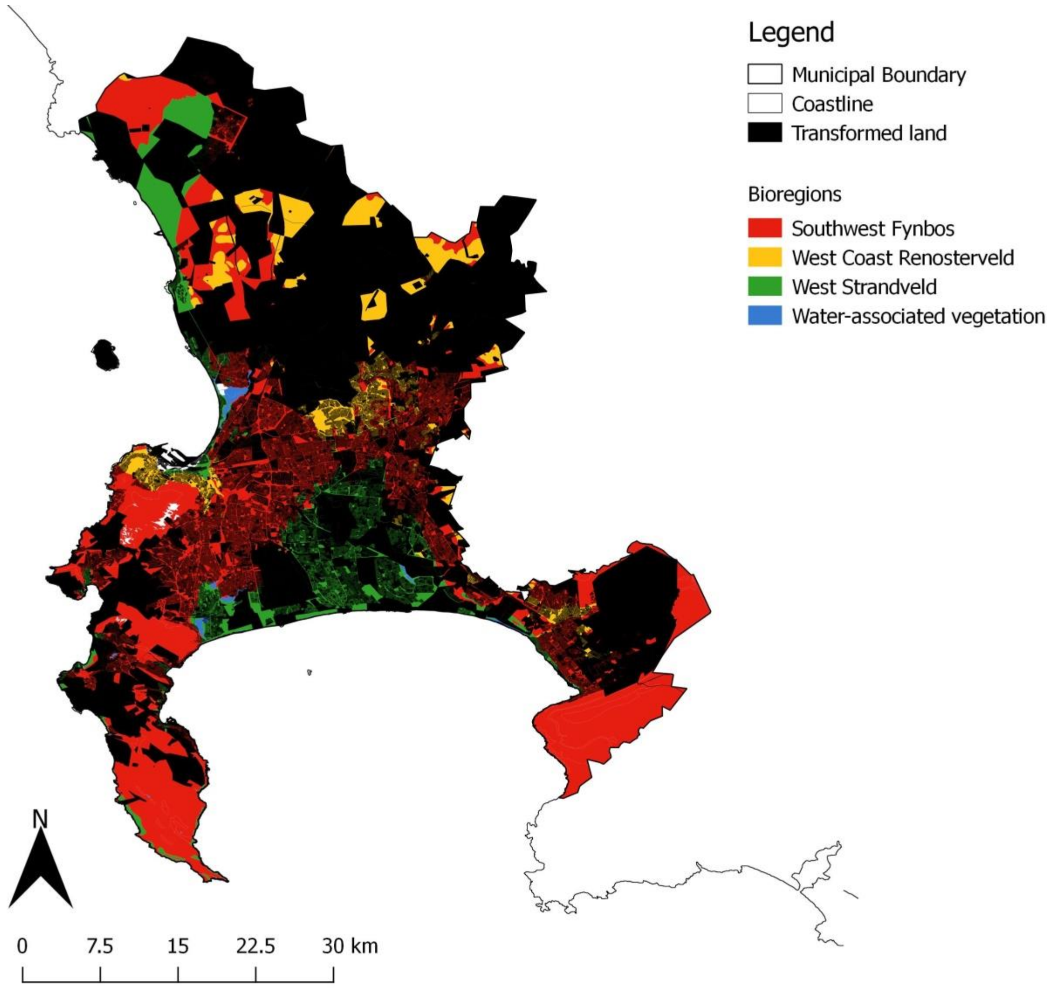


Figure 2.3: Transformed land (urban development, agriculture) and the remaining historically-occurring bioregion areas within the CCT area (adapted from SANBI 2016 and CCT 2019a)

2.2 Study sites

For the purposes of this study only the broader bioregion vegetation classifications were used. Thus southwest Fynbos, West Coast Renosterveld and West Strandveld bioregions are classified broadly as fynbos, renosterveld and strandveld. Wetland, coastal and riverine vegetation are together classified as 'water-associated vegetation' with lawns, plantation and developed areas classified as transformed vegetation. The twelve nature reserves can also be broadly grouped into three categories based on their geographic location, namely northern, central, southern and eastern reserves.

2.2.1 Northern reserves

Table Bay Nature Reserve

This reserve (Figure 2.4) is 880 ha in extent and largely comprised of wetland areas that terminate in the Diep River lagoon which enters the Atlantic Ocean in Table Bay. Ninety-two percent of the reserve is made up of either seasonal pans or permanent wetland and water bodies (including man-made Rietvlei), and the remaining vegetation consists of water-associated (617 ha) and some fynbos communities (16.5 ha). The water edges are dominated by dense reed beds with large areas of grass (101 ha) on the reserve edges. Mid- to high-income housing, a sewerage treatment plant, an industrial area and disturbed land surround the reserve.

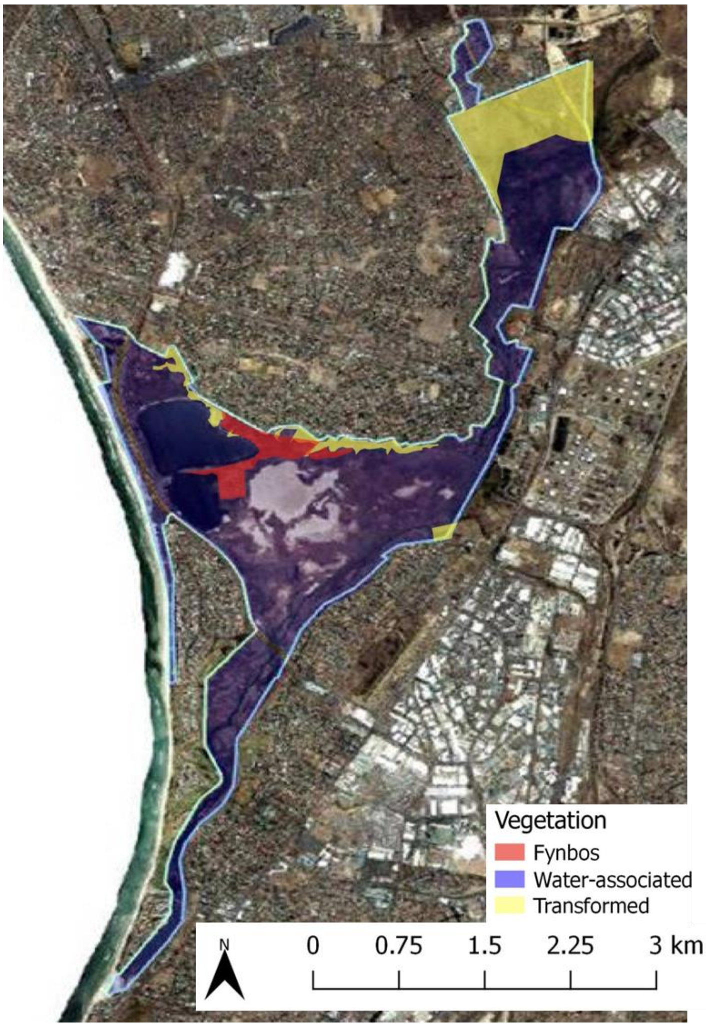


Figure 2.4: Table Bay Nature Reserve vegetation (adapted from SANBI 2016 and CCT 2019a,b). The reserve boundary is indicated in light blue.

Blaauwberg Nature Reserve

The 1 445 ha area of Blaauwberg Nature Reserve (Figure 2.5) is dominated by Strandveld vegetation (790 ha) in the western coastal strip, and renosterveld (115.5 ha) and fynbos (534.5 ha) on Blaauwberg Hill and the extended area to the east. The reserve is under ongoing clearing of the invasive alien plant species *Acacia saligna* (Port Jackson willow), but large stands of this invasive species still exists. The western coastal section is unfenced but separated from the main body of the reserve in the east by the R27, a major arterial road, and game fencing. Surrounding land use includes agriculture, disturbed and Port Jackson-invaded undeveloped land and a small section of high-income residential area.

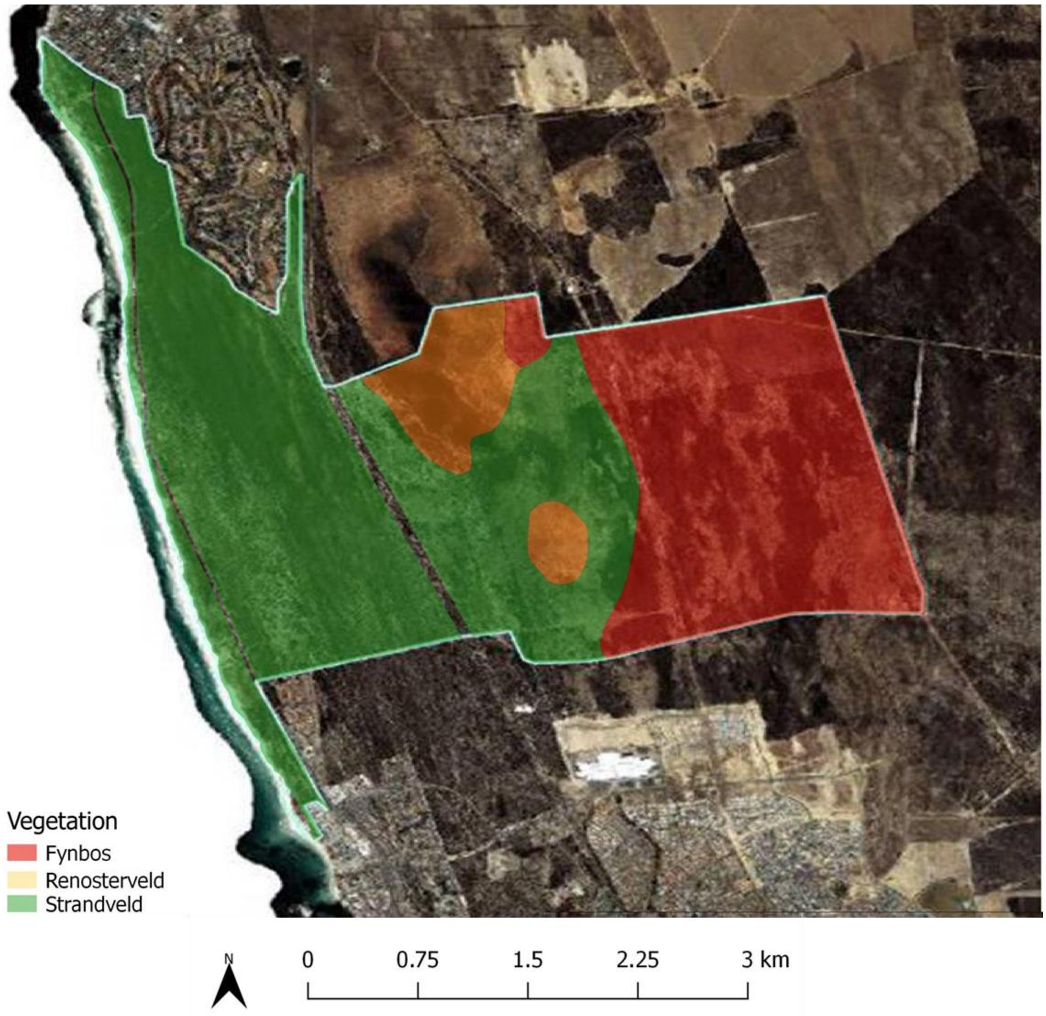


Figure 2.5: Blaauwberg Nature Reserve vegetation (adapted from SANBI 2016 and CCT 2019a,b). The reserve boundary is indicated in light blue.

Witzands Aquifer Nature Reserve

This is the northern-most CCT reserve (1 700 ha), the core of which is a dune field with seasonal wetlands (Figure 2.6) surrounded by strandveld (1652 ha). Cattle fencing surrounds most of the reserve, and a section to the south is separated from the main reserve area by an arterial road and should contain ecotonal fynbos but is largely seasonal wetland area (48 ha). The surrounding land is largely used for subsistence farming and informal housing, and includes an industrial area and natural vegetation. The reserve is frequented by recreational off-road vehicles in the dune area and by wood cutters harvesting invasive alien plant species *Acacia cyclops* (rooikrans).

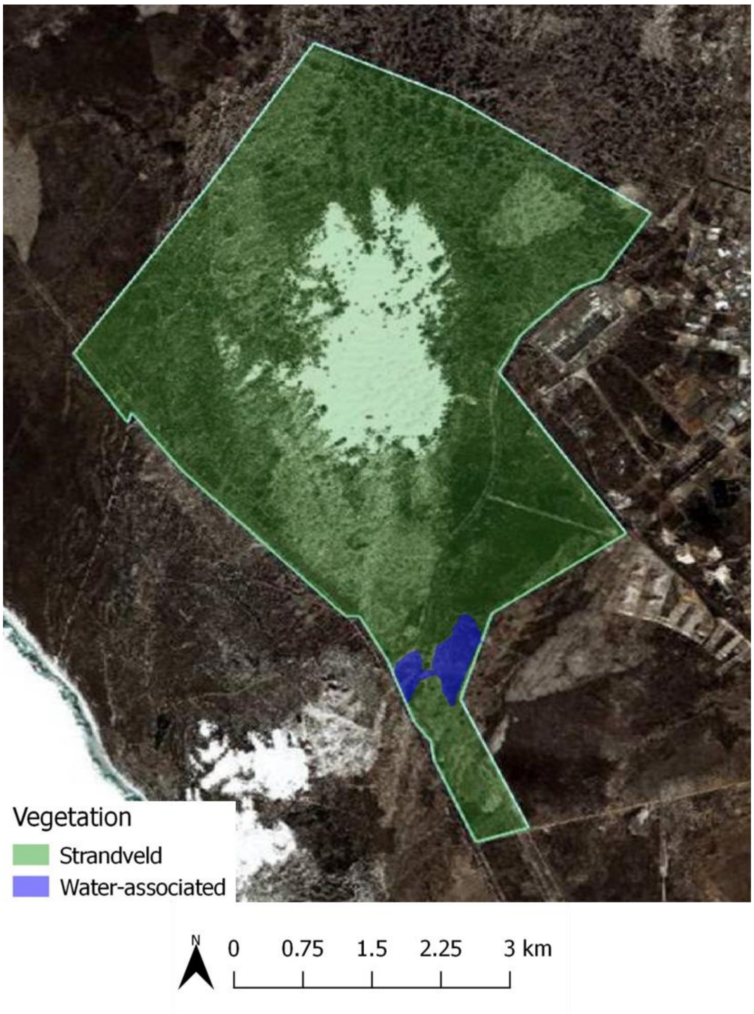


Figure 2.6: Witzands Aquifer Nature Reserve vegetation (adapted from SANBI 2016 and CCT 2019a,b). The reserve boundary is indicated in light blue.

2.2.2 Central reserves

Uitkamp Wetland Nature Reserve

Uitkamp is the smallest of the reserves (32 ha) and primarily comprised of a wetland corridor (24.5 ha) located within a residential area (Figure 2.7) and includes a small patch of ecotonal renosterveld. It shares a tenth of its border with agricultural land. Overhead powerlines run through much of the reserve and borders include the walls and fences of residential and rural properties with some road-side sections having chain-link fencing.

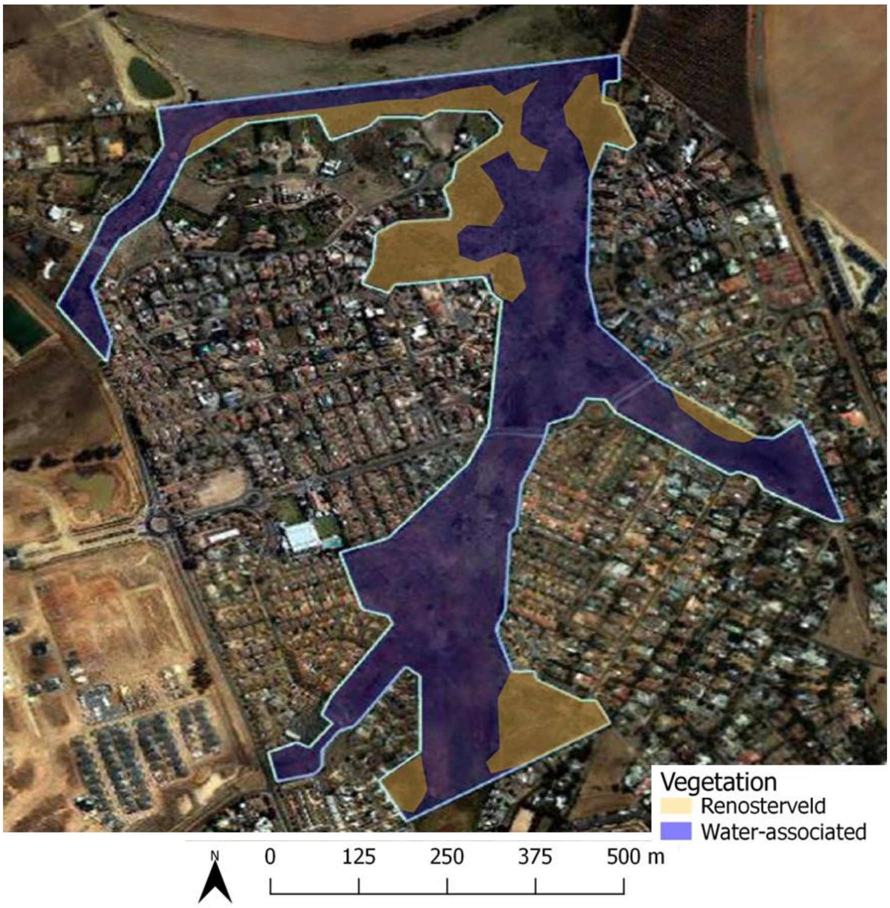


Figure 2.7: Uitkamp Wetland Nature Reserve vegetation (adapted from SANBI 2016 and CCT 2019a,b). The reserve boundary is indicated in light blue.

Bracken Nature Reserve

This reserve is only marginally larger (36 ha) than Uitkamp Nature Reserve (Figure 2.8) and, having previously served as a landfill for municipal waste**,** features an artificial hill. The reserve vegetation falls broadly within renosterveld, although a large proportion (9 ha) is dominated by invasive grasses. Fynbos is predicted to occur in the reserve but the ecotone only begins at the reserve boundary. The reserve is fenced, and the northern and eastern boundaries border industrial and agricultural land uses, with the western and southern boundaries along residential area.

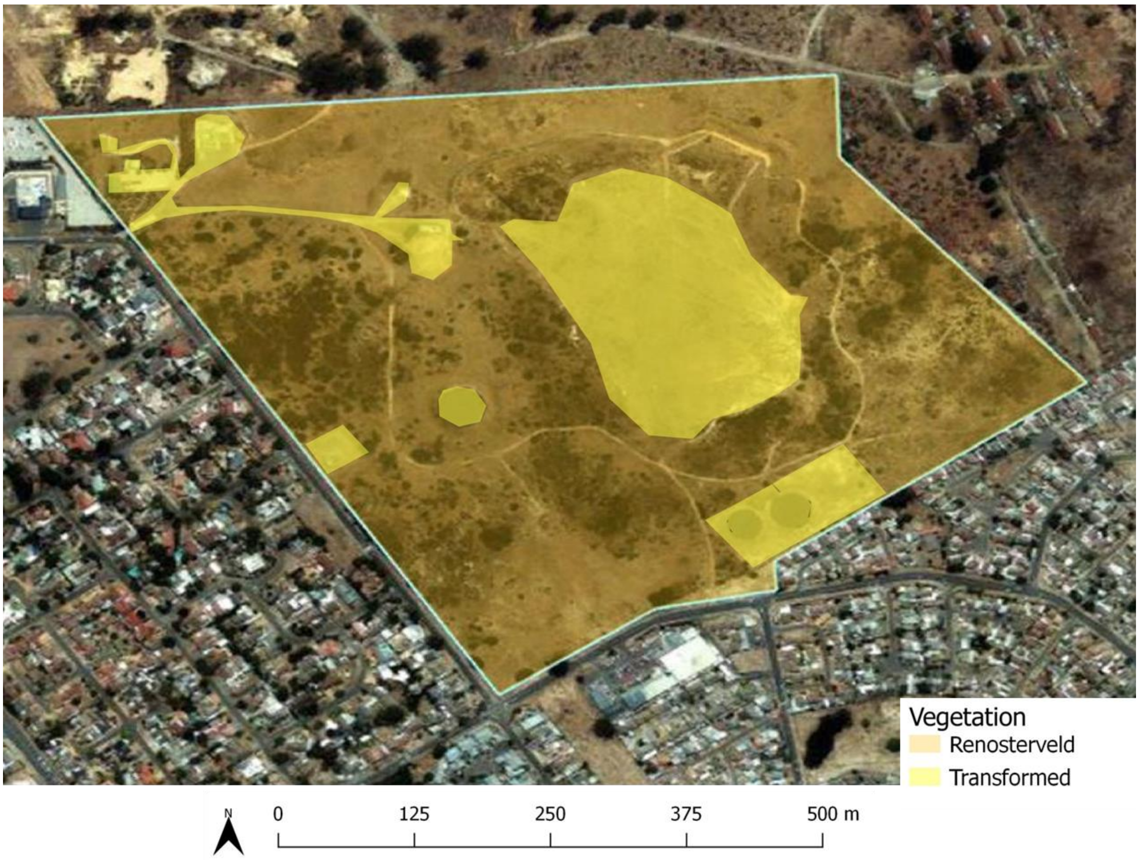


Figure 2.8: Bracken Nature Reserve vegetation (adapted from SANBI 2016 and CCT 2019a,b). The reserve boundary is indicated in light blue.

Tygerberg Nature Reserve

Situated on Tygerberg Hill this reserve (Figure 2.9) covers 388 ha and consists mainly of renosterveld (296.5 ha), of which portions were once ploughed fields, and a very small portion of Cape Flats Sand Fynbos which has been transformed into a pine plantation (82 ha of transformed vegetation collectively). Small streams and rivers run off the slopes and feed several reservoirs and dams within its boundaries which allows for the presence of 9.5 ha of water-associated vegetation. The full boundary of the reserve is fenced and surrounded by suburban housing and unmaintained plantation.

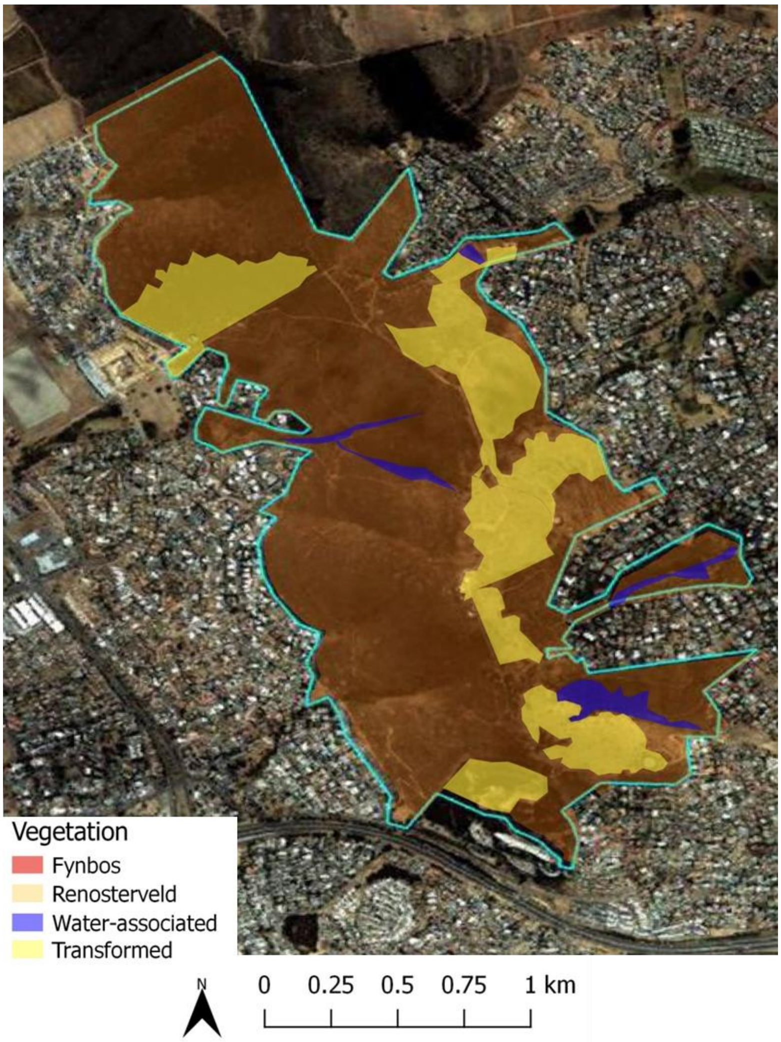


Figure 2.9: Tygerberg Nature Reserve vegetation (adapted from SANBI 2016 and CCT 2019a,b). The reserve boundary is indicated in light blue.

2.2.3 Southern reserves

Kenilworth Racecourse Conservation Area

This reserve is 52 ha in extent (Figure 2.10) and includes a core area of fynbos vegetation (36 ha) with a seasonal wetland (10.5 ha) encircled by a grassed horseracing track (5.5 ha). The surrounding land use is primarily residential and commercial separated from the reserve by a 2.2m concrete wall.

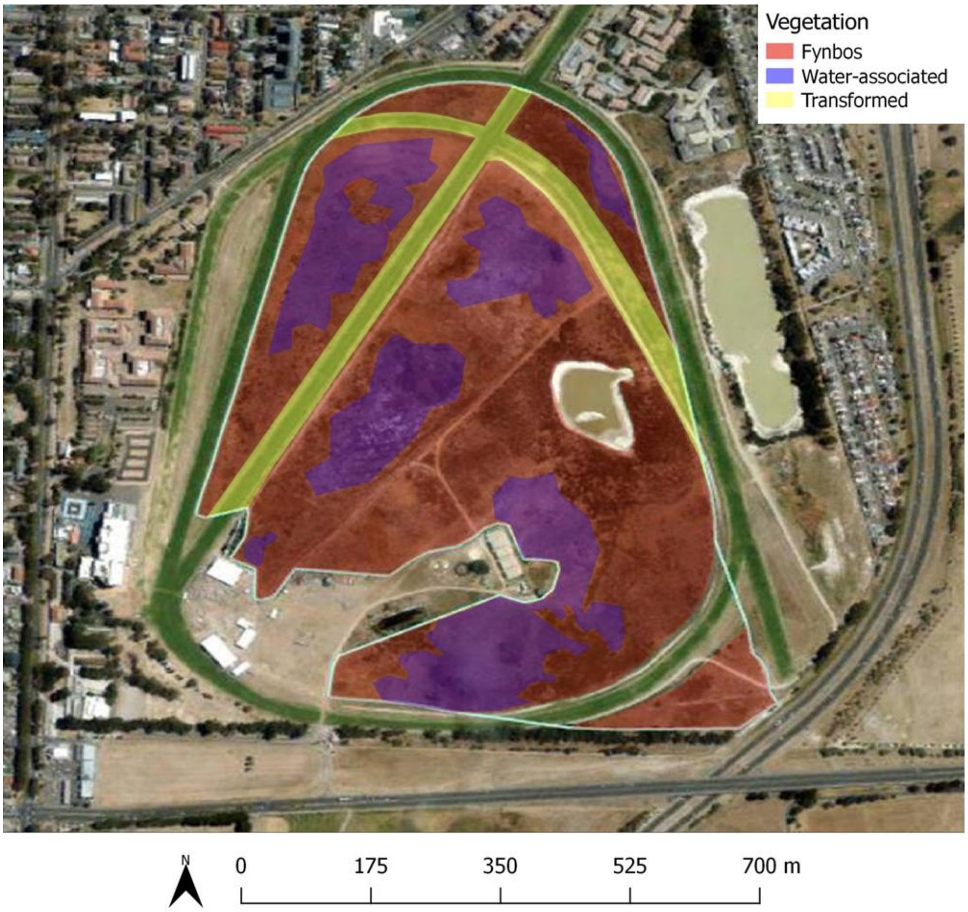


Figure 2.10: Kenilworth Racecourse Conservation Area vegetation (adapted from SANBI 2016 and CCT 2019a,b). The reserve boundary is indicated in light blue.

Zandvlei Estuary Nature Reserve

The 200 ha Zandvlei Estuary Nature Reserve (Figure 2.11) is dominated by a large vlei that includes man-made canals excavated for a marina housing development. A 16 ha man-made island within the vlei is covered with fynbos, and connected to the housing development by a concrete bridge and the main body of the reserve when estuary water levels are low. The low-lying terrestrial section of the reserve consists of seasonal pans and wetland areas (54.5 ha) and includes mostly strandveld surrounded (35.5 ha) by low to middle-income housing, railway lines and a disturbed undeveloped area. The northern section of the reserve is fenced and a 7.5 ha section of the reserve has been transformed into a grass lawn for recreational activities.

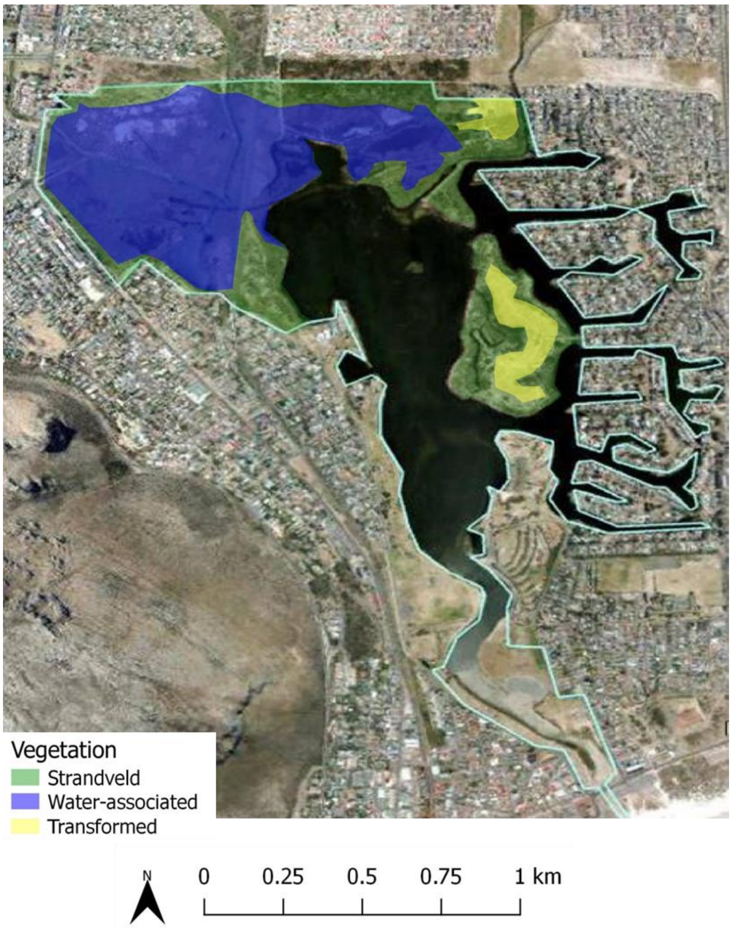


Figure 2.11: Zandvlei Estuary Nature Reserve vegetation (adapted from SANBI 2016 and CCT 2019a,b). The reserve boundary is indicated in light blue.

False Bay Nature Reserve

This reserve (Figure 2.12) consists of the Rondevlei and Zeekoevlei waterbodies which flow into False Bay. The 632 ha reserve includes the Strandfontein sewerage works whose open water bodies and pans (48.7 ha) provide important bird habitat. The dominant vegetation type is strandvleld (139.5 ha), with wetland vegetation (46 ha) adjacent to open water and fynbos (96 ha) comprising the remaining habitat types. The borders of the reserve are fenced as they abut dense mid- to low-income housing, industrial and disturbed undeveloped areas. Some internal fencing exists between major reserve sections and surrounding the sewerage treatment plant.

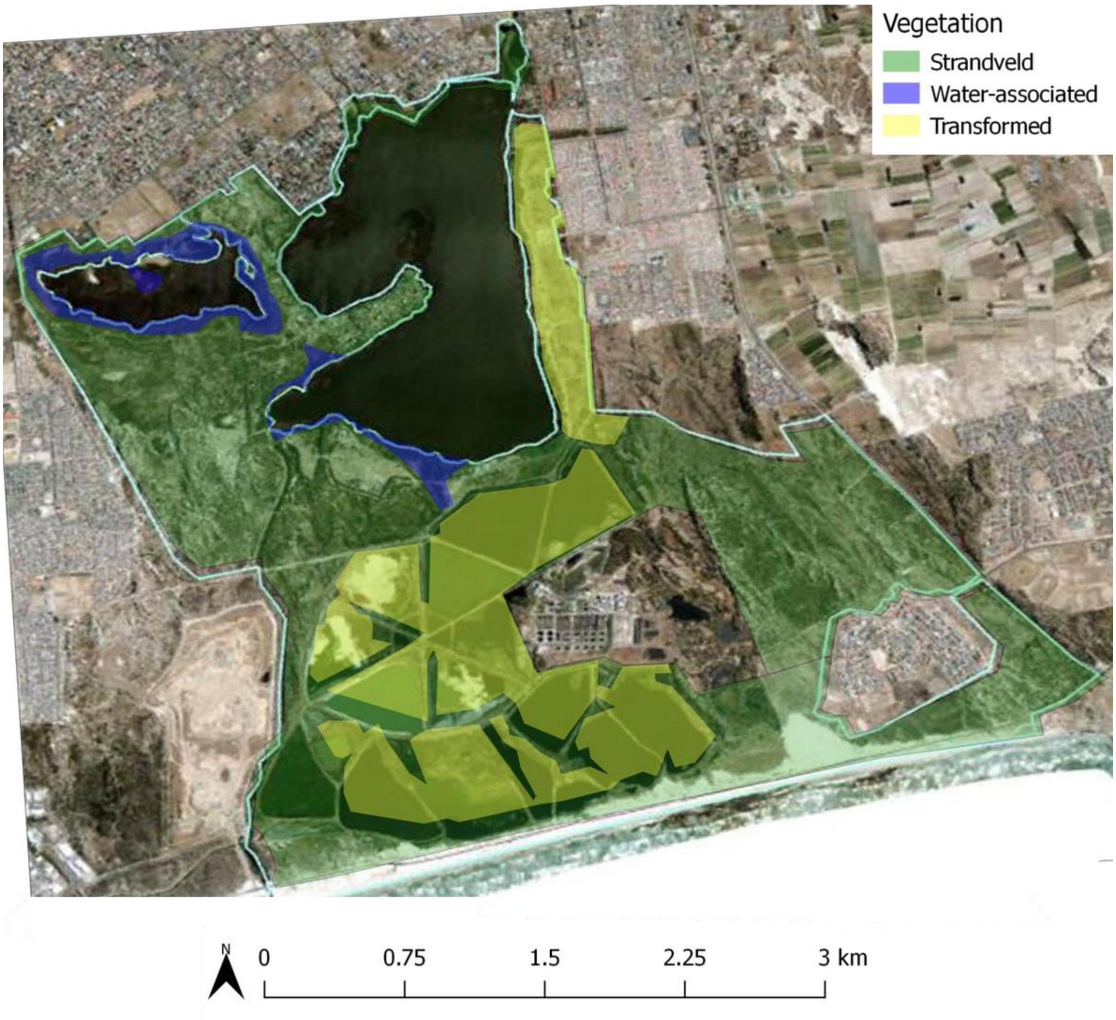


Figure 2.12: False Bay Nature Reserve vegetation (adapted from SANBI 2016 and CCT 2019a,b). The reserve boundary is indicated in light blue.

2.2.4 Eastern reserves

Wolfgat Nature Reserve

This 262 ha reserve (Figure 2.13) includes coastal dunes and strandveld along the False Bay coast. The northern boundary includes informal and low-cost housing while the east and western ends are bordered by natural vegetation. The reserve is unfenced, which also means that human access is unregulated and some illegal poaching of fauna has been recorded.

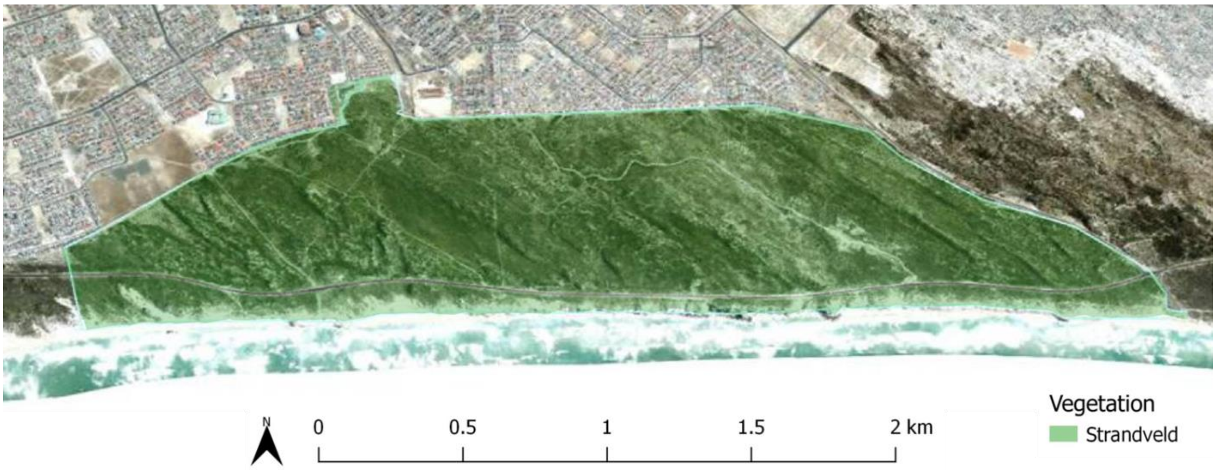


Figure 2.13: Wolfgat Nature Reserve vegetation (adapted from SANBI 2016 and CCT 2019a,b). The reserve boundary is indicated in light blue.

Helderberg Nature Reserve

Situated on the southern slope of the Helderberg mountain, this 402 ha reserve (Figure 2.14) contains both fynbos (370 ha) and a small portion of renosterveld vegetation (8 ha). Permanent river systems, ponds and a municipal reservoir together comprise permanent freshwater aquatic habitat (7 ha), with 8 ha of the reserve comprised of water-associated vegetation and 9 ha of transformed grass lawns. The reserve shares a fenced boundary with up-market housing estate, a golf course and agricultural/forestry land along the eastern and southern borders. The northern border is unfenced and characterised by natural mountain fynbos.

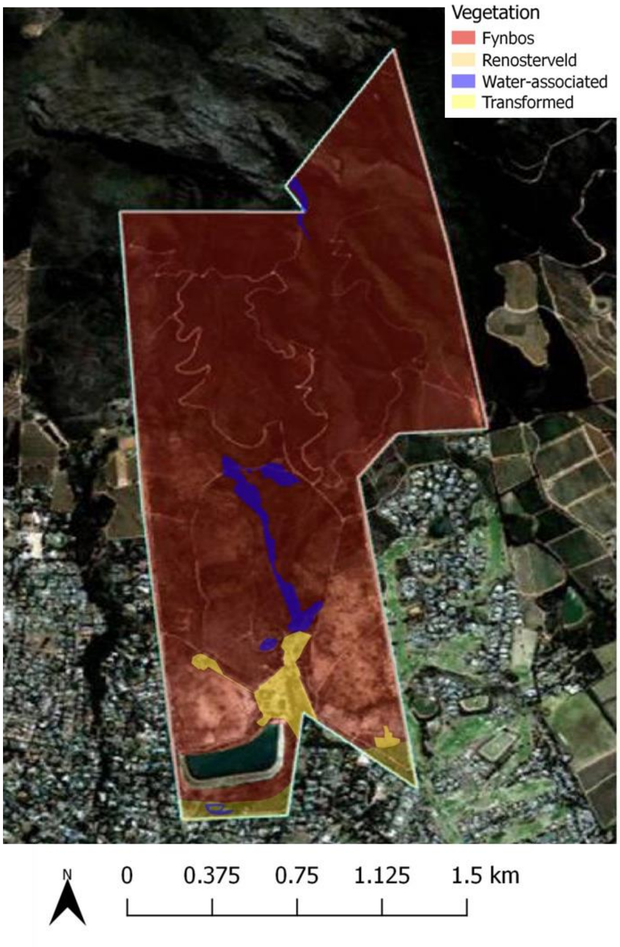


Figure 2.14: Helderberg Nature Reserve vegetation (adapted from SANBI 2016 and CCT 2019a,b). The reserve boundary is indicated in light blue.

Steenbras Nature Reserve

This is the largest of the CCT reserves at 8 400 ha (Figure 2.15), and makes up part of the Kogelberg Biosphere Reserve. The reserve covers the coast and Steenbras basin at Kogel Bay, as well as the mountainous section hosting the Steenbras dam and river system. The majority of the vegetation is fynbos (6752.5 ha), with water-associated vegetation covering 91 hectares. A large section of the reserve (1108.5 ha) has been transformed by pine plantations. The reserve borders are mostly natural vegetation, with some agriculture, forestry, coastline and a very small section of high-income residential (Gordon’s Bay). There are thus very few fences along the reserve boundaries.

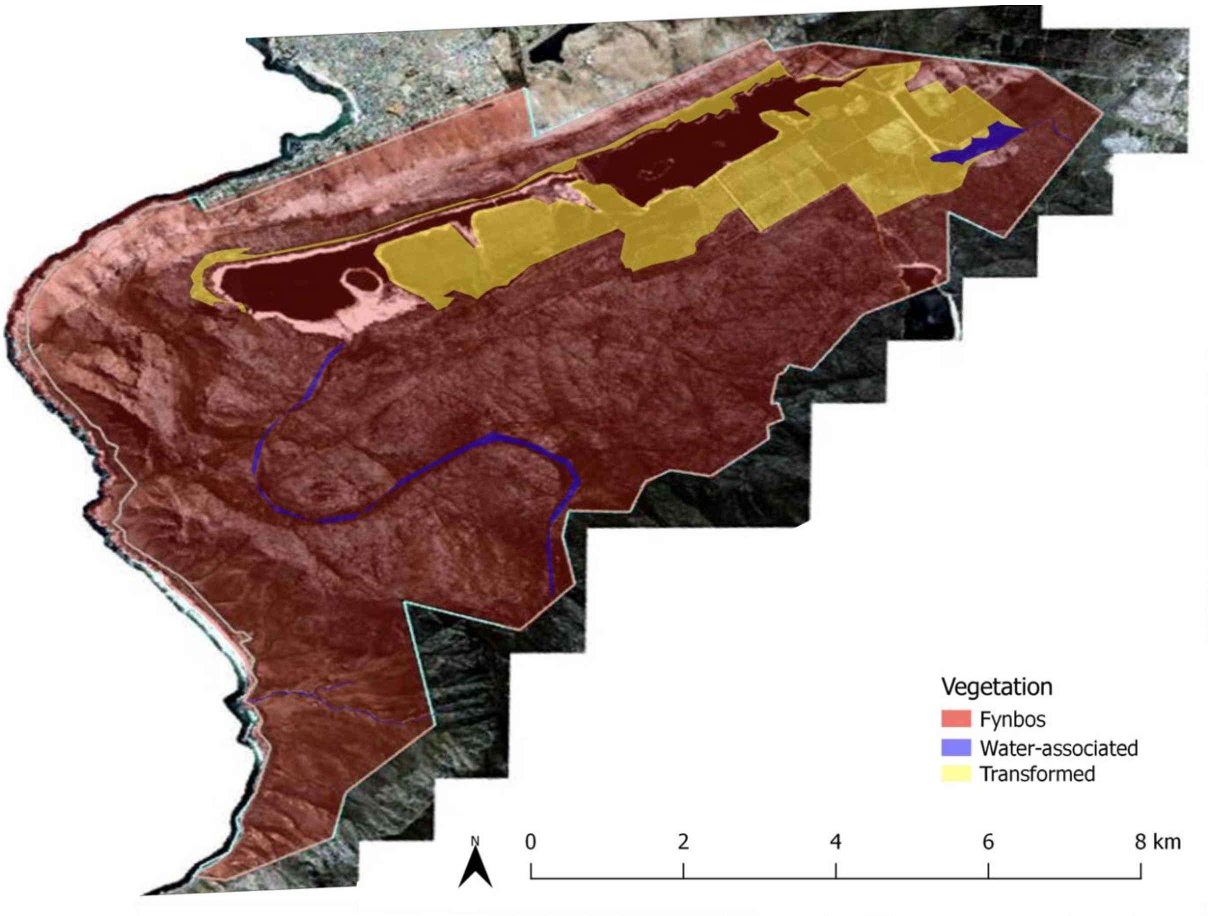


Figure 2.15: Steenbras Nature Reserve vegetation (adapted from SANBI 2016 and CCT 2019a,b). The reserve boundary is indicated in light blue.

2.3 References

City of Cape Town (CCT). 2019a. Open Data Portal: Aerial photography. https://web1.capetown.gov.za/web1/OpenDataPortal/DatasetDetail?DatasetName=Aerial%20photography [25 July 2019].

City of Cape Town (CCT). 2019b. Open Data Portal: Terrestrial Biodiversity Network. http://odp.capetown.gov.za/datasets/4f2d7835518a4e6b8205ce12d77ff463_64 [4 April 2019]

City of Cape Town (CCT). 2019c. *Nature Reserves.* https://www.capetown.gov.za/Family%20and%20home/See-all-city-facilities/Our-recreational-facilities/Nature%20reserves [20 June 2019].

Cowling, R.M., MacDonald, I.A.W. & Simmons, M.T. 1996. The Cape Peninsula, South Africa: physiographical, biological and historical to an extraordinary hot-spot of biodiversity. *Biodiversity & Conservation*, 5(5): 527-550.

Harris, C., Burgers, C. Miller, J. & Rawfoot, F. 2010. O- and H-isotope record of Cape Town rainfall from 1996 to 2008. *South African Journal of Geology,* 113(1): 33–56

Rebelo, A.G., Boucher, C., Helme, N., Mucina, L. & Rutherford, M.C. 2006. Fynbos Biome. In: *The vegetation of South Africa, Lesotho and Swaziland*. Mucina, L. and Rutherford, M.C. (Ed.). Strelitzia 19, South African National Biodiversity Institute, Pretoria

Rebelo, A.G., Holmes, P.M., Dorse, C. & Wood, J. 2011. Impacts of urbanization in a biodiversity hotspot: Conservation challenges in Metropolitan Cape Town. *South African Journal of Botany*, 77(1): 20–35.

South African National Biodiversity Institute (SANBI). 2016. Vegetation Map of South Africa, Lesotho and Swaziland. http://bgis.sanbi.org/SpatialDataset/Detail/18 [28 March 2017]

CHAPTER THREE:
ASSESSING AND MONITORING MEDIUM AND LARGE MAMMAL COMMUNITY COMPOSITION ACROSS CITY OF CAPE TOWN RESERVES USING CAMERA TRAPS

3.1 Introduction

Mammal species play an important role in the functioning of ecosystems and maintenance of biodiversity (De Stefano & De Graaf 2003, Kerley et al. 2003, Ceballos et al. 2005, Visconti et al. 2011). With an increase in anthropogenic activity worldwide, many mammal species are being adversely affected contributing to global declines in biodiversity (Ceballos et al. 2005, Ceballos & Ehrlich 2006, Visconti et al. 2011, Pekin & Pijanowski 2012). Traditional conservation efforts for mammal species have focused largely on establishing protected areas in which species are intended to be protected from anthropogenic influences (McDonald et al. 2008), but in urban areas, natural environments tend to be reduced to small, fragmented and often isolated pockets which are broadly impacted by anthropogenic activity. Urbanisation in particular is impacting on natural habitats, population dynamics and ultimately ecosystem functioning of most fauna and flora (De Stefano & De Graaf 2003, Kerley et al. 2003, McDonald et al. 2008, Ceballos et al. 2005, Visconti et al. 2011, Pekin & Pijanowski 2012, Ramesh et al. 2016) by direct impacts such as habitat loss and fragmentation, as well as indirect impacts such as increases in pollution, resource use, human-wildlife contact (and subsequent behavioural changes) and disease exposure (Ceballos & Ehrlich 2006, McDonald et al. 2008, Ordeñana et al. 2010, Visconti et al. 2011).

The Cape Floristic Region (CFR) is a global biodiversity hotspot, hosting a wide variety of vegetation types and high rates of endemism (Kerley et al. 2003, Pressey et al. 2003, Rebelo et al. 2011). While the literature focuses on the CFR's floral diversity, the area also contains a significant diversity of fauna species (Boshoff et al. 2001, Kerley et al. 2003, Pressey et al. 2003). A total of 41 medium and large mammal species are thought to have occurred historically within the CFR (Kerley et al. 2003), but many of the large carnivore and herbivore species such as lion (*Panthera leo*), black rhinoceros (*Diceros bicornis bicornis)* and common eland (*Tragelephus oryx)* were hunted to local extinction within the first 50 years of colonization (Rebelo 1992). While smaller mammal species have been able to persist in the area, rapid urbanisation has become an increasing threat. The CFR experiences the second highest population growth rate within the Mediterranean biome globally (Underwood et al. 2009), thus the threats associated with urbanisation, including the loss and fragmentation of natural habitat are likely to increase and threaten the remaining mammal diversity.

The City of Cape Town is the largest urban area within the CFR and seeks to maintain biodiversity through a network of 17 nature reserves. The small size and isolation of many of these reserves limits their potential to accommodate viable populations of the remaining mammal species. Currently there is also a lack of reliable species lists for these reserves or a standardized monitoring protocol to allow for comparisons of species richness and abundance between reserves and within reserves over time (Olwell et al. 2004). There are data on the historical distribution of medium and large mammals for the greater Cape Town region in addition to opportunistic citizen and reserve staff sightings. However neither of these data sources are regarded as being sufficiently robust for generating species richness estimates in both time and space (Garden et al. 2006, O'Brien 2008).

A number of methods could be used to assess remaining mammalian species within the nature reserves of the CCT, including distance sampling from roads and sign surveys. However most of these methods tend to be biased towards species that are willing to travel on open paths and roads (Giman et al. 2007). Remote-sensing camera traps (cameras) are becoming increasingly popular as a tool for medium to large mammal surveys, and have in many cases greatly improved our ability to detect a diverse array of medium and large mammals which can then be used to generate species richness, diversity, abundance and behavioural parameters (Kelly 2008, Rowcliffe et al. 2008, Tobler et al. 2008, Ordeñana et al. 2010, Rovero et al. 2010). Camera traps are particularly useful for detecting evasive, nocturnal and/or rare species (Rowcliffe et al. 2008, Tobler et al. 2008, Ordeñana et al. 2010, Rovero et al. 2010, Si et al. 2014). Camera trap surveys also tend to be more efficient compared to other methods and hence are more likely to be approved under circumstances where both budgets and labour are limited (Silveira et al. 2003, O'Connell et al. 2010). Surveys can also be conducted over a relatively short time periods, rather than the years required for accurate species list compilation through direct observation only (Kelly 2008, O'Brien 2008).

There is an increasing amount of evidence showing that when studying species richness in an area, the number of camera days (number of cameras multiplied by survey period) are more important than camera spacing or density for obtaining reliable estimates (Kelly 2008, Tobler et al. 2008, O'Brien 2008, Si et al. 2014, Colyn et al. 2017). The majority of studies have recorded 80-90% of the estimated number of species occupying an area within 900 to 1500 camera days (Giman et al. 2007, Tobler et al. 2008, Si et al. 2014). The time required for reliable species estimates has been shown to vary with habitat type and the number of rare/elusive species. Tobler et al. (2008) required 2340 camera days to record 86% of species in Peruvian forest areas, while Trolle and Kery (2005) found 504 camera days sufficient for a section of the Pantanal wetland area in Brazil. Approximately 1000 camera days were sufficient to detect more than 90% of species in the Fynbos shrubland of Cape Peninsula (Colyn et al. 2017). For smaller areas, a higher number of camera traps can be used over a shorter period of time (Si et al. 2014). Together these studies suggest that a compromise between the number of camera traps and the duration of deployment can ensure adequate sampling within time and budgetary constraints.

Camera placement is another important variable to consider when sampling mammals using camera traps. At a broad scale random placement ensures that all habitat types and species have an equal chance of being sampled and the resultant data can then be used to inform distribution and/or occupancy models (O'Brien 2008). Alternatively cameras can be placed to target specific species, but caution has to be exercised when using such data for species richness estimates (Tobler et al. 2008). In areas which include more than one major habitat type, all habitat types should be included in the survey using a random stratified sampling protocol to account for habitat specialists (Tobler et al. 2008, O'Brien 2008).

In this chapter, the primary goal was to assess medium and large mammal species in CCT nature reserves that are larger than 30 ha (n = 12) using camera trap surveys. The specific objectives were to a) determine which medium and large mammal species are still present in CCT nature reserves larger than 30 ha, compared with historic accounts and current presumptions, and b) determine the minimum survey effort required to sufficiently record and monitor medium and large mammal species presence in each CCT reserve.

3.2 Methodology

3.2.1 Study sites

This study was conducted within the City of Cape Town (CCT) municipal area, which is situated within the Cape Floristic Region and experiences a Mediterranean climate of warm, dry summers and cool, wet winters. The study focused on the thirteen CCT nature reserves that are larger than 30 ha and may thus support viable, naturally-occurring populations of medium and large mammals. Edith Stephens Nature Reserve was subjected to repeated unnatural fires during the study period and could not be adequately sampled. This resulted in twelve reserves of different size and vegetation type composition (Table 3.1) being thoroughly surveyed once for this study. Study sites are discussed in more detail in Chapter 2.

Table 3.1: Reserve size and proportional cover (%) of each of the five habitat types identified within each of the surveyed City of Cape Town nature reserves (City of Cape Town 2019). Some proportions do not add up to 100% due to the presence of permanent open water.

Nature Reserve	Size (ha)	Fynbos	Renoster-veld	Strand-veld	Water-associated	Trans-formed
Uitkamp Wetland	32	23.38	-	-	76.62	-
Bracken	36	-	73.88	-	-	26.12
Kenilworth Racecourse	52	66.21	-	-	20.48	11.08
Zandvlei Estuary	200	-	-	17.76	27.20	3.74
Wolfgat	262	-	-	100	-	-
Tygerberg	388	-	76.54	-	2.43	21.03
Helderberg	402	92.35	1.79	-	1.91	2.16
False Bay	632	15.19	-	22.01	7.23	7.71
Table Bay	880	1.86	-	-	70.13	11.54
Blaauwberg	1445	36.50	8.00	55.50	-	-
Witzands Aquifer	1700	-	-	97.19	2.8	-
Steenbras	8400	80.38	-	-	1.08	13.20

3.2.2 Camera trap survey

Ninety un-baited Bushnell® Trophy Cam infrared remote sensing cameras were used to record the presence of medium to large mammal species in each reserve. Sampling took place over 21 months from June 2017 to Feb 2019, and multiple reserves were sampled simultaneously when possible. Infrared cameras were chosen due to high human activity and risk of theft in most of the reserves. Seasonal variation was not expected to influence species richness estimates as there are no known migratory patterns for species in the area. Seasonal movement will also be hindered in the especially fragmented, hard-edged reserves.

Camera placement

Decisions on camera trap placement were made based on the reserve size, habitat heterogeneity, camera theft risk, limited accessibility (e.g. dense reed beds, cliff faces) and obstructions by infrastructure (e.g. parking lots, building clusters). Maps for each reserve were created in QGIS v2.18.23 (QGIS Development Team 2019) to indicate four major habitat types, namely “fynbos”, “renosterveld”, “strandveld” and “water-associated” (wetland, riparian, coastal, etc.) areas. The fynbos, renosterveld and strandveld classifications were based on the bioregion and water-associated vegetation on the azonal classifications demarcated by Rebelo et al. (2006), as well as satellite imagery (CCT 2019) and ground-truthing when walking the reserve on foot. A one x one km grid layer was projected onto each reserve area within QGIS and adjusted to ensure best fit (i.e. grid alignment was adjusted to ensure each reserve area was covered with the fewest grid squares possible). If this protocol under-represented a major habitat type (i.e. fynbos, strandveld, renosterveld, water-associated and transformed vegetation) from the survey then an additional camera was placed in that habitat type to ensure that habitat specialists were sampled (O'Brien 2008). The number of cameras per reserve was limited to at least five, ensuring that a minimum of 1 000 camera days could be achieved within a maximum period of 200 survey days. In Steenbras Nature Reserve (8 400 ha), the number of cameras available was insufficient to cover the whole reserve at once, so a sub-sample of the total area (which included representation of all habitats) was surveyed at the same camera density.

Camera placement was optimised to detect mammals following the methods of Colyn et al. (2017). This involved searching for sign of mammal presence (e.g. scat, spoor, and foraging signs) within a 120 m radius of the grid point. If no signs were found within 120 m of the grid centre then cameras were placed as close to the grid point as possible without compromising camera safety. Each camera was fixed to a wooden pole with the camera lens at 30 cm above ground level (Tobler et al. 2008) and made to face either north or south so as to prevent false triggers and/or over-exposure from direct sunlight (Figure 3.1). Cameras were set to take a burst of three photographs when triggered, with a delay of 30 seconds between trigger events (Colyn et al. 2017). Due to prevailing wind activity and the need to adequately hide each camera to prevent theft, sensitivity was set to medium and vegetation within a one metre arc of the camera lens was cut to reduce vegetation movement triggering the camera. Cameras were serviced every 20 – 30 days, where secure digital high capacity (SDHC) cards were changed and any potential problems with cameras addressed.

Each reserve was surveyed for a minimum 1000 camera days before cameras were removed/moved to another reserve. This was to enable capture of most species (Si et al. 2014, Colyn et al. 2017). If a species accumulation curve for a specific reserve did not reach an asymptote after a 1000 days, the survey period for that reserve was extended to attempt to reach the number of days required for adequate detection of all species, as time and resources allowed.

Figure 3.1: Remote sensing camera trap setup example - camera trap is secured on a wooden pole at 30 cm above ground level.

3.2.3 Data analysis

3.2.3.1 Survey effort

EstimateS 9.1.0 software (Colwell 2013) was used to generate sample-based species accumulation curves for the recorded native species of each reserve and to determine whether the survey effort in each reserve was sufficient. It was also used to deduce what will be approriate sampling effort for future monitoring (Si et al. 2014, Colyn et al. 2017). The curves show the cumulative number of species recorded over sampling effort and were generated using 1000 randomized runs, with number of samples represented by the number of survey days (Olwell et al. 2004, Mann et al. 2015). Non-parametric species richness estimators (incidence coverage estimator (ICE), Chao 2, Jack 1 and Jack 2) were used to estimate how many species may have been missed during sampling. Accumulation and estimator curves for each reserve were compared to determine whether sampling effort for this study was sufficient and to estimate what can be considered as sufficient survey effort for the future monitoring of each area (Chao & Chiu 2016).

A robust estimate of sampling effort required to adequately survey a reserve was considered to be the point where all four estimators reach an asymptote and where the variance between these four estimators was at its lowest. In this way no particular estimator was favoured over another and it reduced any particular estimator's chance of biasing interpretation.

3.2.3.2 Species richness and community composition

Camera trap data were managed using Camera Base 1.7 software (Tobler 2015). Only photographs of non-burrowing mammals exceeding 0.5 kg in weight were used for analysis. Large mole-rat species (i.e. Cape dune mole-rat *Bathyergus suillus* and Cape mole-rat *Georychus capensis*) were thus excluded. Non-native and reintroduced medium and large mammal species were noted but excluded from richness analyses. The recorded native species were compared to two species lists. The first list comprised of species believed to have been present in the general area at the time of European settlement in 1652 based on historic accounts (Boshoff & Kerley 2001). The second list comprised of species presumed to still be present in the respective reserves based on recorded sightings dating back as far as 2012 (five years before the start of the study) that were either logged on the CCT Biodiversity Database or iNaturalist (https://www.inaturalist.org) and iSpot (https://www.ispotnature.org) citizen science databases.

3.3 Results

The sampling protocol resulted in 151 camera placements (Appendix A) over 1 364 survey days across the 12 reserves totalling an area of 14 429 ha. This resulted in a total of 14 876 camera days (Table 3.2). Five cameras were stolen from five different reserves within the sampling period, and in these cases either a new camera was placed in another location within the reserve for the required survey days or the remaining cameras were left *in situ* for longer to make up for the lost camera days. Throughout the study, a total of 13 360 trigger events by medium and large mammals were recorded.

Table 3.2: Number of cameras placed, number of survey days completed and number of viable camera days achieved for each study site

Nature Reserve	Reserve size (ha)	Cameras (n)	Survey days (n)	Viable camera days (n)
Uitkamp Wetland	32	5	200	1000
Bracken	36	5	200	1000
Kenilworth Racecourse	52	5	200	1000
Zandvlei Estuary	200	10	100	1000
Wolfgat	262	10	100	1000
Tygerberg	388	9	112	1008
Helderberg	402	15	133	2000
False Bay	632	15	67	1005
Table Bay	880	12	84	1008
Blaauwberg	1445	15	67	1005
Witzands Aquifer	1700	20	100	1840
Steenbras	8400	30	67	2010

3.3.1 Survey effort

Native species accumulation and richness estimator curves were calculated for each reserve (Figure 3.2). For Kenilworth Racecourse Conservation Area, only one native species (Cape grysbok - *Raphicerus melanotis*) was recorded in 1000 camera days so no curve could be generated. For nine of the remaining 11 reserves, species richness estimator curves converged during the survey period, suggesting sufficient survey effort was achieved. Generally, species richness estimators (namely ICE, Chao 2, Jack 1 and Jack 2) overestimated the observed species richness within the first 300 camera days before falling within one species difference of each other by 1000 camera days, with the exception of Helderberg, Steenbras and Witzands Aquifer nature reserves.

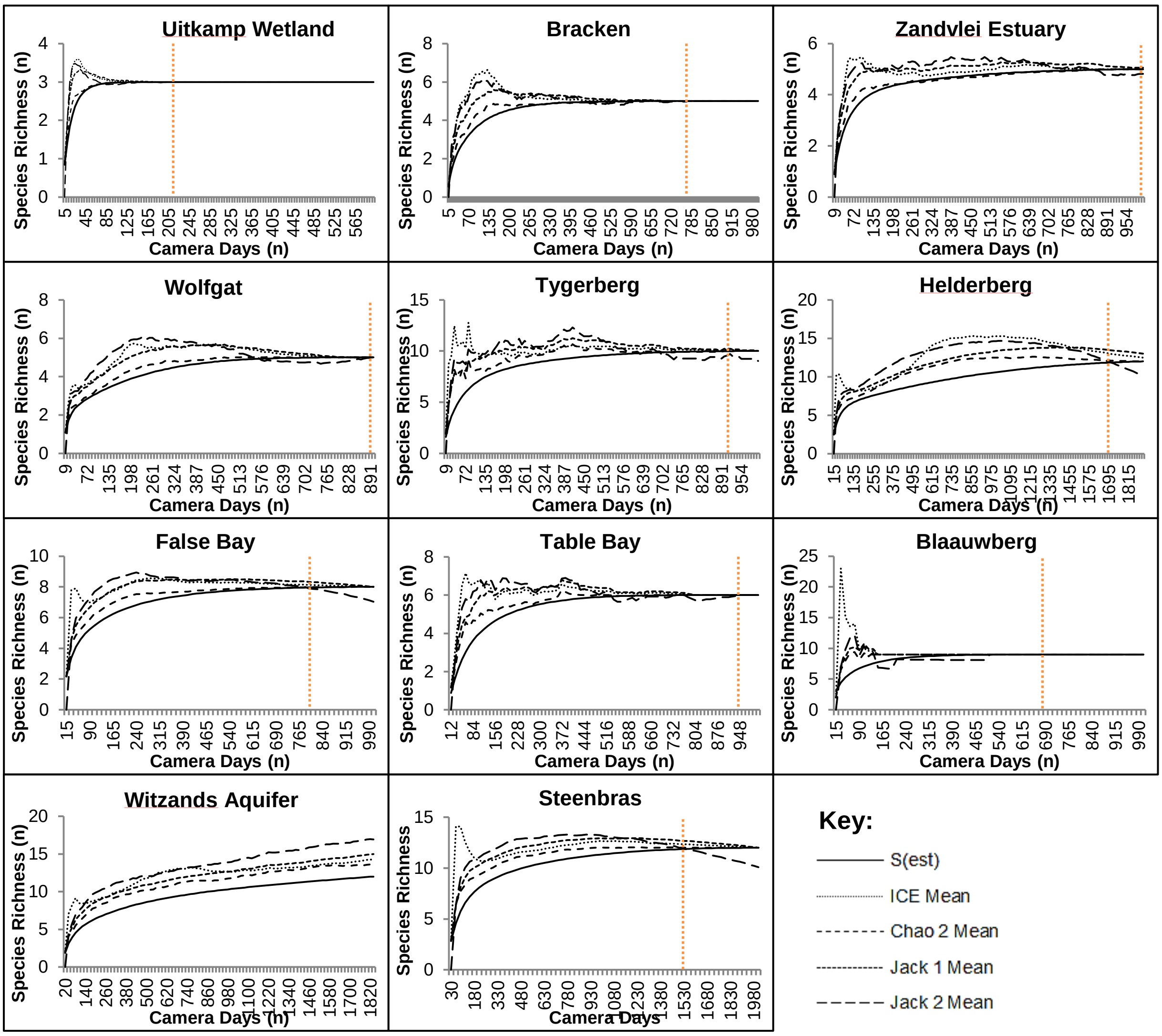


Figure 3.2: Sample-based species accumulation curves [S(est)] and non-parametric species richness estimations for each study site. The orange dotted lines indicate the number of camera days at which variance between all curves is at its lowest.

Curves for Steenbras Nature Reserve converged at 1 980 camera days, although Jack 2 suggested that the number of species present was underestimated by two. For Helderberg Nature Reserve convergence of estimators only started to occur after 1 800 days and for Witzands Aquifer Nature Reserve there was no sign of convergence even at 1 840 days. Helderberg's observed curve fell within one (0.99) species of the estimators, but the Jack 2 mean again underestimated the number of species by two. Witzands Aquifer Nature Reserve's curves indicate sampling was largely insufficient, with the observed accumulation falling short of the lowest estimator by one species and the highest estimator (Jack 2) by five species, which equates to only 70% of the estimated species richness. Overall the Chao 2 mean estimate provided the closest resemblance to the observed species richness, followed by the Jack 2 mean. If the species accumulation curve for Witzands Aquifer is projected at the same trajectory, it suggests that up to 3 000 camera days would be required in order to record the estimated 17 species, but essentially a survey effort of more than 1840 camera days is required.

The number of camera days per reserve required to achieve the minimum variance between species richness estimators at the point where curves asymptote (Table 3.3) ranged from 210 camera days to more than 1840 camera days. When these required camera days are plotted against reserve size (Figure 3.3), a log trendline provides a weak positive correlation ($R^2 = 0.3691$, $p = 0.16$) showing that in smaller reserves, namely Uitkamp Wetland, Bracken and Wolfgat, adequate sampling is reached before 1000 camera days, and larger or more connected reserves such as Steenbras require closer to 2000 camera days in order for sufficient sampling. There are some anomalies, namely the smaller Helderberg Nature Reserve (402 ha) which requires 1710 camera days and larger Blaauwberg Nature Reserve (1440 ha), which requires 660 camera days.

Table 3.3: Lowest variance values between species accumulation and estimator curves and corresponding survey effort (n camera days) per study site, excluding Kenilworth Racecourse and Witzands Aquifer Nature Reserves

Nature Reserve	Reserve size (ha)	Lowest variance value	No. of camera days at lowest variance
Uitkamp Wetland	32	0	210
Bracken	36	0	775
Zandvlei Estuary	200	0.0068	999
Wolfgat	262	<0.0001	891
Tygerberg	388	0.0311	918
Helderberg	402	0.4178	1710
False Bay	632	0.0226	795
Table Bay	880	0	948
Blaauwberg	1445	0	660
Steenbras	8400	0.0915	1530

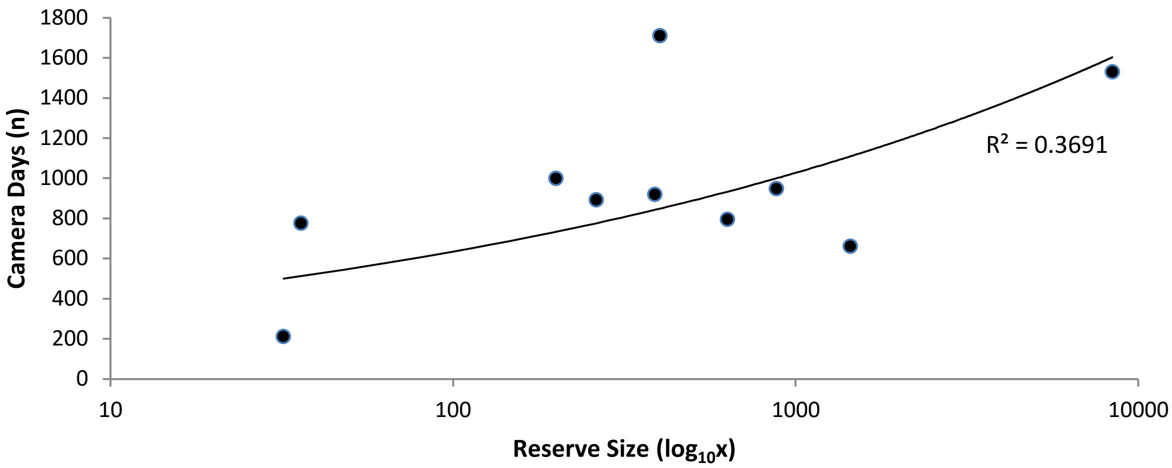


Figure 3.3: Number of camera days with reserve size (in hectares) at which species accumulation curves achieve the lowest variance (excluding Kenilworth Racecourse and Witzands Aquifer Nature Reserves).

3.3.2 Species richness and community composition

Historic/Presumed species

According to historic species lists, 40 medium and large mammal species used to occur within the boundaries of the CCT. Cape hare (*Lepus capensis*) and scrub hare (*Lepus saxatilis*) characteristics were not conspicuous enough to allow differentiation between the species in infrared photographs, so for the purposes of this study, Cape hare and scrub hare will be grouped as "*Lepus* spp.". This left 39 medium and large mammal species with which to compare presumed and recorded results (Appendix B). The total number of native medium and large mammal species still presumed to be present across all reserves and recorded within the past five years before this study commenced (i.e. excluding reintroduced and non-native/domestic species) was calculated at 22 species.

Recorded species

Over the study period, a total of 27 medium and large mammals were recorded across the 12 reserves. Of these, five were non-native species (domestic cat - *Felis sylvestris catus)*, domestic dog - *Canus lupus familiaris*, domestic rabbit - *Oryctolagus cuniculus*, domestic horse - *Equus ferus caballus*, and eastern grey squirrel - *Sciurus carolinensis*), four were reintroduced species (common eland - *Tragelephus oryx*, red hartebeest - *Alcelaphus buselaphus caama* and hippopotamus - *Hippopotamus amphibious*) and 19 were native species. Hewitt's red rock hare (*Pronolagus saundersiae*) was not a species that was predicted to occur in the study area based on database records but was nevertheless recorded in Steenbras Nature Reserve. A total of four species (i.e. grey rhebuck - *Pelea capreolus*, black-backed jackal - *Canis mesomelas*, bat-eared fox - *Otocyon megalotis*, and striped weasel - *Poecilogale albinucha*) were predicted but not recorded in any of the reserves. These results suggest that 49% of the historically occurring species were still present within the surveyed reserve and 86% of the presumed native species richness. When survey results are compared to species expected/recorded on current (2012-2017) databases for each reserve, the survey produced higher species richness totals for seven reserves and lower for the remaining five (Table 3.4). No reserve produced a 100% match between expected and recorded, with "new" species recorded in the majority of the reserves (n = 7). Only Kenilworth Racecourse, Zandvlei Estuary, Wolfgat, False Bay and Blaauwberg nature reserves had fewer observed species than expected. Of the 19 species recorded, 11 were carnivores, seven herbivores and one an ominvore. Small grey mongoose *(Galerella pulverulenta)* occurred in nine of the 12 reserves, large-spotted genet (*Genetta tigrina*) and caracal (*Caracal caracal*) in six of the 12 and the remaining carnivores in five or fewer reserves (Figure 3.4).

Table 3.4: Presumed species richness, recorded species richness and shared species between presumed and recorded lists per study site (see Appendix B for species details)

Nature Reserve	Presumed species richness (n)	Recorded species richness (n)	Shared species (n)
Uitkamp Wetland	2	3	1
Bracken	4	5	4
Kenilworth Racecourse	2	1	1
Zandvlei Estuary	8	5	5
Wolfgat	7	5	5
Tygerberg	6	11	5
Helderberg	11	12	9
False Bay	10	8	8
Table Bay	3	6	2
Blaauwberg	15	9	9
Witzands Aquifer	7	12	7
Steenbras	9	12	8

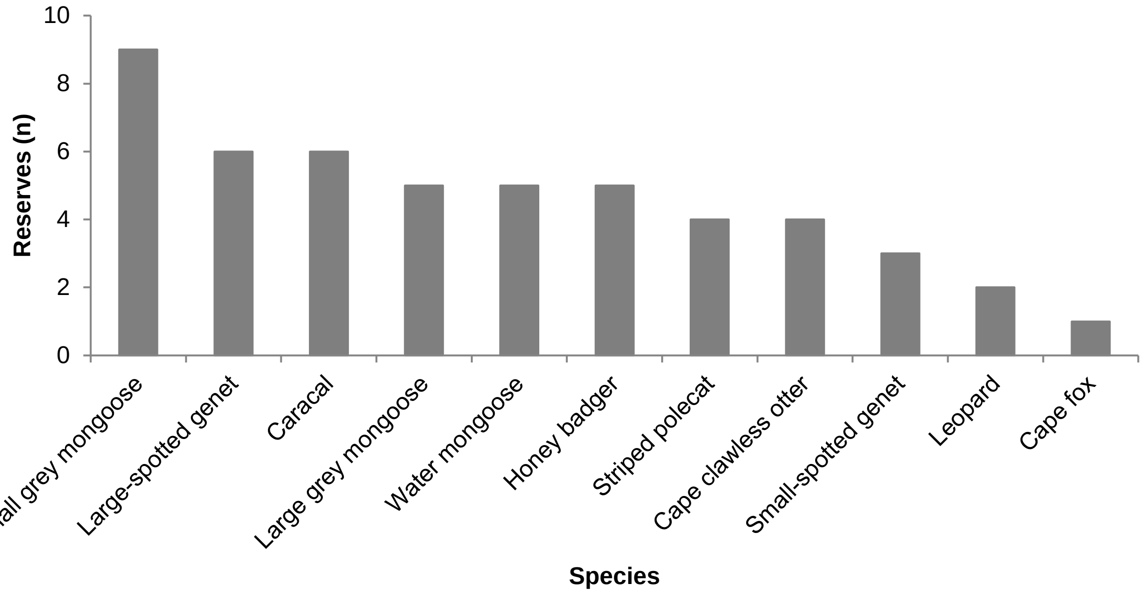


Figure 3.4: Number of reserves in which native carnivore species were recorded

Cape grysbok and Cape porcupine (*Hystrix africaeaustralis*) were the most widely distributed herbivores and were recorded in 11 of the 12 reserves. This was then followed by a marked drop in herbivore common occurance with *Lepus* spp. and common duiker (*Sylvicapra grimmia*) recorded in five reserves and steenbok in four. The rocky habitat associated klipspringer (*Oreotragus oreotragus*) and Hewitt's red rock hare were recorded in only one reserve. The ominvorous chacma baboon (*Papio ursinus*) occurred in two reserves.

3.4 Discussion

With the exception of Witzands Nature Reserve, species accumulation curves and richness estimates indicate that sampling effort was sufficient. Of the 39 medium and large mammal species thought to have historically occurred in the area, a total of 19 (49%) were recorded within the study areas. Larger species such as African elephant (*Loxodonta africana*), black rhinoceros (*Diceros bicornis bicornis*), Cape mountain and plains zebras (*Equus zebra zebra* and *E. quagga*), lion (*Panthera leo*), brown hyena (*Parahyaena brunnea*), spotted hyena (*Crocuta crocuta*) and wild dog (*Lycaon pictus*) were not expected to be recorded, as it is known that they were hunted to local extinction in the 1700s (Rebelo 1992). The absence of large mammals is also to be expected as mammals with large body sizes tend to require large home ranges and are thus more sensitive to habitat loss and fragmentation linked to urban and rural development (McCleery 2010). This trend is particularly true for apex carnivores which generally require extensive connected ranges to complete their life history

(Hansen et al. 2011). Hippopotamus, common eland and red hartebeest, while reintroduced to two reserves (hippopotamus and eland into False Bay Nature Reserve, eland and red hartebeest into Blaauwberg Nature Reserve) were also not expected to be present in local reserves given their large space requirements.

Species potentially present but not recorded over the full study area included grey rhebuck, black-backed jackal, bat-eared fox, yellow mongoose (*Cynictis penicillata*), striped weasel, aardwolf (*Proteles cristata*), aardvark (*Orycteropus afer*) and African wild cat (*Felis sylvestris cafra*). African wild cat was not expected to be recorded in this study, as it is assumed that any surviving cats within the CCT area would have interbred with domestic cats and potentially be locally extinct. Yellow mongoose has been anecdotally sighted at both Tygerberg and Uitkamp Wetland Nature Reserves, but has not been included in any database since 2011 and was not recorded in the study. Aardwolf and aardvark may be disrupted by large-scale development in urban areas with reduced substrate for termite colonies.

Grey rhebuck had previously been reintroduced to both the Tygerberg and Helderberg Nature Reserves to bolster existing population numbers. Despite this the species was not recorded in this study. The last unrecorded sighting in either reserve was in 2017 at Tygerberg Nature Reserve but a local extinction event was thought to have occurred shortly afterward as a carcass was found by reserve staff and no further observations of the species have been made since. This left bat-eared fox, black-backed jackal and striped weasel as presumed to still exist within the CCT area, but not recorded in the study. The last confirmed observation of black-backed jackal within the study sites had been at Blaauwberg Nature Reserve in 2013 which was recorded in the CCT's Biodiversity Database. The current presumed presence of bat-eared fox is largely uncertain as no official records have been logged in any reserve since 2012. Striped weasel was last recorded on the Biodiversity Database at Helderberg Nature Reserve in 2013 and has been recorded elsewhere in the CCT area previously (Child et al. 2016), but it is thought that the study sites may have been at the edge of its range as the majority of sightings tend to occur toward the eastern boundary of the CCT municipal area.

Small grey mongoose, Cape porcupine and Cape grysbok were recorded in the majority of reserves. Globally, species which are able to generalize in terms of food sources and habitat preference such as house mice, squirrels, rabbits, baboons, raccoons, deer and coyotes (De Stefano & De Graaf 2003, Garden et al. 2006, Hoffman & O'Riain 2012, Baker & Harris 2007, Ordeñana et al. 2010, Šálek et al. 2015) seem to thrive in urban areas due to varied combinations of increased food availability and absence of competition and predation pressure from species which are more sensitive to reduced home ranges and disturbance (Newsome et al. 2010, Šálek et al. 2015). This may be likely in the case of porcupine, which readily raid domestic gardens, and small grey mongoose, which are able to predate on a variety of food sources, including small mammals and insects which are readily available in transformed habitats (Cavallini & Nel 1990). Cape grysbok is endemic to the fynbos biome and is a highly selective browser (Kigozi et al. 2008) that may be present mainly due to the availability of fynbos-specific habitat requirements rather than them being adaptable generalists. However, their common occurrence across reserves suggests that they can survive and even thrive within small areas with some protection against anthropogenic pressure.

There was also a higher recorded species richness of carnivores (n = 11) than herbivores (n = 7) at a ratio of 1.6 carnivores to one herbivore species, but that was not very different from historical ratios (1.5:1). Meso-carnivores have been found to be relatively abundant in urban habitat patches, particularly in the absence of large predators which are generally more sensitive to urbanisation. This is due to the larger predators, when present, often restricting meso-carnivore movement, competing with them for resources, or even predating on them (McCleery 2010, Hansen et al. 2011). Despite reduced home ranges and increased population densities, it seems medium-sized omnivores and carnivores are able to adapt feeding and social behaviour to allow for their persistence in human modified landscapes (McCleery 2010, Newsome et al. 2010, Šálek et al. 2015). Herbivores may be less able to adapt their foraging and social behaviour and so may be much more reliant on larger reserves where a greater area of suitable natural habitat may be conserved.

Species richness estimates derived from the Biodiversity Database in conjunction with the iSpot and iNaturalist data are lower at the majority of reserves (n = 7) than recorded using camera trap surveys. This may be because database records are based more on irregular, sporadic and opportunistic sightings than standardised monitoring across sites, as well as the potential for over- or under-recording by individual observers at individual sites. Species which are people-shy may also be under-recorded. By using a standardised sampling method across all study sites, survey effort and results are more comparable than database recordings. Of the 22 species presumed to persist within the reserves, only three were missed (when excluding grey rhebuck due to a possible local extinction event), and each of these were last recorded no later than 2013. This indicates that these species could well be extinct from the study area.

With the exception of Witzands Aquifer, Helderberg and Steenbras Nature Reserves, all species accumulation curves and richness estimators suggested that a survey effort of a

1000 camera days or less is sufficient for the compilation of a medium and large mammal species inventory for the CCT reserves specifically. For the three reserves smaller than 100 ha, it is apparent that survey effort is sufficient well before 1000 camera days. Uitkamp Wetland Nature Reserve's curves reach an asymptote for species richness at 210 camera days, whereas Bracken Nature Reserve (at only 4 more hectares) required 775 camera days. This variation may be related to reserve shape: Uitkamp Wetland has a larger edge in proportion to the total area than Bracken, which provides very little core area for larger mammals (Helzer & Jelinski 1999) and increases detrimental edge effects. Knowing what the minimum survey effort is for these smaller reserves is useful in ensuring surveys can be rolled out effectively without cameras staying *in situ* for longer than necessary, especially in areas where there is a significant risk of theft.

Survey effort in Witzands Aquifer Nature Reserve was insufficient, as indicated by the lack of an asymptote in the species accumulation curve. This is most likely due to single detections of Cape fox (*Vulpes chama*), Cape grysbok and honey badger (*Mellivora capensis*). These results suggest that either survey effort was inadequate, or these species are rare in the study area, possibly moving between reserves and surrounding areas, thus reducing detection probability. Another possible reason for the Witzands Aquifer anomaly is that 44% of the perimeter is bordered by natural habitat (see Chapter 2, Table 2.2), effectively increasing the area of suitable habitat available to wildlife. Together with Helderberg and Steenbras Nature Reserves, it would appear that larger reserves that are well connected to natural areas may increase the effective size of suitable habitat available to wildlife allowing for more transients and lower detection probability of individuals using both the reserve and the neighbouring natural land. While Wolfgat Nature Reserve also has a large proportion of natural boundary, the size and elongated shape of the reserve effectively renders it more of a corridor with reduced core habitat and so required fewer camera days.

3.5 Conclusions

Half of the large- and medium-sized mammal species that historically occurred in the Cape Town area seem to have disappeared since the time Dutch settlers started construction of the first permanent structures 367 years ago. Similar results have been reported in Australia where less than 50% of native species are present within urban areas (McCleery 2010). There are some obvious early casualties associated with urbanisation including mega herbivores (e.g. elephant and black rhino) and large carnivores (e.g. lions, hyena) which all require large areas.

The apparent efficacy of a short, standardised camera trap survey method is an important consideration when routine monitoring is required to measure the impacts of rapid urbanisation on species richness and community composition. To confirm that species not recorded in this study are genuinely absent regular, robust surveys such as those in this study should be repeated regularly. Results from the survey effort trial indicate that up to 1000 camera days is sufficient for most of the studied reserves, but larger more connected reserves require closer to 2000 camera days. The minimum survey effort required as shown by species richness estimators can be used to replicate species presence/absence data and will significantly reduce survey period, especially in smaller reserves. The probability of imperfect detection must be noted (MacKenzie 2005), as reduced survey time lowers the probability of recording rare species, and increased survey effort and more subjective camera placement in specialized habitat would result in a more accurate, albeit biased, species list for reserves. That being said, urban mammal monitoring is time and resource sensitive, and if time, funding, equipment and/or staff availability are limited, using the same camera placements and the minimum survey effort at regular intervals should at least be useful in monitoring species richness trends over time, as long as it is understood that confidence is lost with each “minimum effort” repeat. This could potentially be improved to a certain extent when conducted in conjunction with species richness estimators, verified observations, citizen science and other survey methods, and/or longer surveys (Gotelli & Colwell 2001, MacKenzie 2005).

Not only is accurate and effective recording of species presence imperative for effective conservation efforts, the drivers of distribution patterns must also be understood. It is known that large mammals are the first to be lost with urbanisation and fragmentation, and that omnivores, generalists and meso-carnivores seem better able to adapt to urban spaces (McCleery 2010, Ordeñana et al. 2010, Rovero et al. 2010), but because so many different variables may play varied roles in the persistence of different species, this needs to be further investigated. Fragment size, connectivity, land use practices, disturbance and vegetation also need to be considered to identify associations between mammal species presence, abundance and diversity on an urban landscape scale (Garden et al. 2006, Ordeñana et al. 2010, Rovero et al. 2010). This information can then be used to better streamline conservation efforts on a site-specific and landscape-scale, which forms the basis for the next chapter.

3.6 References

Baker, P.J. & Harris, S. 2007. Urban mammals: What does the future hold? An analysis of the factors affecting patterns of use of residential gardens in Great Britain. *Mammal Review,* 37(4): 297–315.

Boshoff, A.F. & Kerley, G.I.H. 2001. Potential distributions of the medium- to large-sized mammals in the Cape Floristic Region, based on historical accounts and habitat requirements. *African Zoology*, 36(2): 245–273.

Boshoff, A.F., Kerley, G.I.H. & Cowling, R.M. 2001. A Pragmatic Approach to Estimating the Distributions and Spatial Requirements of the Medium- to Large-Sized Mammals in the Cape Floristic Region, South Africa. *Diversity and Distributions*, 7(1): 29–43.

Cavallini, P. & Nel, J.A.J. 1990. The feeding ecology of the Cape grey mongoose, *Galerella pulverulenta* (Wagner 1839) in a coastal area. *African Journal of Ecology*, 28(2):123–130

Ceballos, G. & Ehrlich, P.R. 2006. Global mammal distributions, biodiversity hotspots, and conservation. *Proceedings of the National Academy of Sciences*, 103(51): 19374–19379.

Ceballos, G., Ehrlich, P.R., Soberon, J., Salazar, I. & Fay, J.P. 2005. Global Mammal Conservation: What Must We Manage? *Science*, 309(5734): 603–607.

Chao, A. & Chiu, C.-H. 2016. Nonparametric Estimation and Comparison of Species Richness. *eLS*: 1–11.

Child, M.F., Rowe-Rowe, D., Birss, C., Wilson, B., Palmer, G., Stuart, C., Stuart, M., West, S. & Do Linh San, E. 2016. A conservation assessment of *Poecilogale albinucha*. In: *The Red List of Mammals of South Africa, Swaziland and Lesotho*. Child, M.F., Roxburgh, L., Do Linh San, E., Raimondo, D.& Davies-Mostert, H.T (Ed.) South African National Biodiversity Institute and Endangered Wildlife Trust, Pretoria, South Africa

City of Cape Town (CCT). 2019. *Open Data Portal: Aerial photography*. https://web1.capetown.gov.za/web1/OpenDataPortal/DatasetDetail?DatasetName=Aerial%20photography [25 July 2019].

Colwell, R.K. 2013. *EstimateS version 9.1.0.* Computer software, University of Connecticut, United States of America.

Colyn, R, Radloff, F.G.T. & O'Riain, M.J. 2017. Camera trapping mammals in the scrubland's of the Cape Floristic Kingdom - the importance of effort, spacing and trap placement. *Biodiversity and Conservation*, 27(2): 503-520.

De Stefano, S. & De Graaf, R.M. 2003. Exploring the ecology of suburban wildlife. *Frontiers in Ecology and the Environment*, 1(2): 95–101.

Garden, J., McAlpine, C., Peterson, A., Jones, D. & Possingham, H. 2006. Review of the ecology of Australian urban fauna: A focus on spatially explicit processes. *Austral Ecology*, 31: 126–148.

Giman, B., Stuebing, R., Megum, N., McShea, W.J. & Stewart, C.M. 2007. A camera trapping inventory for mammals in a mixed use planted forest in Sarawak. *Raffles Bulletin of Zoology*, 55(1): 209–215.

Gotelli, N.J. & Colwell, R.K. 2001. Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. *Ecology Letters*, 4: 379–391.

Hansen, A.J., Knight, R.L., Marzluff, J.M., Powell, S., Gude, P.H. & Jones, K. 2011. Effects of Exurban Development on Biodiversity: Patterns, Mechanisms, and Research Needs. *Ecological Applications*, 15(6): 1893–1905.

Helzer, C.J. & Jelinski, D.E. 1999. The relative importance of patch area and perimeter-area ratio to grassland breeding birds. *Ecological Applications*, 9(4): 1448–1458.

Hoffman, T.S. & O'Riain M.J. 2012. Troop size and human-modified habitat affect the ranging patterns of a chacma baboon population in the Cape peninsula, South Africa. *American Journal of Primatology*, 74(9): 853-863.

Kelly, M.J. 2008. Design, evaluate, refine: Camera trap studies for elusive species. *Animal Conservation*, 11(3): 182–184.

Kerley, G.I.H., Pressey, R.L., Cowling, R.M., Boshoff, A.F. & Sims-Castley, R. 2003. Options for the conservation of large and medium-sized mammals in the Cape Floristic Region hotspot, South Africa. *Biological Conservation*, 112(1-2): 169–190.

Kigozi, F., Kerley, G. I. H. & Lessing, J. S. 2008. The diet of Cape grysbok (*Raphicerus melanotis*) in Algoa Dune Strandveld, Port Elizabeth, South Africa. *South African Journal of Wildlife Research*, 38(1): 79-81.

MacKenzie, D.I. 2005. What Are the Issues with Presence-Absence Data for Wildlife Managers? *Journal of Wildlife Management*, 69(3): 849–860.

Mann, G.K.H., Lagesse, J. V., O'Riain, M.J. & Parker, D.M. 2015. Beefing Up Species Richness? The Effect of Land-Use on Mammal Diversity in an Arid Biodiversity Hotspot. *African Journal of Wildlife Research*, 45(3): 321–331.

McCleery, R. 2010. Urban Mammals. In: *Urban Ecosystem Ecology*, Agronomy Monographs. 55. ASA, CSSA, SSSA, Madison, WI. 87-102

McDonald, R.I., Kareiva, P. & Forman, R.T.T. 2008. The implications of current and future urbanization for global protected areas and biodiversity conservation. *Biological Conservation*, 141(6): 1695–1703.

Newsome, S.D., Ralls, K., Job, C.V.H., Fogel, M.L. & Cypher, B.L. 2010. Stable isotopes evaluate exploitation of anthropogenic foods by the endangered San Joaquin kit fox (*Vulpes macrotis mutica*). *Journal of Mammalogy*, 91(6): 1313–1321.

O'Brien, T.G. 2008. On the use of automated cameras to estimate species richness for large- and medium-sized rainforest mammals. *Animal Conservation*, 11(3): 179–181.

O'Connell, A., Nichols, J. & Karanth, K. U. 2010. *Camera traps in animal ecology: Methods and analyses*. A. F. O'Connell, J. D. Nichols, & K. Ullas Karanth (Ed.). Springer.

Olwell, R.O.K.C., Ao, C.H.X.U.A.N.M. & Hang, J.I.N.G.C. 2004. Interpolating , Extrapolating , and Comparing Incidence-Based Species Accumulation Curves. *Ecology*, 85(10): 2717–2727.

Ordeñana, M. A, Crooks, K.R., Boydston, E.E., Fisher, R.N., Lyren, L.M., Siudyla, S., Haas, C.D., Harris, S., Hathaway, S. A, Turschak, G.M., Miles, A.K. & Van Vuren, D.H. 2010. Effects of urbanization on carnivore species distribution and richness. *Journal of Mammalogy*, 91(6): 1322–1331.

Pekin, B.K. & Pijanowski, B.C. 2012. Global land use intensity and the endangerment status of mammal species. *Diversity and Distributions*, 18(9): 909–918.

Pressey, R.L., Cowling, R.M. & Rouget, M. 2003. Formulating conservation targets for biodiversity pattern and process in the Cape Floristic Region, South Africa. *Biological Conservation*, 112(1-2): 99–127.

Rebelo, A.G. 1992. Red Data Book species in the Cape Floristic Region: threats, priorities and target species. *Transactions Royal Society of South Africa,* 48(1): 55–86.

Rebelo, A.G., Holmes, P.M., Dorse, C. & Wood, J. 2011. Impacts of urbanization in a biodiversity hotspot: Conservation challenges in Metropolitan Cape Town. *South African Journal of Botany*, 77(1): 20–35.

Rovero, F., Tobler, M. & Sanderson, J. 2010. Camera trapping for inventorying terrestrial vertebrates. In: *Manual on field recording techniques and protocols for All Taxa Biodiversity Inventories and Monitoring*. Eymann, J., Degreef, J., Häuser, C., Monje, J.C., Samyn, Y., Van den Spiegel, D. (Ed.). Abc Taxa, Vol. 8 (Part 1), 100 –128.

Rowcliffe, J.M., Field, J., Turvey, S.T. & Carbone, C. 2008. Estimating animal density using camera traps without the need for individual recognition. *Journal of Applied Ecology*, 45: 1228–1236.

Šálek, M., Drahníková, L. & Tkadlec, E. 2015. Changes in home range sizes and population densities of carnivore species along the natural to urban habitat gradient. *Mammal Review*, 45(1): 1–14.

Si, X., Kays, R. & Ding, P. 2014. How long is enough to detect terrestrial animals? Estimating the minimum trapping effort on camera traps. *PeerJ*, 2: e374.

Silveira, L., Jácomo, A.T.A. & Diniz-Filho, J.A.F. 2003. Camera trap, line transect census and track surveys: A comparative evaluation. *Biological Conservation*, 114(3): 351–355.

Tobler, M. 2015. *Camera Base 1.7*. http://www.atrium-biodiversity.org/tools/camerabase [4 April 2019].

Tobler, M.W., Carrillo-Percastegui, S.E., Leite Pitman, R., Mares, R. & Powell, G. 2008. An evaluation of camera traps for inventorying large- and medium-sized terrestrial rainforest mammals. *Animal Conservation*, 11(3): 169–178.

Trolle, M. & Kery, M. 2005. Camera-trap study of ocelot and other secretive mammals in the northern Pantanal. *Mammalia*, 69(3-4): 405-412.

Underwood, E.C., Viers, J.H., Klausmeyer, K.R., Cox, R.L. & Shaw, M.R. 2009. Threats and biodiversity in the Mediterranean biome. *Diversity and Distributions*, *15*(2): 188-197.

Visconti, P., Pressey, R.L., Giorgini, D., Maiorano, L., Bakkenes, M., Boitani, L., Alkemade, R., Falcucci, A., Chiozza, F. & Rondinini, C. 2011. Future hotspots of terrestrial mammal loss. *Philosophical Transactions of the Royal Society of London B: Biological Sciences*, 366(1578): 2693–2702.

CHAPTER FOUR:
VARIATION IN MEDIUM AND LARGE MAMMAL COMMUNITIES IN CITY OF CAPE TOWN NATURE RESERVES

4.1 Introduction

Medium and large mammal species richness is an important indicator of ecosystem functioning and health (Augustine & McNaughton 1998, Kerley et al. 2003, Ordeñana et al. 2010). To ensure healthy mammal species richness in protected areas it is important to understand what factors negatively influence species richness patterns (Ramesh et al. 2016, Gonçalves et al. 2018). This in turn demands effective monitoring of mammals, an understanding of their ecological role and the subsequent prioritization of management and conservation actions that seek to improve or maintain richness and community composition (Ramesh et al. 2016).

At a global scale climatic variables such as temperature and evapotranspiration are the most important correlates of mammal species richness (Torres-Romero & Olalla-Tárraga 2015, Ramesh et al. 2016). Limits of temperature tolerance constrain the geographic range of most mammal species (Ramesh et al. 2016), while evapotranspiration influences primary productivity and bottom up processes (Torres-Romero & Olalla-Tárraga 2015). At both a regional and a landscape scale, climate and geological features are the most important drivers of species richness patterns (Torres-Romero & Olalla-Tárraga 2015, Ramesh et al. 2016). On smaller scales, anthropogenic activity has influenced species richness patterns at all scales due to habitat loss and fragmentation, pollution, alien species introduction, resource depletion and human-wildlife conflict (Ceballos & Ehrlich 2006, McDonald et al. 2008, Ordeñana et al. 2010, Visconti et al. 2011). Most large cities globally are located in areas of high biodiversity and endemism, and consequently urban land use has had a disproportionate impact on mammal species richness (McDonald et al. 2008, Visconti et al. 2011, Pekin & Pijanowski 2012). Despite this many species persist in small urban and peri-urban reserves with select species thriving in human modified habitats (De Stefano & De Graaf 2003, Garden et al. 2006, Baker & Harris 2007, Ordeñana et al. 2010, Hoffman & O'Riain 2012, Šálek et al. 2015). Urbanisation also tends to affect trophic level dynamics within the urban environment and human activity often supplements food sources (Pickett et al. 2011, Saito & Koike 2013). This increases the abundance of certain species and can eliminate others, changing trophic level structure and community composition (De Stefano & De Graaf 2003, Picket et al. 2011).

Understanding which species survive in such reserves and how they are impacted by varying levels of fragmentation, isolation and neighbouring land use is important to ensure the persistence of the last remaining species and ecosystem functioning in the remaining natural spaces (Fischer and Lindenmayer 2007, Ramesh et al. 2016). In general the species richness of small, fragmented and isolated reserves is affected by fragment size (Diamond 1975, De Stefano & De Graaf 2003, Kerley et al. 2003, Ceballos et al. 2005, Visconti et al. 2011, Matthies et al. 2017, Gonçalves et al. 2018), fragment shape (Diamond 1975), habitat heterogeneity (Ramesh et al. 2016, Matthies et al. 2017), connectivity to additional suitable habitat (Diamond 1975, Stevens et al. 2006, Correa Ayram et al. 2016) and surrounding land use (De Stefano & De Graaf 2003, Ceballos et al. 2005, McDonald et al. 2008, Visconti et al. 2011, Pekin & Pijanowski 2012, Mann et al. 2015, Torres-Romero & Olalla-Tárraga 2015, Gonçalves et al. 2018).

Garden et al. (2006) reviewed demographic data on urban mammal species, and found that population size at a landscape level was significantly affected by patch size, vegetation and habitat type, and fragmentation. Patch size affects the number of individuals and species it can support (Diamond 1975, Turgeon & Kramer 2012) and the smaller the patch becomes, the more likely local extinction events will occur as risk of extinction is usually related to population size (Diamond 1975). For this reason, one large patch is generally preferable to many small patches over the landscape (Diamond 1975).

Patch shape also plays an important role in species presence and therefore richness (Diamond 1975, Ramesh et al. 2016, Matthies et al. 2017). The shape of an area determines the amount of core habitat available and the amount of exposed edge, with a longer edge in relation to core area allowing for more exposure to anthropogenic effects in an urban context (Herse et al. 2018). Some species respond positively to increased edge and the resulting variation in habitat (Diamond 1975), but many species are sensitive to edge effects and so require a larger core area (Hardt and Forman 1989, Ramesh et al. 2016).

Connectivity refers either to the spatial relationship between patches in a landscape (structural connectivity) or the ability of the landscape to facilitate the movement of species between patches (functional connectivity), and is considered important for species population persistence within fragmented landscapes (Diamond 1975, Correa Ayram et al. 2016). When patch size is small, a species' persistence may rely solely on its ability to disperse, which depends largely on the type of habitat adjacent to and between patches (Diamond 1975, Brooker et al. 1999, Söndgerath & Schröder 2002, Turgeon & Kramer 2012). Diamond et al. (1975) and Turgeon and Kramer (2012) suggest that the number of species a patch can hold is a balance between rates of extinction and immigration, i.e. if animals are able to move into and between areas, local extinction may be reduced. Connectivity is also affected by permeability of patch boundaries (Stevens et al. 2006). Physical barriers (e.g., walls, fences) are important for reducing collisions with vehicles and unregulated access by domestic animals and people. However such barriers also prevent movement between patches (Garden et al. 2006, Ordeñana et al. 2010).

Mammal community composition should also be considered as certain species may be more vulnerable to, or alternatively more adaptable to the effects of urbanisation (Bateman & Fleming 2012). This means that naturally-occurring species community composition might be disrupted, influencing how species interact with each other and the environment (Lowry et al. 2013). Large predatory mammals tend to have large home ranges which are invariably reduced and fragmented by urban development (Kerley et al. 2003, Ordeñana et al. 2010). The removal of large predators reduces both competition and predation for smaller predators and prey species, which may then persist at higher numbers (Bateman & Fleming 2012, Lowry et al. 2013). Species which typically thrive in human transformed areas, such as house mice, foxes, rabbits, raccoons, deer and coyotes (De Stefano & De Graaf 2003, Garden et al. 2006, Ordeñana et al. 2010, Šálek et al. 2015), tend to be those which are able to generalize in terms of food sources and habitat preference, and further benefit from the absence of competition and predation (Hoffmann & O'Riain 2012, Šálek et al. 2015). Certain non-native species (mostly domestic species) also respond positively (Kerley et al. 2003, Ordeñana et al. 2010), indicating that another implication of urbanisation is an increase in alien invasive fauna with subsequent effects on local mammal populations (Bernardo & Melo 2013).

Mammal conservation in the City of Cape Town area, internationally recognised for its exceptional floral species richness (Rebelo et al. 2011), is particularly challenging. Flat, fertile soils in proximity to freshwater were rapidly developed firstly for farming and subsequently for housing (Anderson & O'Farrell 2012). Most wildlife populations are thus restricted to marginal habitats including wetlands and mountain habitat where development costs were high or prohibited (Anderson & O'Farrell 2012). Consequently only small, fragmented and irregularly-shaped patches were set aside as nature reserves within the Cape Flats region of Cape Town (Rebelo et al. 2011). Despite this fragmentation and low proportion of protected habitat (approximately 9% excluding Table Mountain National Park), the CCT area still maintains an extraordinary wealth of biodiversity as highlighted in April 2019 when citizen science aided in the recording of 4 157 individual fauna, flora and fungal species to win the iNaturalist City Nature Challenge (iNaturalist Network 2019).

To best conserve the medium and large mammal species still remaining within the CCT boundary and more specifically the reserves of the CCT, a clear understanding of current species richness patterns and the potential drivers thereof is necessary. In this chapter the species composition results obtained for the camera trap study conducted across 12 of the CCT nature reserves (Chapter 3) are used to explore species richness patterns and the drivers thereof.

4.2 Methodology

4.2.1 Study sites

Eleven of the 12 City of Cape Town nature reserves surveyed using camera traps in Chapters 2 and 3 were used in this study. Kenilworth Race course was excluded as it only had one species – *Raphicerus melanotis* (Cape grysbok) – present which was reintroduced in the late 2000s.

4.2.2 Species richness

The total number of medium and large mammal species estimated to be present in each of the 11 nature reserves was established using camera trap surveys at each site (see Chapter 3 for detail). Only native medium and large mammal species believed to have persisted naturally without the aid of reintroduction are included in this study. Each nature reserve was sampled for a minimum of 1 000 camera days and if species richness estimators (ICE, Chao 2, Jack 1, Jack 2) generated from downloaded data did not indicate that all potential species had been recorded, effort was increased to a maximum of 2 010 days.

Species richness estimates were considered robust when all four estimators had reached an asymptote and were the same or lower than the observed richness values. Where the observed species richness value was lower than species richness estimators after maximum survey effort, the mean species richness estimate at that maximum survey effort was used for further analysis.

4.2.3 Species richness predictor variables

Satellite imagery (City of Cape Town 2019a), vegetation shapefiles (SANBI 2016) and official CCT mapped boundaries (City of Cape Town 2019b) together with ground-truthing and reserve manager liaison were used to extract variables hypothesised to influence species richness patterns in small urban reserves. Variables included reserve size (Diamond 1975, Matthies et al. 2017), area-perimeter ratio (Diamond et al. 1975, Lagro 1991, Helzer &

Jelinski 1999, Ewers & Didham 2007, Herse et al. 2018), vegetation heterogeneity (Ramesh et al. 2016, Matthies et al. 2017), connectivity (Diamond 1975, Stevens et al. 2006, Turgeon & Kramer 2012, Correa Ayram et al. 2016), and permanent freshwater aquatic habitat. Larger, more connected reserves with a greater area-perimeter ratio and habitat heterogeneity and the presence of permanent water were predicted to individually and collectively have a positive effect on species richness.

Reserve size and area-perimeter ratio

The perimeter (km) and area (km^2) of each reserve were calculated using QGIS v2.18.23 software (QGIS Development Team 2019). An area-perimeter ratio was then determined for each reserve by dividing the area of the reserve by the perimeter length. This proportion is considered to be indicative of the ratio of core habitat relative to edge habitat (Helzer & Jelinski 1999) with high ratio values indicating a greater proportion of interior habitat relative to edge habitat.

Vegetation heterogeneity

Vegetation across all of the CCT reserves, excluding human infrastructure, was divided into five broad vegetation categories. The bioregion descriptions of Rebelo et al. (2006), namely Southwest Fynbos, West Coast Renosterveld and West Strandveld, were condensed into “fynbos”, “renosterveld” and “strandveld” categories respectively. Wetland or coastal (azonal) vegetation was classed as “water-associated” vegetation (Rebelo et al. 2006) and transformed vegetated areas (i.e. mowed grassland areas or plantation) was included as the fifth category labelled “transformed” vegetation.

The area (ha) of each of the five habitat types within each reserve was calculated using GIS vegetation shapefiles (SANBI 2016) and satellite photography for transformed areas (see Chapter 3). These absolute values were then used to calculate a Shannon-Wiener diversity index for each reserve to appraise large-scale vegetation heterogeneity (Matthies et al. 2017) using the formula:

$$\boldsymbol{H' = -\sum_{i=1}^{s} p_i \ln p_i}$$

where H = habitat diversity index, s = number of habitat types present, and p_i = proportion of the total area (ha) of the reserve (Shannon 1948). The higher the index value, the more diverse or heterogeneous the reserve is believed to be from vegetation type perspective.

Connectivity

There are a number of methods available to quantify both structural and functional connectivity between two potentially habitable areas (Stevens et al. 2006). These methods can be complex and depend largely on the arrangement of patches in relation to the size, shape and quality of corridors within a larger landscape matrix, as well as the size and dispersal behavioural of the species within reserves (Brooker et al. 1999, Tischendorf & Fahrig 2000, Kindlmann & Burel 2008, Crooks et al. 2011, Bateman & Fleming 2012, Correa Ayram et al. 2016). Because Cape Town is already densely developed, most of the few existing corridors connect reserves to patches which are often smaller than the reserves themselves, making traditional connectivity measures difficult. For the purposes of this study, connectivity is simply quantified as the length (km) of reserve boundary directly adjacent to natural or ecologically functional habitat that can facilitate the movement of medium to large mammals to or from the reserve. Disturbed or transformed non-native vegetation was also considered as functional habitat if perceived to mimic the structure of native vegetation and hence provide some form of cover as a refuge and to facilitate movement (e.g. stands of invasive alien plants).

Permanent freshwater aquatic habitat

If a reserve possessed at least one permanent wetland such as a perennial river/stream and/or large waterbody (dam, wetland, etc.) it was considered as having suitable habitat for a water-associated medium to large mammal species (e.g. Cape Clawless otter – *Aonyx capensis*). Results were classified as binary indicator variables, with presence of permanent waterbodies indicated by a score of 1 and absence by a score of 0. The consideration of perennial water sources rather than non-perennial prevented the under-representation of water-associated species such as otter.

4.2.4 Analyses

Non-metric dimensional scaling (NMDS) ordination was conducted using the vegan package in R v3.5.3 (R Core Team 2019) to provide a visual representation of similarities between reserves based on medium and large mammal community structure (using Jaccard index for species presence/absence) and the species richness predictor variables. Pairwise plots and correlation coefficients were used to determine which covariates may be correlated with species richness, as well as to identify potential covariate collinearity. Collinearity between covariates was tested for in the *car* package (Fox & Weisberg 2019) using variance inflation factors (VIF) with species richness as the response variable. Any covariates with VIF scores of >5 and correlation coefficients of >7 were considered to have collinearity and were modelled separately (Dormann et al. 2013).

Because covariates contained no random effects and only one value per site, simple linear regression models were used to assess drivers of species richness. Linear regression models were run using various combinations of covariates and due to small sample size (n = 11) only two covariates were included in each model. Models were ranked according to second-order Akaike information criterion (AICc) calculated using the *AICcmodavg* package (Mazerolle 2019). The models with the lowest ∆AICc scores (difference between the model and lowest AICc score) were considered likely to predict species richness (Burnham et al. 2004). P-values and F-statistics were also compared to verify which model was most parsimonious in predicting species richness.

4.3 Results

4.3.1 Species richness

Nineteen native, medium to large mammal species were recorded across the 11 nature reserves (Table 4.1). Species richness estimators reached asymptotes and matched observed species richness at all reserves except at Helderberg and Witzands nature reserves (Chapter 3, Figure 3.2), so mean species richness estimates for these were calculated at 13 and 15 respectively.

Estimated medium and large mammal species richness across the 11 study sites ranged from three (Uitkamp Wetland Nature Reserve) to 15 species (Witzands Aquifer Nature Reserve). There was also a higher recorded species richness of carnivores (n = 11) than herbivores (n = 7), and only one omnivore (chacma baboon). Mean and median estimated species richness across the reserves was eight species, with the mode at five species. Cape porcupine was present in all 11 reserves, with Cape grysbok and small grey mongoose also being quite common with records in 10 and nine of the reserves respectively. Klipspringer and Hewitt's red rock hare were only recorded in Steenbras Nature Reserve and Cape fox only in Witzands Nature Reserve.

Table 4.1: Species presence (marked “X”), observed species richness and estimated species richness (based on estimators ICE, Chao 2, Jack 1, Jack 2 - see text) for each of the 11 City of Cape Town Nature Reserves included in this study. Species are listed in order of most common occurrence and diet is indicated by “H” (herbivore), “C” (carnivore) and “O” (omnivore). Cape and scrub hare are grouped for the purposes of this study. Reserves are listed in order of species richness.

Species	**Common Name**	**Diet**	**Uitkamp**	**Bracken**	**Zandvlei**	**Wolfgat**	**Table Bay**	**False Bay**	**Blaauwberg**	**Tygerberg**	**Steenbras**	**Helderberg**	**Witzands**
Hystrix africaeaustralis	Cape porcupine	H	X	X	X	X	X	X	X	X	X	X	X
Raphicerus melanotis	Cape grysbok	H		X	X	X	X	X	X	X	X	X	X
Galerella pulverulenta	Small grey mongoose	C		X	X	X		X	X	X	X	X	X
Genetta tigrina	Large spotted genet	C			X	X		X		X	X	X	
Caracal caracal	Caracal	C					X	X	X	X		X	X
Sylvicapra grimmia	Common duiker	H							X	X	X	X	X
Lepus capensis/saxatilis	Cape/scrub hare	H				X		X			X	X	X
Herpestes ichneumon	Large grey mongoose	C	X				X			X		X	X
Mellivora capensis	Honey badger	C							X	X	X	X	X
Atilax paludinosus	Water mongoose	C	X		X		X	X		X			
Aonyx capensis	Cape clawless otter	C					X	X		X	X		
Ictonyx striatus	Striped polecat	C		X					X			X	X
Raphicerus campestris	Steenbok	H		X					X				X
Genetta genetta	Small spotted genet	C							X	X			X
Panthera pardus	Leopard	C									X	X	
Papio ursinus	Chacma baboon	O									X	X	
Oreotragus oreotragus	Klipspringer	H									X		
Pronolagus saundersiae	Hewitt's red rock hare	H									X		
Vulpes chama	Cape fox	C											X
Observed species richness			3	5	5	5	6	8	9	11	12	12	12
Estimated species richness			3	5	5	5	6	8	9	11	12	13	15

4.3.2 Species richness predictor variables

Reserve sizes ranged from 0.32 to 84 km^2 (mean = 13.07 ± 23.03) (Table 4.2). Area to perimeter proportions varied from 0.05 for Uitkamp to 1.69 for Steenbras (mean = 0.46 ± 0.48), with values for the largest reserves (Witzands and Steenbras) showing the largest area in proportion to reserve edge (>1). Blaauwberg, Witzands and Steenbras are the only three reserves with an area to perimeter proportion of more than 0.6. Uitkamp Wetland Nature Reserve had the lowest connectivity with only 0.79 km (11%) of its boundary bordering functional habitat suitable for dispersal. In comparison, Witzands shares 22.6 km (81%) of its border with suitable habitat.

Table 4.2: Species richness predictor variable values for the respective reserves. Area to perimeter ratio was calculated using reserve size (km^2) relative to boundary length (km), connectivity refers to the distance of boundary line shared with functional dispersal habitat, habitat heterogeneity of the respective reserves was calculated from the proportional representation of five different habitat types present in each reserve and expressed as a Shannon-Wiener diversity index value, the presence-1 or absence-0 of permanent freshwater aquatic habitat for aquatic or semi-aquatic mammals are indicated under aquatic habitat (see text for more detail).

Nature Reserve	Size (km^2)	Area : Perimeter ratio	Connectivity (km)	Heterogeneity (SWDI)	Aquatic habitat
Uitkamp	0.32	0.05	0.79	0.54	1
Bracken	0.36	0.14	1.13	0.57	0
Zandvlei	2.00	0.11	2.33	0.89	1
Wolfgat	2.62	0.29	1.72	0.00	0
Tygerberg	3.88	0.22	2.29	0.62	1
Helderberg	4.02	0.36	5.66	0.29	1
False Bay	6.32	0.28	8.84	1.28	1
Table Bay	8.80	0.24	4.93	0.51	1
Blaauwberg	14.45	0.61	14.18	0.90	0
Witzands	17.00	1.09	22.60	0.13	0
Steenbras	84.00	1.69	47.69	0.47	1
Mean	13.07	0.46	10.20	0.56	-
SD	23.03	0.48	13.46	0.35	-

Vegetation heterogeneity scores varied from 1.28 in Table Bay to 0 for Wolfgat which had only one habitat type present (strandveld). The mean vegetation heterogeneity score was 0.56 ± 0.35. Permanent freshwater aquatic habitat was present at seven of the 11 reserves.

4.3.3 Non-metric multidimensional scaling of predictor variables

The NMDS ordinance revealed a strong association between reserve size, connectivity and area-perimeter ratio as explanatory variables (Figure 4.1), all of which are highest in Steenbras and Helderberg. These two reserves had nine species in common, five of which (klipspringer, Hewitt's red rock hare, leopard – *Panthera pardus*, chacma baboon and honey badger) had a strong relationship with the variables area-perimeter ratio, connectivity and reserve size.

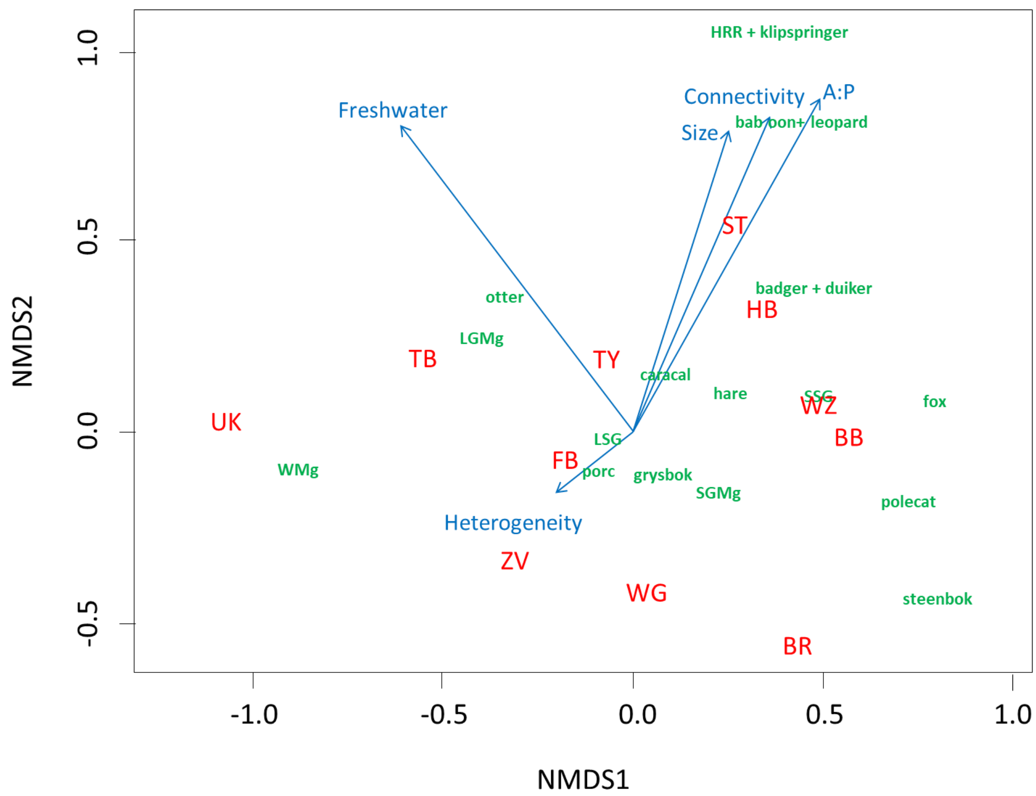


Figure 4.1: Non-metric dimensional scaling (NMDS) plot of reserve similarity using species presence and species richness predictor covariates. Reserves are indicated by capital letters (red): UK (Uitkamp Wetland), TB (Table Bay), FB (False Bay), TY (Tygerberg), ZV (Zandvlei Estuary), WG (Wolfgat), ST (Steenbras), HB (Helderberg), BR (Bracken), WZ (Witzands Aquifer) and BB (Blaauberg). Blue lines indicate the strength of predictor variables: Heterogeneity (vegetation heterogeneity), Freshwater (presence of permanent freshwater aquatic habitat), Size (reserve size), Connectivity (boundary connectivity) and A:P (area-perimeter ratio). Species are indicated in green text with abbreviations: WMg (water mongoose), LMg (large grey mongoose), SMg (small grey mongoose), LSG (large-spotted genet), SSG (small-spotted genet), otter (Cape clawless otter), porc (porcupine), grysbok (Cape grysbok), fox (Cape fox), polecat (striped polecat), hare (Cape/scrub hare), HRR (Hewitt's red rock hare), baboon (chacma baboon), badger (honey badger) and duiker (common duiker). A "+" indicates that species overlap, i.e. have the same centre point.

Uitkamp Wetland Nature Reserve, with the smallest area, the lowest area-perimeter ratio and very low connectivity, does not associate closely with any other reserves. However both freshwater habitat and the presence of large grey mongoose make it most similar to Table Bay and Tygerberg Nature Reserves. False Bay, Zandvlei and Wolfgat cluster at the bottom of the plot and all three reserves comprise of large proportions of strandveld vegetation and have four mammal species (porcupine, grysbok, small grey mongoose, large-spotted genet) in common. Witzands and Blaauwberg group together with high area-perimeter ratio and connectivity values. Together with Bracken, Witzands and Blaauwberg reserves have no permanent freshwater aquatic habitat and share the rarer steenbok and striped polecat. Unsurprisingly the NMDS plot shows a strong association between Cape clawless otter and reserves with permanent fresh water aquatic habitat. Large grey mongoose clusters close to otters but is not known to associate closely with aquatic habitat (Palomares & Delibes 1990).

Strong correlations are evident between area-perimeter ratio, connectivity and reserve size (correlation coefficients of >0.9, Table 4.3) and support the NMDS plot that places these three covariates together. This is confirmed by the VIF scores for reserve size, area-perimeter ratio and connectivity which produce values >5, indicative of strong collinearity. These covariates were thus modelled separately from each other (Table 4.3). The correlation matrix also indicates a significant positive correlation between area-perimeter ratio and species richness (0.675).

Table 4.3: Correlation coefficients between the response variable (species richness) and predictor variable data ranges, namely reserve size, heterogeneity, permanent freshwater aquatic habitat ("water"), area-perimeter ratio and boundary connectivity (see Table 4.2 for values). Variance inflation factors (VIF) for each predictor variable indicate covariate collinearity if greater than 5.

	Species richness	Reserve size (Ha)	Hetero-geneity	Freshwater	A-P ratio	Connectivity
Species Richness	1	-	-	-	-	-
Reserve Size	0.437	1	-	-	-	-
Heterogeneity	-0.237	-0.097	1	-	-	-
Water	-0.028	0.146	0.355	1	-	-
A-P Ratio	0.675	0.911	-0.254	-0.114	1	-
Connectivity	0.591	0.961	-0.137	0.004	0.983	1
VIF score	N/A	36.137	2.741	1.777	167.945	295.770

Eight different linear regression models were run using individual variables first in order to rank model fit and thus discard the least significant collinear models. Additional models were then run using combinations of the remaining covariates (Table 4.4). Due to small sample size (n = 11), no more than two covariates were combined in models.

Table 4.4: Models ranked according to AICc scores and other selection criteria (p-value, F-statistic and residual standard error), with species richness as the response variable and area-perimeter ratio, connectivity, reserve size, presence of permanent freshwater aquatic habitat and heterogeneity as covariates (see Table 4.2 for values).

Model formula	AICc	∆AICc	P-value	F-Stat	SE
SR = $\beta_0 + \beta_1$A-P Ratio	63.015	0	0.023	7.547	3.056
SR = $\beta_0 + \beta_1$Connectivity	64.981	1.966	0.055	4.839	3.342
SR = $\beta_0 + \beta_1$Reserve Size	67.385	4.37	0.179	2.122	3.728
SR = $\beta_0 + \beta_1$A-P Ratio + β_2Heterogeneity	68.160	5.145	0.085	3.417	3.228
SR = $\beta_0 + \beta_1$A-P Ratio + β_2Freshwater	68.202	5.187	0.086	3.388	3.234
SR = $\beta_0 + \beta_1$Heterogeneity	69.079	6.064	0.483	0.534	4.026
SR = $\beta_0 + \beta_1$Freshwater	69.705	6.691	0.936	0.007	4.142
SR = $\beta_0 + \beta_1$Heterogeneity + β_2Freshwater	74.275	11.26	0.254	0.782	4.262

The most parsimonious model explaining species richness was area-perimeter ratio (p = 0.023). It also yielded the highest F-statistic (7.547) and lowest standard error (3.056). It was a stronger fit than the collinear covariates of connectivity (ΔAICc = 1.966) and reserve size (ΔAICc = 4.370). The next best fitting model was the combination of area-perimeter ratio and heterogeneity (ΔAICc = 5.145), but was similar in predictive power to a combination of area-perimeter ratio and permanent freshwater aquatic habitat (ΔAICc = 5.187). Both of these models are different from the minimum AIC score by >4, thus there was not much support for either model (Burnham et al. 2004). Water and heterogeneity considered separately were not found to be significant predictors of species richness (ΔAICc > 6), and even less so when combined (ΔAICc = 11.260).

When richness was plotted as a function of area-perimeter ratio (Figure 4.2), relatively few observed values fell within the 95% confidence intervals of the model, but an R^2 value of 0.4561 (p = 0.023) indicated a significantly positive linear relationship between species richness and area-perimeter ratio.

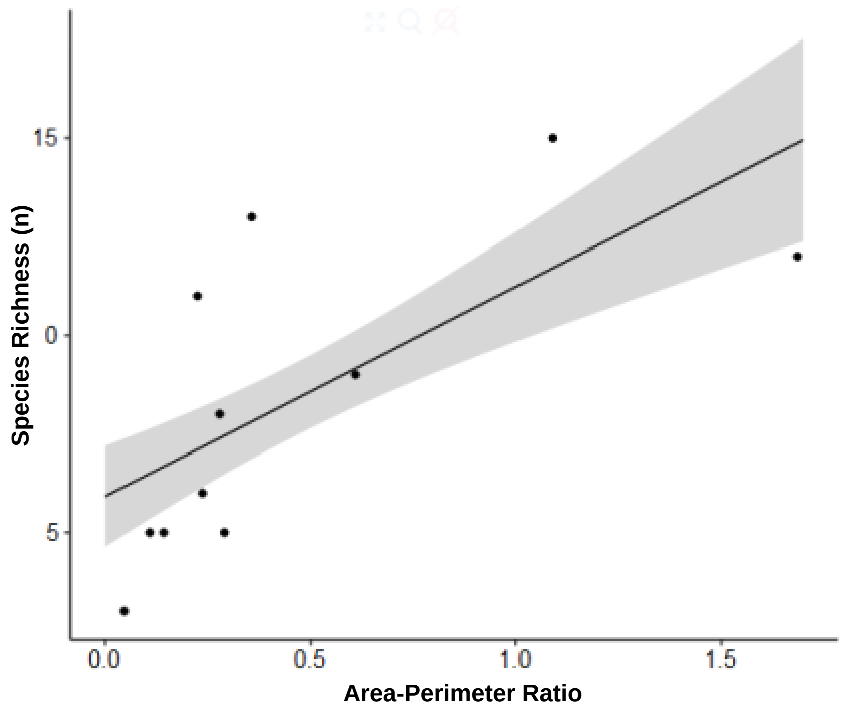


Figure 4.2: Linear model (solid line) fit where estimated species richness is a function of area-perimeter ratio (R^2 = 0.4561, p = 0.023). Shaded area indicates 95% confidence intervals and points indicate estimated species richness values.

4.4 Discussion

Carnivore species richness was higher than herbivores across all reserves. All but one of the carnivore species (viz. leopard) were classified as either small or medium carnivores.

Mesopredators persist and even thrive in human modified environments (Prugh et al. 2009, McCleery 2010, Bateman & Fleming 2012). The global rise of mesopredators is largely due to the absence of large predators which usually restrict the movement and/or predate on mesopredators but also because of abundant prey species in human dominated landscapes which in turn benefit from abundant food resources (Prugh et al. 2009, McCleery 2010, Hansen et al. 2011, Bateman & Fleming 2012, Bernardo & Melo 2013, Ramesh et al. 2016). Urban populations of commensal wildlife often exhibit reduced home ranges, which drives higher population densities in response to access to abundant anthropogenic feed sources (McCleery 2010, Newsome et al. 2010, Bateman & Fleming 2012, Šálek et al. 2015).

Small grey mongoose, Cape porcupine and Cape grysbok were the most common species in the study areas (>80 % of reserves). Both small grey mongoose and porcupine are classified as adaptable generalists which are predicted to survive better in fragmented, transformed landscapes (De Stefano & De Graaf 2003, Garden et al. 2006, Baker & Harris 2007, Ordeñana et al. 2010, Lowry et al. 2013, Šálek et al. 2015). These species seem to thrive in peri-urban areas due to varied combinations of increased food availability and absence of competition and predation pressure from species which are more sensitive to reduced home ranges and disturbance (Newsome et al. 2010, Bateman & Fleming 2012, Šálek et al. 2015). Thus porcupine and small grey mongoose can forage in peri-urban landscapes including residential gardens and public open spaces while taking refuge in small protected areas (Cavallini & Nel 1990). Cape grysbok by contrast is a fynbos endemic and a highly selective browser (Kigozi et al. 2008) that is unlikely to be able to exploit food in anthropogenic landscapes. They may be present mainly due to the availability of fynbos-specific habitat requirements rather than then being adaptable generalists, but may also benefit from reduced predation pressure in small urban reserves although caracal which are also present in many reserves are known to predate on them (Leighton et al. in press).

Hewitt's red rock hare and klipspringer were only recorded at Steenbras Nature Reserve, which is one of only two reserves that have their preferred habitat of bare rocky outcrops (Druce et al. 2009, Matthee et al. 2016). Chacma baboon and leopard were only recorded at Helderberg and Steenbras nature reserves, which both have rugged rocky terrain and high connectivity with large (>200 000 ha) protected areas. Leopards in particular have large home ranges (up to 900 km^2) within the Western Cape mountain habitat (Martins & Harris 2013), and could not persist in small isolated reserves but may pass through them provided they are connected to larger areas with suitable habitat. The honey badger is another species with large home range requirements (Begg et al. 2005) and was recorded at two large reserves (Tygerberg and Blaauwberg) with high area-perimeter ratios and connectivity to non-urban land uses.

Strong positive correlations were found between species richness and variables such as reserve size, connectivity and area to perimeter ratio. However, collinearity was also found between the predictors of species richness viz. reserve size, connectivity and area to perimeter ratio. The best predictor of high species richness proved to be reserve area to perimeter ratio (Table 4.3 and Figure 4.2), suggesting that reserves with the largest amount of core area relative to edge will be able to accommodate the highest number of species. This relationship is well supported by theory and substantiated by empirical proof for a number of taxa (Diamond 1975, Helzer & Jelinski 1999, Orrock et al. 2003, Ewers & Didham 2007, Nams 2011, Herse et al. 2018). Area-perimeter ratio was largest at Witzands and Steenbras and both of these reserves showed relatively high species richness. However, two outliers were evident, namely Tygerberg and Helderberg, both of which have relatively small area-perimeter ratios but still yielded high species richness scores (n = 11 and n = 13 respectively). This may be attributed to their high levels of connectivity with other natural habitat, noting that if a large proportion of a reserve boundary is connected to suitable habitat and the boundary is permeable, then the core area is effectively increased (Stevens et al. 2006).

Herbivore species richness also increased with increasing core area size, and consequently the largest reserves had the highest herbivore species richness (Blaauwberg = 4, Witzands Aquifer = 5, Steenbras = 5). Helderberg, which despite not being as large as the above reserves is connected to a larger conservation area, also had four herbivore species. Conversely, carnivore species richness does not seem to follow the same pattern. Only five carnivore species were recorded at Steenbras Nature Reserve although it has the largest area-perimeter ratio. Smaller reserves had higher carnivore species richness in comparison, perhaps due to the availability of anthropogenic food sources (Newsome et al. 2010, Bateman & Fleming 2012, Šálek et al. 2015). Although leopard was also recorded at Helderberg Nature Reserve which has a healthy carnivore community (n = 7 spp.), it was a single event, suggesting that leopard are transient rather than permanent residents. Tygerberg, with a low perimeter-area ratio (0.22) but good boundary connectivity, had the most carnivore species (n = 8), suggesting that some mesopredator species are either unaffected by, or respond positively to large edges which allow for movement (Diamond 1975, Nams 2011).

Vegetation heterogeneity showed a weak negative correlation (-0.237) with species richness (Table 4.3). This contradicts a body of literature suggesting that increased habitat heterogeneity positively influences species richness in an area (Ramesh et al. 2016, Matthies et al. 2017). While the fynbos, strandveld and renosterveld vegetation classifications used here differ in floral species composition, they may provide similar structural cover (low shrubland) for species (Rebelo et al. 2006). Separating vegetation types as fynbos, strandveld and renosterveld in this context might thus only be relevant to specialist herbivores and rather meaningless for mesopredators or generalist herbivores in terms of functional habitat. Inclusion of additional parameters such as vegetation structure or prey availability may therefore be more relevant when exploring species richness patterns and is something that future studies should consider including. It is also possible that reserve size and the area-perimeter ratio are such important drivers of species richness in small reserves that the effects of vegetation type are largely negated. Presence of permanent freshwater aquatic habitat explains the presence of select species such as otter and water mongoose.

4.5 Conclusions

Mammal species richness varied widely within the 11 City of Cape Town nature reserves surveyed in this study but collectively they currently provide refugia for 19 mammal species. A large core area of good habitat (irrespective of bioregion type) was the best predictor of high species richness across the reserves and large reserves that are well connected had the highest overall species richness. As predicted, generalists (e.g. porcupine and grey mongoose) were the most common species and carnivores were better represented (n = 11) than herbivores (n = 7). Species with large home ranges such as leopard seem to be most associated with reserves with high area to perimeter ratios and connectivity. Habitat specialists including klipspringer and Hewitt's red rock hare, and Cape clawless otter and water mongoose were only present in those reserves that provided rocky outcrops or permanent freshwater aquatic habitat respectively.

Overall the results presented here provide few surprises and suggest that small, fragmented natural landscapes within urban areas provide important refugia for a number of mammalian wildlife species. Consequently, and as predicted by theory small, isolated reserves such as Uitkamp had the lowest overall richness while large well, connected reserves had the highest. As has been demonstrated through the Durban Metropolitan Open Space System (Roberts 1994), maintaining species richness in urban areas is best done by establishing large reserves and stewardship sites with good connectivity to natural land. Urban conservationists can therefore improve species richness of urban protected areas by seeking to maximise the area to perimeter ratio, ensuring connectivity of reserves to other natural land and being cognisant of the needs of species that are habitat specialists. Further reduction of reserve or patch size might greatly reduce core area size with grave consequences for species richness and an increased risk of local extinctions, particularly for herbivores and edge-sensitive species. Existing small and fragmented reserves may be able to prevent further species loss if existing connectivity is maintained, but the only option for increasing species richness may be to increase core area, which can only be achieved by expanding to include surrounding area containing suitable habitat. While the CCT Biodiversity Network aims to increase patch size and connectivity (Holmes et al. 2012), the CCT area is already developed to the extent that relatively little viable municipal land remains. Because of this, the acquisition of new land for expansion of conservation areas may be difficult in the context of the CCT area, but the development of stewardship agreements with neighbouring land owners may assist in securing existing corridors and suitable habitat to maintain species richness. To do this effectively, adjacent areas chosen should ideally contribute to enlarging the reserve core area rather than lengthening of edge habitat by inclusion of small perpendicular strips of land or narrow corridors.

4.6 References

Anderson, P.M.L. & O'Farrell, P.J.O. 2012. An ecological view of the history of the City of Cape Town. *Ecology and Society*, 17(3): 28.

Augustine, D. & McNaughton, S. 2010. Regulation of shrub ungulates by native browsing dynamics Regulation on East African rangeland. *Journal of Applied Ecology*, 41: 45–58.

Baker, P.J. & Harris, S. 2007. Urban mammals: What does the future hold? An analysis of the factors affecting patterns of use of residential gardens in Great Britain. *Mammal Review*, 37(4): 297–315.

Baranyi, G., Saura, S., Podani, J. & Jordán, F. 2011. Contribution of habitat patches to network connectivity: Redundancy and uniqueness of topological indices. *Ecological Indicators*, 11(5): 1301–1310.

Bateman, P.W. & Fleming, P.A. 2012. Big city life: Carnivores in urban environments. *Journal of Zoology*, 287(1): 1–23.

Begg, C.M., Begg, K.S., Toit, J.T. Du & Mills, M.G.L. 2005. Spatial organization of the honey badger *Mellivora capensis* in the southern Kalahari: home-range size and movement patterns. *Journal of Zoology*, 265: 23–35.

Bernardo, P.V. dos S. & Melo, F.R. de. 2013. Assemblage of medium and large size mammals in an urban Semideciduous Seasonal Forest fragment in Cerrado biome. *Biota Neotropica*, 13(2): 76–80.

Brooker, L., Brooker, M. & Cale, P. 1999. Animal Dispersal in Fragmented Habitat: Measuring Habitat Connectivity, Corridor Use, and Dispersal Mortality. *Conservation Ecology*, 3(1): 4.

Burnham, K.P., Anderson, D.R. & Anderson, D.R. 2004. Multimodel Inference: Understanding AIC and BIC in Model Selection. *Sociological Methods Research*, 33(2):

261–304.

Cavallini, P. & Nel, J.A.J. 1990. The feeding ecology of the Cape grey mongoose, *Galerella pulverulenta* (Wagner 1839) in a coastal area. *African Journal of Ecology*, 28(2):123–130.

Ceballos, G. & Ehrlich, P.R. 2006. Global mammal distributions, biodiversity hotspots, and conservation. *Proceedings of the National Academy of Sciences*, 103(51): 19374–19379.

Ceballos, G., Ehrlich, P.R., Soberon, J., Salazar, I. & Fay, J.P. 2005. Global Mammal Conservation: What Must We Manage? *Science*, 309(5734): 603–607.

City of Cape Town (CCT). 2019a. *Open Data Portal: Aerial photography*. https://web1.capetown.gov.za/web1/OpenDataPortal/DatasetDetail?DatasetName=Aerial%20photography [25 July 2019].

City of Cape Town (CCT). 2019b. *Open Data Portal: Terrestrial Biodiversity Network*. http://odp.capetown.gov.za/datasets/4f2d7835518a4e6b8205ce12d77ff463_64 [4 April 2019]

Correa Ayram, C.A., Mendoza, M.E., Etter, A. & Salicrup, D.R.P. 2016. Habitat connectivity in biodiversity conservation: A review of recent studies and applications. *Progress in Physical Geography*, 40(1): 7–37.

Crooks, K.R., Burdett, C.L., Theobald, D.M., Rondinini, C. & Boitani, L. 2011. Global patterns of fragmentation and connectivity of mammalian carnivore habitat. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 366(1578): 2642–2651.

De Stefano, S. & De Graaf, R.M. 2003. Exploring the ecology of suburban wildlife. *Frontiers in Ecology and the Environment*, 1(2): 95–101.

Diamond, J.M. 1975. The island dilemma: Lessons of modern biogeographic studies for the design of natural reserves. *Biological Conservation*, 7(2): 129–146.

Dormann, C.F., Elith, J., Bacher, S., Buchmann, C., Carl, G., Carr, G., Garc, J.R., Gruber, B., Lafourcade, B., Leit, P.J., Tamara, M., Mcclean, C., Osborne, P.E., Der, B.S., Skidmore, A.K., Zurell, D. & Lautenbach, S. 2013. Collinearity : a review of methods to deal with it and a simulation study evaluating their performance. *Ecography*, 36: 27–46.

Druce, D.J., Brown, J.S., Kerley, G.I.H., Kotler, B.P., MacKey, R.L. & Slotow, R. 2009. Spatial and temporal scaling in habitat utilization by klipspringers (*Oreotragus oreotragus*) determined using giving-up densities. *Austral Ecology*, 34(5): 577–587.

Ewers, R.M. & Didham, R.K. 2007. The effect of fragment shape and species' sensitivity to habitat edges on animal population size: Contributed papers. *Conservation Biology*, 21(4): 926–936.

Fischer, J., Lindenmayer, D.B. 2007. Landscape modification and habitat fragmentation: a synthesis. *Global Ecology and Biogeography*, 16: 265–280.

Fox, J. & Weisberg, S. 2019. *An {R} Companion to Applied Regression*. Third Edition. Thousand Oaks CA: Sage. https://socialsciences.mcmaster.ca/jfox/Books/Companion/

Garden, J., McAlpine, C., Peterson, A., Jones, D. & Possingham, H. 2006. Review of the ecology of Australian urban fauna: A focus on spatially explicit processes. *Austral Ecology*, 31: 126–148.

Gonçalves, M.S.S., Gil-Delgado, J.A., Gosálvez, R.U., López-Iborra, G.M., Ponz, A. & Velasco, Á. 2018. Seasonal differences in drivers of species richness of waders in inland wetlands of La Mancha Húmeda Biosphere Reserve. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 28(6): 1414–1423.

Hansen, A.J., Knight, R.L., Marzluff, J.M., Powell, S., Gude, P.H. & Jones, K. 2011. Effects of Exurban Development on Biodiversity: Patterns, Mechanisms, and Research Needs. *Ecological Applications*, 15(6): 1893–1905.

Hardt, R.A. & Forman, R.T.T. 1989. Boundary Form Effects on Woody Colonization of Reclaimed Surface Mines. *Ecology*, 70(5): 1252–1260.

Helzer, C.J. & Jelinski, D.E. 1999. The relative importance of patch area and perimeter-area ratio to grassland breeding birds. *Ecological Applications*, 9(4): 1448–1458.

Herse, M.R., With, K.A. & Boyle, W.A. 2018. The importance of core habitat for a threatened species in changing landscapes. *Journal of Applied Ecology*, 55(5): 2241–2252.

Hoffman, T.S. & O'Riain M.J. 2012. Troop size and human-modified habitat affect the ranging patterns of a chacma baboon population in the Cape peninsula, South Africa. *American Journal of Primatology*, 74(9): 853-863.

Holmes, P.M., Rebelo, A.G., Dorse, C. & Wood, J. 2012. Can Cape Town's unique biodiversity be saved? Balancing conservation imperatives and development needs. *Ecology and Society*, 17(2): 28.

iNaturalist Network. 2019. City Nature Challenge 2019. https://www.inaturalist.org/projects/city-nature-challenge-2019 [20 October 2019]

Kerley, G.I.H., Pressey, R.L., Cowling, R.M., Boshoff, A.F. & Sims-Castley, R. 2003. Options for the conservation of large and medium-sized mammals in the Cape Floristic Region hotspot, South Africa. *Biological Conservation*, 112(1-2): 169–190.

Kigozi, F., Kerley, G. I. H. & Lessing, J. S. 2008. The diet of Cape grysbok (*Raphicerus melanotis*) in Algoa Dune Strandveld, Port Elizabeth, South Africa. *South African Journal of Wildlife Research*, 38(1): 79-81.

Kindlmann, P. & Burel, F. 2008. Connectivity measures: A review. *Landscape Ecology*, 23(8): 879–890.

Lagro, J. 1991. Assessing patch shape in landscape mosaics. *Photogrammetric Engineering & Remote Sensing*, 57(3): 285–293.

Lowry, H., Lill, A. & Wong, B.B.M. 2013. Behavioural responses of wildlife to urban environments. *Biological Reviews*, 88(3): 537–549.

Mann, G.K.H., Lagesse, J.V., O'Riain, M.J. & Parker, D.M. 2015. Beefing Up Species Richness? The Effect of Land-Use on Mammal Diversity in an Arid Biodiversity Hotspot. *African Journal of Wildlife Research*, 45(3): 321–331.

Martins, Q. & Harris, S. 2013. Movement , activity and hunting behaviour of leopards in the Cederberg mountains, South Africa. *African Journal of Ecology*, 51: 571–579.

Matthee, C., Wilson B., Robinson, T.J. & Child, M.F. 2016. A conservation assessment of Pronolagus spp. In: *The Red List of Mammals of South Africa, Swaziland and Lesotho*. Child, M.F., Roxburgh, L., Do Linh San, E., Raimondo, D., Davies-Mostert, H.T. (Ed.). South African National Biodiversity Institute and Endangered Wildlife Trust, South Africa.

Matthies, S.A., Rüter, S., Schaarschmidt, F. & Prasse, R. 2017. Determinants of species richness within and across taxonomic groups in urban green spaces. *Urban Ecosystems*, 20(4): 897–909.

Mazerolle, M.J. 2019. *AICcmodavg: Model selection and multimodel inference based on (Q)AIC(c)*. R package version 2.2-2. https://cran.r-project.org/package=AICcmodavg

McCleery, R. 2010. Urban Mammals. In: *Urban Ecosystem Ecology*, Agronomy Monographs. 55. ASA, CSSA, SSSA, Madison, WI. p. 87-102

McDonald, R.I., Kareiva, P. & Forman, R.T.T. 2008. The implications of current and future urbanization for global protected areas and biodiversity conservation. *Biological Conservation*, 141(6): 1695–1703.

Nams, V.O. 2011. Emergent properties of patch shapes affect edge permeability to animals. *PLoS ONE*, 6(7): e21886.

Newsome, S.D., Ralls, K., Job, C.V.H., Fogel, M.L. & Cypher, B.L. 2010. Stable isotopes evaluate exploitation of anthropogenic foods by the endangered San Joaquin kit fox (*Vulpes macrotis mutica*). *Journal of Mammalogy*, 91(6): 1313–1321.

Ordeñana, M. A, Crooks, K.R., Boydston, E.E., Fisher, R.N., Lyren, L.M., Siudyla, S., Haas, C.D., Harris, S., Hathaway, S. A, Turschak, G.M., Miles, A.K. & Van Vuren, D.H. 2010. Effects of urbanization on carnivore species distribution and richness. *Journal of Mammalogy*, 91(6): 1322–1331.

Orrock, J.L., Danielson, B.J., Burns, M.J. & Levey, D.J. 2003. Spatial ecology of predator-prey interactions: Corridors and patch shape influence seed predation. *Ecology*, 84(10): 2589–2599.

Palomares, F. & Delibes, M. 1990. Habitat preference of large grey mongooses *Herpestes ichneumon* in Spain. *Acta Theriologica*, 35(1): 1–6.

Pekin, B.K. & Pijanowski, B.C. 2012. Global land use intensity and the endangerment status of mammal species. *Diversity and Distributions*, 18(9): 909–918.

Prugh, L.R., Stoner, C.J., Epps, C.W., Bean, W.T., Ripple, W.J., Laliberte, A.S. & Brashares, J.S. 2009. The Rise of the Mesopredator. *BioScience*, 59(9): 779–791.

QGIS Development Team. 2019. *QGIS Geographic Information System*. Open Source Geospatial Foundation Project. http://qgis.osgeo.org

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

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

Rebelo, A.G., Boucher, C., Helme, N., Mucina, L. & Rutherford, M.C. 2006. Fynbos Biome. In: *The vegetation of South Africa, Lesotho and Swaziland*. Mucina, L. and Rutherford, M.C. (Ed.). Strelitzia 19, South African National Biodiversity Institute, Pretoria

Rebelo, A.G., Holmes, P.M., Dorse, C. & Wood, J. 2011. Impacts of urbanization in a biodiversity hotspot: Conservation challenges in Metropolitan Cape Town. *South African Journal of Botany*, 77(1): 20–35.

Roberts, D.C. 1994. The Design of an Urban Open-space Network for the City of Durban (South Africa). *Environmental Conservation*, 21(1): 11–17.

Šálek, M., Drahníková, L. & Tkadlec, E. 2015. Changes in home range sizes and population densities of carnivore species along the natural to urban habitat gradient. *Mammal Review*, 45(1): 1–14.

Shannon, C.E. 1948. A Mathematical Theory of Communication. *Bell System Technical Journal*, 27(3): 379–423.

Söndgerath, D. & Schröder, B. 2002. Population dynamics and habitat connectivity affecting the spatial spread of populations - A simulation study. *Landscape Ecology*, 17(1): 57–70.

South African National Biodiversity Institute (SANBI). 2016. *Vegetation Map of South Africa, Lesotho and Swaziland*. http://bgis.sanbi.org/SpatialDataset/Detail/18 [28 March 2017]

Stevens, V.M., Leboulengé, É., Wesselingh, R.A. & Baguette, M. 2006. Quantifying functional connectivity: Experimental assessment of boundary permeability for the natterjack toad (*Bufo calamita*). *Oecologia*, 150(1): 161–171.

Sullivan, T.., Sullivan, S. & Lindgren, P. 2001. Stand structure and small mammals in young lodgepole pine forest: 10-year results after thinning. *Ecological Applications*, 11(4): 1151–1173.

Tischendorf, L. & Fahrig, L. 2000. On the usage and measurement of landscape connectivity. *Oikos*, 90(1): 7–19.

Torres-Romero, E.J. & Olalla-Tárraga, M. a. 2015. Untangling human and environmental effects on geographical gradients of mammal species richness: A global and regional evaluation. *Journal of Animal Ecology*, 84(3): 851–860.

Turgeon, K. & Kramer, D.L. 2012. Compensatory immigration depends on adjacent population size and habitat quality but not on landscape connectivity. *Journal of Animal Ecology*, 81(6): 1161–1170.

Visconti, P., Pressey, R.L., Giorgini, D., Maiorano, L., Bakkenes, M., Boitani, L., Alkemade, R., Falcucci, A., Chiozza, F. & Rondinini, C. 2011. Future hotspots of terrestrial mammal loss. *Philosophical Transactions of the Royal Society of London B: Biological Sciences*, 366(1578): 2693–2702.

CHAPTER FIVE:
RESEARCH FINDINGS AND MANAGEMENT IMPLICATIONS

5.1 Introduction

Transformation of land through urbanisation has a negative impact on biodiversity, ecosystem functioning and ultimately ecosystem services essential for human recreation and wellbeing (Anderson & O'Farrell 2012, O'Farrell et al. 2012, Pickett et al. 2011, Cheesbrough et al. 2019). Despite this many animals persist in the urban matrix and may even thrive in small protected areas within or on the edge of urban areas (McKinney 2002, De Stefano & De Graaf 2003, Garden et al. 2006, Baker & Harris 2007, Ordeñana et al. 2010, Pickett et al. 2011, Lowry et al. 2013, Šálek et al. 2015). Conservation of remnant populations of medium and large mammal species in the urban environment is important, not only for the sake of the species themselves, but also for the persistence of ecosystem processes directly and indirectly linked to their presence and behaviour (Kerley et al. 2003, De Stefano & De Graaf 2003, Ceballos et al. 2005, Visconti et al. 2011). A first step in preserving wildlife in protected areas is deriving methods for monitoring species presence and population trends that will enable evaluation of conservation efforts and associated management actions (Van der Ree 2004, Anderson & O'Farrell 2012, Fischer et al. 2012, Saito & Koike 2013).

The City of Cape Town (CCT) Biodiversity Management Branch manages 17 urban and peri-urban nature reserves. Historical accounts suggest that 39 medium and large mammal species were once present in the area, but it is unclear how many of these native species persist within the CCT's current network of nature reserves. Variables influencing the presence of mammal species in the different reserves are also unknown, limiting the ability of management to achieve their goal of conserving the largest number of species possible in existing protected areas and possibly motivating for new reserves. While the primary aim of this study was thus to develop and use a standardised monitoring protocol to determine the medium and large mammal species community composition within the respective CCT reserves, the secondary aim was to understand how reserve-level variables that might positively or negatively influence species community composition. The latter is essential to improving conservation and management strategies of small protected areas that typically suffer the combined challenges of fragmentation, isolation and small size and thus need ongoing human intervention to sustain healthy mammal populations.

5.2 Survey protocol

The hardening of the boundaries of urban nature reserves and ongoing development of the surrounding land increases deleterious edge effects and reduces connectivity with other parcels of natural land. Understanding which species are most affected by these anthropogenic changes requires repeated, cost effective and reliable surveys (Kerley et al. 2003, Garden et al. 2006, Cilliers & Siebert 2012). Camera traps are an excellent tool for conducting faunal surveys in protected areas (Kelly 2008, Rowcliffe et al. 2008, Tobler et al. 2008, Ordeñana et al. 2010, Rovero et al. 2010, Colyn et al. 2017) and were used in this study to sample medium and large mammal species presence in the twelve CCT nature reserves larger than 30 ha (Chapter 3).

Results support a growing body of literature that reveals smaller reserves as the most susceptible to anthropogenic effects and consequent local extinctions (Diamond 1975, Turgeon & Kramer 2012). Monitoring mammals in small reserves requires a low survey effort which means it is feasible to conduct annual monitoring of species to assess whether interventions (e.g. reintroductions) are viable in such reserves. While larger reserves require a much higher survey effort they also have higher species richness and appear less prone to local extinctions. The sampling protocol developed in this study (see Chapter 3 for protocol and Appendix A for camera placement details) and survey effort (summarised in Table 5.1) are considered to serve as realistic and achievable long term monitoring tool for reserve managers.

Table 5.1: Lowest recommended camera trapping survey effort to be used for effective species richness estimates across 11 CCT nature reserves. Cameras should be placed at the same locations to reduce detection bias (see Appendix A for details) and should be *in situ* for the number of camera days (number of cameras multiplied by number of actual survey days) shown to provide a reliable estimate of species richness in each reserve.

Nature Reserve	No. Cameras	Survey period (days)	No. Camera days
Uitkamp Wetland	5	50	250
Bracken	5	160	800
Zandvlei Estuary	10	100	1000
Wolfgat	10	90	900
Tygerberg	9	106	950
Helderberg	15	117	1750
False Bay	15	54	800
Table Bay	12	80	950
Blaauwberg	15	47	700
Witzands Aquifer	20	>93	>1850
Steenbras	30	52	1550

5.3 Species richness patterns

Historical accounts suggest that 39 medium and large mammal species (when Cape and scrub hare are grouped as *Lepus* spp.) were present in the general area of the CCT at the time of Dutch settlement in the 17th century (Boshoff & Kerley 2001). More recently the CCT Biodiversity Database, and citizen science portals iNaturalist and iSpot suggested a total of 22 medium and large mammal species in the 12 surveyed CCT reserves over a five year period preceding this study (2012 to 2017). A recent study by Okes and O'Riain (2019) highlighted the value of opportunistic citizen sightings in recording rare and elusive species over longer time intervals relative to a single formal survey. However it is important to stress that citizen sighting data bases are confounded by detection bias, misidentification and unequal sampling effort and can thus provide a distorted perspective of how well select species are surviving within protected areas. This study, using camera traps, recorded 19 native and non-reintroduced species present in the 12 CCT reserves, which is 86% of the species in the existing databases and 49% of the historically-occurring species.

Species in the existing databases that were not recorded in this study included: bat-eared fox (*Otocyon megalotis*), striped weasel (*Poecilogale albinucha*), black-backed jackal (*Canis mesomelas*) and grey rhebuck (*Pelea capreolus*). Only one species that is currently not in the existing data bases was recorded in this study, viz. Hewitt's red rock hare (*Pronolagus saundersiae*). The most recent records for bat-eared fox, striped weasel and black-backed jackal were in 2013, suggesting they may have disappeared from the area rather than being present but not detected in this study. The status of grey rhebuck is uncertain as a local extinction event is thought to have occurred at Tygerberg Nature Reserve, where the species was recorded in the databases, shortly before the camera trap survey was conducted. A better grasp of the loss of these species is needed and their disappearance accentuates the importance of understanding what drives current species persistence and richness patterns to avoid further loss.

Recorded species richness across reserves ranged from three to 12 species and generally met the expectation that larger reserves will have higher species richness (Chapter 4). A higher richness of carnivores (n = 11) than herbivores (n = 7) was also recorded, although this ratio (1.6:1) was not dissimilar to that of historical records (1.5:1). Two generalist species, the Cape porcupine (*Hystrix africaeaustralis*) and small grey mongoose (*Galerella pulverulenta*), were found in most reserves, but surprisingly a habitat specialist, the Cape grysbok (*Raphicerus melanotis*), was as widely distributed, and seems capable of persisting in even small fragments of natural habitat.

5.4 Drivers of species richness patterns

Species richness patterns were further explored by comparing camera trap survey results with the size, connectivity, area-perimeter ratio, vegetation heterogeneity and presence of permanent freshwater aquatic habitat of all reserves (Chapter 4). While reserve size and connectivity were positively correlated to species richness, linear models showed that area-perimeter ratio had the strongest association with species richness, suggesting that reserves with a larger core area compared to edge length are able to support higher species richness, irrespective of vegetation type. It is thus not simply a matter of increasing the size of reserves to improve species richness and species persistence probabilities, as the shape of any potential acquisitions must also be considered (Diamond 1975, Helzer & Jelinski 1999, Herse et al. 2018). Narrow or irregularly shaped land pieces with high edge proportions might add little value without significant core area (Helzer & Jelinski 1999, Herse et al. 2018).

Results suggest that it is also important to consider area – perimeter ratio when any potential reserve size reductions, boundary alterations or land-use change on a boundary are considered. Any change or loss that will significantly affect the core area of the reserve might have disproportional effects on sensitive species persistence and overall species richness (Hardt and Forman 1989, Helzer & Jelinski 1999, Orrock et al. 2003, Ewers & Didham 2007, Nams 2011, Herse et al. 2018). If the edge is increased, reserves are more susceptible to anthropogenic effects and edge-sensitive species may no longer be able to persist (Diamond 1975, Nams 2011, Ramesh et al. 2016). Connectivity, as calculated here by length of reserve boundary adjacent to suitable dispersal habitat, proved to be important for species richness with reserves described as having a low area-perimeter ratio having more species than expected. This trend is apparent because natural land or suitable habitat adjacent to a reserve effectively increases the size of the core habitat area (Stevens et al. 2006) which has a positive effect on species richness.

Non-metric multidimensional scaling of species richness with reserve level variables and all 19 mammal species provided a useful visual summary of how these variables influence the presence of different species and how different species may share a preference for select variables. Thus Cape clawless otter (*Aonyx capensis*) and water mongoose (*Atilax paludinosus*) are noticeably absent from reserves without permanent freshwater aquatic habitat and klipspringer (*Oreotragus oreotragus*), baboon (*Papio ursinus*) and rock rabbits were only found in reserves with access to steep, rocky habitat. Another widely supported trend evident in this study was the absence of large predators, e.g. the leopard (*Panthera pardus*), from all small reserves and medium sized reserves with poor connectedness.

Together these findings suggest that while increasing the size of already isolated reserves may not be an option, where connectivity to suitable habitat exists it should be maintained or even improved by securing the land and converting it to protected status, via means such as stewardship agreements with neighbouring landowners. Steenbras and Helderberg are currently connected to large expanses of protected and conserved land, and similar connectivity should be prioritised in the form of stewardship agreements with landowners bordering Witzands, Blaauwberg and Tygerberg to ensure the persistence of honey badger (*Mellivora capensis*) and improve chances of leopard, black backed jackal and striped polecat (*Ictonyx striatus*) returning to the area. Caracal (*Caracal caracal*) and Cape/scrub hare (*Lepus capensis/saxatilis*) also associate (but to a lesser degree) with larger reserve size, better connectivity and higher area-perimeter ratio. Klipspringer (*Oreotragus oreotragus*) and Hewitt's red rock hare were strongly associated with area-perimeter ratio, but as they are only found in Steenbras Nature Reserve this is most likely due to preferred rocky habitat which is not found elsewhere in the city. Striped polecat do not associate strongly with any of the measured reserve attributes, but all the reserves where they are present are situated on the edges of the CCT municipal boundary.

Neither small- nor large-spotted genet were found in the two smallest reserves and only appeared in reserves ≥ 200 ha (i.e. Zandvlei), suggesting that they are vulnerable to extinction in small fragments. Neither species was found in Table Bay, although this may be due to the large proportion of water-associated vegetation that reduces the cover provided by fynbos, strandveld or renosterveld habitat (see Chapter 2 for Table Bay site description). Without more information on why they are absent from the smaller reserves, it is unclear as to whether reintroduction of genet into small reserves is feasible or desirable. Interestingly, large- and small-spotted genet seldom overlapped, with the former detected in the southern and eastern reserves, the latter only in the northern reserves, and some overlap in the centrally-located Tygerberg Nature Reserve.

None of the reserve attributes used to interrogate drivers of species richness patterns seemed to explain the occurrence of steenbok (*Raphicerus campestris*), common duiker (*Sylvicapra grimmia*) or Cape fox (*Vulpes chama*) and this leaves avenues for further exploration. Duiker occurs in five reserves and steenbok in only three, and they overlap in two of the reserves. In all the reserves they co-occur with Cape grysbok. How these three small browsing antelope species (≤18 kg) use the CCT landscape might be particularly interesting to explore.

5.5 Conclusions

The study shows that while a number of historically-occurring medium and large mammal species have already been lost in the CCT area, nearly half of these species have been able to persist despite urbanisation. CCT nature reserves with high area to perimeter ratios and, where this is low, high connectivity to suitable habitat are better able to support medium and large mammal species richness and species persistence. Further reduction of core habitat within existing CCT reserves may greatly reduce species richness and potentially encourage local extinctions, particularly for herbivores and edge-sensitive species. Reserve expansions and/or stewardship agreements that will significantly increase core area should be prioritised and current connectivity retained, or improved, to increase and sustain species richness. Regular monitoring of medium and large mammals species presence using a standardised sampling protocol should be put in place to guard against further species loss. Camera traps with a placement protocol as used in this study together with the minimum camera day effort estimates presented here for each reserve should provide comparable results.

5.6 References

Anderson, P.M.L. & O'Farrell, P.J.O. 2012. An ecological view of the history of the City of Cape Town. *Ecology and Society*, 17(3): 28.

Baker, M.A. & Brown, J.S. 2013. Foraging and habitat use of common duikers, *Sylvicapra grimmia*, in a heterogeneous environment within the Soutspanberg, South Africa. *African Journal of Ecology*, 52: 318–327.

Boshoff, A.F. & Kerley, G.I.H. 2001. Potential distributions of the medium- to large-sized mammals in the Cape Floristic Region, based on historical accounts and habitat requirements. *African Zoology*, 36(2): 245–273.

Ceballos, G., Ehrlich, P.R., Soberon, J., Salazar, I. & Fay, J.P. 2005. Global Mammal Conservation: What Must We Manage? *Science*, 309(5734): 603–607.

Cheesbrough, A.E., Garvin, T. & Nykiforuk, C.I.J. 2019. Everyday wild: Urban natural areas, health, and well-being. *Health and Place*, 56(January): 43–52.

Cilliers, S.S. & Siebert, S. 2012. Urban Ecology in Cape Town: South African Comparisons and Reflections. *Ecology and Society*, 17(3): 33.

Colyn, R., Radloff, F.G.T. & O'Riain, M.J. 2017. Camera trapping mammals in the scrubland's of the Cape Floristic Kingdom - the importance of effort, spacing and trap placement. *Biodiversity and Conservation*, 27(2): 503-520.

De Stefano, S. & De Graaf, R.M. 2003. Exploring the ecology of suburban wildlife. *Frontiers in Ecology and the Environment*, 1(2): 95–101.

Diamond, J.M. 1975. The island dilemma: Lessons of modern biogeographic studies for the design of natural reserves. *Biological Conservation*, 7(2): 129–146.

Ewers, R.M. & Didham, R.K. 2007. The effect of fragment shape and species' sensitivity to habitat edges on animal population size: Contributed papers. *Conservation Biology*, 21(4): 926–936.

Fischer, J.D., Cleeton, S.H., Lyons, T.P., Miller, J.R., Fischer, J.D., Cleeton, S.H. & Timothy, P. 2012. Urbanization and the Predation Paradox: The Role of Trophic Dynamics in Structuring Vertebrate Communities. *BioScience*, 62(9): 809–818.

Garden, J., McAlpine, C., Peterson, A., Jones, D. & Possingham, H. 2006. Review of the ecology of Australian urban fauna: A focus on spatially explicit processes. *Austral Ecology*, 31: 126–148.

Gotelli, N.J. & Colwell, R.K. 2001. Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. *Ecology Letters*, 4: 379–391.

Hardt, R.A. & Forman, R.T.T. 1989. Boundary Form Effects on Woody Colonization of Reclaimed Surface Mines. *Ecology*, 70(5): 1252–1260.

Helzer, C.J. & Jelinski, D.E. 1999. The relative importance of patch area and perimeter-area ratio to grassland breeding birds. *Ecological Applications*, 9(4): 1448–1458.

Herse, M.R., With, K.A. & Boyle, W.A. 2018. The importance of core habitat for a threatened species in changing landscapes. *Journal of Applied Ecology*, 55(5): 2241–2252.

Holmes, P.M., Rebelo, A.G., Dorse, C. & Wood, J. 2012. Can Cape Town's unique biodiversity be saved? Balancing conservation imperatives and development needs. *Ecology and Society*, 17(2): 28.

Kerley, G.I.H., Pressey, R.L., Cowling, R.M., Boshoff, A.F. & Sims-Castley, R. 2003. Options for the conservation of large and medium-sized mammals in the Cape Floristic Region hotspot, South Africa. *Biological Conservation*, 112(1-2): 169–190.

Kelly, M.J. 2008. Design, evaluate, refine: Camera trap studies for elusive species. *Animal Conservation*, 11(3): 182–184.

Lowry, H., Lill, A. & Wong, B.B.M. 2013. Behavioural responses of wildlife to urban environments. *Biological Reviews*, 88(3): 537–549.

McDonald, R.I., Kareiva, P. & Forman, R.T.T. 2008. The implications of current and future urbanization for global protected areas and biodiversity conservation. *Biological Conservation*, 141(6): 1695–1703.

McKinney, M.L. 2002. Urbanization, biodiversity and conservation. *BioScience*, 52(10): 883–890.

Nams, V.O. 2011. Emergent properties of patch shapes affect edge permeability to animals. *PLoS ONE*, 6(7): e21886.

O'Farrell, P.J., Anderson, P.M.L., Le Maitre, D.C. & Holmes, P.M. 2012. Insights and opportunities offered by a rapid ecosystem service assessment in promoting a conservation agenda in an urban biodiversity hotspot. *Ecology and Society*, 17(3): 27.

Okes, N.C. & O'Riain, M.J. 2019. Can opportunistic citizen sightings assist in the monitoring of an elusive, crepuscular mammal in an urban environment? *Urban Ecosystems*, 22(3): 483–492.

Orrock, J.L., Danielson, B.J., Burns, M.J. & Levey, D.J. 2003. Spatial ecology of predator-prey interactions: Corridors and patch shape influence seed predation. *Ecology*, 84(10): 2589–2599.

Ordeñana, M. A, Crooks, K.R., Boydston, E.E., Fisher, R.N., Lyren, L.M., Siudyla, S., Haas, C.D., Harris, S., Hathaway, S. A, Turschak, G.M., Miles, A.K. & Van Vuren, D.H. 2010. Effects of urbanization on carnivore species distribution and richness. *Journal of Mammalogy*, 91(6): 1322–1331.

Pickett, S.T.A., Cadenasso, M.L., Grove, J.M., Boone, C.G., Groffman, P.M., Irwin, E., Kaushal, S.S., Marshall, V., McGrath, B.P., Nilon, C.H., Pouyat, R. V, Szlavecz, K., Troy, A. & Warren, P. 2011. Urban ecological systems: Scientific foundations and a decade of progress. *Journal of Environmental Management*, 92(3): 331–362.

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

Rebelo, A.G., Holmes, P.M., Dorse, C. & Wood, J. 2011. Impacts of urbanization in a biodiversity hotspot: Conservation challenges in Metropolitan Cape Town. *South African Journal of Botany*, 77(1): 20–35.

Rovero, F., Tobler, M. & Sanderson, J. 2010. Camera trapping for inventorying terrestrial vertebrates. In: *Manual on field recording techniques and protocols for All Taxa Biodiversity Inventories and Monitoring*. Eymann, J., Degreef, J., Häuser, C., Monje, J.C., Samyn, Y., Van den Spiegel, D. (Ed.). Abc Taxa, 8(1): 100 –128.

Rowcliffe, J.M., Field, J., Turvey, S.T. & Carbone, C. 2008. Estimating animal density using camera traps without the need for individual recognition. *Journal of Applied Ecology*, 45: 1228–1236.

Saito, M. & Koike, F. 2013. Distribution of Wild Mammal Assemblages along an Urban-Rural-Forest Landscape Gradient in Warm-Temperate East Asia. *PLoS ONE*, 8(5): e65464.

Šálek, M., Drahníková, L. & Tkadlec, E. 2015. Changes in home range sizes and population densities of carnivore species along the natural to urban habitat gradient. *Mammal Review*, 45(1): 1–14.

Stevens, V.M., Leboulengé, É., Wesselingh, R.A. & Baguette, M. 2006. Quantifying functional connectivity: Experimental assessment of boundary permeability for the natterjack toad (*Bufo calamita*). *Oecologia*, 150(1): 161–171.

Tobler, M.W., Carrillo-Percastegui, S.E., Leite Pitman, R., Mares, R. & Powell, G. 2008. An evaluation of camera traps for inventorying large- and medium-sized terrestrial rainforest mammals. *Animal Conservation*, 11(3): 169–178.

Turgeon, K. & Kramer, D.L. 2012. Compensatory immigration depends on adjacent population size and habitat quality but not on landscape connectivity. *Journal of Animal Ecology*, 81(6): 1161–1170.

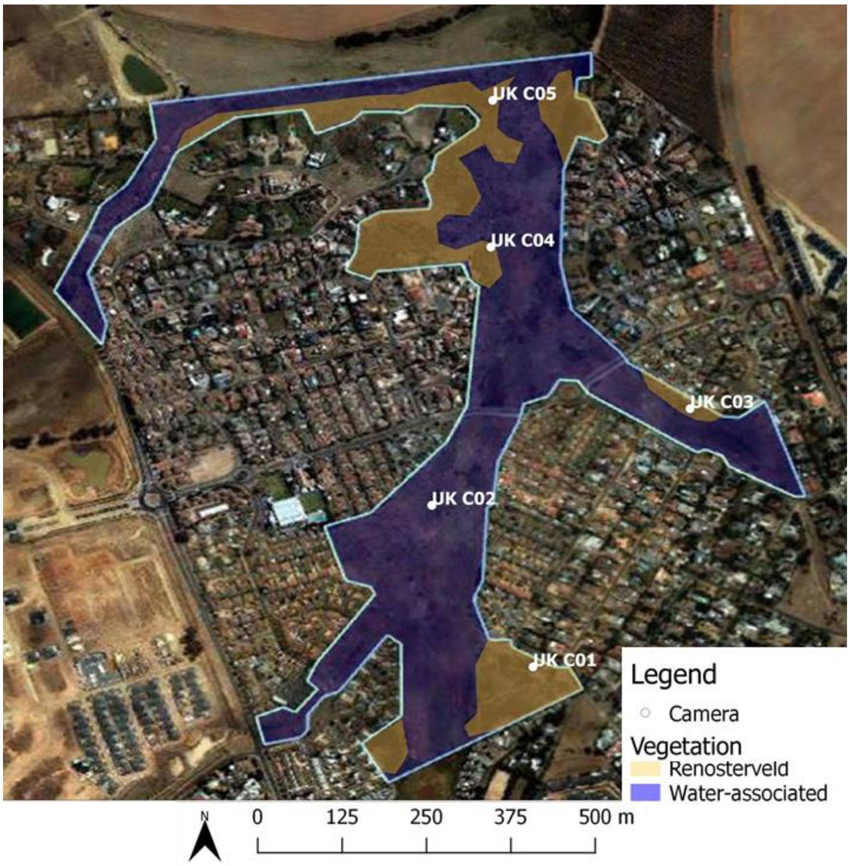
Van der Ree, R. 2004. The impact of urbanisation on the mammals of Melbourne - do atlas records tell the whole story or just some of the chapters? In *Urban Wildlife: more than meets the eye*. D. Lunney & S. Burgin, (Ed.). Mosman, NSW: Royal Zoological Society of New South Wales: 195–204.

Visconti, P., Pressey, R.L., Giorgini, D., Maiorano, L., Bakkenes, M., Boitani, L., Alkemade, R., Falcucci, A., Chiozza, F. & Rondinini, C. 2011. Future hotspots of terrestrial mammal loss. *Philosophical Transactions of the Royal Society of London B: Biological Sciences*, 366(1578): 2693–2702.

APPENDICES

APPENDIX A: Camera trap placements in each of the 12 surveyed City of Cape Town nature reserves. For each study site (listed as 1-12 in order of reserve size), a map of camera placements is provided including 1 x 1 km^2 grids (white lines), followed by a table of the exact latitude and longitude of each camera as placed in the study. Grids were not used for Uitkamp Wetland, Bracken and Kenilworth Racecourse reserves as they are all smaller than 1 km^2

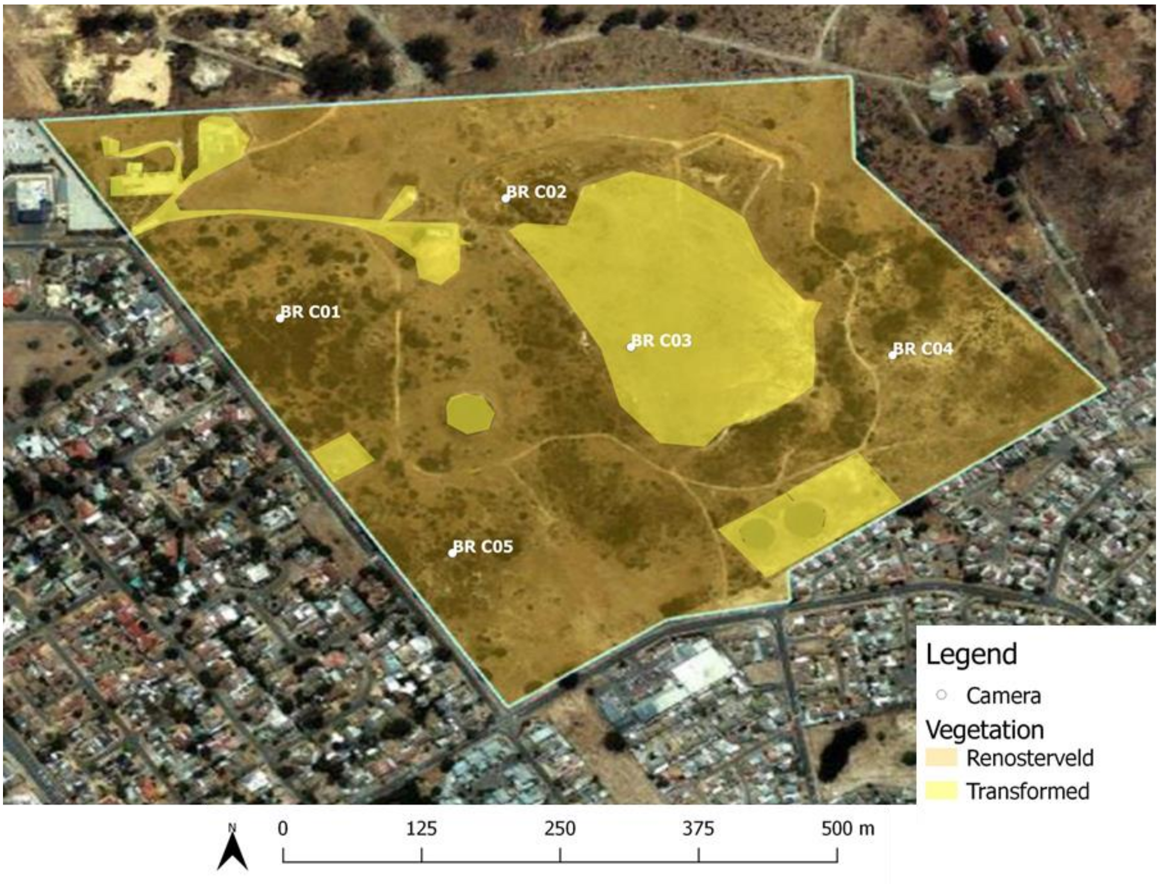
1. Uitkamp Wetland Nature Reserve (32 ha)



Camera coordinates:

Camera Placement	Latitude (decimal degrees)	Longitude (decimal degrees)
UK C01	-33.82122	18.64076
UK C02	-33.81861	18.63913
UK C03	-33.81705	18.64329
UK C04	-33.81444	18.64008
UK C05	-33.81207	18.64011

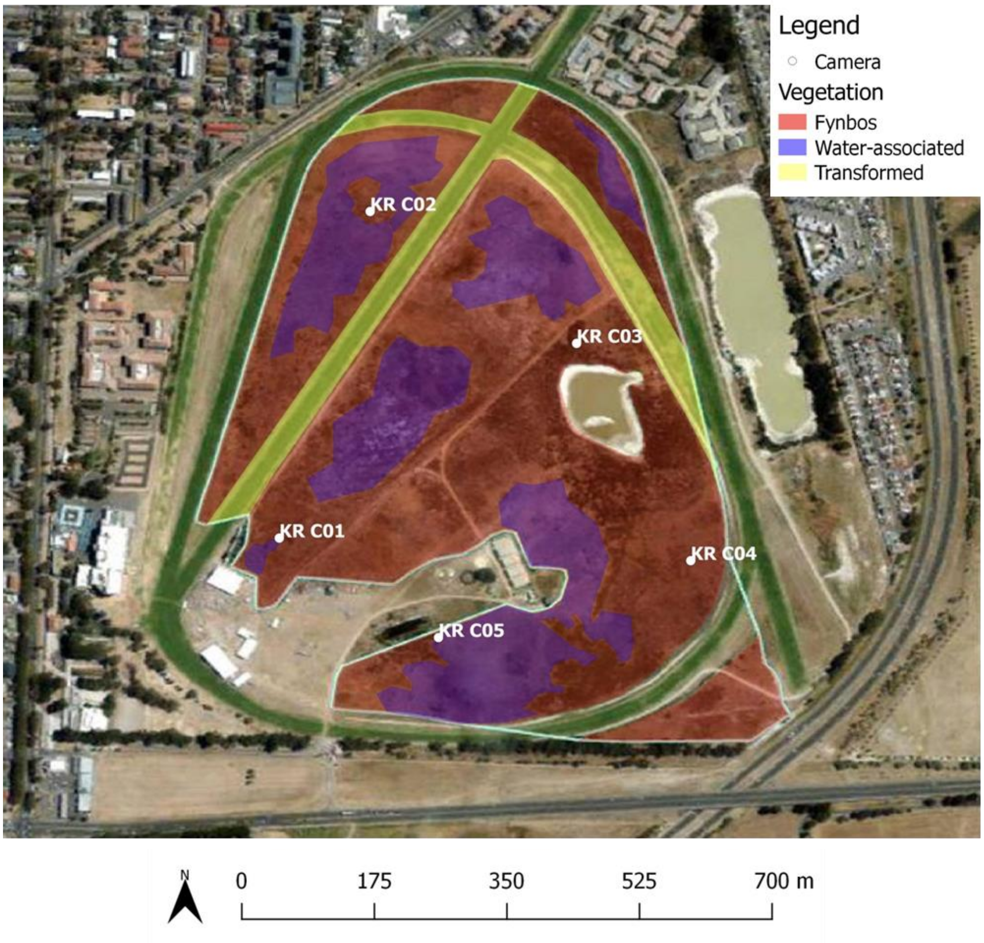
2. Bracken Nature Reserve (36 ha)



Camera coordinates:

Camera Placement	Latitude (decimal degrees)	Longitude (decimal degrees)
BR C01	-33.87907	18.70995
BR C02	-33.87790	18.71215
BR C03	-33.87935	18.71337
BR C04	-33.87943	18.71592
BR C05	-33.88136	18.71163

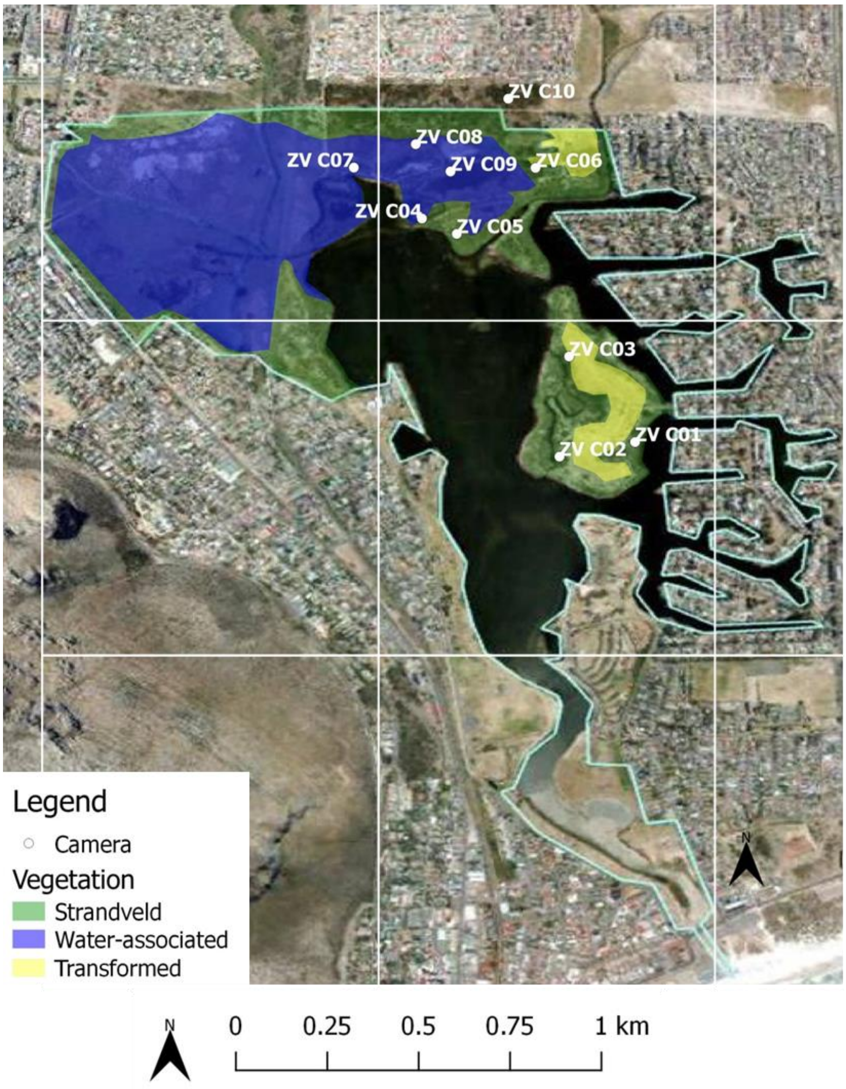
3. Kenilworth Racecourse Conservation Area (52 ha)



Camera coordinates:

Camera Placement	Latitude (decimal degrees)	Longitude (decimal degrees)
KR C01	-33.99850	18.48188
KR C02	-33.99387	18.48317
KR C03	-33.99574	18.48610
KR C04	-33.99882	18.48772
KR C05	-33.99991	18.48414

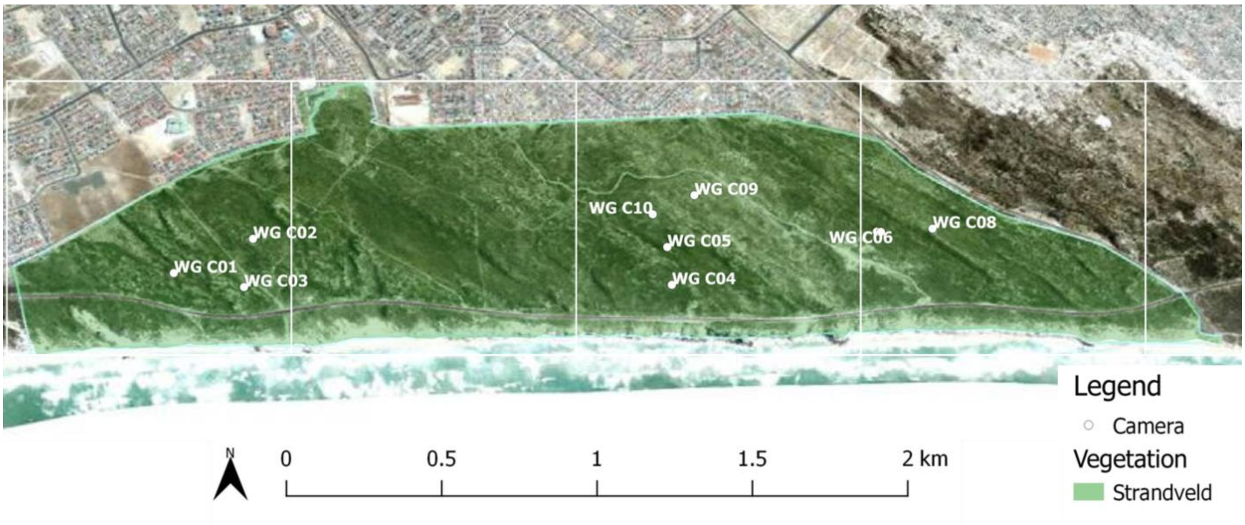
4. **Zandvlei Estuary Nature Reserve (200 ha)**



Camera coordinates:

Camera Placement	Latitude (decimal degrees)	Longitude (decimal degrees)
ZV C01	-34.08996	18.47316
ZV C02	-34.09039	18.47092
ZV C03	-34.08742	18.47120
ZV C04	-34.08334	18.46684
ZV C05	-34.08379	18.46787
ZV C06	-34.08184	18.47021
ZV C07	-34.08183	18.46482
ZV C08	-34.08114	18.46666
ZV C09	-34.08194	18.46769
ZV C10	-34.07979	18.46940

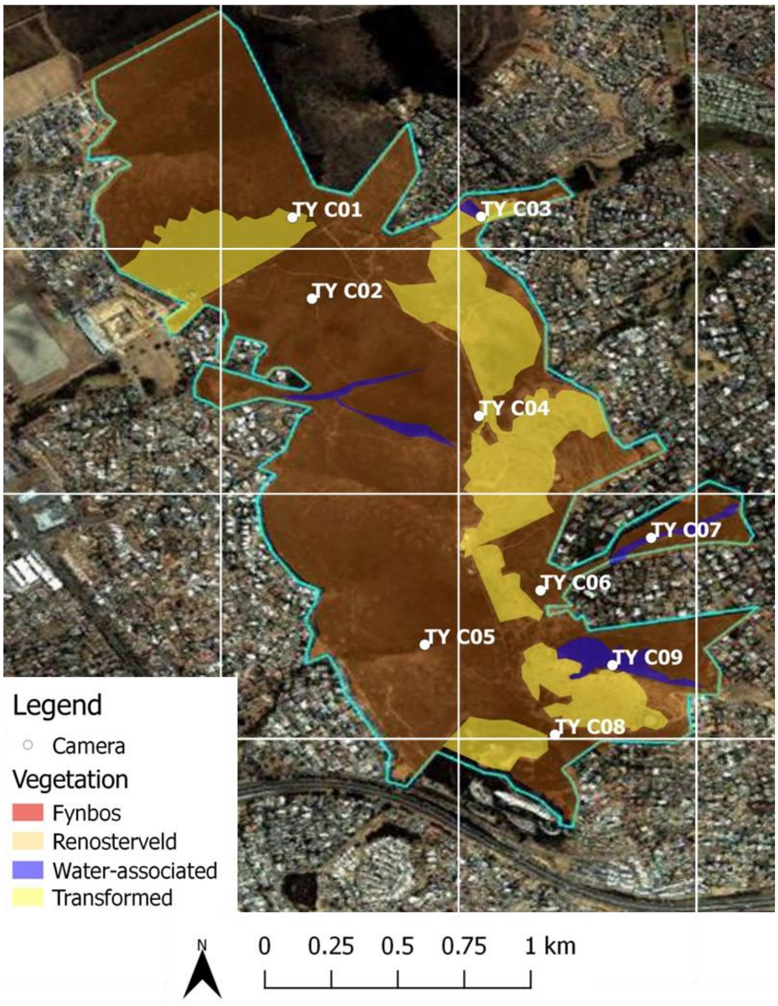
5. Wolfgat Nature Reserve (262 ha)



Camera coordinates:

Camera Placement	Latitude (decimal degrees)	Longitude (decimal degrees)
WG C01	-34.07073	18.63343
WG C02	-34.06953	18.63621
WG C03	-34.07122	18.63590
WG C04	-34.07114	18.65088
WG C05	-34.06981	18.65072
WG C06	-34.06928	18.65807
WG C07	-34.06928	18.65821
WG C08	-34.06917	18.66001
WG C09	-34.06800	18.65167
WG C10	-34.06866	18.65021

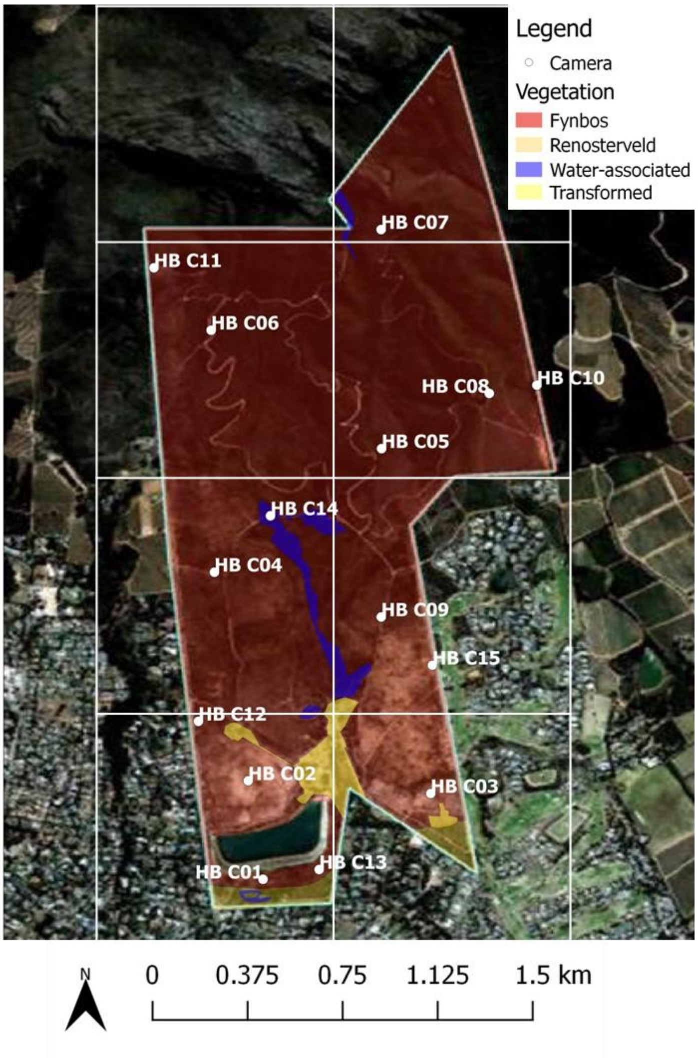
6. Tygerberg Nature Reserve (388 ha)



Camera coordinates:

Camera Placement	Latitude (decimal degrees)	Longitude (decimal degrees)
TY C01	33.86177	18.58849
TY C02	33.86509	18.58929
TY C03	33.86174	18.59620
TY C04	33.86989	18.59612
TY C05	33.87924	18.59390
TY C06	33.87702	18.59863
TY C07	33.87487	18.60315
TY C08	33.88292	18.59922
TY C09	33.88008	18.60157

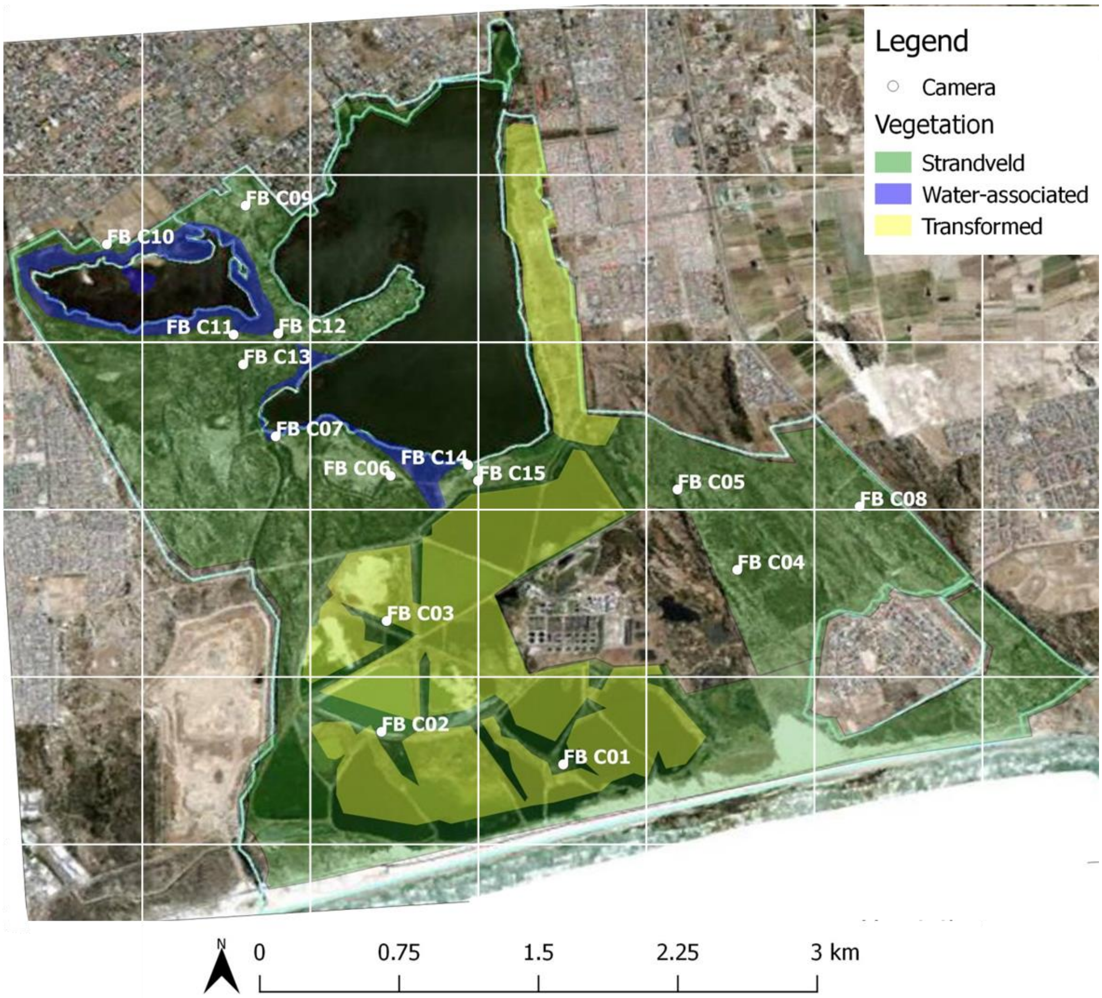
7. Helderberg Nature Reserve (402 ha)



Camera coordinates:

Camera Placement	Latitude (decimal degrees)	Longitude (decimal degrees)
HB C01	-34.06976	18.87024
HB C02	-34.06554	18.86960
HB C03	-34.06606	18.87738
HB C04	-34.05659	18.86815
HB C05	-34.05130	18.87530
HB C06	-34.04622	18.86800
HB C07	-34.04194	18.87525
HB C08	-34.04891	18.87987
HB C09	-34.05850	18.87529
HB C10	-34.04859	18.88195
HB C11	-34.04356	18.865569
HB C12	-34.06298	18.867469
HB C13	-34.06932	18.872635
HB C14	-34.05418	18.870553
HB C15	-34.06057	18.877481

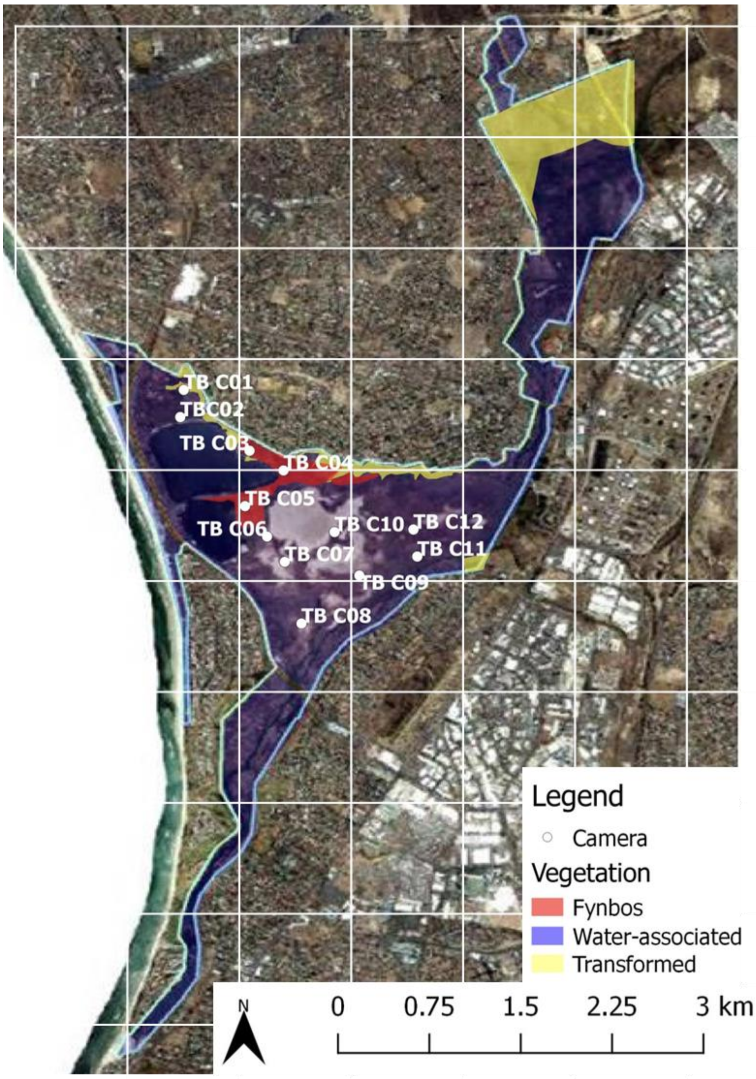
8. False Bay Nature Reserve (634 ha)



Camera coordinates:

Camera Placement	Latitude (decimal degrees)	Longitude (decimal degrees)
FB C01	-34.09049	18.52161
FB C02	-34.08859	18.51102
FB C03	-34.08212	18.51132
FB C04	-34.07914	18.53176
FB C05	-34.07445	18.52827
FB C06	-34.07365	18.51155
FB C07	-34.07136	18.50485
FB C08	-34.07545	18.53889
FB C09	-34.05790	18.50309
FB C10	-34.06015	18.49500
FB C11	-34.06542	18.50239
FB C12	-34.06536	18.50499
FB C13	-34.06716	18.50295
FB C14	-34.07303	18.51607
FB C15	-34.07395	18.51665

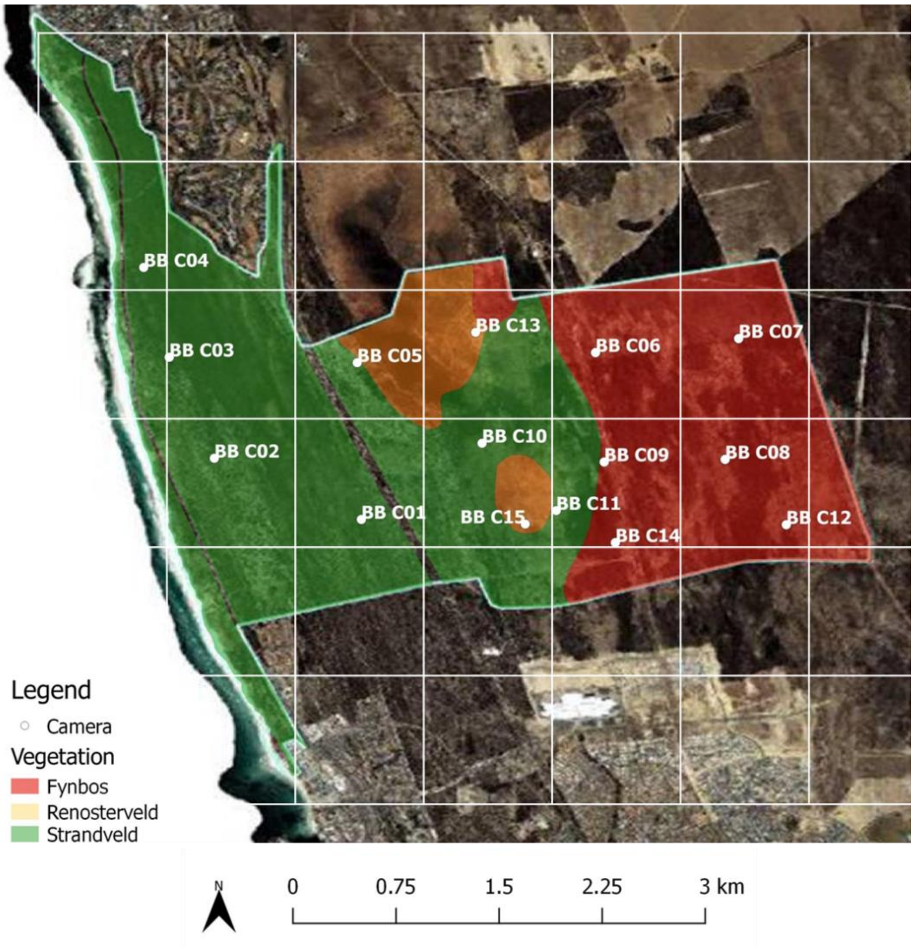
9. Table Bay Nature Reserve (880 ha)



Camera coordinates:

Camera Placement	Latitude (decimal degrees)	Longitude (decimal degrees)
TB C01	-33.83199	18.48927
TB C02	-33.83440	18.48896
TB C03	-33.83738	18.49509
TB C04	-33.83912	18.49810
TB C05	-33.84228	18.49469
TB C06	-33.84497	18.49664
TB C07	-33.84721	18.49821
TB C08	-33.85267	18.49969
TB C09	-33.84842	18.50481
TB C10	-33.84457	18.50262
TB C11	-33.84674	18.50994
TB C12	-33.84435	18.50960

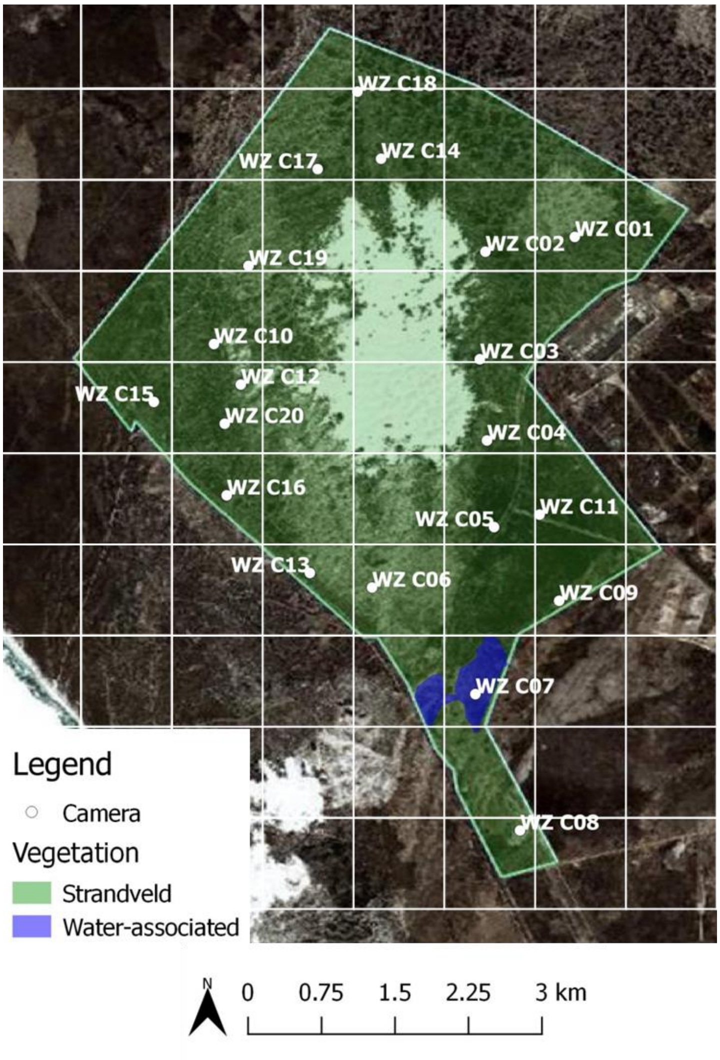
10. Blaauwberg Nature Reserve (1 445 ha)



Camera coordinates:

Camera Placement	Latitude (decimal degrees)	Longitude (decimal degrees)
BB C01	-33.77142	18.46114
BB C02	-33.76661	18.44958
BB C03	-33.75867	18.44604
BB C04	-33.75161	18.44403
BB C05	-33.75910	18.46081
BB C06	-33.75830	18.47954
BB C07	-33.75720	18.49079
BB C08	-33.76670	18.48973
BB C09	-33.76690	18.48023
BB C10	-33.76543	18.47062
BB C11	-33.77072	18.47645
BB C12	-33.77183	18.49454
BB C13	-33.75671	18.47012
BB C14	-33.77324	18.48110
BB C15	-33.77178	18.47402

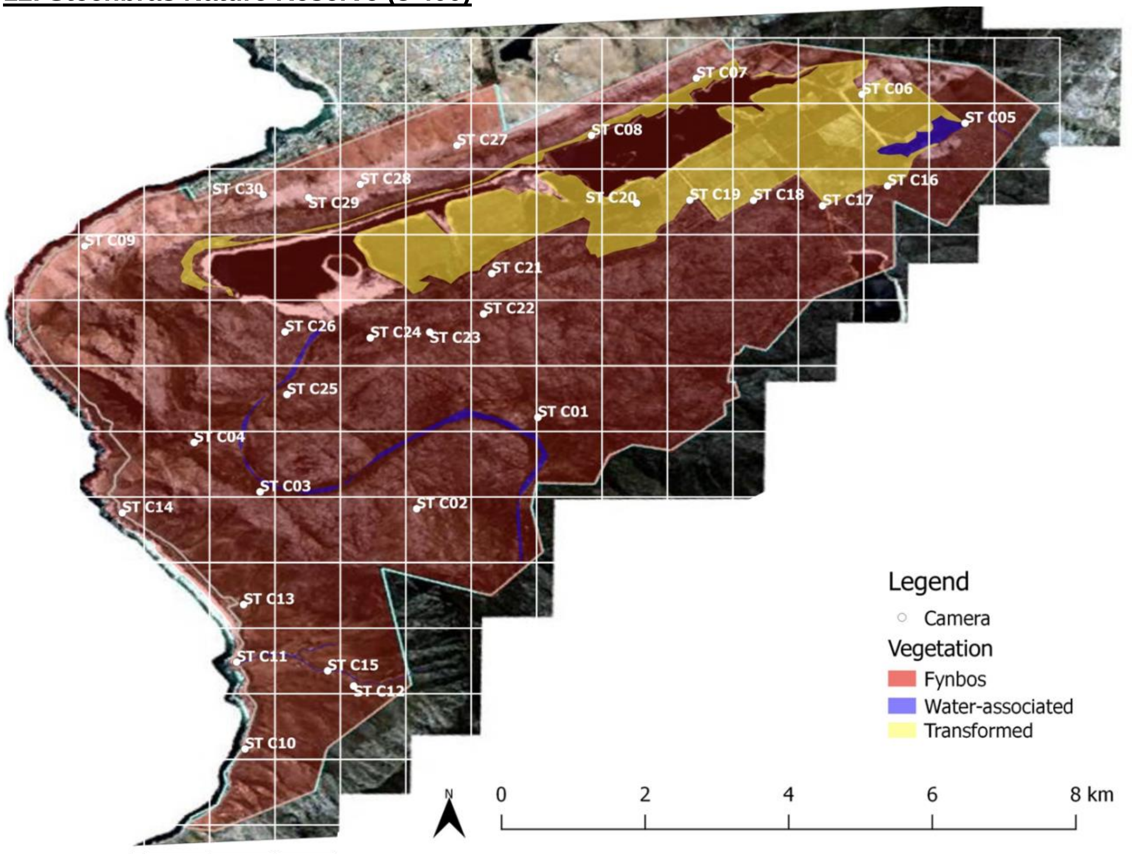
11. Witzands Aquifer Nature Reserve (1 700 ha)



Camera coordinates:

Camera Placement	Latitude (decimal degrees)	Longitude (decimal degrees)
WZ C01	-33.57995	18.45857
WZ C02	-33.58158	18.44874
WZ C03	-33.59342	18.44813
WZ C04	-33.60236	18.44891
WZ C05	-33.61189	18.44969
WZ C06	-33.61855	18.43627
WZ C07	-33.63029	18.44764
WZ C08	-33.64529	18.45255
WZ C09	-33.62001	18.45685
WZ C10	-33.59175	18.41887
WZ C11	-33.61053	18.45470
WZ C12	-33.59620	18.42182
WZ C13	-33.61695	18.42943
WZ C14	-33.57135	18.43726
WZ C15	-33.59812	18.41226
WZ C16	-33.60841	18.42027
WZ C17	-33.57249	18.43028
WZ C18	-33.56396	18.43467
WZ C19	-33.58314	18.42268
WZ C20	-33.60052	18.42006

12. Steenbras Nature Reserve (8 400)



Camera coordinates:

Camera Placement	Latitude (decimal degrees)	Longitude (decimal degrees)
ST C01	-34.20676	18.89976
ST C02	-34.22072	18.88125
ST C03	-34.21814	18.85736
ST C04	-34.21061	18.84731
ST C05	-34.16192	18.96496
ST C06	-34.15753	18.94921
ST C07	-34.15506	18.92394
ST C08	-34.16378	18.90792
ST C09	-34.18066	18.83058
ST C10	-34.25734	18.85505
ST C11	-34.24409	18.85378
ST C12	-34.24774	18.87164
ST C13	-34.23533	18, 85484
ST C14	-34.22132	18.83632
ST C15	-34.24542	18.86767
ST C16	-34.17147	18.95312
ST C17	-34.17451	18.94321
ST C18	-34.17367	18.93265
ST C19	-34.17364	18.92294
ST C20	-34.17412	18.91483
ST C21	-34.18483	18.89272
ST C22	-34.19100	18.89147
ST C23	-34.19380	18.88325
ST C24	-34.19466	18.87418
ST C25	-34.20329	18.86146
ST C26	-34.19374	18.86115
ST C27	-34.16531	18.88740
ST C28	-34.17125	18.87264
ST C29	-34.17325	18.86475
ST C30	-34.17281	18.85784

APPENDIX B – Historically-occurring medium and large mammal species list for the CCT nature reserves. Species presumed (P) to persist within each reserve based on Biodiversity database, iNaturalist and iSpot observations are indicated in light grey, and species recorded (R) in this study are indicated in dark grey. Species are listed alphabetic order by common name.

		Uitkamp		Bracken		Kenilworth		Zandvlei		Wolfgat		Tygerberg		Helderberg		False Bay		Table Bay		Blaauwberg		Witzands		Steenbras		No. reserves	
Species	Common Name	P	R	P	R	P	R	P	R	P	R	P	R	P	R	P	R	P	R	P	R	P	R	P	R	P	R
Orycteropus afer	Aardvark																									0	0
Proteles cristata	Aardwolf																									0	0
Loxodonta africanus	African elephant																									0	0
Felis sylvestris cafra	African wild cat																									0	0
Otocyon megalotis	Bat-eared fox																									1	0
Diceros b. bicornis	Black rhinoceros																									0	0
Canis mesomelas	Black-backed jackal																									1	0
Parahyaena brunnea	Brown hyena																									0	0
Aonyx capensis	Cape clawless otter																									4	4
Vulpes chama	Cape fox																									3	1
Raphicerus melanotis	Cape grysbok																									11	11
Equus z. zebra	Cape mountain zebra																									0	0
Hystrix africaeaustralis	Cape porcupine																									9	11
Lepus capensis/saxatilis	Cape/scrub hare																									4	5
Caracal caracal	Caracal																									5	6
Papio ursinus	Chacma baboon																									1	2
Sylvicapra grimmia	Common duiker																									6	5
Pelea capreolus	Grey rhebuck																									1	0
Pronolagus saundersiae	Hewitt's red rock hare																									0	1
Mellivora capensis	Honey badger																									3	5
Oreotragus oreotragus	Klipspringer																									1	1
Herpestes ichneumon	Large grey mongoose																									2	5
Genetta tigrina	Large spotted genet																									4	6
Panthera pardus	Leopard																									2	2
Panthera leo	Lion																									0	0

Species	Common Name	Uitkamp		Bracken		Kenilworth		Zandvlei		Wolfgat		Tygerberg		Helderberg		False Bay		Table Bay		Blaauwberg		Witzands		Steenbras		No. reserves	
		P	R	P	R	P	R	P	R	P	R	P	R	P	R	P	R	P	R	P	R	P	R	P	R	P	R
Equus quagga	Plains zebra																									0	0
Leptailurus serval	Serval																									0	0
Galerella pulverulenta	Small grey mongoose																									9	9
Genetta genetta	Small spotted genet																									3	3
Crocuta crocuta	Spotted hyena																									0	0
Raphicerus campestris	Steenbok																									5	3
Ictonyx striatus	Striped polecat																									4	4
Poecilogale albinucha	Striped weasel																									1	0
Atilax paludinosus	Water mongoose																									3	5
Lycaon pictus	Wild dog																									0	0
Cynictis penicillata	Yellow mongoose																									0	0
Total Native		2	3	4	5	2	1	8	5	7	5	6	11	11	12	10	8	3	6	15	9	7	12	9	12		
Reintroduced Species																											
Taurotragus oryx	Common eland																									2	2
Hippopotamus amphibius	Hippopotamus																									1	1
Alcelaphus b. caama	Red hartebeest																									1	1
Total Reintroduced		0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	2	0	0	2	2	0	0	0	0		
Non-native Species																											
Damaliscus p. pygargus	Bontebok																									1	0
Felis sylvestris catus	Domestic cat																									2	9
Canus lupus familiaris	Domestic dog																									4	2
Equus ferus caballus	Domestic horse																									0	1
Oryctolagus cuniculus	Domestic rabbit																									1	1
Sciurus carolinensis	Eastern grey squirrel																									2	1
Total Non-Native		2	1	0	2	0	2	2	1	1	0	2	3	1	1	1	1	1	1	1	1	1	0	0	1		