The ecology of large-spotted genets within an urban landscape

Craig D. Widdows

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ABSTRACT

Urbanization is one of the most damaging and rapidly expanding forms of anthropogenic landscape modification and is having profound consequences on biodiversity worldwide. The global increase in urbanization has resulted in exclusion of many carnivore species from human-altered landscapes due to a variety of anthropogenic impacts. However, despite the negative impacts of urbanization on carnivores, certain species such as large-spotted genets (*Genetta tigrina*) exhibit an ability to persist within urban areas. Despite their extensive distribution range, large-spotted genets are poorly studied in comparison to other African carnivores, with a handful of studies conducted on genetics, activity patterns and diet. Furthermore, no studies have focused on their ecology in an urban environment.

There have been increasing reports of large-spotted genets within urban areas throughout KwaZulu-Natal, South Africa. The mosaic of patches of native vegetation within this urban landscape provides habitats for a variety of wildlife species. The main aim of the study was to investigate the ecology of large-spotted genets within an urban landscape and to determine what factors facilitate their ability to persist in an urban environment.

Residential interviews were conducted to ascertain information pertaining to behavioural observations, land use as well as wildlife conflict and public perceptions of genets. Chi-square (χ^2) goodness-of-fit tests were used to determine significant differences in the frequency of responses. Domestic pet food was the main food item genets were observed eating. The most cited cause of genet fatalities were attacks by domestic animals and collisions with vehicles. Genets were reported using roof spaces for resting during daylight hours and as locales for breeding. The majority of respondents expressed positive attitudes towards genets in urban areas; negative views stemmed from concerns of disease transfer and impacts on wild bird populations.

We investigated the relationship between occurrence of large-spotted genets with various environmental variables believed to influence their site occupancy and detection in an urban environment. Presence/absence data was collected from 28 camera trap stations between June 2012 and October 2013 in Kloof, KwaZulu-Natal, South Africa. Average estimated occupancy of urban genets was 0.62 ± 0.14 with a detection probability 0.19 ± 0.03. Model selection indicated that bush cover and placement of camera traps on wide paths negatively influenced large-spotted genet occupancy. Both winter and fringe habitats influenced the site occupancy of large-spotted genets positively. Furthermore, bush cover was negatively associated with detection probability of urban genets.

The dietary composition and the influence of predictable feeding stations on urban large-spotted genets were investigated through scat analysis. Scats were collected on a monthly basis over a year from eleven midden sites. Invertebrates were the dominant prey items consumed with small mammals also forming an important component of the diet. Significant seasonal variation was recorded with the presence of birds and pollution in the scats with highest recorded frequencies during spring and winter respectively. During winter, when invertebrate abundance declined, urban genets increased their intake of anthropogenic refuse resulting in plastic, elastic bands and various other waste products present in the scats.

We investigated the effect of anthropogenic structures on the roost temperatures of large-spotted genets. Roost temperatures were recorded using i-Button$^{®}$ temperature loggers at known genet roosts in anthropogenic structures as well as in natural roost sites (tree hollows and rocky overhangs). Over the seasons temperatures varied significantly between months and among different roosts. However, anthropogenic roost temperatures were significantly higher than ambient temperatures throughout the study period. Furthermore, anthropogenic roosts had higher

temperatures (with lower variability) than natural roost sites. This study indicated the importance of anthropogenic structures as daytime roosts and for breeding for large-spotted genets within an urban matrix.

Given their adaptability and apparent success within the urban environment, we investigated the movement ecology of large-spotted genets within the urban landscape. Large-spotted genets were captured and fitted with global positioning system mobile transmitters. Although seven individuals were trapped, detailed data were only obtained for two individuals. Minimum convex polygons (MCPs), 95% and 50% kernel density estimates (KDEs) were calculated for a male and female large-spotted genet. The 95% kernel density home range sizes showed relatively small home range sized for both individuals, with an area of 48.2 ha (female) and 17.5 ha (male). Habitat selection within home ranges indicated that the male large-spotted genet preferred urban residential habitat while the female genet avoided nearby croplands. Both individuals used most of the habitats within their study area. Variable habitat used by both individuals in this study confirmed the species' ability to adapt to the urban mosaic of habitats.

This thesis showed the importance of a variety of factors on the distribution of large-spotted genets in landscapes where natural habitats are threatened by changing land use and increasing human populations. Furthermore, this thesis illustrated the ability of genets to live in an urban mosaic landscape by using a variety of anthropogenic resources.

PREFACE

The data described in this thesis were collected in Pietermaritzburg, Republic of South Africa from April 2012 to August 2015. Experimental work was carried out while registered at the School of Life Sciences, University of KwaZulu-Natal, Pietermaritzburg, under the supervision of Professor Colleen T. Downs.

This thesis, submitted for the degree of Doctor of Philosophy in the College of Agriculture, Science and Engineering, University of KwaZulu-Natal, Pietermaritzburg campus, represents original work by the author and has not otherwise been submitted in any form for any degree or diploma to any University. Where use has been made of the work of others, it is duly acknowledged in the text.

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Craig Widdows
November 2015

I certify that the above statement is correct and as the candidate's supervisor I have approved this thesis for submission.

……………………………..

Professor Colleen T. Downs
Supervisor
November 2015

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DECLARATION 1 - PLAGIARISM

I, Craig D. Widdows, declare that

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

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

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DECLARATION 2 - PUBLICATIONS

DETAILS OF CONTRIBUTION TO PUBLICATIONS that form part and/or include research presented in this thesis.

Publication 1

CD Widdows & CT Downs

Genets in the city: Community observations and perceptions of large-spotted genets in an urban environment.

Author contributions:

CDW conceived paper with CTD. CDW collected and analysed data, and wrote the paper. CTD contributed valuable comments to the manuscript.

Publication 2

CD Widdows, T Ramesh & CT Downs

Factors affecting the distribution of large spotted genets (*Genetta tigrina*) in an urban environment in South Africa

Author contributions:

CDW and TR conceived paper with CTD. CDW collected data, CDW was assisted by TR with data analyses. CW wrote the paper. CTD & TR contributed valuable comments to the manuscript.

Publication 3

CD Widdows & CT Downs

A genet drive-through. Are large-spotted genets using urban areas for "fast food"? A dietary analysis.

Author contributions:

CDW conceived paper with CTD. CDW collected and analysed data, and wrote the paper. CTD contributed valuable comments to the manuscript.

Publication 4

CD Widdows & CT Downs

Urban roost temperature of large-spotted genets: The effect of anthropogenic structures.

Author contributions:

CDW conceived paper with CTD. CDW collected and analysed data, and wrote the paper. CTD contributed valuable comments to the manuscript.

Publication 5

CD Widdows, M Pfeiffer & CT Downs

Home range and habitat use of large-spotted genets in the urban landscape of Kloof, South

Africa

Author contributions:

CDW conceived paper with CTD. CDW collected and analysed data, and wrote the paper. CDW was assisted by MP with data analyses. CTD contributed valuable comments to the manuscript.

Signed: ……………………………………………………

Craig D. Widdows
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Contents

CHAPTER 1

Introduction

The urban environment

Currently urbanization is increasing at a prolific rate and is the fastest growing type of land use worldwide, with over 3 billion people living in urban areas worldwide (Bokony et al. 2012; Murray & St Clair 2015). Urbanization is the anthropogenic conversion of land into commercial, industrial and residential uses and is currently one of the most important threats to biodiversity worldwide (McKinney 2002; Randa & Yunger 2006; Sol et al. 2013; Murray & St Clair 2015). Urbanization is increasing at an alarming rate and according to Lubowski et al. (2006) the amount of urban land between 1945 and 2002 within the United States of America quadrupled in area. It dramatically changes the composition and diversity of biotic communities and ecosystem structure as well as the direct depletion of natural habitats and ecosystems (Baker & Harris 2007; Bonier et al. 2007; Evans et al. 2009; Salek et al. 2015). This anthropogenic process further influences taxa habitat selection and use as well as individual fitness (Lewis et al. 2015). Therefore, urbanization has vital implications for biodiversity conservation at a global scale (Salek et al. 2015).

Urban areas comprise of a habitat mosaic of buildings of varying density, road infrastructure and isolated patches of natural vegetation (Swaddle et al. 2015; Soulsbury & White 2015). The fragmented nature of urban habitats isolates populations by preventing connectivity between patches (Magle et al. 2012; Soulsbury & White 2015). This can have a negative effect on the demographic and genetic variability of these populations (Magle et al. 2012; Newsome et al. 2015). Urban landscapes are characterised by high densities of human populations and large concentrations of food, water and pollution (light, noise and chemical) (Baker & Harris 2007;

McCleery & Parker 2011). Pollution is one of the primary aspects driving anthropogenic environmental change (Swaddle et al. 2015).

One particularly well documented impact of urban areas is light pollution. Ecological light pollution emanating from stadiums, factories, and streetlights, for example, alters the natural light regimes in ecosystems (Gehrt et al. 2010; Stone et al. 2015). The ecological implications of artificial night lighting are becoming increasingly apparent (Baker & Harris 2007; Van Langevelde et al. 2011; Davies et al. 2012; Stone et al. 2015). Artificial light has important consequences for nocturnal species because it alters predator-prey interactions, foraging, reproduction, communication and competition between individuals (Longcore & Rich 2004; Gehrt et al. 2010). For example, the artificial illumination of habitats has negatively affected the reproduction, social interactions and predator avoidance in many frog species (Buchanan 2006). Artificial lights attract many insects and moths, and some bat species such as *Vespadelus darlingtoni* and *Pipistrellus pipistrellus* use these artificial lights as foraging sites (Acharya & Fenton 1999; Arlettaz et al. 2000; Eisenbeis 2006; Hale et al. 2015; Stone et al. 2015). Artificial light induces a sensory overload that inhibits a moth's response to ultrasound and bat echolocation calls. Furthermore, the cost of foraging for bats is significantly reduced as insects become concentrated at predictable sites (Rydell 1992; Acharya & Fenton; 1999; Hale et al. 2015). However, not all bats benefit from light pollution. For example, in southern Britain, the commuting behaviour of *Rhinolophus hipposideros* from their roosts to feeding sites was delayed or alternative routes were used when artificial lights were set up along the flight paths used by this bat species (Stone et al. 2009). The change in commuting behaviour was attributed to the vulnerability of slow-flying species such as *R. hipposideros* to predation by birds of prey such as owls in artificially lit areas. Furthermore, suboptimal routes (reduced shelter or increased distance to foraging areas) selected by *R.*

hipposideros in response to artificially lit areas will further impact individual fitness (Stone et al. 2009).

As urban populations and the associated areas increase, levels of noise pollution emanating from urban areas has increased at an alarming rate (Swaddle et al. 2015). In a study conducted by Chepesiuk (2005), 65% of the European population is exposed to anthropogenic sound levels exceeding 55dB. The commercial shipping industry is responsible for 16 fold increases in ocean noise levels (Hildebrand 2009). The change in noise levels can alter communication, foraging dynamics, vigilance and animal physiology (Francis & Barber 2013).

Urban-rural gradient/mosaic

Native habitats and subsequently native taxa are largely absent in the centre of many human-modified landscapes (Randa & Yunger 2006; Kertson et al. 2011). Urban landscapes typically reflect a gradient of land use which varies from the highly urbanized city centres through to the suburban areas and rural landscapes (Riem et al. 2012; Salek et al. 2015). Alternatively, urban areas form a mosaic of natural and anthropogenic features. Habitat structure and resource availability change along this gradient. Changes in abiotic factors such as light, noise, pollution, traffic and microclimate conditions, human activity and intensity of habitat fragmentation also occur along this gradient (Salek et al. 2015). Urban-gradient studies indicate that for many taxa, the number of native species decrease and the number of non-native species increase towards the centre of urban areas (McKinney 2002). Urbanization not only depletes natural habitats but also alters landscape heterogeneity, which can subsequently have significant effects on the structure and function of affected ecosystems (Chace & Walsh 2004; Randa & Yunger 2006; Jung & Kalko

2010). As a result, many of these species have withdrawn into reduced ranges in response to the spread of urban environments (Bateman & Fleming 2012).

The presence of natural areas such as greenbelts, vegetated gardens, parks and conservancies within urban areas are essential for sustaining wildlife populations and can increase urban biodiversity (Savard et al. 2000; Bateman & Fleming 2012; Shultz et al. 2012). These connections of green spaces within urban areas act as refugia for a variety of species and can also alleviate the impact of urbanization on wildlife populations (Baker & Harris 2007; Ordeñana et al. 2010; Fleming & Bateman 2012). Urban green spaces provide long-term habitat stability in a rapidly changing urban environment (LaPoint et al. 2015). Riparian habitats along streams and waterways can present wildlife with protected corridors with reduced human activity within urban areas (Savard et al. 2000). It is imperative that urban planning and wildlife conservation techniques be implemented to reduce the negative impacts of urbanization on biodiversity. A good example of such techniques is the Metropolitan Open Space System (MOSS) which is a network of open spaces within urban areas in South Africa (Donaldson-Selby et al. 2007). This system ensures the sustained protection of flora and fauna as well as the protection of environmental services.

Implications to urban living

Despite the generally negative impacts of urbanization, urban ecosystems are becoming increasingly important for local biodiversity (Magle et al. 2012; Lewis et al. 2015). With the increasing depletion of natural habitats, urban resources are likely to become increasing important for wildlife, thus driving wildlife into urban areas (Bateman & Fleming 2012). This trend is reflected in the literature with an increase in the publication of scientific studies in urban wildlife research and urban ecology in the last 20 years (Magle et al. 2012). Urban ecology is not only the

study of the interaction between the biotic and abiotic factors in an urban environment, but also encompasses the ability of humans and ecological processes to coexist in a human-dominated environment (Marzluff et al. 2008; Cilliers & Siebert 2012). Positive interactions between residents and wildlife living in urban areas can reinforce the importance of biodiversity conservation (Saito & Koike 2015). However, there are a variety of threats for wildlife moving through and living in urban landscapes such as their vulnerability to human persecution, collisions with motor vehicles, or attacks by domestic animals (Magle et al. 2012; Parker & Nilon 2012). Within the urban environment, networks of roads separate green areas, thus increasing the probability of vehicle collisions (Murray et al. 2015). Collisions with motor vehicles account for a large proportion of urban carnivore deaths. In a study conducted by Harris et al. (1992), ± 50 000 badgers (*Meles meles*) were killed by motor vehicles each year in Britain. Collisions are not confined to small carnivores, with black bears (*Ursus americanus*) and coyotes (*Canis latrans*) also succumbing to road deaths (Chamberlain & Leopold 2001; Beckman & Lackey 2008). Other human-wildlife conflicts in urban areas include damage to lawns and landscaping, and the spread of diseases between wildlife populations and from wildlife to domestic pets and humans (Magle et al. 2012). Urban populations may live in close proximity to each other, resulting in a greater exposure and transmission of diseases such as rabies and bovine tuberculosis (Harris et al. 1992; Ordeñana et al. 2010; Saito & Koike 2015). Urban species such as racoon, kit foxes (*Vulpes macrotis*) and striped skunks (*Mephitis mephitis*) have resulted in the spread of zoonotic diseases (diseases transferred from animals to humans) within urban areas (Gehrt et al. 2010). This conflict becomes more acute when livestock and poultry are involved, as the main cause of mortality in many feral carnivore species is the anthropogenic actions taken (killing and poisoning) to prevent the attacks on livestock (Garrote et al. 2013).

Species response to urbanization

Responses to urbanization vary among species, with some dominating urban landscapes while others avoid these areas (Fischer et al. 2012; Newsome et al. 2015). There are few examples of indigenous species being able to survive in highly urbanized areas due to their inability to adapt to the rapid changes in their environment associated with urbanization (Bonier *et al.* 2007; Barnagaud et al. 2011; Threlfall et al. 2011). Specialist species are particularly vulnerable to environmental change, whereas more generalist species tend to adapt to habitat modification (Cilliers & Siebert 2012). A study conducted by Pauw & Louw (2012) supported this by indicating a decline in specialist species such as the malachite sunbird (*Nectarinia famosa*) in metropolitan areas. The reduction of specialist species in urban areas could result in biotic homogenization (Cilliers & Siebert 2012). Species that have taken up this anthropogenic niche exhibit significant ecological, behavioural and demographic plasticity (McCleery & Parker 2011; Lowry et al. 2013). These adaptations are consistent across taxa and include reduced vigilance, reduced dispersal and home ranges, increased population densities, and altered diel activity patterns (Parker & Nilon 2012; Newsome et al. 2015). Shifts in diel activity patterns to avoid human activity have been documented in various carnivore species, such as black bears (Beckman & Berger 2003), coyotes, and bobcats (*Lynx rufus*) (Tigas et al. 2002; Baker et al. 2007). These characteristics aid in the successful colonization and adaptation to novel anthropogenic environments (Newsome et al. 2015). In general, most avian and mammalian species respond negatively to increased levels of urbanization, as they are unable to adapt to and exploit urban centres (Randa & Yunger 2006). According to Blair (2001), seven resident summer bird species were documented in California's central business district as opposed to the 21 avian species recorded in the city's outer limits. The

more developed sectors of the urban-rural gradient are characterised by long-term habitat loss and high rates of local extinctions (Randa & Yunger 2006).

Despite the many challenges of living in urban environments, there has been an increasing tendency for some avian and mammalian species to colonize urban habitats (Parker & Nilon 2012; Poessel et al. 2012; Murray et al. 2015; Saito & Koike 2015). These species use the urban landscape and its ecological opportunities with some species thriving in these urban areas by using a variety of natural and anthropogenic resources (Poessel et al. 2012; Sol et al. 2013). Avian species such as the house sparrow (*Passer domesticus*) and hadeda ibis (*Bostrychia hagedash*) survive and even thrive in urban areas (Macdonald et al. 1986; De Laet & Summers-Smith 2007). The house sparrow is unique among many wild avian species in its close association with humans and has expanded its range worldwide, mainly in urban areas (De Laet & Summers-Smith 2007). These "urban utilisers" are able to use the large collection of ecological niches, abundant anthropogenic food resources, and novel anthropogenic habitats (Fischer et al. 2015; Salek et al. 2015). However, the use of anthropogenic food resources can have negative impacts on urban species. In a study conducted by Heiss et al. (2009), American crow's (*Corvus brachyrhynchos*) nestlings reared in an urban landscape exhibited nutritional limitations (lower blood protein and calcium concentrations) due to the high levels of anthropogenic food items in their diet (Heiss et al. 2009).

Urban carnivores

Urbanization has had a negative impact on many large mammalian carnivores, which have low reproductive rates, extensive home ranges, high prey biomass requirements and increased susceptibility to human persecution (Randa & Yunger 2006; Kertson et al. 2011; Salek et al. 2015).

These ecological characteristics make them particularly vulnerable to human-dominated landscapes and associated activities which may result in population decline (Beckmann & Berger 2003; Salek et al. 2015). As a result, many carnivore species actively avoid highly populated urban landscapes (Bateman & Fleming 2012; Murray et al. 2015). However, some medium-sized, generalist carnivores such as the red fox (*Vulpes vulpes*) and coyote, have successfully exploited many urban environments in various parts of the world (Randa & Yunger 2006; Gehrt 2007; Poessel et al. 2012; Statham et al. 2012). This can be attributed to the opportunistic, adaptable, and generalistic behaviour exhibited by these canid species (Poessel et al. 2012). Consequently, canid population densities in urban areas can be as much as fifteen times higher than in unspoiled rural habitats due to human-derived resources (Bino et al. 2011). For example, behavioural changes such as reduced territories, tolerance of man, and exploitation of anthropogenic food sources have resulted in high urban population densities of red fox in many European countries (Contesse et al. 2004; Francis & Chadwick 2012). The red fox has the most expansive geographic range of any wild carnivore and has colonized 114 cities in 56 countries worldwide (Bateman and Fleming 2012). Human settlements are providing adequate habitats, shelter and food sources for red fox (Contesse et al. 2004). Urban populations have been so successful in urban areas that in Zurich, Switzerland, there is a greater population density of red foxes in urban areas as opposed to rural habitats (Contesse et al. 2004). Subsequently, the home range of red fox is smaller in urban environments as opposed to rural habitats (Gehrt et al. 2010). This reduction in home range size has also been documented in raccoons (*Procyon lotor*) (Prange et al. 2004). High concentrations of anthropogenic food resources are readily available and scavenged by urban fox populations (Contesse et al. 2004). More than half the stomach contents of red fox in Zurich, Switzerland contained anthropogenic food resources (Contesse et al. 2004). Exploitation of human-derived

resources has a profound effect on the reproductive success of red fox and other canids in human settlements (Contesse et al. 2004; Bino et al. 2010). The reproduction and development of red foxes differs in urban and rural habitats. Red fox are classified a socially monogamous, with each group producing one litter of cubs each year (Gehrt et al. 2010). However, multiple litters are common in urban fox populations due to the high productivity of the urban environment (Gehrt et al. 2010). Urban males and females have also been observed mating with individuals outside of their family groups (Iossa et al. 2008; Gehrt et al. 2010).

Coyotes have become common in many urban areas across the United States of America (Gehrt 2007). In the 1990's there was a fifteen-fold increase in the number of coyotes that were removed from the metropolitan area of Chicago annually (Gehrt 2011; Bateman & Fleming 2012). The flexibility of their diet, social and behavioural ecology has facilitated their colonization of urban environments (Bateman & Fleming 2012). They have been documented as having smaller home ranges in urban compared with rural habitats and this may contribute to their establishment and persistence in metropolitan areas (Atwood et al. 2004; Gehrt 2007; Gehrt et al. 2010). Activity patterns of coyotes have shifted from typical diurnal and nocturnal movements to mainly nocturnal activity patterns in urban areas (Randa & Yunger 2006). In a study conducted by Murray et al. (2015), coyotes with an urban landscape in Canada, adjusted their activity patterns (peak activity at midnight) to reduce overlap with human and vehicular activity. Collared individuals that were killed on roads were most active during dusk (peak human activity hours) and by contrast collared individuals that survived were most active and crossed roads during midnight. This indicated behaviour flexibility as individuals with peak activity late at night crossed roads more frequently, thus not avoiding these areas but adapting their behaviour to exploit the areas. Bobcats (Tigas et al. 2002; Lowry et al. 2013), grizzly bears (*Ursus arctos*) (Northrup et al. 2012) and leopards

(*Panthera pardus*) (Odden et al. 2014) also have shifted their activity to peak late in the night so as to avoid human and vehicle activity. Benefits of the change in diel activity are firstly, a reduction in human activity at night that results in a lower probability of encountering humans (Gehrt et al. 2010) and secondly, traffic volumes are lower at night, allowing coyotes to easily cross roads with reduced mortality due to collisions (Gehrt et al. 2010). Coyotes, grey foxes (*Urocyon cinereoargenteus*) and bobcats make use of the urban environment. However, these species will preferentially select habitats that minimize encounters with humans (Fischer et al. 2012; Gehrt et al. 2012; Lowry et al. 2013). Similarly, urban red fox and coyote populations exist at higher densities than those in rural habitats due to increased anthropogenic food resources (Gehrt et al. 2010). Both these species demonstrate variation in their diets dependent on local and seasonal food availability (Randa & Yunger 2006). American raccoons and black bears also have increased their occurrence and persistence in urban areas in the last decade (Bateman & Fleming 2012). Raccoons are able to reach higher densities in urban areas as opposed to natural habitats by making use of a variety of anthropogenic resources (Prange et al. 2003).

Another urban resource that carnivores readily exploit are anthropogenic structures. Urban carnivores often use anthropogenic structures for secure shelters and breeding sites (Bateman & Fleming 2012). In some instances, these anthropogenic structures are being used even when natural alternatives such as green belts are available (Lowry et al. 2013). Japanese badgers (*Meles anakuma*) exploit abandoned buildings as secure shelters (Kaneko *et al*. 2006). Raccoons, brushtail possums (*Trichosurus vulpecula*), and Virginia opossums (*Didelphis virginiana*) also make use of anthropogenic structures such as buildings, sewers and chimneys as alternative resting and breeding sites (Prange et al. 2004; Wright et al. 2012; Lowry et al. 2013). Medium-sized carnivores such as coyotes, red foxes and kit foxes (*Vulpes macrotis*) readily exploit vacant plots of land,

parks, and other patches of vegetation as protective cover (Gehrt et al. 2010; Bateman & Fleming 2012).

Another factor that may facilitate the high densities of urban medium-sized carnivores is the absence of apex predators from urban environments resulting in a lack of top-down regulation (Fischer et al. 2012). Apart from the more common northern hemisphere urban carnivores, certain small to medium carnivores such as the small-spotted genet (*Genetta genetta*), slender mongoose (*Galerella sanguinea*) and yellow mongoose (*Cynictis penicillata*) have been observed in urban landscapes in South Africa. Larger African carnivores such as the spotted hyena (*Crocuta crocuta*) and striped hyena (*Hyaena hyaena*) have also been known to forage around urban areas in Ethiopia and Israel respectively (Abay et al. 2011; Yirga et al. 2015). Response rates of spotted hyena were significantly higher around urban garbage dumps as opposed to agricultural landscapes within the districts of Tigray, Ethiopia (Yirga et al. 2015). However, these species have not reached the cosmopolitan status as some of their North American counterparts.

The majority (94%) of urban carnivore research has been conducted in North America and Europe, focusing on species including coyote, red fox, grey fox, raccoon, black bear and opossum (Table 1). There has been a lack of urban carnivore studies conducted in the southern hemisphere, with little attention focusing on the rapidly urbanizing and high biodiversity continent of Africa (Magle et al. 2012). As southern Africa becomes more urbanized it is imperative that studies determine the effect human-modified landscapes are having on carnivore species.

Large-spotted genets

Large-spotted genets (*Genetta tigrina*) belong to the family Viverridae and are small, exclusively nocturnal carnivores that have a relatively wide distribution in South Africa (Rowe-Rowe 1978;

Smithers 1983; Roberts et al. 2007). Their status according to the Red Data Book of southern Africa is Least Concern and current threats include poaching for traditional medicine in South Africa (Gaubert & Hoffmann 2008). However, very little is known about the home range and habitat use of these nocturnal carnivores. Due to their diminutive size and elusive habitats, most Viverridae and Herpestidae have not been studied in the same detail as other carnivores in southern Africa (Ramesh & Downs 2014). Large-spotted genets are normally solitary, however, females may be accompanied by a mate during the breeding season or by the young of the previous litter (Rautenbach 1982). Their body is elongated with spots and other dark markings that vary from black to a rusty colour. Large-spotted genet are a relatively small carnivore weighing between 1.5 and 3.2 kg, and have a long white-ringed tail. They are adapted to a semi-arboreal lifestyle and rest during the daylight hours in dense trees, hollow logs, and under tree roots (Maddock 1988; Roberts et al. 2007). In KwaZulu-Natal (KZN), they have been recorded using anthropogenic structures such as outbuildings and roofs of houses as resting places (Skinner & Chimimba 2005). Both large-spotted and small-spotted genet (*Genetta genetta*) show a high tolerance capacity to human activity and have been recorded within urban landscapes (Virgos & Casanova 1997). According to Ramesh and Downs (2014), the occurrence of *G. tigrina* in rural KZN was positively associated with human abundance. The absence of large carnivores within urban landscapes could further contribute to the expansion of genet populations within urban areas through a reduction in competition and intraguild predation (Roemer et al. 2009).

Large-spotted genets use middens, which are usually in large branches in trees (Rowe-Rowe 1978; Skinner & Chimimba 2005). They are found in a range of habitats and show preference for riverine habitats, and well-covered woodland habitats close to permanent water (Rowe-Rowe 1992). Ramesh and Downs (2014) indicated that the occurrence of *G. tigrina* was

negatively associated with higher attitudes. Similar trends have been reported for *Genetta genetta* which inhabit low altitudinal habitats (Virgos and Casanovas 1997). Large-spotted genet association with low altitudinal habitats is likely due to thermal restrictions coupled with their insectivorous diet (Virgos & Casanovas 1997).

Large-spotted genets have an opportunistic and generalist diet consisting of invertebrates, particularly insects, small rodents, reptiles, birds and wild fruits with amphibians, arachnids and myriapods also recorded in the diet (Rowe-Rowe 1992). Large-spotted genets breed seasonally in the summer months from September to March with a gestation period of 70-77 days (Smithers 1983; Skinner & Chimimba 2005). Females litter in hollows of trees, rocky areas, and even in the roofs of houses (Skinner & Chimimba 2005). Large-spotted genet females have been recorded using roofs to rear their young in urban areas in Gauteng, South Africa (Rowe-Rowe 1992; Skinner & Chimimba 2005). On average 2-3 young are born per litter and their eyes open after 10 days of being born (Maddock 1988). The canine teeth emerge after four weeks and the young are weaned after ±10 weeks (Rowe-Rowe 1992; Skinner & Chimimba 2005). The skin and tail of *G. tigrina* are used as traditional medicine in the treatment of eyes, as well as decorating traditional garments and charms (Rowe-Rowe 1992).

Hybridization has been recorded within the *Genetta* genus, particularly within South Africa (Gaubert et al. 2005). *Genetta tigrina* and *G. maculata* form part of the large-spotted genet complex (Gaubert et al. 2003). The morphological similarities between *G. tigrina* and *G. maculata* has provided uncertainty regarding the assessment of taxonomic boundaries. In addition, delimitations within *G. maculata* are difficult to determine because of high variability in coat patterns, and several species/subspecies have been proposed for southern African populations (Crawford-Cabral & Fernandes 2001). This is particularly true within coastal KwaZulu-Natal

(KZN) where hybridization and range overlap have been recorded for *G. tigrina* and *G. maculata*. Individuals captured and assessed from KZN bore a mosaic of coat patterns between *G. tigrina* and *G. maculata* (Gaubert et al. 2005). Furthermore, a specimen from Umgeni Valley Game Ranch (KZN) had a *maculata*-like coat pattern but based on discrete morphometric characters and discriminant scores, this individual displayed morphometric features which characterized *G. tigrina*. The validity of the species status of *G. maculata* and *G. tigrina* is confused due to reports of crossbreeding within captivity (Gray 1971). As a consequence of the uncertainties within the large-spotted genet complex within KZN, we generally referred to large-spotted genets as *G. tigrina* in the current study. This was particularly useful when utilising citizen science as guide books for the area refer to large-spotted genets as *G. tigrina*. Furthermore, residents would have experienced difficulties differentiating between *G. tigrina*, *G. maculata* or hybrids within this complex.

Motivation for this study

Urbanization is one of the most damaging and rapidly expanding forms of anthropogenic landscape modification and is having profound consequences on biodiversity worldwide (Brearley et al. 2010; McCleery & Parker 2011; Murray et al. 2015). Rapid human population growth coupled with the loss and fragmentation of habitat by urbanisation is fast becoming an increasing global problem. Demographic assessments estimate that there will be a 50% increase in the human population by the year 2050 and half of these people will live in urban areas (Jung & Kalko 2010). Africa and Asia are predicted to the main contributors to the rise in global urban populations (Saito & Koike 2015). These statistics have huge implications for taxa that inhabit urban areas or habitats on the fringe of urban areas. The reduction of pristine, undisturbed habitats is likely to result in an

increased importance of urban landscapes and resources for the conservation of wild taxa (Bateman & Fleming 2012). Therefore, it is imperative that we understand the ecological processes that operate within urban landscapes (Rodewald et al. 2013). Urban ecosystems have the ability to connect local citizens with their local biodiversity and subsequently create awareness of these expanding ecosystems (Savard et al. 2000; Magle et al. 2012). Focusing management and conservation initiatives in urban areas will allow local citizens to develop a deeper appreciation for their urban environment and can enhance residents experience with nature (Saito & Koike 2015). Furthermore, it is important to understand why certain species avoid metropolitan areas while others reach peak densities in urban environments (Bateman & Fleming 2012; Sol et al. 2013). By understanding more about the biology of these species it will allow the determination of characteristics favouring urban persistence, this is likely to assist small carnivore conservation and management (Bateman & Fleming 2012).

This study aimed to provide a window into the ecology of large-spotted genets in an urban environment and what characteristics of this urban landscape are potentially influencing the persistence of large-spotted genet populations. Although urban wildlife biology is an increasing discipline, the overall number of publications in this field remains relatively low (Magle et al. 2012). The majority of urban wildlife research has been conducted in North America, Europe and Australia with little attention focusing on the rapidly urbanizing and high biodiversity continent of Africa (Magle et al. 2012). Most of the urban ecological research in South Africa has focused on birds, with little focus on small carnivores, in particular large-spotted genets. Furthermore, large-spotted genets have a wide distribution within southern Africa but there is limited data pertaining to the species distribution (Ramesh & Downs 2014). Understanding more about the urban biology

of large-spotted genets is likely to aid the management and conservation of this nocturnal species in the rapidly urbanizing landscape within KZN.

The main aim of this study was to investigate the ecology of large-spotted genets in an urban environment. Furthermore, the study aimed to determine the factors that facilitate genet populations within an urban environment. It was expected that the urban landscape would have abundant food and shelter resources for these small carnivores. Wildlife conflict was expected to be high within the highly populated urban area and community perceptions would be dependent on individual interactions with genets.

Arrangement of thesis

The thesis is arranged as chapters prepared for publication in peer-reviewed journals, and so some repetition in the chapters was unavoidable. The respective hypotheses are presented in each chapter. There are five experimental chapters:

Chapter 2: Genets in the city: Community observations and perceptions of large-spotted genets in an urban environment.

Chapter 3: Factors affecting the distribution of large spotted genets (*Genetta tigrina*) in an urban environment in South Africa.

Chapter 4: A genet drive-through: Are large-spotted genets using urban areas for "fast food"? A dietary analysis.

Chapter 5: Urban roost temperature of large-spotted genets (*Genetta tigrina*): The effect of anthropogenic structures.

Chapter 6: Home range and habitat use of large-spotted genets in the urban landscape of Kloof, South Africa.

Finally, the thesis has a concluding chapter that summarizes the various components of this study.

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Table

Table 1. Studies focusing on urban ecology of carnivores in various countries.

Study Species	Latin name	Location	Authors
Raccoon	*Procyon lotor*	North America	Hoffmann & Gottschang (1977)
Coyote	*Canis latrans*	North America	Atkinson & Shackleton (1991)
Skunk, Raccoon, Red Fox	*Mephitis mephitis, Procyon lotor, Vulpes vulpes*	Canada	Rosatte et al. (1991)
Raccoon, Skunk, Opossum	*Procyon lotor, Mephitis mephitis, Didelphis virginiana*	North America	Clark (1994)
Coyote	*Canis latrans*	North America	Quinn (1997)
Red fox	*Vulpes vulpes*	Canada	Adkins & Stott (1998)
Red fox	*Vulpes vulpes*	Australia	Marks & Bloomfield (1999)
Coyote	*Canis latrans*	North America	Grinder & Krausman (2001)
Bobcat, Coyote	*Lynx rufus, Canis latrans*	North America	Tigas et al. (2002)
Raccoon	*Procyon lotor*	North America	Prange et al. (2003)
Bob cat, Coyote	*Lynx rufus, Canis latrans*	North America	Riley et al. (2003)
Coyote	*Canis latrans*	North America	Atwood et al. (2004)
Red Fox	*Vulpes vulpes*	North America	Contesse et al. (2004)
Coyote, Raccoon, Skunk	*Canis latrans, Procyon lotor, Mephitis mephitis*	North America	Gehrt (2004)
Coyote	*Canis latrans*	North America	Morey (2004)
Dingo	*Canis lupus dingo*	Australia	Allen (2006)
Coyote, Raccoon, Red Fox	*Canis latrans, Procyon lotor, Vulpes vulpes*	North America	Randa & Yunger (2006)
Bobcat, Grey Fox	*Lynx rufus, Urocyon cinereoargenteus*	North America	Riley (2006)
European badger	*Meles meles*	Japan	Kaneko et al. (2006)
Red fox	*Vulpes vulpes*	North America	Baker et al. (2007)
Review	-	Britain	Baker & Harris (2007)
Coyote	*Canis latrans*	North America	Carrillo et al. (2007)
Coyote	*Canis latrans*	North America	Farrar (2007)

Black bear	*Ursus americanus*	North America	Beckmann & Lackey (2008)
European badger	*Meles meles*	United Kingdom	Davison et al. (2008)
European badger	*Meles meles*	United Kingdom	Davison et al. (2009)
Coyote	*Canis latrans*	North America	Gehrt et al. (2009)
Review	-	North America	Gehrt et al. (2010)
Raccoon, Bobcat, Opossum	*Procyon lotor, Lynx rufus, Didelphis virginiana*	North America	Ordeñana et al. (2010)
Cougar	*Puma concolor*	North America	Kertson et al. (2011)
Review	-	-	Bateman & Fleming (2012)
Grey Fox	*Urocyon cinereoargenteus*	North America	Cooper et al. (2012)
Raccoon	*Procyon lotor*	North America	Grazer et al. (2012)
Grey fox	*Urocyon cinereoargenteus*	North America	Kapfer & Kirk (2012)
Virginia opossum	*Didelphis virginiana*	North America	Wright et al. (2012)
Leopard	*Panthera pardus*	India	Odden et al. (2014)
Bobcat and Puma	*Lynx rufus, Puma concolor*	Colorado	Lewis et al. (2015)
Coyote	*Canis latrans*	Canada	Murray et al. (2015)
Coyote	*Canis latrans*	Chicago	Newsome et al. (2015)
Review	*Vulpes vulpes, Canis latrans, Lynx rufus, Procyon lotor*	North America	Salek et al. (2015)
Spotted hyena	*Crocuta crocuta*	Ethiopia	Yirga et al. (2015)

CHAPTER 2

Genets in the city: Community observations and perceptions of large-spotted genets in an urban environment

C. WIDDOWS AND C.T. DOWNS[*]

School of Life Sciences, University of KwaZulu-Natal, Private Bag X01, Scottsville,

Pietermaritzburg, KwaZulu-Natal, 3209, South Africa

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Abstract

The global increase in urbanization has resulted in exclusion of many carnivore species from human-altered landscapes due to a variety of anthropogenic impacts. However, despite the negative impacts of urbanization on carnivores, certain species exhibit an ability to survive within urban areas. There have been increasing reports of large-spotted genets (*Genetta tigrina*) within urban areas throughout KwaZulu-Natal, South Africa. Consequently, interviews were conducted in the suburbs of Kloof/Hillcrest, KwaZulu-Natal to establish information pertaining to behavioural observations, land use, wildlife-conflict and public perceptions of genets for their presence in urban areas. Walking between areas (during foraging bouts or between roost sites) was the main genet activity observed. Domestic pet food was the main food item genets were observed eating. The most cited cause of genet fatalities were attacks by domestic dogs and collisions with vehicles. Genets were reported using roof spaces for resting during daylight hours and for breeding. Respondents reported providing supplementary food such as meat, chicken and pet food to urban genets. The majority of respondents expressed positive attitudes towards genets in urban

Kloof. Negative views stemmed from concerns of disease transfer (e.g. rabies) and impacts on wild bird populations. This study illustrated the adaptability of genets to live in urban areas and their ability to utilise anthropogenic resources. Furthermore, the study identified the various threats to genets in an urban environment.

Key words: Carnivore, genet, land use, perceptions, urbanization

Correspondent: downs@ukzn.ac.za

Introduction

Currently urbanization is increasing at a prolific rate and is the fastest growing form of land use worldwide (Murray and St Clair 2015). This global phenomenon has resulted in a dramatic loss of natural landscapes and associated biodiversity (Lowry et al. 2013; Francis et al. 2015). Urbanization is attributed as the most significant cause of species loss in North America (Marzluff et al. 2001). Urban environments are generally characterized by high densities of human populations, varying densities of buildings and large concentrations of food, water, and pollution (light, noise and chemical) (Baker and Harris 2007; McCleery and Parker 2011). Anthropogenic conversion of landscapes has resulted in the direct depletion of natural ecosystems as well as the alteration of biotic communities' composition and diversity (Baker and Harris 2007; Evans et al. 2009; Murray and St Clair 2015).

Increase in urbanization has resulted in many species living in close proximity to human populations (Stott et al. 2015). Urbanization also impacts landscape heterogeneity, which can subsequently have significant effects on ecosystem structure and functioning (Chace and Walsh 2004; Soulsbury and White 2015). The homogenization of habitats is a cause of local extinctions within these novel landscapes (Francis et al. 2015). The composition and complexity of human-

modified landscapes, as well as the fragmented nature of these landscapes, isolates wildlife populations by preventing connectivity between patches of suitable habitats (Magle et al. 2012; Parker and Nilon 2012). These patches become increasingly isolated and degraded as urban areas expand (Francis et al. 2015). This can have a negative effect on both the demographic and genetic variability of wildlife populations (Magle et al. 2012). In a study conducted by Heiss et al. (2009), American crow (*Corvus brachyrhynchos*) nestlings reared in an urban landscape exhibited nutritional limitations (lower blood protein and calcium concentrations) due to the high levels on anthropogenic resources in their diet (Heiss et al. 2009).

Urban landscapes typically reflect a gradient of land use or mosaic from the highly urbanized metropolitan centres through to the suburban areas and rural (less altered) landscapes (McKinney 2002; Riem et al. 2012), the result of which creates a mosaic of roads, gardens, residential houses and industrial buildings throughout the urban landscape (Pickett and Cadenasso 2008; Soulsbury and White 2015). Particularly in South Africa, the gradient of land use correlates with the rugged terrain where development is difficult and the land is left unchanged creating a mosaic. Irregular terrain makes building, transportation, cultivation, and subsequently urban development very difficult and these areas often remain free from development (Nunn and Puga 2012). The degree of land use and habitat modification is generally lower with increasing distance from the urban core (Salek et al. 2014). This can be used to quantify the effects of urbanization on wildlife communities and subsequently determine what areas within this gradient these species are utilizing (Riem et al. 2012; Salek et al. 2014). For many taxa, the number of native species decreases and the number of non-native species increases towards the centre of urban areas (McKinney 2002). Indigenous habitats are often absent within city centres, and these areas of the urban-rural gradient are associated with the highest extinction rates (McKinney 2002; Randa and

Yunger 2006). Furthermore, species richness is negatively impacted across the urban-rural gradient, with the lowest species richness recorded in the urban centre (Soulsbury and White 2015). The presence of patches of natural green areas such as gardens, parks, cemeteries, vacant plots, and conservancies within urban areas are essential for sustaining wildlife populations and can increase urban biodiversity (Savard et al. 2000; Shultz et al. 2012). These connections of green spaces within urban areas act as refugia for a variety of species and can also alleviate the impact of urbanization on wildlife populations (Baker and Harris 2007; Ordeñana et al. 2010; Bateman and Fleming 2012). Furthermore, urban green spaces provide long-term habitat stability in a rapidly changing urban environment (Shultz et al. 2012). The successful movement of species through this mosaic of natural and anthropogenic habitats and the ability to utilise the resources within these habitats will have an impact of the population dynamics within this novel environment (Scott et al. 2015).

As urbanization rates continue to rise, urban ecosystems and resources are becoming increasingly important for local biodiversity, attracting some wildlife species into human-altered landscapes (Magle et al. 2012). In some instances, urban environments support higher levels of biodiversity than surrounding habitats (Stott et al. 2015). However, there are a variety of threats for wildlife manoeuvring through and living in urban landscapes such as their vulnerability to human persecution, collisions with motor vehicles or attacks by domestic animals (Magle et al. 2012; Parker and Nilon 2012; Murray and St Clair 2015). Other human-wildlife conflict in urban areas includes damage to lawns and landscaping, the spread of diseases between wildlife populations, and anthropogenic reactions (poisoning or killing) to the death of livestock (Magle et al. 2012; Parker and Nilon 2012).

Responses to urbanization vary among species but as mentioned, it generally has a negative impact on species diversity with many unable to survive within urban landscapes (Fischer et al. 2012; Newsome et al. 2015). This has resulted in the exclusion of certain species from the urban environment (Randa and Yunger 2006; Newsome et al. 2015). This is especially true for most carnivores that require large home ranges and are prone to conflict and persecution by humans (Kertson et al. 2011; Bateman and Fleming 2012). However, despite the negative impacts of urbanization on mammalian carnivores, various species are able to survive, reproduce and thrive in human-altered ecosystems (Bateman and Fleming 2012). These species exhibit significant behavioural, demographic and ecological plasticity (Lowry et al. 2013; Newsome et al. 2015). These adaptations include altered diel activity patterns, exploitation of new resources, reduced vigilance, reduced home ranges and increased population densities (Parker and Nilon 2012). Therefore, the persistence of urban carnivores is largely dependent on the permeability and connectivity of the landscape and the ability to adapt to this novel environment (Francis et al. 2015).

The knowledge of habitat preferences of mammals is essential for their effective management and conservation (Kauhala and Salonen 2012). In many instances ecological and behavioural information central to the development and implementation of wildlife management strategies are limited (Jones 2003). This has resulted in a surge of research focusing on urban landscapes. As humans are an important component of the urban ecosystem, it is important to integrate human perceptions into conservation strategies (McIntyre et al. 2000). This can be achieved through the implementation of techniques such as community-based questionnaire surveys, to determine the public perception of wildlife and management programs.

There has been an increasing trend employing questionnaires in ecological studies over the past decade (Grey-Ross et al. 2010). A study conducted by White et al. (2005), indicated that 168 questionnaires were used in ecological journals between 1991 and 2003, 19 of which were conducted in Africa. These surveys and questionnaires enable the quantification of human behaviour towards various ecological and conservation strategies (White et al. 2005; Grey-Ross et al. 2010). Recently, the quantification of public perceptions has played an integral role in the conversion of ecology into conservation and management strategies (McIntyre et al. 2000; White et al. 2005). It is this amalgamation of both scientific information and public perceptions that form the framework for the effective management of ecosystems (Holsman and Peyton 2003).

Questionnaires engage local communities with wildlife conservation and management, allowing for the expression of public opinions (McIntyre et al. 2000; Frank et al. 2015). This expression plays a key role in the ecological education and awareness to minimize wildlife conflict (White et al. 2005). Questionnaires provide valuable information on both present and past impacts of management strategies as well as providing qualitative data in studies focusing on anthropogenic impacts (hunting) and behaviour towards wildlife (Frank et al. 2015). Furthermore, residents that have lived in an area for an extended period can provide information pertaining to the abundance and diversity from the past, as well as possible insight in environmental trends (White et al. 2005). Questionnaires can be conducted in a variety of ways such as postal surveys, telephonic and electronic (e mail or web based sites) or in-person interviews (White et al. 2005).

There have been increasing reports of large-spotted genets (*Genetta tigrina*) within urban landscapes throughout KwaZulu-Natal (KZN), South Africa. Due to their elusive nature, small size and nocturnal habits, large-spotted genets along with other members of the Viverridae family

have not been studied in the same depth as other carnivores (Eisenberg 1981; Do Linh San et al. 2013). They have a broad diet feeding on a wide variety of invertebrates, small rodents, reptiles birds and wild fruits (Skinner and Chimimba 2005; Roberts et al. 2007). They are adapted to a semi-arboreal lifestyle and rest during the daylight hours in trees, hollows or under roots. They have been recorded using anthropogenic structures such as roofs and outbuildings as roosts and as locales to rear young within urban areas in KZN and Gauteng (Rowe Rowe 1992; Skinner and Chimimba 2005).

Large-spotted genets are distributed throughout most of southern Africa from dense shrub cover and forested areas to coastal and riverine habitats (Gaubert et al. 2005; Skinner and Chimimba 2005). However, the distributions of most African Viverridae and Herpestidae are speculated due to the lack of recent scientific studies (Skinner and Chimimba 2005). Despite this extensive range, large-spotted genets are poorly studied in comparison to other African small carnivores. The limited studies conducted on *G. tigrina* in Africa have focused on diet (Roberts et al. 2007) movements and activity patterns in Orno National Park, Ethiopia (Ikeda et al. 1982; Fuller et al. 1990) as vectors for disease (Purnell et al. 1970) and genetics (Gaubert et al. 2004, 2005) with no studies focusing on their ecology in an urban environment.

Hybridization has been recorded within the *Genetta* genus, particularly within South Africa (Gaubert et al. 2005). *Genetta tigrina* and *G. maculata* form part of the large-spotted genet complex (Gaubert et al. 2005). The morphological similarities between *G. tigrina* and *G. maculata* has provided uncertainty regarding the assessment of taxonomic boundaries. This is particularly true within coastal KZN where hybridization and range overlap have been recorded for *G. tigrina* and *G. maculata*. As a consequence of the uncertainties within the large-spotted genet complex within KZN, we generally referred to large-spotted genets as *G. tigrina* in the current study. This

was particularly useful when utilising citizen science as guide books for the area refer to large spotted genets as *G. tigrina*. Furthermore, residents would have experienced difficulties differentiating between *G. tigrina*, *G. maculata* or hybrids within this complex.

As urban areas continue to expand, the ability to recognize and account for human perceptions in the translation of ecology into management, is likely to become increasingly important for the conservation of urban wildlife populations (White et al. 2005; Frank et al. 2015). Land-use intensification is on the rise, resulting in the conversion of native landscapes into homogenous human-altered habitats (Mulwa et al. 2012). As human populations continue to increase the conservation of species and the effect of varying degrees of land-use on wildlife populations is likely to become increasingly important. The aim of the study was to solicit community-based information using a questionnaire mainly, in order to investigate the distribution of large-spotted genets within the urban environment of the greater Durban district, KwaZulu-Natal, South Africa. A further aim was to investigate their use of anthropogenic structures for resting, breeding and foraging. The study was also designed to investigate possible areas of conflict, habitat use and residents' attitudes towards *G. tigrina* to augment future conservation strategies within this urban environment.

Materials and Methods

We conducted the study in the urban suburbs of Kloof and Hillcrest in KwaZulu-Natal, South Africa. According to the 2011 census Kloof and Hillcrest have a population of 29,704 and 13, 329 respectively (Statistics South Africa 2011). The altitude of the study site area was $\geq$ 560 m ASL. Although some of the region's natural vegetation (KwaZulu-Natal Sandstone Sourveld scattered with low shrubs and patches of riverine forests) exists, in the large Durban Metropolitan Open

Space System (D'MOSS) areas on the city outskirts, numerous patches of natural vegetation remain throughout the urban landscape (Mucina and Rutherford 2006). This is particularly evident within the Kranzkloof Nature Reserve located within the western border of Kloof. However, transformation due to plantations, cultivations and urban development still occur (Mucina and Rutherford 2006). The mosaic of native patches of vegetation within the urban landscape provides habitats for a variety of wildlife species. As a result of urban development and the establishment of gardens, many alien invasive trees such as *Cinnamomum camphora* and *Melia azedarach* are found in the Kloof-Hillcrest area.

Questionnaire survey- We placed a popular article into numerous papers, local conservancy newsletters and websites requesting the public to report recent sightings of large-spotted genets, including the location, time and behavioural observations. The article explained the reason for the study and also included a picture of a large-spotted genet to assist with correct identification. Public sightings were collected between April 2012 to July 2013 throughout KwaZulu-Natal, however this study focused on Kloof/Hillcrest and the surrounding areas.

The questionnaire presented 15 mainly open-ended questions to assess various aspects of the ecology of large-spotted genets with urban Kloof/Hillcrest as well as the public perception of large-spotted genets. This allowed participants to elaborate or expand on their answers. Questions elicited information on where large-spotted genets were observed and their noticeable activities as well as possible management strategies for urban genet populations. The use of anthropogenic food resources and shelter by *G. tigrina* were ascertained by the frequency of reports. The questionnaire survey was conducted between April 2012 and February 2013.

We conducted all interviews with the participation and consent of the respondents. Participants were identified by responses to the public petition and also by attending various meetings and conservancy talks. Due to the public attention the project had gathered, a large number of participants were identified and questioning was conducted via e-mail in the majority of the cases. However, interviews were also conducted with respondents at various meetings, presentations and house visits.

Data analysis- All responses to survey questionnaires, interviews and sightings were collated and stored in an electronic data base. Qualitative responses were categorized and ranked for statistical analysis. Descriptive statistics and Chi-square analysis were conducted to compare resident's observations and responses using Statistica V.7 (Tulsa, OK, USA).

Results

A total of 212 public sightings of *G. tigrina* were recorded between March 2012 and June 2013 (Fig.1). Sightings of *G. tigrina* were distributed throughout Kloof/Hillcrest across a variety of land use types (Fig. 1). Furthermore, 64.0 % (n = 136) of the sightings occurred in areas more than 1 km from the Kranzkloof Nature Reserve protected area. Most respondents (52.0 %) reported that they rarely encountered *G. tigrina* on their properties compared with others who saw them more frequently (31% monthly, 11% weekly or 6% daily). Public sightings of large-spotted genets in urban landscapes are not limited to Kloof, with 85 sightings in Durban and 36 sightings in Pietermaritzburg (Fig. 2). Other sightings were reported in Figure 2.

A total of 367 qualitative interviews were conducted with residents of Kloof, KwaZulu-Natal. More males (59.0 %) responded to the interviews than females, but there was no significant difference between the two genders response rate ($\chi^2 = 0.376$, df = 1, p > 0.05).

Behavioural observations of large-spotted genets in the urban environment

Despite genets' primarily nocturnal nature, 11% of respondents reported observing *G. tigrina* in daylight hours. The main large-spotted genet behaviour reported was walking between areas (48%) followed by sitting and feeding (Table 1). The main food items large-spotted genets were recorded eating were domestic pet food (24.5 %), followed by wildlife species (21.3 %), and scavenging in dustbins and black plastic bags (9.4 %). A further five respondents reported *G. tigrina* feeding on flying ants and geckos around both house and street lights. Respondents also identified other forms of behaviour (Table 1). According to respondents, majority (93%) of genet grooming took place above the ground in either trees (63%) or anthropogenic structures (21%). Respondents also reported genets drinking out of dog bowls (n = 6), pools (n = 4), and bird baths (n = 2).

The use of anthropogenic structures by large-spotted genets

A total of 48 respondents (13%) reported having *G. tigrina* living in their roofs with a further 83 respondents (22%) unsure as access to the roof was restricted (Table 2). The majority of respondents (44%) were positive about the genets living in their roof spaces ($\chi^2 = 8.2$, df = 2, p <0.05). In contrast 26% of participants were negative about them living in their roofs and many (77%) mentioned the presence of a midden and the smell that accompanied it to be the main negative factor. A few respondents (n = 4) also reported the destruction of air-conditioning ducts as a negative factor associated with genets living in their roof space.

Respondents cited *G. tigrina* using their roofs for a period of 6 -12 months (38 %) as well as a further 27% indicating that genets lived in their roofs for a period of more than 2 years (Table 2). Furthermore, 35% of respondents who cited genets in their roof confirmed that breeding occurred within the roof spaces. Significantly more respondents reported a genet litter size of two individuals (χ^2= 45.3, df =3, p < 0.05), while reports of a litter size of three and four were only recorded on two and one occasion(s) only.

Feeding of large-spotted genets

The majority (79.6 %) of respondents did not place supplementary food out specifically for genets. However, a total of 33 (14.0 %) of participants placed specific food out for genets and only 6.4 % of respondents did not want to share information on their food provisioning for fear of being reported; all of these respondents lived in housing where pets were generally banned. Fifteen respondents (45.0 %) who placed supplementary food out for genets did so on a nightly basis, while others (21.0 %) provided food on a weekly basis or erratically (34.0 %). When questioned about the type of food fed to genets, significantly more respondents, 51% (χ^2 = 21.24, df = 3, P < 0.05) used chicken (hearts/gizzards and necks) and a further 21.0 % provided leftovers from a meal. Interestingly a number of respondents (24.0 %) provided domestic pet food while only two respondents provided fruit as supplementary food. When questioned about the number of individuals that were fed, the respondents cited 1-5 individuals. The majority (39.0 %) of participants cited feeding only one individual with 12.0 % citing that they fed three individuals. Residents reported varied opinions with regards to predictably feeding the genets. The majority (78.0 %) of residents that provided supplementary food expressed no negative views about predictably feeding the genets. Residents felt "sorry" for the urban individuals as they felt that the

genets' habitat was being taken over by the urban landscape and they provided food for them to ease this environmental pressure. However, residents that did not feed large-spotted genets (73.0 %) expressed concerns that feeding these individuals will result in a dependence on this reliable and predictable food source. A high percentage (19.7 %) expressed neutral opinions about feeding urban genets with only 4.3% that opted not to respond.

Community perceptions and wildlife conflict

The majority of respondents expressed positive attitudes towards genets in urban Kloof (60.5%) and many felt "privileged" to be in such close proximity to these nocturnal animals (χ^2 = 18.2, df = 2, p < 0.05). However, a portion of respondents expressed concerns about the increased sightings of the genets (22%) within the urban landscape. The most cited cause for concern of genets was the impact they have on wild bird populations and the domestic poultry attacks (59%). The latter was significantly higher than other conflicts (χ^2 = 21.8, df = 3, p < 0.05) reported by respondents (Table 3). Eleven sightings of large-spotted genets attacking various bird species and raiding nests were reported. These species include hadeda ibis (*Bostrychia hagedash*, n = 6), laughing dove (*Spilopelia senegalensis*, n = 3) and southern masked weavers (*Ploceus velatus*, n = 3). Respondents who owned poultry (n = 8) were negative as many birds had been killed by the genets. In one instance a single individual was reported to have killed 16 bantam chickens in a single night. Several participants who owned poultry admitted to privately trapping the genets (n = 7) and translocating them to various nature conservancies while a few admitted to shooting them with pellet guns. Numerous participants mentioned growing concerns that genets may carry diseases (29.1 %), particularly rabies (RABV) (Table 3).

A total of 51 (13.9%) respondents reported large-spotted genet fatalities between March 2012 and August 2013. The most cited cause for genet deaths within the urban landscape was attacks by domestic dogs, reported by 25 respondents (49.0 %) and this was significantly higher than other causes ($\chi^2 = 13.81$, df = 3, p < 0.05). Collisions with motor vehicles were cited by 17 respondents (33.0 %) and collisions occurred on both municipal and national roads throughout Kloof area. Poisoning and shooting of genets with pellet guns were cited and confirmed by only 4 participants (7.8%).

Management

Significantly more participants (64%) indicated that further research on large-spotted genets in urban areas is required than those who disagreed (19%) ($\chi^2 = 9.61$, df = 1, p < 0.05). Educational public talks and the erection of wildlife road signs were the most commonly reported management plans.

Discussion

The feedback from residents in Kloof, KZN provides a preliminary perspective of perceptions of large-spotted genets in an urban environment, their sighting frequency and their use of anthropogenic resources (food and structures). The frequency (212) and wide distribution of genet sightings throughout Kloof/Hillcrest suburbs, indicated the success of this species to colonize urban areas. Over the past decade several carnivores such as red fox (*Vulpes vulpes*), raccoon (*Procyon lotor*), coyote (*Canis latrans*) and European badgers (*Meles meles*) have reached high densities in urban areas (Prange et al. 2003; Gehrt et al. 2010; Bateman and Fleming 2012). Interestingly, the presence of large-spotted genets within an urban environment is not confined to

Kloof, KZN, with five other localities within KZN confirming their presence within urban areas. On a larger scale, red foxes (*Vulpes vulpes*) have colonized over 114 cities in no less than 56 countries (Bateman and Fleming 2012). Furthermore, genet populations in the greater Durban Metropolitan area are displaying similar characteristics to the Kloof, KZN, genet population including the use of anthropogenic structures for roosting during daylight hours and the utilization of anthropogenic food resources. This indicates a high degree of adaptation to the urban environment. This is further supported by the high sighting frequency of large-spotted genets across the urban landscape, especially in areas that have little connection or access to green areas.

Observations of large-spotted genet population

According to Skinner and Chimimba (2005), large-spotted genets are strictly nocturnal with movement during the day being irregular behaviour. However, 11% of respondents reported observing large-spotted genets during the day within the urban landscape of Kloof, KZN. This change in diel activity within the urban environment has also been recorded in coyotes (Randa and Yunger 2006), bobcats (*Lynx rufus*) (Lowry et al. 2013) and black bears (*Ursus americanus*) (Beckman & Berger 2003). However, both bobcat and coyote shifted their activity to late in the night when both human and vehicle traffic is significantly lower (Gehrt et al. 2010). A lack of large carnivores and reduced predation pressure coupled with high quality food resources available in urban areas, may be factors facilitating this diurnal activity in large-spotted genets.

Observations of *G. tigrina* scavenging through dustbins and black plastic bags could indicate the opportunistic nature of these urban genets. However, these finding are not isolated to KZN, with large-spotted genets also recorded scavenging around campsites in Dwesa Nature Reserve, Eastern Cape (Roberts et al. 2007). Genets may benefit from the abundance of high-

energy content food found in anthropogenic refuse, and its predictability as a food source may subsequently reduce foraging costs. Bin raiding has been recorded in striped skunks (*Mephitis mephitis*) and raccoons (Hoffman and Gottschang 1977). Carnivores that readily exploit anthropogenic waste include side-striped jackal (*Canis adustus*), black-backed jackal (*Canis mesomelas*) (Loveridge and Macdonald 2003) and coyotes (Gehrt 2007). According to Saunders et al. (1993) up to 60% of an urban red fox's diet is comprised of scavenged food items. However, the consumption of scavenged anthropogenic waste can result in the ingestion of high levels of indigestible pollutants (plastic, elastic and paper). In a study conducted by Newsome et al. (2010), anthropogenic waste was recorded within 12.5% of the scats urban kit foxes. Food packaging material has been recorded in the scats of urban coyotes (Morey et al. 2007), raccoons and kit foxes (*Vulpes macrotis*) (Newsome et al. 2015).

Not only were genets readily exploiting anthropogenic food by scavenging in bins but were also using the high-quality and high-energy content diet of domestic pet food. Similar results have been recorded in the urban red fox populations of Zurich, where more than 50% of stomachs examined contained anthropogenic food items (Contesse et al. 2004). As domestic pet food is a predictable food source, large-spotted genets living in the urban landscape may target this high energy food resource and subsequently reduce the amount of energy expended foraging, locating and capturing prey items. This indicates high levels of behavioural adaptation to a novel and predictable food source. The presence of high-energy content food resources may be a factor driving the increasing urban population and alteration in diel activity patterns of large-spotted genets.

Large-spotted genet fatalities

Perceived threats to large-spotted genets varied among respondents, with the majority citing attacks by domestic dogs as the primary threat. The majority (60%) of genet fatalities caused by domestic pets were juvenile genets. Youngsters accompany adults on foraging bouts after being weaned at ± 10 weeks (Skinner and Chimimba 2005) and this makes them vulnerable to attacks, as they are not as agile as adults. Genets also have a musty odour (Skinner and Chimimba 2005) and dogs appear to be very sensitive to this odour. This often identifies the individuals' position (C. Widdows Pers. obs.).

Collisions with motor vehicles were also reported by respondents and the danger of this is reinforced by the fact that 44% of public sightings of large-spotted genets occurred on roads. In most areas within the urban landscape of Kloof, a network of roads separate green areas and green belts. In order to get to these areas, individuals would need to cross roads and subsequently run the risk of collisions with vehicles. Collisions with motor vehicles account for a large proportion of urban carnivore deaths worldwide. In a study conducted by Harris et al. (1992), ± 50 000 badgers (*Meles meles*) are killed by motor vehicles each year in Britain. Further studies indicate larger carnivore species such as black bears and coyotes also succumb to death on public roads (Beckman and Lackey 2008). Over 80 coyotes were killed on roads between 2013 and 2014 in the city of Alberta, Canada (Murray and St Clair 2015).

The use of anthropogenic structures by large-spotted genets

The high percent of respondents who identified genets breeding in their roofs indicates the importance of roof spaces to genets. The use of anthropogenic structures by *G. tigrina* may provide safer daylight resting places for both adults and juveniles as they are unlikely to be detected by

predators such as crowned eagles (*Stephanoaetus coronatus*). The roof space may also provide an area for juveniles to move around with a reduced detection probability by predators, thus allowing adults to forage for prolonged periods. In a study conducted by Kaneko et al. (2006), Japanese badgers (*Meles anakuma*) readily exploited abandoned buildings as shelters. Raccoons and brushtail possums (*Trichosurus vulpecula*) also make use of buildings and roof spaces as alternative resting and breeding sites (Prange et al. 2004; Russell et al. 2011). The positive attitude of the majority (44%) of respondents to genets living in their roof spaces could have a positive effect on the growth and expansion of the urban genet population. The portion of negative views mainly pertained to middens present in the roof space. In some instances, both defecation and urination in one spot caused seepage into residents' homes. Tolerance to this was very low and entrances that genets used to get into the roof were subsequently blocked to prevent them entering the roof space.

Feeding of large-spotted genets

Despite most respondents not placing supplementary food for genets, 14% of respondents did so on a regular basis and it is possible that this figure will rise in the future. According to Roper (2010), 29% of residents surveyed in Brighton (United Kingdom) provided supplementary food for a range of mammals such as foxes and badgers. Furthermore, over half of the residents ensured that food was set out on a nightly basis (Roper 2010). Similar results were recorded in this study, where 45% of respondents provided supplementary food for urban genets on a nightly basis. A study by Roberts et al. (2007) indicated that the main prey items of large-spotted genets in the Dwesa Nature Reserve (Eastern Cape, South Africa) were invertebrates. However, the main food item fed to genets was meat, particularly chicken (hearts, gizzards and necks) and this provides a

larger amount of protein per night than what might be obtained naturally. The energy content provided by meat would be much higher than that provided by invertebrates in rural environments. Furthermore, six respondents reported feeding between 2-3 individuals at the same time and this is irregular behaviour as *G. tigrina* are normally solitary (Skinner and Chimimba 2005). The high-energy content food provided by residents coupled with the reduced energetic costs could outweigh the costs associated with encountering other individuals. Therefore, the provision of supplementary food items by a portion of residents may have a significant effect on the foraging behaviour of urban genets. Newsome et al. (2015) proposed that the provision of anthropogenic food resources can have an influence on the sociality of urban carnivores. In a study conducted by Herrero (1983) solitary black bears were observed communally foraging at a garbage dump in Jasper National Park, Canada.

Community perceptions of large-spotted genets

Various urban green spaces such as the Kranzkloof Nature Reserve, Kloof Gorge, and Everton conservancy in Kloof, KZN, place most residents relatively near natural environments and animals. It was therefore not surprising that the majority (61%) of respondents were positive about the presence of genets within the urban landscape. This perception may have a positive influence on the future growth and expansion of this urban population. Similar positive attitudes towards the presence of urban wildlife has been recorded with respect to coyotes (Spacapan 2013), brushtail possum (*Trichosurus Vulpecula*) (Russell et al. 2011), European badger (Harris and Skinner 2002) and bobcat (Harrison 1988). Red fox populations have expanded into urban areas in Britain, North America, Poland and Switzerland (Randa and Yunger 2006). A similar trend has been seen in raccoons where urban populations have higher densities than those in rural areas (Contesse et al.

2004; Prange et al. 2004). These high densities have been achieved through the reduction in the size of home ranges within urban landscapes for both *V. vulpes* and *P. lotor* (Prange et al. 2004). Both species have a generalist diet and exploit the high concentrations of anthropogenic resources found in urban areas (Bateman and Fleming 2012). Similarly, coyotes are also common in urban areas across the United States of America and exhibit urban adaptations such as reduced home range size in comparison with rural conspecifics and a shift in diel activity patterns. Carnivores successfully living in urban areas are not limited to the few mentioned but also include striped skunk, kit fox, grey fox (*Urocyon cinereoargenteus*), black bear (*Ursus americanus*) and bobcat (*Lynx rufus*) (Tigas et al. 2002; Gehrt et al. 2010).

The 22% of residents who held negative views on large-spotted genets considered them to be harmful to wild bird populations. However, most reported attacks on bird species recorded were on hadeda ibis (*Bostrychia hagedash*) and laughing dove (*Spilopelia senegalensis*), both species associated with human altered habitats. The bulk of respondents who held negative views were poultry owners who had experienced previous losses due to genets. Their tolerance was very low, and several respondents (n = 3) admitted to shooting large-spotted genets.

The increase in frequency and public awareness of rabies in KZN (Malerczyk *et al.* 2010) could explain the levels of concern expressed by some respondents regarding the expansion of genets into urban areas. According to Coetzee and Nel (2007), the majority of animal and human rabies cases recorded in South Africa have occurred in KZN. Rabies has been recorded in the Viverridae and Herpestidea, and most noticeably in the yellow mongoose (*Cynictis penicilata*) (Nel et al. 1997). Furthermore, rabies has been recorded in small-spotted genets (*Genetta genetta*) in South Africa, with over 192 rabies cases diagnosed between 1928 and 2000 (Bishop et al. 2003). Further studies have recorded rabies in *G. genetta* in the Serengeti and Ngorongoro districts of

Tanzania and human infection through bites have occurred (Lembo et al. 2008). Cases of rabies within South Africa are low and this could be a result of the shy nature of *G. tigrina* and *G. genetta*. However, the increasing population of large-spotted genets in urban Kloof, KZN, may result in an increase in contact with residents or domestic animals and the greater potential for disease transfer.

Management

The majority of participants (64%) indicated that further research on the urban population of *G. tigrina* is required and were interested in gaining more knowledge of genet biology. Respondents felt that education programs such as public talks could help dispel many uncertainties about large-spotted genets. Furthermore, individuals expressed that erection of road signs near green belts requesting a reduction in speed may increase drivers' awareness and subsequently reduce genet fatalities.

Large-spotted genets within the suburbs of Kloof/Hillcrest are successfully living in an urban landscape and this could be attributed to their adaptable and generalist behaviour. However, this study had identified threats facing large-spotted genets living within urban environments. Although current threats are numerous, all are directly related to humans and the anthropogenic conversion of natural habitats. Collaboration between various stakeholders as well as the education of residents is critical to ensure the protection of urban genet populations throughout KwaZulu-Natal.

Future research should estimate the population size and density of genets within the suburbs of Kloof/Hillcrest. This will allow comparisons to be made with wild populations to determine the effect of human-altered landscapes on genet population sizes.

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Figure Legends

Fig. 1. Locations of the 212 public sightings of large-spotted genets (*Genetta tigrina*) within Kloof (star) and Hillcrest suburban areas, between March 2012 and June 2013. The N3 highway is the main access road that runs parallel to both Kloof and Hillcrest. The star indicates the centre of Kloof.

Fig. 2. Areas in KwaZulu-Natal where large-spotted genets (*Genetta tigrina*) have been sighted by the general public, between April 2012 and February 2013.
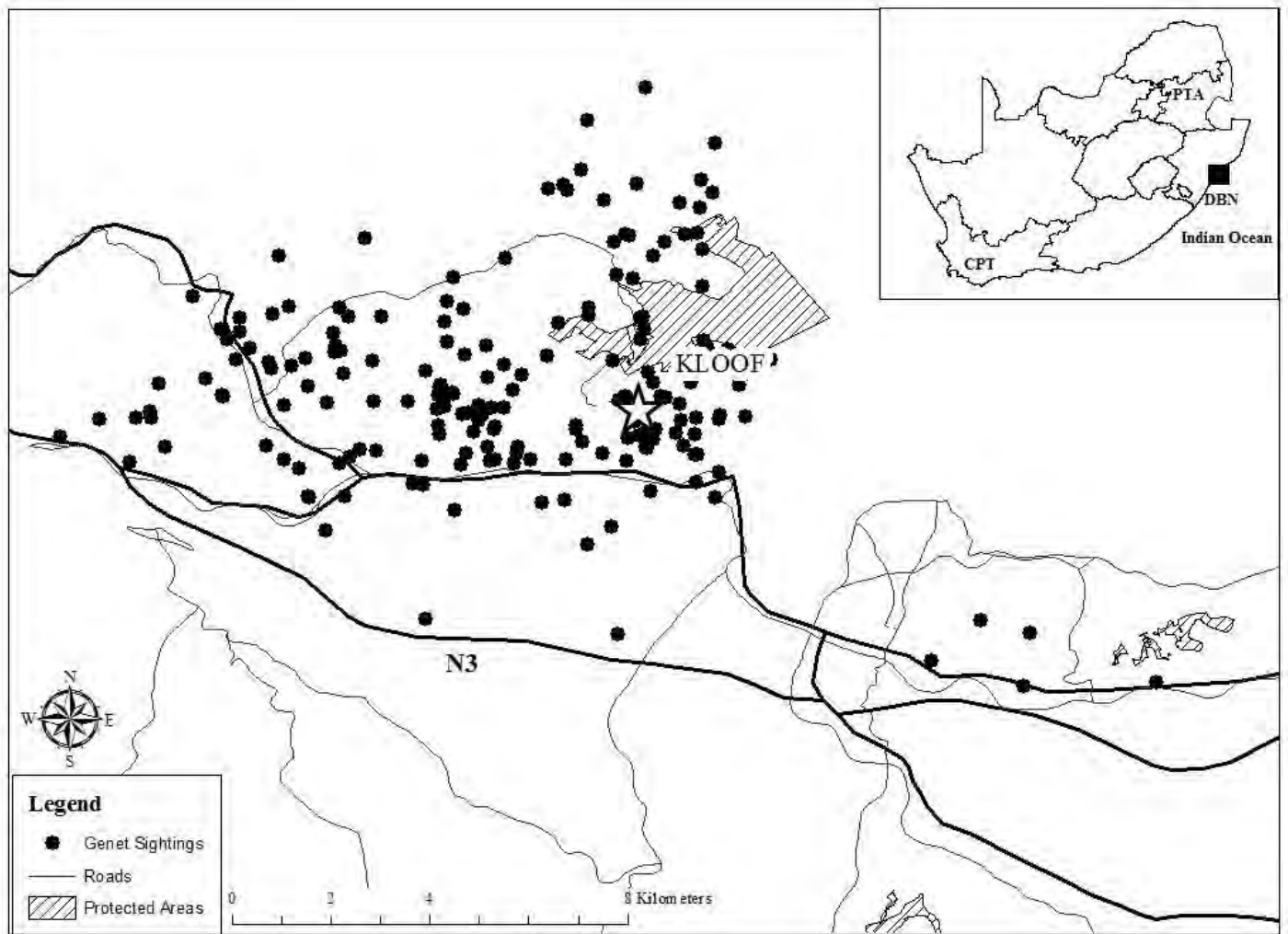
Figures

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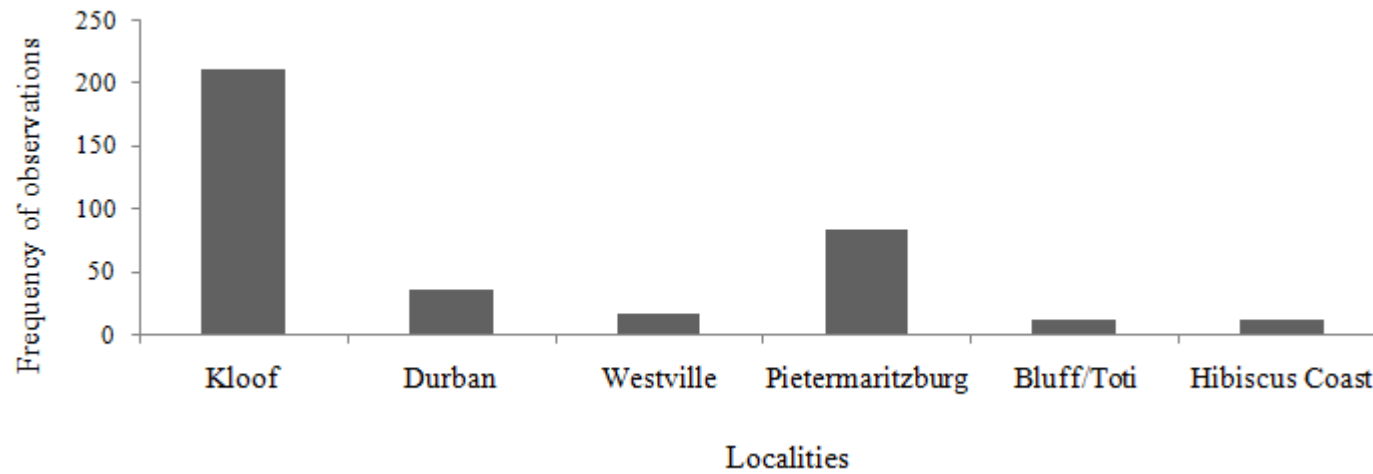
Kloof.

Fig. 2. Areas in KwaZulu-Natal where large-spotted genets (*Genetta tigrina*) have been sighted by the general public, between April 2012 and February 2013.

Tables

Table 1. Frequencies (%) and number of recorded activities of large-spotted genets by respondents in Kloof, KwaZulu-Natal. Results compiled from sightings reported between April 2012 and July 2013.

Observed activities	N	Frequency (%)
Moving	102	48.1
Sitting	41	19.3
Grooming	15	7.1
Eating	27	12.7
Hunting	18	8.5
Fighting	9	4.3

Table 2. Frequencies (%) and number of recorded activities of large-spotted genets using anthropogenic structures in Kloof, KwaZulu-Natal.

Question	Possible answers	N	Frequency (%)
Do you have genets living in your roof?	Yes	48	13.08
	No	236	64.31
	Unsure	83	22.61
How long have genets been using your roof?	1-6 month	6	12.50
	6-12 month	18	37.50
	1-2 years	3	6.25
	More than 2 years	13	27.08
	Unsure	8	16.67
Have you recorded breeding in your roof?	Yes	17	35.42
	No	19	39.58
	Unsure	12	25.00

Table 3. Responses from participants in Kloof, regarding a question from the survey which asked what issues residents perceived would accompany increasing large-spotted genet populations.

Large-spotted genet issues	Frequency (%)
Disease transmission	29.1
Influence on domestic and wild bird population	41.8
Midden	18.5
Attack on human or domestic pets	10.7

CHAPTER 3

Factors affecting the distribution of large spotted genets (*Genetta tigrina*) in an urban environment in South Africa

Craig D. Widdows, Tharmalingam Ramesh, Colleen T. Downs

School of Life Sciences, University of KwaZulu-Natal, Private Bag X01, Scottsville, Pietermaritzburg, 3209, KwaZulu-Natal, South Africa.

Correspondence Email: Downs@ukzn.ac.za

Abstract

Aspects influencing the distribution patterns of mammals are particularly important for species living in human altered landscapes. The current study investigated the relationship between occurrence patterns of large spotted genets (*Genetta tigrina*) with various environmental variables believed to influence their detection and site occupancy in an urban environment. Presence/absence data was collected from 28 camera trap stations between June 2012 and October 2013 in Kloof/Hillcrest suburbs, Durban, South Africa. Average estimated occupancy of *G. tigrina* was 0.62 ± 0.14 with a detection probability 0.19 ± 0.03. The naïve occupancy was 0.607. Model selection indicated that bush cover and placement of camera traps on wide paths negatively influenced *G. tigrina* occupancy. Both winter and fringe habitats influenced the site occupancy of *G. tigrina* positively. Furthermore, bush cover was negatively associated with detection probability of *G. tigrina*. The current camera surveys provided baseline data for long-term species

observations within suburban Kloof/Hillcrest. The present study indicated the importance of a variety of factors on the distribution of *G. tigrina*, particularly in landscapes where natural habitats are threatened with changing land use and increasing human populations.

Keywords Camera trap; Detection probability; Genet; Habitat use; Occupancy; Small carnivore; Urban

Introduction

Over the past few decades, there has been an increase in the human population living in urban areas and subsequently an increase in the amount of land urban areas occupy (Riem et al. 2012). Urbanization results in the anthropogenic conversion and altering of natural habitats into commercial, industrial and residential uses and has been linked to a worldwide decline in native biodiversity (Adams 2005; Randa and Yunger 2006; Riem et al. 2012). Native vegetation is replaced with anthropogenic structures (buildings, pavement) and often non-native vegetation (Riem et al. 2012). Furthermore, the reduction in available habitats for wildlife subsequently reduces species richness, abundance, distribution, and genetic diversity (Bettigole et al. 2013). As urbanization continues to alter and degrade natural habitats, animals are forced into close proximity to human populations (Bateman and Fleming 2012; Lowry et al. 2013). The urban landscape is characterized by reduced landscape heterogeneity which influences both ecosystem functioning and structuring and the introduction of environmental stressors such as pollutants and elevated human activity (Randa and Yunger 2006; Jung and Kalko 2010). This often results in many species withdrawing into reduced ranges in response to spread of urban environments (Bateman and Fleming 2012).

Urban landscapes often reflect a gradient of land use varying from the highly urbanized city centers through to the suburban areas and less altered rural landscapes (Riem et al. 2012). The ability of these patches to sustain wildlife populations varies spatially between open residential areas and highly developed central business districts or industrial areas (Baker and Harris 2007). The degree of habitat modification and land use generally declines with increasing distance from the highly urbanized city center (Randa and Yunger 2006). This creates an urban-rural gradient that can be assessed to determine the effect that increasing urbanization is having on biotic communities. Urban-gradient studies generally reflect a reduction of native species and an increase of non-native species towards the center of urban areas (McKinney 2002). Native habitats and species are often eliminated from the city centers and these areas are associated with high rates of extinction (McKinney 2002; Randa and Yunger 2006). Furthermore, species richness decreases across the urban-rural gradient, with the lowest species richness recorded in urban centers (McKinney 2002).

Increasing distance from the urban center often results in a decline in habitat modification. This gives rise to a mosaic of natural and semi-natural habitats such as residential gardens, golf courses, parks, green belts and conservancies (patches of semi-natural habitats) with a lower land use (Baker and Harris 2007; Maruani and Amit-Cohen 2007). The presence of these natural green areas provide numerous habitat possibilities for wildlife, are essential for sustaining wildlife populations and can increase urban biodiversity (Savard et al. 2000; Shultz et al. 2012; Bateman and Fleming 2012). Connections of relatively undisturbed habitat patches within urban areas act as refugia for a variety of species and can also alleviate the impact of urbanization on wildlife populations (Baker and Harris 2007; Ordenana et al. 2010; Fleming and Bateman 2012). Urban green spaces can provide long-term habitat stability in a rapidly changing urban environment and

often support high species diversity (McKinney 2002; Shultz et al. 2012). However, in most instances patches of remaining natural habitats are highly isolated with no connection to other patches thus creating abrupt boundaries that generally hinder species diversity (McKinney 2002; Randa and Yunger 2006).

A good knowledge of species abundance, distribution and occurrence is imperative for effective planning and implementation of conservation strategies (Tobler et al. 2008; Mugerwa et al. 2012). Furthermore, aspects that influence species distribution patterns such as competition, predation, topography and land-use changes, provide important knowledge for future conservation and ecological studies (Macdonald and Rushton 2003; Mugerwa et al. 2012; Msuha et al. 2012). Recent monitoring techniques have implemented camera trapping to determine animal presence, abundance and habitat use (Stein et al. 2008; Mugerwa et al. 2012). Camera trapping is particularly useful in habitats that present difficult field conditions for other assessment methods, and provide important information that would otherwise be difficult to obtain using traditional field techniques (Cutler and Swan 1999; Abi-Said and Amr 2012; Anile et al. 2012; McCallum 2012). Camera trapping data have recently been used to estimate the landscape capacity to support wildlife through occupancy modeling (MacKenzie et al. 2006; Long et al. 2010; Ramesh et al. 2012; Bettigole et al. 2013; Ramesh et al. 2013). Occupancy modeling is a monitoring technique based on the detection/non detection of a target species at a particular site and allows researchers to estimate the probability of occurrence (ψ) at any number of locations, based on the habitat characteristics of each site (MacKenzie et al. 2006; Ramesh et al. 2012; Bettigole et al. 2013). These detection/non detection surveys allow for the assessment and distribution of secretive species by detecting changes in their occupancy (Ramesh et al. 2012).

The large-spotted genet (*Genetta tigrina*) is a nocturnal generalist carnivore adapted to a

semi arboreal lifestyle (Roberts et al. 2007). It is distributed throughout most of southern Africa in a variety of habitats including savannah, woodlands, forest and is adaptable to human settlements (Rowe Rowe 1992; Skinner and Chimimba 2005; Roberts et al. 2007; Ramesh and Downs 2014; Widdows and Downs 2015). However, the distributions of most members of African Viverridae are speculated due to the lack of recent scientific surveys (Skinner and Chimimba 2005). Despite their extensive range they are poorly studied in comparison to other small African carnivores (yellow mongoose and meerkat (*Suricata suricatta*)). This is mainly due to their nocturnal and elusive habits coupled with their small size. The limited studies conducted on *G. tigrina* in Africa have focused on diet (Roberts et al. 2007; Widdows and Downs 2015), habitat use (Ramesh and Downs 2014), movements and activity patterns in Orno National Park, Ethiopia (Ikeda et al. 1982; Fuller et al. 1990), as vectors for disease (Purnell et al. 1970) and genetics (Gaubert et al. 2005; Gaubert et al. 2009) with no studies focusing on their distribution in an urban environment.

Research focusing on mammals in urban landscapes in South Africa is very limited particularly in the province of KwaZulu-Natal. Wildlife communities and humans in Kloof/Hillcrest suburbs, Durban, South Africa (Fig. 1) live in close proximity due to the large conservancy situated within this urban landscape. Habitat requirements and landscape features influencing *G. tigrina* have not been studied in an occupancy framework. In the current study, we used camera trap data to estimate the site occupancy and detection probabilities of *G. tigrina* within an urban landscape of Kloof/Hillcrest. This analysis has enhanced the understanding of the habitat requirements of urban genets and provided basic ecological data for population management and conservation.

Methods

Study area

The study was conducted in the urban suburbs of Kloof/ Hillcrest (S 29.785982, E 30.829841) in Durban, KwaZulu-Natal, South Africa. The mean annual minimum and maximum temperatures are 13.9ºC and 24ºC respectively and the mean annual rainfall is 974 mm (http://en.climate-data.org/location/27097/). Most rainfall occurs in the summer. Frost is infrequent whereas mist is common and provides additional moisture (Mucina and Rutherford 2006). The altitude of the study site is ≥ 560 m ASL. The vegetation in the study area is dominated by KwaZulu-Natal Sandstone Sourveld scattered with low shrubs and patches of riverine forests (Mucina and Rutherford 2006). Six-eight percent of this grassland biome has been transformed due to plantations, cultivations and urban development (Mucina and Rutherford 2006). This landscape has numerous urban nature reserves such as the Kranzkloof Nature Reserve, Glenholme, Ipithi and Springside Nature Reserves.

Camera traps

We used five digital camera traps (LtI Acorn ® 6210MC, China) triggered by passive infrared sensors, set up at 28 sites during June 2012 – October 2013. Each camera was mounted to a fixed point such as a tree, cross beam, fence or a pole placed in the ground. We set camera height between 30 cm and 40 cm above the ground in order to increase the probability of detecting all mammalian species that are encountered. Cameras were placed in relatively concealed areas to avoid theft or vandalism. In instances where camera sensors were triggered by unwarranted motion

by vegetation or foliage, the cameras were subsequently elevated up to one meter and angled down (Stein et al. 2008). Furthermore, vegetation was cleared around the camera sites to minimize false triggers and avoid blank images and to facilitate the capturing of unobstructed images. The cameras were operational continuously over the 24 h cycle recording the date and time of each photograph. The cameras were programmed to delay one minute between successive photographs and the motion sensor was programmed to "medium sensitivity. This time interval prevented numerous images and videos during one triggering episode. Videos allow for accurate identification of smaller mammals. All camera trap sites were marked and georeferenced using a Global Positioning System (GPS, Garmin Nuvi) (Fig. 1). Each site was visited once a week to check the suitability of the selected site and to verify that the cameras were working normally. In areas with a high site activity, some cameras were required to be checked more frequently. Over the study period each camera remained at a site for 21 nights (SD = 1.37). Photos from the camera traps were downloaded onto a laptop (Lenovo G530, China) in the field on a weekly basis. These weekly visits also ensured the continuous operation of the cameras, routine maintenance and to replace the batteries in the cameras if required.

Habitat measurements

We determined habitat use of *G. tigrina* by measuring habitat features around each camera location. The variables were selected to test various landscape characteristics concerned with the species habitat use. Percent canopy cover, leaf litter and bush cover were evaluated in a 30-m radius plot from the center of each camera location. Canopy cover was defined as percent of ground covered by the horizontal projection of tree crowns within the plot (Ramesh et al. 2013). Leaf litter

was visually assessed in 1×1 m subplots placed in four cardinal directions and mean of these four values represented the entire plot. The three different habitat types were scaled according to the increasing human disturbance, namely residential (situated within the urban landscape with no connection to green areas), fringe (situated on the border of residential properties and green belts) and greenbelt habitats (situated within green belts). Surface water bodies, both natural (rivers and streams) and artificial (pools and water features), and tarred roads were plotted and the Euclidean distance function in ArcGIS 9.3 was used to calculate the distance (meters) from each camera location to the nearest water source and tarred roads. The presence (1) and absence (0) of domestic dogs (*Canis domesticus*) and cats (*Felis catus*) at a camera location and the placement of camera-trap on a narrow trail or large trail were also recorded. Certain locations comprised on degraded grasslands and this may influence the occurrence of large spotted genets. Therefore, we recorded degraded grassland habitats at each camera site as presence (1) and absence (0) scale. As the study covered all seasons, we added all seasons under presence (1) and absence scale (0) in our analysis. A relative abundance index (RAI) of domestic pets (dogs and cats) was estimated from each camera site.

Data analyses

Genetta tigrina occupancy was estimated using a likelihood based method (MacKenzie et al. 2002). A detection history for large spotted genets was created for each camera location, consisting of binary values with '1' indicating species detection during the sampling occasion and '0' indicating non detection (Otis et al. 1978). The event of photographing an individual from the camera traps operating at a single camera station was considered to be an independent

record/detection. On some occasions, individuals were captured repeatedly at a camera station during a time period (≤ 1 min). To avoid pseudo-replications, the first capture of the individual within one minute time frame was considered as an independent record/detection (O'Brien et al. 2003). Photographs detecting multiple individuals were counted as a single detection. It was assumed that each site was independent and no individual would move between sites during the survey period. We considered each camera location as an independent site and each sampling occasion was treated as a temporal repeat of the survey. The program PRESENCE 5.5 (Hines 2006) was used to model site occupancy and detection probability of *G. tigrina* with its covariates. We also included covariates measured at camera points: log transformed RAI for domestic dogs and cats. We hypothesized that site occupancy and detection were related to the given covariates.

We used continuous site covariates: canopy cover, leaf litter, bush cover, land-use type, distance to the nearest water source (both artificial and natural), and roads as well as four categorical variables presence/absence of domestic dogs, cats, narrow trail, and large trail. Categorical variables were recorded as '1', where the particular variable was present and '0' when absent. Correlations among independent variables were tested to avoid problems with multicollinearity (Graham 2003). All continuous covariates were log transformed prior to the analysis. A global model was generated that contained all potential covariates for occupancy and allowed detection probability (p) to vary by all covariates. P was initially modeled and then followed by ψ. The potential covariates for occupancy were allowed to vary, individually or in combination, whereas detection was either maintained in the global model or remained constant, i.e.,ψ(covariate)p(covariate), or ψ(covariate)p(.). The simplest model was also considered, where both occupancy and detection probability remained constant, ψ(.)p(.). This information was then used to rank and compare candidate models (MacKenzie et al. 2002).

Models with ΔAICc≤2 were considered to have strong support (Burnham and Anderson 2002). Model selection, calculation of model weights, and averaging of parameters were conducted according to Burnham and Anderson (2002). We examined covariates that best explained overall *G. tigrina* occupancy (ψ). We ran 10,000 bootstrap samples ($ĉ>1.0$ indicated that there was more variation in the observed data than expected) and Pearson goodness-of fit tests (White and Burnham 1999) to assess the fit of the models to the data. Due to the ratio of sample sizes (n) to the maximum number of estimated parameters (k) being less than 40, models were ranked according to AICc (AIC adjusted for small sample size). The model-averaged parameters were calculated using Akaike weights (ωi) for both proportion of sites used and detection probabilities. In order to determine the relative influence of each covariate on occurrence, model weights were summed over all models containing the particular covariate.

Results

A total of 118 independent photographs captured *G. tigrina* across all camera sites from 582 camera trap nights. Overall *G. tigrina* was detected at 17 camera trap locations producing a naïve occupancy of 0.607. Based on a model with all parameters held constant, the site occupancy and detection probability of *G. tigrina* was 0.61 ± 0.09 and 0.20 ± 0.02 respectively. There was no major difference between the naïve occupancy and the estimated occupancy of *G. tigrina* (Table 1). In addition, a range of other mammals were identified using the camera traps (Appendix 1).

Four of the variables measured were significantly associated with *G. tigrina* occurrence ($\Delta AIC_c \leq 2$, Table 2). The goodness-of-fit test for the global model showed no lack of fit ($ĉ = 0.6$). The model that produced the greatest support for *G. tigrina* occurrence ($\Delta AIC_c = 0$) was

ψ(BC+WP+WN+F),p(BC)(Table 1, highest w_i = 0.115). This suggests that fringe habitats (β = 1.71 ± 1.26) and the winter season (β = 1.98 ± 1.41) positively influenced occupancy, and variation in bush cover (β = -2.32 ± 1.40, R^2 = 0.47) and wide paths (β = -2.24 ± 1.51) negatively influenced occupancy. In this model, the detection probability of *G. tigrina* was 0.19 ± 0.029 (Table 1).

Of the 10 top ranked models (ΔAIC$_c$ ≤ 2) occupancy of three models was positively influenced by camera locations at fringe sites and one model was influenced by distance to roads positively. In all top 10 models (ΔAIC$_c$ ≤ 2) the detection probability of *G. tigrina* was negatively influenced by increasing bush cover (β = -0.41 ± 0.25, Table 2).

The summed model weights for each factor with respect to occupancy of *G. tigrina* were bush cover (wi = 0.87), wide path (wi = 0.45), winter (wi = 0.47), fringe habitats (wi = 0.42), distance to road (wi = 0.18) and distance to natural water source (wi = 0.07). Across all models, the variables that best predicted detection probability of *G. tigrina* was bush cover (wi = 0.82), whereas there was little support for the distance to road (wi = 0.056) (Table 1). Therefore, the average estimated site occupancy (0.62 ± 0.14) and detection probability (0.19 ± 0.03) of *G. tigrina* were selected as the final estimates. This corresponded to a difference of 1.3 % from the naïve site occupancy. The detection probability (0.19 ± 0.03) of *G. tigrina* was low.

Discussion

This study indicated the importance of a variety of landscape factors that influence the distribution and detection probability of *G. tigrina*, despite having a broad distribution in southern Africa. The importance of these factors is of particular relevance for the implementation of conservation

strategies within altered human landscapes.

Areas of high bush cover are thought to provide adequate refuge and cover and subsequently reduce predation risk. However, our modeling indicated that the occurrence of *G. tigrina* was negatively associated with increasing bush cover. This result is contrary to previous studies that have indicated the preference of *G. tigrina* to well-covered, woodland habitats (Rowe-Rowe 1992; Skinner and Chimimba 2005). Previous research focusing on *G. tigrina* has occurred in protected areas, which are occupied by other large carnivores. Large carnivores can exert interspecific and predation pressure on the occurrence of small carnivores such as *G. tigrina* (Virgos and Casanovas 1997; Caro and Stoner 2003). However, human persecution and habitat loss associated with urban areas has resulted in the exclusion of many large carnivores from the suburbs of Kloof/Hillcrest. The reduced predation pressure could allow *G. tigrina* to move through a variety of habitats and not be confined to areas with high bush cover. Furthermore, *G. tigrina* is a visual predator and rely on their keen eye sight (Skinner and Chimimba 2005). Areas with high bush cover could inhibit the ability of *G. tigrina* to locate and capture prey. With the reduction of bush cover in urban landscapes due to habitat loss and modification, the reduced reliance on this habitat type could be a contributing factor to the success of *G. tigrina* within urban Kloof/ Hillcrest. Our results suggest that areas with high bush cover hinder the site occupancy and distribution of *G. tigrina* within an urban landscape. Furthermore, there has been increasing reports of *G. tigrina* using anthropogenic structures such as buildings as daytime resting places further reducing the reliance on areas with high bush cover.

Wide paths had a negative effect on the site occupancy of *G. tigrina*. The avoidance of wide paths could be a strategy to avoid humans and domestic dogs that are more likely to use wider paths as opposed to narrow paths. In a study conducted in a California urban nature reserve by

George and Crooks (2006), the detection and frequency probabilities of bobcats (*Lynx rufus*) declined on wider paths that had high human and domestic dog activity. This indicates spatial displacement due to human and domestic animal activity.

In combination with bush cover and wide paths, *G. tigrina* occupancy was positively associated with fringe habitats. This suggests that the spatial design of this habitat may be structuring the distribution patterns of *G. tigrina* in urban landscapes. These habitats have not experienced the high modification associated with the urban centers. The positive association is not surprising as fringe habitats provide numerous habitat possibilities, refugia, reduced human and domestic animal activity and provide alleviation from the effects of urbanization (Savard et al. 2000; Bateman and Fleming 2012; Shultz et al. 2012). Furthermore, fringe habitats allow *G. tigrina* to access the urban environment where they are able to make use of a variety of anthropogenic resources such as food, shelter and supplementary feeding stations provided by residents. It is the ability to colonize and adapt to both natural and urban landscapes that has facilitated the urban genet population and expansion into the suburbs of Kloof/Hillcrest.

Genetta tigrina occupancy experienced seasonal influences and was positively associated with winter. During winter the main prey items of *G. tigrina* was invertebrates, becoming less abundant (Scholtz and Holm 1986; Anu et al. 2009). This reduction in food availability may result in *G. tigrina* increasing their foraging bouts and distances in search of alternative food resources in the urban environment. This trend was observed when analyzing the scats of *G. tigrina* in the suburbs of Kloof/Hillcrest. During winter there was a reduction in invertebrates and a significant increase in anthropogenic waste in the scats. The reduction in the main prey items of *G. tigrina* and subsequent increased reliance on less desirable food items could account for the increased site occupancy of *G. tigrina* during winter (Widdows and Downs 2015).

The detection probability of *G. tigrina* was low (0.19 ± 0.03) and this could be due to several factors. Firstly, *G. tigrina* are considered to be semi arboreal in nature (Skinner and Chimimba 2005) and as a result may avoid moving on the ground where camera traps were positioned. Secondly, camera trap height and position may influence detection probability and site occupancy of *G. tigrina* due to their diminutive body size. Furthermore, interspecies interactions and spatial avoidance with humans and domestic pets may also have influenced the detection probability and site occupancy.

Bush cover had a negative influence on the detection probability of *G. tigrina*. In a study conducted by Ramesh and Downs (2014) bush land had a positive association with *Genetta tigina* detection probability as this habitat type provides shelter and reduced predation risk. Throughout the suburbs of Kloof/Hillcrest, *G. tigrina* have been observed utilizing anthropogenic structures such as roofs for day time resting sites. The availability of this novel resting sites and subsequent benefits (reduced predation risk and presence of prey items associated with human dwellings) may reduce the reliance on areas with high bush cover.

As human populations continue to rise and urban urbanization leads to a continuous decline in natural habitats, it is imperative to conduct surveys of carnivores living in close proximity to human settlements, in order to gain a greater understanding of the factors that influence their distribution. Occupancy modeling provides this information by calculating the relationship between species and various habitat characteristics (Bailey et al. 2004). Our occupancy modeling provided an insight into the landscape features that influence *G. tigrina* within an urban landscape. Future research could include other mammals that were documented during the urban survey such as caracal (*Caracal caracal*), blue duiker (*Philantomba monticola*) and porcupine (*Hystrix*

africaeaustralis) in order to obtain information on their occurrence within a human dominated landscape.

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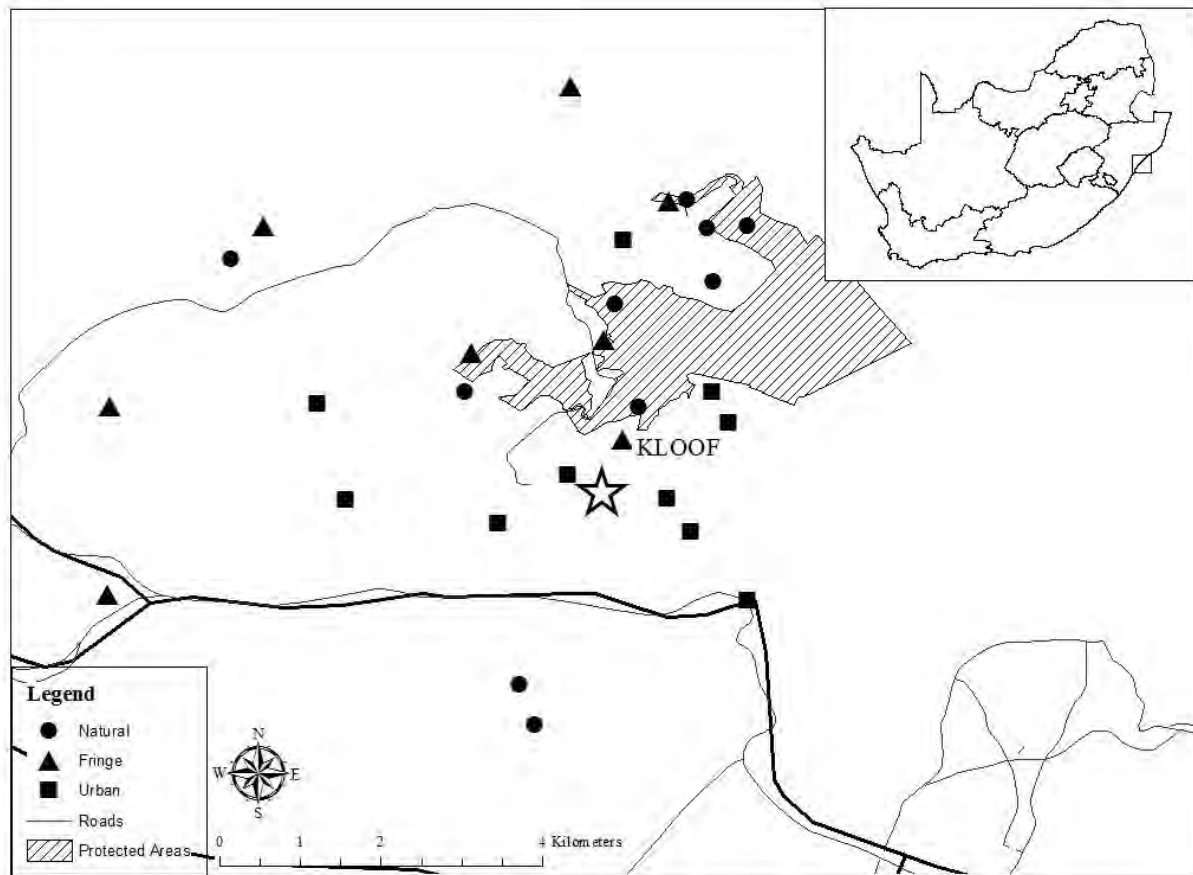
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Legends for Figures

Fig. 1 Camera trap locations in urban (situated within the urban landscape with no connection to green areas), fringe (situated on the border on green belts) and natural (situated within green belts) sites across an urban landscape in Kloof, Durban, KwaZulu-Natal, South Africa.

Fig. 2 The relationship between bush cover (%) and occupancy probability of *Genetta tigrina* based on parameter estimates from top models (ΔAIC < 2).

Fig. 3 The relationship between bush cover (%) and detection probability of *Genetta tigrina* based on parameter estimates from top models (ΔAIC < 2).
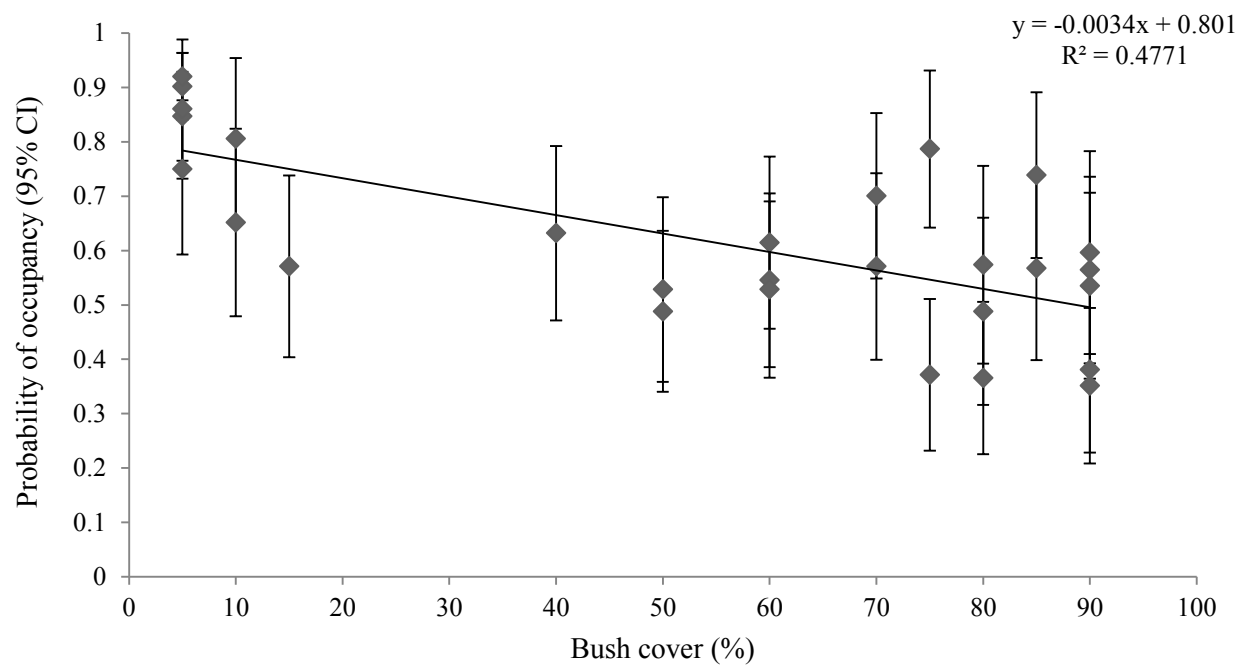
Figures

Fig. 1 Camera trap locations in urban (situated within the urban landscape with no connection to green areas), fringe (situated on the border on green belts) and natural (situated within green belts) sites across an urban landscape in Kloof, Durban, KwaZulu-Natal, South Africa.

Fig. 2 The relationship between bush cover (%) and occupancy probability of *Genetta tigrina* based on parameter estimates from top models (ΔAIC < 2).
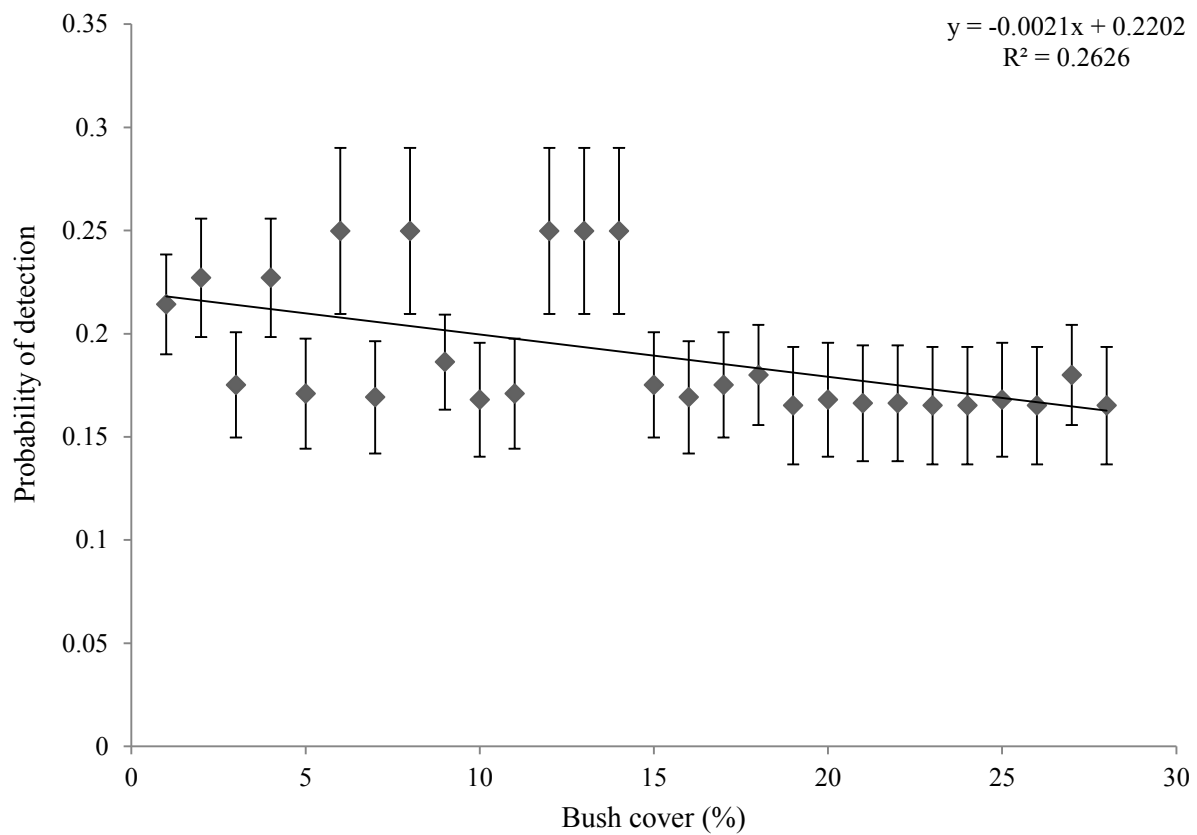
Fig. 3 The relationship between bush cover (%) and detection probability of *Genetta tigrina* based on parameter estimates from top models (ΔAIC < 2).

Tables

Table 1 Summary of AICc model selection and parameter estimates of site occupancy (ψ) and detection for *Genetta tigrina* in the urban

landscape of Kloof/Hillcrest, KwaZulu–Natal between June 2012 and October 2013.

Model	AIC	ΔAIC	AIC wgt	Model Likelihood	No.Par.	2LL	$\psi \pm$ SE	$p \pm$ SE
psi(BC+WP+WN+F),p(BC)	398.07	0.00	0.115	1.000	7	384.07	0.619 ± 0.179	0.190 ± 0.029
psi(BC+WP),p(BC)	398.17	0.10	0.109	0.951	5	388.17	0.615 ± 0.142	0.190 ± 0.029
psi(BC+WP+WN),p(BC)	398.28	0.21	0.104	0.900	6	386.28	0.618 ± 0.162	0.190 ± 0.029
psi(BC),p(BC)	398.54	0.47	0.091	0.791	4	390.54	0.615 ± 0.118	0.191 ± 0.029
psi(WN+BC),p(BC)	398.87	0.80	0.077	0.670	5	388.87	0.615 ± 0.142	0.191 ± 0.029
psi(BC+WP+F),p(BC)	398.94	0.87	0.075	0.647	6	386.94	0.614 ± 0.161	0.191 ± 0.029
psi(WN+DRD),p(BC)	399.19	1.12	0.066	0.571	5	389.19	0.628 ± 0.150	0.187 ± 0.030
psi(BC+F),p(BC)	399.34	1.27	0.061	0.530	5	389.34	0.615 ± 0.143	0.191 ± 0.029
psi(WN+DRD+F),p(BC)	399.35	1.28	0.061	0.527	6	387.35	0.622 ± 0.167	0.189 ± 0.030
psi(BC+DNW+WP),p(BC)	399.87	1.80	0.047	0.407	6	387.87	0.616 ± 0.163	0.191 ± 0.029
psi(.),p(.)	400.16	2.09	0.041	0.352	2	396.16	0.613 ± 0.093	0.199 ± 0.022

psi(F),p(.)	400.46	2.39	0.035	0.303	3	394.46	0.613 ± 0.123	0.199 ± 0.022
psi(WN+F),p(BC+DRD)	400.90	2.83	0.028	0.243	6	388.9	0.618 ± 0.147	0.190 ± 0.035
psi(DNW),p(.)	400.96	2.89	0.027	0.236	3	394.96	0.613 ± 0.125	0.200 ± 0.022
psi(BC+R+F),p(BC)	401.26	3.19	0.023	0.203	6	389.26	0.615 ± 0.169	0.191 ± 0.029
psi(WN+F),p(DRD)	401.51	3.44	0.021	0.179	5	391.51	0.612 ± 0.145	0.200 ± 0.029
psi(.),p(BC)	402.46	4.39	0.013	0.111	2	398.46	0.628 ± 0.096	0.195 ± 0.022
psi(.),p(DRD)	403.31	5.24	0.008	0.073	2	399.31	0.613 ± 0.093	0.212 ± 0.021

Abbreviations: Delta Alkaike Information Criterion (ΔAIC), twice the Log likelihood (2LL), number of parameters (No. Par.), estimated occupancy (ψ), estimated detection probability (p), bush cover (BC), wide path (WP), winter (WN), fringe habitat (F), distance to road (DRD) and distance to nearest water source (DNW).

Table 2 Untransformed parameter estimates and associated standard error (SE) for explanatory variables from the top occupancy and detection probability models for *Genetta tigrina*.

Site Occupancy			Site Detection Probability		
Covariates	Estimate	Standard error	Covariates	Estimate	Standard error
Intercept (p)	3.90	2.51	Intercept (ψ)	-0.81	0.37
BC	-1.95	1.20	BC	-0.41	0.25
WP	-1.78	1.21			
WN	1.79	1.25			
F	1.30	1.08			

Abbreviations: Estimated occupancy (ψ), estimated detection probability (p), bush cover (BC), wide path (WP), winter (WN) and fringe habitat (F).

Appendix 1 List of independent photographs of mammals captured in camera traps in the study sites, including their threatened status according to the Red Data Book of South African Mammals (Friedmann and Daly 2004).

Mammals	Scientific Name	Captured Images	Threatened Status
Banded mongoose	*Mungos mungo*	7	Least Concern
Blue duiker	*Philantomba monticola*	104	Vulnerable
Burchell's zebra	*Equus burchelli*	24	Least Concern
Bushbuck	*Tragelaphus scriptus*	205	Least Concern
Bush pig	*Potamochoerus larvatus*	24	Least Concern
Caracal	*Felis caracal*	15	Least Concern
Common grey duiker	*Sylvicapra grimmia*	125	Least Concern
Domestic dog	*Canis domesticus*	287	—
Feral cat	*Felis catus*	52	—
Greater cane rat	*Thryonomys swinderianus*	5	Least Concern
Rock hyrax	*Procavia capensis*	47	Least Concern
Large spotted genet	*Genetta tigrina*	118	Least Concern
Large grey mongoose	*Herpestes ichneumon*	4	Least Concern
Cape porcupine	*Hystrix africaeustralis*	71	Least Concern
Red duiker	*Cephalophus natalensis*	1	Least Concern
Slender mongoose	*Galerella sanguinea*	20	Least Concern
Thick-tailed bushbaby	*Otolemur crassicaudatus*	11	Least Concern
Vervet monkey	*Chlorocebus pygerythrus*	97	Least Concern
Water mongoose	*Atilax paludinosus*	94	Least Concern
White-tailed mongoose	*Ichneumia albicauda*	43	Least Concern

Errata

Line 1710: Full stop deleted after "site"

Line 1829: "bring" changed to "being"

Line 1825: Space included before "The"

Line 1873: Journal reference order corrected (Virgos and Casanovas 1997; Caro and Stoner 2003)

Line 1893: These habitats have not experienced the high modification associated with the urban centers.

Line 1946: Journal abbreviation corrected "Wildl Biol"

Line 1953, 2002: Journal abbreviation corrected "Mamm Rev"

Line 1988: Journal abbreviation corrected "Landsc Ecol"

Line 2000, 2036: Journal abbreviation corrected "Landsc Urban Plan"

Line 2005: Journal abbreviation corrected "Wildl Res"

Line 2019: Journal abbreviation corrected "Wildl Monogr"

Line 2026: Journal abbreviation corrected "Eur J Wildl Res"

CHAPTER 4

A genet drive-through: Are large spotted genets using urban areas for "fast food"? A dietary analysis

Craig D. Widdows & Colleen T. Downs*

School of Life Sciences, University of KwaZulu-Natal, Private Bag X01, Scottsville, Pietermaritzburg, 3209, KwaZulu-Natal South Africa.

Correspondence email: Downs@ukzn.ac.za

Correspondence

Prof CT Downs

School of Life Sciences, University of KwaZulu-Natal

P. Bag X01, Scottsville, 3209, Pietermaritzburg

Telephone: 27 (0) 33 260 5127/04, Cell: 27 (0) 829202026, Fax: 27 (0) 33 260 5105

Email: downs@ukzn.ac.za

Abstract

Knowledge of an urban carnivore's foraging behavior is vital to understanding its ecology. This is particularly important as urbanization continues to convert natural habitats into human-altered landscapes. Over the past few years there have been increasing reports of large spotted genets (*Genetta tigrina*) foraging within suburbs of towns and cities in KwaZulu-Natal, South Africa. Consequently, we investigated the dietary composition of urban *G. tigrina* using scat analyses, and the influence of predictable supplementary feeding stations on their feeding behavior in the

suburbs of Kloof/Hillcrest, KwaZulu-Natal. Prey items with the highest relative frequency of occurrence were invertebrates (42.5%). In particular, their scats found in anthropogenic structures such as roofs and out buildings were dominated by cockroaches (Blattodea). Small mammals also formed an important component of the diet. Significant seasonal variation in diet was recorded. The highest recorded relative frequency of occurrence of reptiles in scats was during spring (8.6%). The highest recorded relative frequency of occurrence of anthropogenic refuse in the scats was in winter (12.7%) with pieces of plastic, elastic bands and cardboard present in the scats. Uncommon genet behavior recorded at artificial feeding stations included diurnal feeding patterns and multiple individuals feeding with no signs of aggression. The presence of pet food, invertebrates associated with anthropogenic structures, and anthropogenic pollution/waste in the diet of urban genets, as well as their ability to use supplemental feeding stations highlights their adaptability to make use of temporally or locally available food resources within the urban environment.

Keywords Urban environment; Small carnivore; Scat analysis; Anthropogenic refuse; *Genetta tigrina*; Supplementary feeding

Introduction

Anthropogenic habitat modification degrades and alters natural ecosystems and is generally a threat to biodiversity worldwide (Lee et al. 2013). Consequences of urbanization include the replacement of natural habitats with infrastructure, fragmentation of landscapes, reduction in genetic diversity within meta-populations, as well as further loss of genetic connectivity across landscapes (Lee et al. 2013; Lowry et al. 2013). Further risks of the urban environment to vertebrates include persecution by humans, conflict with domestic pets, and collisions with motor

vehicles; all often with fatal consequences (Merkle et al. 2013). Species that are unable to adapt to human-modified environments may be excluded from these habitats (Lowry et al. 2013). Despite many challenges however, urban landscapes often offer widespread, high-energy content forage for some wildlife, drawing these species into them (Merkle et al. 2013). Furthermore, these resources can be more reliable and less likely to undergo seasonal fluctuations as opposed to naturally occurring resources (Bateman and Fleming 2012; Lowry et al. 2013). This provides secure, year round, food sources for urban wildlife that have the ability to exploit it. These species tend to exhibit various behavioral adjustments in response to human-modified landscapes (Sol et al. 2013). For example, black bears (*Ursus americanus*) observed in urbanized environments were active for fewer hours per night and had shorter hibernation periods than wild populations (Beckmann and Berger 2003).

The ability to exploit urban resources usually requires behavioral adaptations and favours species that express behavioral flexibility (Lowry et al. 2013). Various studies indicate that urban species are able to respond more rapidly to novel food opportunities and resources than species inhabiting non-urbanized landscapes (Sol et al. 2013). Urban food resources available include human refuse, crops, road kill, domestic pets, deliberate feeding and synanthropic rodents and birds (Bateman and Fleming 2012). Red foxes (*Vulpes vulpes*) and raccoons (*Procyon lotor*) readily exploit anthropogenic food by foraging at rubbish sites and scavenging in bins (Contesse et al. 2004; Bateman and Fleming 2012). In a study conducted by Contesse et al. (2004) more than 50% of urban fox stomachs examined, contained anthropogenic food items. Individuals readily exploiting refuse have a high probability of consuming pollutants such as plastic, rubber or paper, and the faeces of urban populations of raccoons, coyotes (*Canis latrans*) and stone marten (*Martes foina*) have been found to contain a variety of anthropogenic refuse (Hoffman and Gottschang

1977; Gehrt 2007; Lukasik and Alexander 2012). Deliberate feeding of wildlife in urban areas is also a common practice. The negative influences of these predictable supplementary feeding sites include altering of normal activities, nutritional setbacks and most notably habituation to humans (Lowry et al. 2013).

The diet of carnivores is affected by a variety of factors such as abundance of prey items, interactions with others predators and nutritional demands (Atkinson et al. 2002; Loveridge and Macdonald 2003; Van der Merwe et al. 2009). Knowledge of the diet of mammalian predators is fundamental for understanding their foraging behavior, population dynamics and habitat use (Caryl et al. 2012). Furthermore, an accurate description of a predator's diet is vital for a holistic assessment of its role in the ecosystem, including impact on prey populations, level of competition with other predators, structure of food webs and their trophic interactions (Herbst and Mills 2010; Klare et al. 2011; Caryl et al. 2012; Mbizah et al. 2012). Consequently, dietary analyses are instrumental in the establishment of carnivore conservation and management plans (Klare et al. 2011; Mbiza et al. 2012). However, accurately calculating the dietary composition, quantity, and quality is difficult, as direct feeding observations are often impossible under field conditions (Klare et al. 2011; Steyaert et al. 2012). Dietary analyses have played an integral role in the development of numerous carnivore management plans, especially for economically important or endangered species or those where there is human wildlife conflict (Klare et al. 2011; Mbizah et al. 2012). It is imperative to understand the ecological role of carnivores in ecosystems to enable the effective implementation of conservation strategies (Roberts et al. 2007), and in understanding their persistence in different habitats, such as urban areas.

The large spotted genet (*Genetta tigrina*) has been described as a generalist carnivore; widespread across much of South Africa, it occurs in a variety of habitats including savannah, woodlands, forest and is adaptable to human settlements (Skinner and Chimimba 2005; Roberts et al. 2007). Despite its generally widespread distribution, little research has been conducted on *G. tigrina*. The dietary composition of *G. tigrina* is generally broad and includes invertebrates, particularly insects as an important food source as well as small rodents (Muridae), reptiles, plant material and wild fruits (Rowe-Rowe 1996; Mills and Hes 1997, Skinner and Chimimba 2005, Roberts et al. 2007). Large spotted genets also attack domestic and free ranging poultry in farmlands and aviaries in urban areas resulting in human-wildlife conflict (Rowe-Rowe 1996). In most instances it is a single individual that returns to the poultry farms, which it identifies as a freely available and predictable food source (Skinner and Chimimba 2005). The generally catholic and opportunistic diet of *G. tigrina* is assumed to be influential in their wide spread distribution in Southern Africa (Roberts et al. 2007).

As urban landscapes continue to expand in KwaZulu-Natal, South Africa, there has been an increase in sightings of *G. tigrina* in urban areas (Widdows unpublished data). The diet of urban populations of *G. tigrina* has not been previously studied. Consequently, to understand the persistence of *G. tigrina* in urban areas we used scat analysis to examine their dietary composition in an urban area in KwaZulu-Natal. We collected scats over 12 months to determine seasonal variations in the dietary composition of urban genets. We expected dietary composition of urban *G. tigrina* to be broad with no seasonal variation.

Materials and Methods

Study area

The study was conducted in the urban area of Durban, KwaZulu-Natal, South Africa, particularly the Kloof/Hillcrest suburbs. This area is a mosaic of degrees of urban development interspersed with green areas (Roberts 1994; eThekwini Municipality 2007). The mean annual minimum and maximum temperature are 13.9ºC and 24ºC respectively; the mean annual rainfall is 974 mm (http://en.climate-data.org/location/27097/). Most rainfall occurs in the summer. Frost is infrequent whereas mist is common and provides additional moisture (Mucina and Rutherford 2006). The altitude of the study site is ≥ 560m ASL. The vegetation in the study area is dominated by KwaZulu-Natal sandstone sourveld scattered with low shrubs and patches of riverine forests (Mucina and Rutherford 2006). Most (68%) of this grassland biome has been transformed due to forestry plantations, cultivation and urban development (Mucina and Rutherford 2006). In addition, as a result of urban development and the establishment of gardens, many alien invasive plant species such as *Cinnamomum camphora* are found in the suburbs of Kloof/Hillcrest.

Data collection

Modern assessments of dietary composition in carnivores use a variety of visual techniques such as scat analysis, stomach content analysis and genetic techniques such as DNA barcoding and stable isotope analysis (Di Domenico et al. 2012; Steyaert et al. 2012). Scat analysis though, is the most popular technique employed to investigate their dietary composition. It is inexpensive, relatively quick to apply, non-destructive (does not required the animal to be destroyed) and is practical for obtaining large sample sizes (Di Domenico et al. 2012; Romero et al. 2012; Marucco

et al. 2008). Scat analysis involves the identification and quantification of natural and unnatural "ingesta" that have passed through the digestive system of the animal in question (Trites and Joy 2005). Therefore, scats provide information on dietary composition and allow for feeding habits to be continuously monitored (Trites and Joy 2005). Scat analysis is a non-invasive technique and one that is compatible with the elusive lifestyle of many carnivore species (Marucco et al. 2008). Therefore, this technique is most useful when direct observations of foraging are not possible (Klare et al. 2011). Consequently, we opted to use scat analysis to determine dietary composition of urban *G. tigrina.*

Before scat collection of urban *G. tigrina* began, we placed an article in the local Durban newspapers (electronic and hard copy) and with various urban conservancies asking the public for information on whether they had genets on their property. Thereafter we visited and inspected possible sites based on information received. We identified midden sites by an accumulation of faeces in prominent locations such as in house roofs, overturned pots and on elevated mounds in suburbia (Roberts et al. 2007). The identification of the faeces was based on odor, color, morphology and dimensions (Stuart and Stuart 1998; Skinner and Chimimba 2005). This was further supported by the presence of associated field signs such as spoor, digital images collected using camera traps placed in the area and recent public sightings of *G. tigrina*. Water mongoose (*Atilax paludinosus*), have an overlapping range and similar faeces to that of *G. tigrina* and also makes use of middens. However, the presence of fresh water crab shells in the scats is characteristic of *A. paludinosus* and so these were not confused with scats of *G. tigrina*.

We collected faecal samples from latrine or midden sites of *G. tigrina* that were found. Thereafter we collected genet scats at monthly intervals between June 2012 and June 2013 at the

regular midden sites identified. We classified seasons as spring (September-November), summer (December-February), autumn (March-May) and winter (June-August) to parallel changes in the climate and vegetation (Kamler et al. 2012). During the first month of scat collection (June 2012) old faeces were identified by their pale coloration and lack of odor, and were subsequently excluded from the study. The scats from each midden were individually placed into Ziploc bags (26.8 cm x 27.9 cm) and numbered with the date of the collection, the GPS coordinates of the midden site and the location of the midden site (anthropogenic structure, garden, reserve). In addition some scats were also opportunistically collected whilst walking along hiking and game paths in the Kranzkloof Reserve, a green area within the urban landscape.

Scat-content analysis

After collection we refrigerated the *G. tigrina* scat samples to avoid decomposition before analysis. In the laboratory, we placed each scat sample into a 250 ml glass beaker with boiling water to soften the sample. The sample was then washed through a 1mm sieve to separate the prey items such as insects, vertebrate bones and hairs from other organic material. The remains were examined under a dissecting microscope (E. Leitz Wetzlar, Germany) with 10x Magnification. The presence and frequency of each prey category were recorded (Roberts et al. 2007). We initially categorized prey items into six food classes to allow for comparison: small mammals, birds, invertebrates, reptiles, vegetation, and anthropogenic pollution or waste. Invertebrate prey remains were further identified to order using the keys in Scholtz and Holm (1986). Invertebrate prey samples were identified from diagnostic fragments such as head capsules, wings, legs, shells and mouth parts. Small mammals were identified by their teeth to species level (Bowland and Bowland

1989). Birds, reptiles and vegetation were not identified to lower taxonomic levels (Braczkowski et al. 2012). Any prey items from the various categories that were unidentified were recorded as unknown. Anthropogenic items (e.g. plastic, elastic bands) were separated and identified.

We extracted the hair samples from *G. tigrina* scats, then used a microscope to identify them to species level using cuticular scale patterns and groove characters. The samples were removed using fine forceps and placed into labeled Petri dishes. The hairs were then washed in acetone and dried on filter paper. Finally, hair samples were placed on slides and thinly coated with transparent nail polish, creating cuticular imprints (Klare et al. 2010). Prey identification was based on the microstructure of the hairs from observations of scale patterns and cross sections. These microscopic characteristics, coupled with the physical appearance of hairs were compared with various reference collections (Keogh 1983) for species identification.

Monitoring genet behavior at predictable urban feeding sites

We used camera traps (LtI Acorn ® 6210MC, China) triggered by passive infrared sensors to monitor *G. tigrina* activity at six artificial/supplementary feeding stations in urban gardens of Kloof/Hillcrest. Feeding stations were defined as properties with landowners that placed food out on a nightly basis for *G. tigrina*. All these feeding stations had been operating and feeding *G. tigrina* for more than six months with the individuals accustomed to human activity in the vicinity. Camera traps were mounted on poles within 5 m of the feeding stations with the sensitivity set to "medium" and the time of each image was recorded. During the processing of images, both the behavior and number of individuals observed were recorded for each site. Furthermore, we

recorded the type and quantity of food used as well as the time at which the food was put out at each site. Non-target species that used the feeding stations were also recorded.

Data analyses

We analyzed *G. tigrina* scat contents for both seasonal and overall dietary trends (Loveridge and MacDonald 2003). The dietary composition of *G. tigrina* was expressed in two complementary ways namely, percentage of occurrence (PO) and relative frequency of occurrence (RFO) of the different components. The percentage of occurrence was calculated as the number of faecal samples in which a species or taxonomic group occurred, multiplied by 100/ the total number of faecal samples (n) (Kalle et al. 2012). This provided a measure of how often *G. tigrina* feed on each dietary component (Loveridge and MacDonald 2003). The relative percentage of occurrence was calculated as the number of occurrences of a species or taxonomic group, multiplied by 100/ total occurrences of all prey items in all the samples (Loveridge and MacDonald 2003). This provided a measure of the relative importance of each food item in the overall diet of *G. tigrina* (Loveridge and MacDonald 2003). Both measures should be treated with caution as they produce errors; scats collected in succession may contain common particles from a single, large meal thus causing them to be overestimated (Atkinson et al. 2002). Furthermore, small, commonly consumed prey items such as invertebrates may also be overestimated in the dietary analysis (Atkinson et al. 2002). However, the influence of these errors was minimized due to the substantial sample size and the collection of scats from multiple midden sites.

The number of images of *G. tigrina* provided a measure of activity (Presence/absence, feeding observations, intraspecific and interspecific interactions) at the feeding stations. Individuals were identified by spot markings using camera trap images, to determine the number of individuals repeatedly exploiting the feeding stations. Images recorded at the feeding stations were separated into six time categories at 3h increments starting at 15:00 and ending at 09:00. The total activity in the six time categories were compared in order to determine preferential feeding times and assess how feeding times may influence activity levels. This was conducted between June 2012 and September 2013. Feeding stations were separated into two categories, those where food was put down at a set time every night (n = 3) and those where the feeding times varied (n = 3).

Statistical analyses

As the data were not normally distributed, comparisons between dietary components were conducted using Friedman ANOVA. Seasonal comparisons were conducted using the non-parametric Kruskal-Wallis test (Statistica V9.0, StatSoft 2009). Comparisons in activity levels among the time categories were conducted using Analysis of Variance (ANOVA). A Tukey's post-hoc test was conducted to determine among which time categories were significant differences recorded. ORIANA statistical program was used to construct circular histograms representing *G. tigrina* activity patterns at feeding stations across all time categories (ORIANA V4, Kovach Computing Services 2013).

Results

Urban diet composition

A total of 591 *G. tigrina* scats were collected from 11 middens over one year. The lowest number of scats collected in June (n = 38) and the highest number of scats collected in December (n = 62). A total of 14 taxa were identified: four orders of insects, four species of small mammals, birds, reptiles, vegetation and anthropogenic waste were recorded in the scats (Table 1). There was a significant difference among the overall dietary constituents recorded in the scats of *G. tigrina* (ANOVA, χ^2 = 48.26555, df = 5, P < 0.05). Invertebrates, the most common food items, accounted for more than 42.5% (± 6.9%) of the items collected from the scats (Fig. 1). Within the invertebrates, three orders of insects dominated the diet, namely Blattodea, Isoptera and Coleoptera, which together comprised 33.5% (± 4.4%) of the total prey throughout the year. The scats of *G. tigrina* living in anthropogenic structures, however, were dominated by Blattodea. Small mammals accounted for 20.8% (± 2.0%) of items collected from the scat samples (Fig. 1). Reptiles and birds appeared in a low proportion of the overall diet comprising of 5.6% and 11.5% of prey items respectively (Fig. 1). Anthropogenic waste including refuse comprised a greater percentage of the diet than reptiles, forming 6.1% of the annual diet of *G. tigrina* (Fig. 1). A large portion of the scats of *G. tigrina* contained the remains of pet food, identified by a fine sand-like substance in broken down scats. This was further supported by a variety of observations of *G. tigrina* feeding out of dog bowls (13 observations) or residents feeding *G. tigrina* with domestic animal food (8 observations).

Seasonal variations

The relative frequency of occurrence of invertebrates and mammals recorded in urban *G. tigrina* scats across seasons ranged from 32.2 - 51.9% and 17.1 - 23.1% respectively (Fig. 2). However, this variation was not significantly different for invertebrates (H = 6.180, P = 0.1032) nor mammals (H = 2.590, P = 0.4593) between seasons. Birds comprised 0 - 17.7% of prey items consumed throughout the year (Fig. 2). However, this variation was not significantly different between seasons (H = 3.410, P = 0.3326). Despite vegetation constituting between 9 and 20.8% of the prey items recorded in the scats between seasons, this variation was not significant (H = 6.231, P = 0.1009). There was a significant difference in the frequency of occurrence of reptiles recorded in the scats between seasons (H = 8.035, P = 0.0453) with the highest frequency recorded in spring, 8.5% (± 3.3%) and summer, 7.9% (± 0.6%) (Fig.2). Amount of anthropogenic pollution in the diet was significantly different between seasons (H = 8.741546, P =.0329). The highest relative frequency of occurrence of anthropogenic pollution in scats was in winter comprising 12.7% (± 2.4%) of prey items recorded (Fig. 2).

Predictable feeding stations

The main type of food regularly provided at artificial feeding stations in urban gardens was chicken comprising of hearts, gizzards, necks and feet. Average daily mass of supplementary food provided per feeding table was 190 g with the highest quantity of food provided being 400 g (20 chicken hearts/gizzards). Two of the six feeding stations provided ±400 g of chicken nightly.

The earliest and latest time that *G. tigrina* were documented visiting supplementary feeding stations were 15:10 and 07:37 respectively (Fig. 3). Genets most often visited between 18:00 - 20:59 with a total of 703 images captured (117.6 ± 75.45) (Fig. 3). There was no significant difference in genet activity with time categories at the feeding stations where food was put out at

varied times (ANOVA, $F = 1.99$, $P = 0.152$). In comparison, there was a significant difference in *G. tigrina* activity with time categories at the feeding stations where food was provided at a set time (ANOVA, $F = 22.05$, $P < 0.05$). Genets were more active at the feeding stations between 18:00-20:59 than any other time which coincided with the food provision (post-hoc Tukey, $P < 0.05$). Individuals were identified by spot markings using camera trap images, to determine the number of individuals repeatedly exploiting the feeding stations. In most instances a single *G. tigrina* regularly visited and consumed all of the available food. However, two and three adult individuals were recorded feeding together at two of the feeding stations in particular and showed no aggression towards each other. Both feeding stations were opposite a wooded area and no dogs were present on the properties (C. Widdows pers. obs.). Furthermore, at predictable feeding stations individuals were recorded on a daily basis during daylight hours, sometimes as early as 15:10. One of the feeding stations placed food out in the mornings between 05:00 – 06:00 and *G. tigrina* were recorded feeding between 06:00 and 07:45.

Other mammalian species were also recorded visiting the supplementary feeding stations. These included the nocturnal water mongoose and white-tailed mongoose (*Ichneumia albicauda*); and diurnal slender mongoose (*Galerella sanguinea*), banded mongoose (*Mungos mungo*), vervet monkey (*Chlorocebus pygerythrus*) and rock hyrax (*Procavia capensis*). Rock hyrax was the only species that was not observed feeding on remains, and this is likely due to their herbivorous diet (Kotler et al. 1999; Skinner and Chimimba 2005).

G. tigrina individuals did not exhibit aggression at feeding stations. They were also observed feeding with other species, e.g. water mongoose, with no signs of aggression (n = 6). Similar trends observed in altered activity around feeding stations by *G. tigrina*, were also

observed in water mongoose at two stations. Individuals were initially recorded between 24:00 and 2:00, but once the individuals located the food resource they were recorded as early as 18:00. Individuals arrived 6 h earlier at feeding stations to ensure they consumed the food before *G. tigrina* arrived. Possible competition between these two species for the food provision may result in both nocturnal species becoming more active during daylight hours.

Discussion

Although *G. tigrina* is described as a generalist carnivore, previous studies in non-urban areas recorded invertebrates as the largest component of their diet with Coleoptera and Orthoptera the dominant invertebrates in terms of frequency of occurrence and volume (Roberts et al. 2007). Other invertebrate species in their diet include Isoptera, Lepidoptera, Arachnida and Myriapoda (Skinner and Chimimba 2005). The mammalian component of their diet consists of a variety of Muridae of which *Mastomys* spp. and *Otomys* spp. form a high proportion (Skinner and Chimimba 2005, Roberts et al. 2007). Other vertebrates recorded in smaller proportions of their diet include reptiles (common striped skink (*Mabuya striata*), flap-necked chameleon (*Chamaeleo dilepis*), Kirk's rock agama (*Agama kirkii*), boomslang (*Dispholidus typus*), amphibians and fish (Skinner and Chimimba 2005; Roberts et al. 2007).

Similarly in the current study, scat analyses showed that urban *G. tigrina* were generalists with a diverse diet and seasonal variation in some dietary components. In addition, the most common prey items consumed by urban *G. tigrina* were invertebrates. Furthermore, there was no significant difference in insects consumed between seasons by *G. tigrina*. Despite the reduction in abundance of insects during the cold winter periods (Scholtz and Holm 1986; Anu et al. 2012),

invertebrates remained their dominant prey item in winter. This suggests that *G. tigrina* are utilizing insect species that are associated with urban environments during these periods. In particular, their scats found in anthropogenic structures such as roofs and out buildings were dominated by cockroaches (Blattodea). The importance of this species in their diet is probably related to its high abundance in human settlements (Mahmoud et al. 2013), although this was not quantified in the current study.

Although our results showed that invertebrates were an important prey for urban *G. tigrina*, frequency of occurrence calculations may overemphasize the importance of smaller prey items in the diet of carnivores (Klare et al. 2011). Relative biomass is considered a more reliable dietary index (Klare et al. 2011) but the main aim of the study was to determine the dietary composition of urban genets as a population as no prior research had been conducted. Furthermore, we were unable to estimate relative biomass because in some instances multiple individuals made use of a single midden site.

Mammals were the second most common prey item recorded in the diet of urban *G. tigrina*. However, the species richness was low with only four species recorded from the scats. This may reflect the relatively low small mammal species richness within urban areas (Downs unpublished data). There was significant variation in the consumption of reptiles by urban *G. tigrina* with the highest rates of consumption in spring and summer, which was consistent with the findings of Rosalino and Santos-Reis (2002). This seasonal increase may result from sustained reptile activity during dusk in warmer periods, thus bringing them into contact with *G. tigrina*. The variable skinks (*Mabuya varia*) which were identified by scale patterns in the scats, were consumed at high levels. Like cockroaches, *M. varia* are common around human-altered landscapes (Widdows pers. obs.).

Birds comprised a low proportion (11.5%) of the diet of urban *G. tigrina*. Similarly, birds made up only 1% of the diet of *G. tigrina* in Dwesa Nature Reserve, Eastern Cape (Roberts et al. 2007). However, these South African findings are contrary to that on *Genetta genetta* in Spain, where birds comprised between 45.5% and 88.4% of prey items consumed (Virgós et al. 1996). In the current study, birds were most commonly found in the diet during winter (14.1%) and spring (12.9%) but this increase was not significant. Roberts et al. (2007) also documented an increased importance of birds in the diet of *G. tigrina* during spring. This increase may be due to the greater susceptibility of breeding birds and the presence of juveniles over the breeding season. This was further supported by sightings of *G. tigrina* raiding hadeda ibis (*Bostrychia hagedash*) and masked weaver (*Ploceus velatus*) nests in the greater Durban district (C. Widdows pers. obs.).

The presence of grass in the scats of urban *G. tigrina* is consistent with other carnivore species such as *Canis mesomelas, C. adustus* (Loveridge and Macdonald 2003) and *C. latrans* (Lukasik and Alexander 2012; Watts and Alexander 2012). Grass may be present to aid in digestion, to stimulate vomiting and to dislodge hair from the intestines (Roberts et al. 2007). Grass was often present in scats that contained large amounts of fur and bones and this observation was also recorded in the scats of *C. mesomelas* (Loveridge and Macdonald 2003). The presence of grass and vegetation in the diet of genets has also been recorded in *G. maculata* (Stuart and Stuart 2003) and *G. rubiginosa* (Engel 1998) with 39.8% and 13.7% respectively.

Anthropogenic pollution/waste provided a reliable alternative food source (relative frequency of occurrence 6.13%) utilized by urban *G. tigrina*, particularly in winter. The nocturnal nature of this species facilitates refuse scavenging when human activity is relatively low. The highest relative frequency of occurrence of anthropogenic pollution/waste was during winter (12.7 %). As mentioned during winter other prey items, such as invertebrates, generally become less

abundant (Scholtz and Holm 1986; Anu et al. 2012). This reduction in winter food availability may result in *G. tigrina* using these alternative urban food resources. Furthermore, during summer and spring we detected little to no anthropogenic pollution/waste in their scats when prey items such as invertebrates are more abundant. Use of anthropogenic resources, such as a refuse, have also been recorded in the diet of *G. genetta* in Spain (Rosalino and Santos-Reis 2002). Increased anthropogenic refuse in the diet of *C. adustus* than *C. mesomelas* was attributed to their movements and proximity to areas with human activity (Loveridge and MacDonald 2003). Significant amounts of refuse in scats have also been recorded in carnivores such as red fox (Contesse et al. 2004), coyote (Gehrt 2007; Lukasik and Alexander 2012) and raccoons (Hoffman and Gottschang 1977). Contesse et al. (2004) found that more than 50% of stomach contents of red fox in urban Zurich contained anthropogenic refuse. The ability to use anthropogenic refuse allows these species to live in close proximity to human-modified landscapes and exploit this abundant new resource. A wide variety of anthropogenic waste was recorded in the diet of stone martens living in Hungary, including indigestible material such as nylon, string, straw, wool, styrofoam, rubber, paper and aluminum foil (Lanszki et al. 2009).

The ability of *G. tigrina* to exploit urban resources is a behavioral adaptation to the urban environment and exemplifies their opportunistic feeding habits. This species has identified a novel, energy rich food source that is present throughout the year. As the abundance other food items such as invertebrates and mammals decrease, *G. tigrina* appear to target anthropogenic resources to supplement their diet. However, the amount of plastic, rubber, aluminum and string that we recovered from the scats was high, and this quantity of indigestible material passing through the gut may have negative impacts of the health and bodily functioning (turning of the gut or tearing the digestive tract) of individuals. This also indicates a high level of behavioral flexibility (Lowry

et al. 2013).

A large portion of the scats of *G. tigrina* contained the remains of pet food. Scats that contained domestic animal food contained few other prey items. This could indicate the large consumption of domestic animal food when accessible. When domestic food was present in the stomach of red fox in Zurich, it was the dominant food item recovered from the stomach contents (Contesse et al. 2004). Urban *G. tigrina* may be targeting this high energy food resource and subsequently reduce their energy expenditure required for locating, foraging and capturing food items, as pet food is generally predictably available each night. Targeting such a predictable food source further indicates the behavioral adaptation of *G. tigrina* to an urban environment's resources. However, the use of predictable food sources such as anthropogenic pollution/waste and domestic pet food by urban genets has resulted in conflict with humans and domestic pets. The most cited cause of genet fatalities within the suburbs of Kloof/Hillcrest were attacks by domestic animals and collisions with vehicles (Widdows unpublished data).

Predictable supplementary feeding of urban carnivores is not unique to the current study, and is described in most areas where carnivores live in close proximity to humans (Lowry et al. 2013). Roper (2010) indicated that 29% of residents surveyed in Brighton, England provided food for foxes, badgers and various other mammals. Furthermore, over half of these residents provided food every night. In the current study, two of six feeding stations regularly provided ± 400g of chicken nightly. In most instances a single *G. tigrina* consumed all of the available food. Similar trends have been recorded in California where a resident placed on average ±7 kg of chicken, beef and turkey to a high density of urban red foxes (Lewis et al. 1993). This is a large quantity of food being consumed by a single individual on a nightly basis, with more than a third of their body weight being consumed in a short period of time.

Feeding stations that provided food at set times had little variation in the activity and behavior of *G. tigrina*; most activity at these coincided with the times that food was provided between 18:00 and 19:00. It appears that individuals have identified the times at which these food resources are regularly provided and consequently arrive at the stations close to these time periods to ensure they secure the maximum benefit from these resources. So the provisioning of food by urban residents has resulted in significant changes to the behavior of *G. tigrina* within an urban landscape.

According to Skinner and Chimimba (2005), *G. tigrina* are exclusively nocturnal with any movements or activity during daylight hours being considered atypical behavior. However, at predictable feeding stations individuals were recorded on a daily basis during daylight hours. This is unusual behavior for a nocturnal species and appears to be a consequence of the predictability of supplementary food provision. Furthermore, the general absence of apex predators in the urban environment has opened up a new diel niche that *G. tigrina* appear to be exploiting, allowing genets to move around with reduced predation risks during daytime. The benefits of utilizing the high quality food resources offered at feeding stations is greater than the risks involved in daylight moving. Individuals who are able to exploit this resource will save considerable amounts of energy as the energetic costs required to locate, capture and kill prey are significantly reduced. As missed-opportunities costs also are reduced, individuals can spend more time performing alternative fitness-enhancing activities and less time foraging (Mitchell et al. 1990). Furthermore, the reduction in foraging time results in a decline in the probability of encountering a predator, which subsequently further decreases predation risk and energetic costs. This could possibly explain the diurnal feeding patterns shown by certain individuals and change in their feeding ecology in the urban environment. This also shows a high level of behavior plasticity that is considered a

prerequisite for a successful urban carnivore (Bateman and Fleming 2012). Behavioral adaptations from provisioning of food have also been documented in Eastern chipmunks (*Tamias striatus*), who in natural habitats exhibit elevated activity during spring and autumn and a reduction in activity levels during summer whereas individuals living in human-modified landscapes in Massachusetts showed no seasonal reduction in activity (Ryan and Larson 1976). This has been attributed to the stable supply of anthropogenic resources available in residential area and is a behavioral adaptation to urban life.

Genetta tigrina are generally regarded as solitary (Skinner and Chimimba 2005), but up to three adults were recorded feeding together regularly with no signs of aggression between individuals. It is possible that the benefits of the high quality food outweigh the energetic costs that would be expended in an aggressive display, and more, the risk of injuries that may follow. Even if individuals are foraging with others, the resource requires no foraging effort and this may outweigh the costs of aggression. The lack of aggression of *G. tigrina* at feeding stations is not limited within the species as they fed together with other species showing no aggression. Again, the high energy content available at feeding station and predictably of feeding appears to be having a significant influence on the foraging ecology of *G. tigrina*.

Bateman and Fleming (2012) defined the "ideal urban carnivore" as expressing high adaptability and flexibility in diet, social behavior, and movements. *Genetta tigrina* in this urban study have expressed the above characteristics, with behaviors observed being contrary to available literature and rural populations. In summary, urban *G. tigrina* have a broad dietary niche and showed high levels of adaptability in using anthropogenic food and refuse during the stressful winter periods and exploiting predictable food sources such as domestic pet food and feeding

stations. Furthermore, shifts in the diel movements and feeding as well as increased tolerance of competitors were observed at feeding stations. The ability of urban genets to utilize anthropogenic food resources has resulted in their persistence within urban areas. However, the costs associated with the increased trends towards urban living (conflict with humans, domestic pets and collisions with vehicles) results in high rates of mortality in urban areas. As human populations grow, urban areas will continue to expand. Therefore, it is imperative that we understand the urban ecology of urban carnivores to ensure improved conservation and management strategies.

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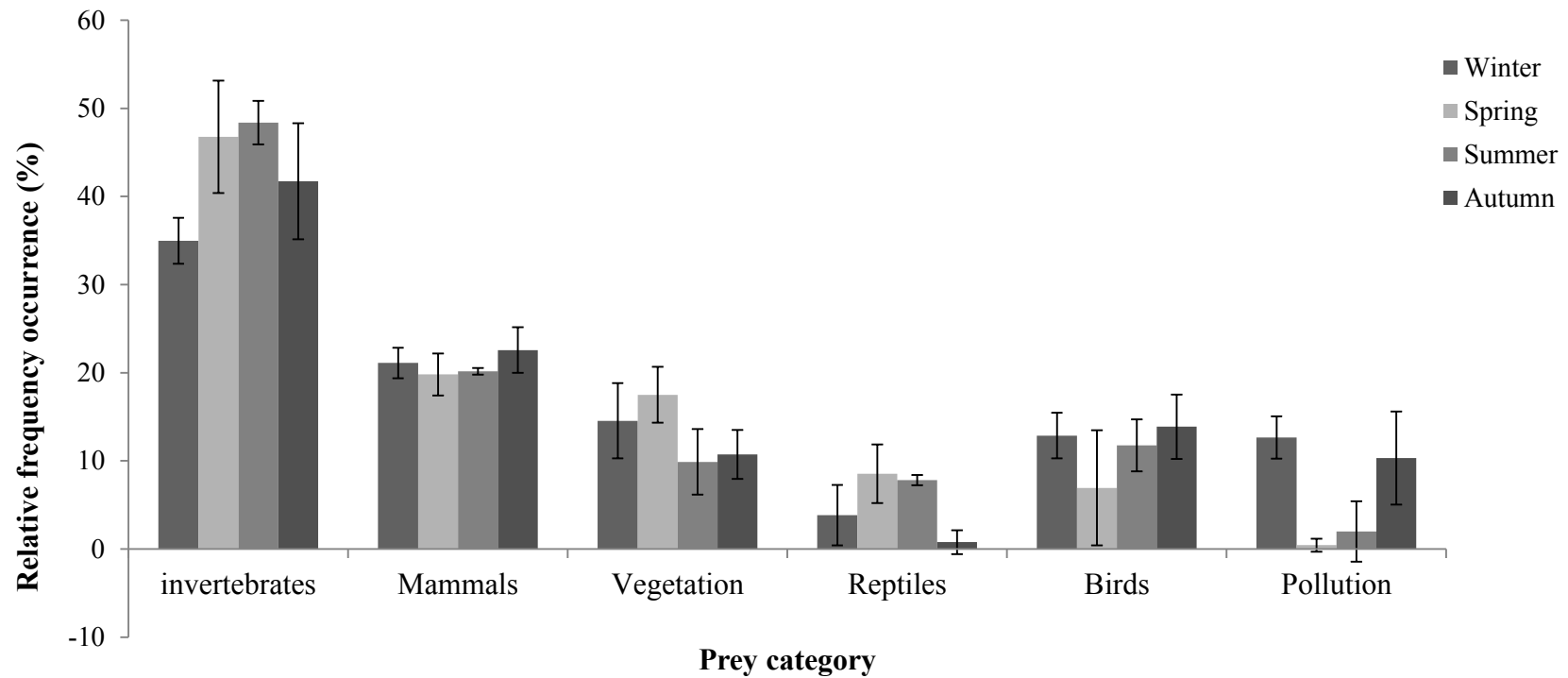
Fig. 1 The mean (± SD) annual relative frequency of occurrence (%) of all prey categories recovered from the scats of *Genetta tigrina* in an urban environment.

Fig. 2 The mean (± SD) relative frequency of occurrence (%) of all prey categories recovered from the scats of *Genetta tigrina* across all four seasons in an urban environment.

Fig. 3 Activity patterns of *Genetta tigrina* at predictable feeding stations (n = 6) situated within the urban landscape of Kloof, KwaZulu-Natal.
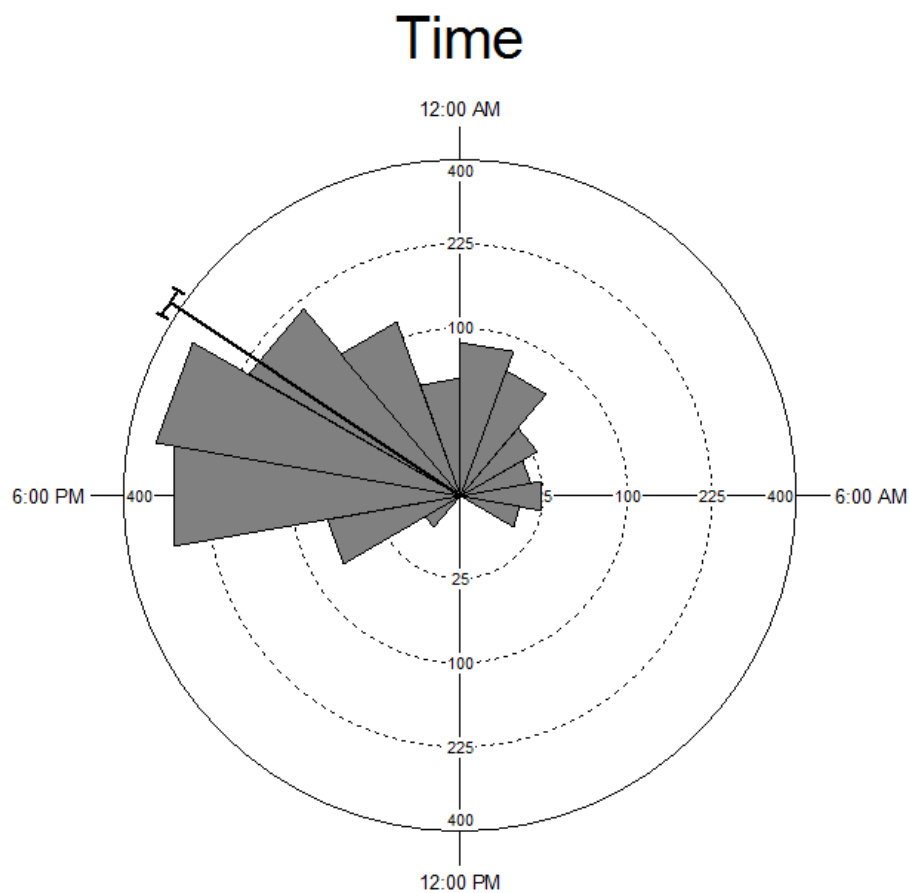
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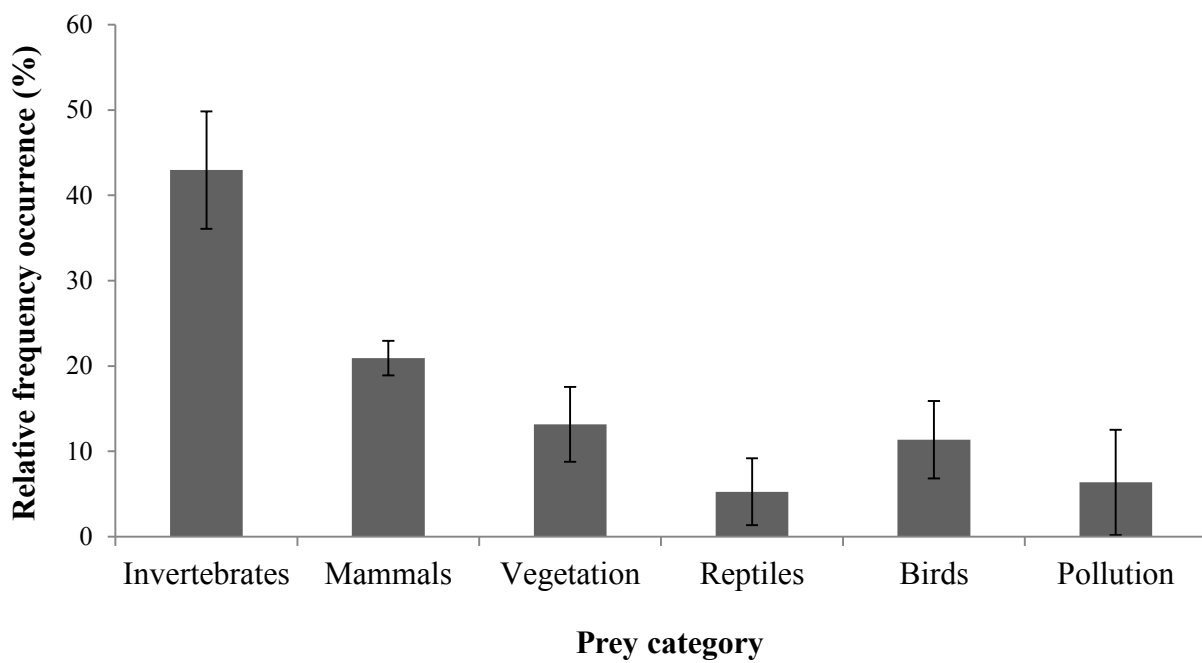
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Fig. 3 Activity patterns of *Genetta tigrina* (reflected as camera trap images per hour) at predictable feeding stations (n = 6) situated within the urban landscape of Kloof, KwaZulu-Natal.

Tables

Table 1 Prey items recorded in the scats of *Genetta tigrina* in the urban landscape of Kloof/Hillcrest, KwaZulu–Natal between June 2012 and June 2013.

Food item	PO	RPO
Invertebrates	79.69	42.47
Coleoptera	42.13	22.45
Dictyoptera	20.81	11.09
Orthoptera	10.15	5.41
Lepidoptera	6.60	3.52
Mammals	39.08	20.83
Rattus sp.	19.79	10.56
Mastomys natalensis	9.64	5.14
Dendromus sp.	7.28	3.87
Mus minutoides	2.37	1.26
Vegetation	25.21	13.44
Reptiles	10.49	5.59
Birds	21.65	11.54
Pollution	11.5	6.13

Errata

Line 2282: Full stop added at end of sentence

Line 2418: Scietific name deleted

Line 2471: "Dwesa Nature Reserve" corrected

Line 2621: Journal abbreviation corrected "Carni Conserv"

Line 2643: Journal abbreviation corrected "Nat Somogy"

CHAPTER 5

Urban roost temperature of large-spotted genets: The effect of anthropogenic structures

C.D. Widdows and C.T. Downs[*]

School of Life Sciences, University of KwaZulu-Natal, Private Bag X01, Pietermaritzburg,

3209, South Africa

[*]Tel: 033 260 5127

  Fax: 033 260 5105

[*]Correspondence Email: downs@ukzn.ac.za

ABSTRACT

Generally, large-spotted genets (*Genetta tigrina*) use tree hollows and canopies as daytime roosts during their inactive phase. However, there has been an increasing tendency for individuals to make use of anthropogenic structures, such as roofs, within urban landscapes in KwaZulu-Natal, South Africa. This study investigated the roosting thermal dynamics of large-spotted genets within the urban suburbs of Kloof. Roost temperatures were recorded with *i*-Button[®] temperature loggers at known large-spotted genet roosts in anthropogenic structures as well as in natural roost sites. Over the year temperatures varied significantly between months and among different roosts. However, anthropogenic roost temperatures were higher than ambient temperatures throughout the study period. Furthermore, anthropogenic roosts had higher temperatures (with lower variability) than natural roost sites. This study highlighted the

importance of anthropogenic structures as daytime roosts for large-spotted genets within an urban mosaic. However, high temperatures experienced during the summer can be detrimental to juvenile large-spotted genets resulting in plasticity of breeding behaviour and a switch to producing young in cooler months.

Keywords: *Genetta tigrina*, Urban landscape, Roost temperature.

Introduction

Urbanization is currently the fastest growing type of landscape modification globally, with recent estimates predicting that 60% of the global human population will be living in urbanized landscapes by 2030 (Lowry et al*.,* 2013; Murray and St. Clair, 2015). This anthropogenic conversion, alteration, and fragmentation of natural habitats has resulted in a dramatic loss of biodiversity worldwide (Jung and Kalko, 2011; Murray and St. Clair, 2015). This reduction of natural habitats has subsequently forced many species to live in close proximity to human settlements (Bateman and Fleming, 2012; Lowry et al., 2013). Responses to urbanization vary among species with some thriving while others avoid and are subsequently excluded from urbanised landscapes (Fischer et al., 2012; Newsome et al., 2015). Species that are able to live within urban areas are often generalists that are able to adapt to anthropogenic landscapes and exploit urban resources (Sol et al., 2013). These species exhibit significant ecological, behavioural and demographic plasticity (McCleery and Parker, 2011; Lowry et al., 2013). These adaptations include reduced wariness, reduced dispersal and home ranges, generalist diet, increase in population densities and altered diel activity patterns (Parker and Nilon, 2012). The urban landscape provides a variety of novel resources for species that are able to persist in anthropomorphic environments (Newsome et al., 2015). However, there is an array of threats

facing wildlife living in urban landscapes, such as their vulnerability to human persecution, collisions with motor vehicles, or attacks by domestic animals (Magle et al., 2012).

Roost sites, such as tree hollows and cavities under root systems, provide a variety of benefits for taxa including thermoregulation, denning, breeding and evasion from predators (Isaac et al., 2014). More than 360 mammal and bird species make use of these roost sites worldwide (Davis et al., 2014). As urban areas continue to expand, the availability of natural roost sites for fauna are declining and this can have an impact on their breeding ecology and subsequent persistence within this environment (Carvalho et al., 2014; Davis et al., 2014). This is particularly true for solitary carnivores where each roost site is generally used by a single individual at a time, with the exception of females and cubs during the breeding season (Carvalho et al., 2014). Urban expansion is replacing natural roost sites with anthropogenic alternatives including roofs and outbuildings (Isaac et al., 2014). Faunal species spend a large portion of their time roosting and as a result roost selection and behaviour provides an important mechanism for energetic optimization (Gruebler et al., 2014). The reduction of natural roost sites such as tree hollows can be overcome through the use alternative roosting options within an individual's range.

Urban carnivores use anthropogenic structures for secure shelters, roosting and breeding sites (Bateman and Fleming, 2012). In some instances, anthropogenic structures are utilized even when natural alternatives such as green belts are available (Lowry et al., 2013). The use of man-made structures as resting areas is not isolated to carnivores; certain urban bat species such as *Epomophorus wahlbergi*, *Eptesicus fuscus*, *Eptesicus serotinus* and *Vespertilio murinus* make use of anthropogenic structures for roost sites (Neubaum et al., 2007; Mazurska and Ruczynski, 2008; Monadjem et al., 2011). Day roosts provide buffering from severe climatic conditions and provide protection from predators (Moussy, 2011). Temperature plays a significant role in roost selection, as microclimate is considered an important parameter for

endotherms (Kerth et al., 2001; Gruebler et al., 2014). The selection of a thermally suitable

resting and breeding site reduces thermoregulatory costs. This has implications for individual

fitness, as low roost temperatures during juvenile growth and development stages can result in

long-term impairments to fitness (Kerth et al., 2001; Moussy, 2011). Furthermore, this allows

for the relocation of conserved energy to other important processes. This is particularly true for

certain bat species that require efficient balancing of their energy budget due to their

energetically costly mode of locomotion (Moussy, 2011). Other features that influence roost

selection include proximity to water and food resources (Moussy, 2011), risk of depredation

and competition with other species utilizing similar roosts (Kerth et al., 2001).

Large-spotted genets (*Genetta tigrina*) are semi-arboreal, nocturnal carnivores that use

tree and root hollows as well as rock overhangs for their daylight roosts (Skinner and

Chimimba, 2005). These are also roosts used for the rearing of young, as they provide

protection against inclement weather and reduce the risk of predation. However, large-spotted

genets have been recorded using anthropogenic structures such as roofs, eaves, and

outbuildings as daylight roosts within the urban landscape (Widdows and Downs, 2015). The

use of these anthropogenic roosts isolate the individuals' body from the external climatic

conditions and the resultant decline in thermal conductance increases the roost cavity

temperature (Davis et al., 2014). Furthermore, the presence of geyser tanks (boilers) within

roof spaces will increase the internal roost temperature. It is therefore important to determine

the roost characteristics and associated benefits and risks of anthropogenic roosts for large-

spotted genets.

Hybridization has been recorded within the *Genetta* genus, particularly within South

Africa (Gaubert et al. 2005). *Genetta tigrina* and *G. maculata* form part of the large-spotted

genet complex (Gaubert et al. 2005). The morphological similarities between *G. tigrina* and *G.

maculata* has provided uncertainty regarding the assessment of taxonomic boundaries. This is

particularly true within coastal KZN where hybridization and range overlap have been recorded for *G. tigrina* and *G. maculata*. As a consequence of the uncertainties within the large-spotted genet complex within KZN, we generally referred to large-spotted genets as *G. tigrina* in the current study. This was particularly useful when utilising citizen science as guide books for the area refer to large-spotted genets as *G. tigrina*. Furthermore, residents would have experienced difficulties differentiating between *G. tigrina*, *G. maculata* or hybrids within this complex.

The study investigated roosting dynamics of large-spotted genets in the urban environment of Kloof/Hillcrest, KwaZulu-Natal (KZN), South Africa, using temperature as the response variable. It was predicted that roosts in anthropogenic structures would have higher temperatures than natural roosts in hollows of trees and vegetation. The higher temperatures recorded within the anthropogenic roosts may have thermoregulatory benefits as well as reduced predation risk for *G. tigrina*. It was also predicted that anthropogenic roosts would have more stable temperatures than natural roosts and less seasonal temperature fluctuations.

Materials and methods

Study area

We conducted the study in the suburbs of Kloof/Hillcrest (S -29.781215, E 30.829669) in KZN, South Africa. The mean annual rainfall of the area is 947 mm and the mean maximum and minimum temperature are 24°C and 13.9°C respectively (http://en.climate-data.org/location/27097/). This landscape has several urban conservancies and protected areas such as Springside, Ipithi, Glenholme and Kranzkloof nature reserves within the urban mosaic.

Roost temperatures

Anthropogenic roost locales used by large-spotted genets were identified during a survey conducted between April 2012 and July 2013. A pilot study was conducted in four

anthropogenic roost sites between August and October 2015. Natural roost locales were identified by public sightings of large-spotted genet within urban conservancies and these roost sites were ground truthed to confirm active use by large-spotted genets.

Roost temperatures were recorded with calibrated data logger *i*-Buttons® (Model DS 1922L ± 0.06°C, Dallas Semiconductor, Sunnyvale, CA) at seven large-spotted genet roost sites within the suburbs of Kloof, KZN (Fig. 1). A single *i*-Button® was placed at each roost site and programmed to record ambient temperature at 15 min intervals continuously. Four *i*-Buttons® were placed in roosts in anthropogenic structures (roof spaces) and three *i*-Buttons® in natural roost sites including tree hollows (n = 2) and a rock overhand (n = 1). All four roosts identified in anthropogenic structures occurred in roofs where large-spotted genets were active in the roost sites throughout the study. Genet presence was confirmed by fresh droppings and a midden within the roof space or sightings of large-spotted genets by the home owners (Fig. 2). Of the three natural roost sites, one was in the canopy of a *Kigelia africana* while the second was located in a large hollow in a *Ficus sur* tree. The third natural roost site occurred within a rock outcrop. Upon observation, all natural roosts showed recent signs of large-spotted genets. The *i*-Buttons® were placed in areas of roofs where large spotted-genets appeared most active to record accurate roost site temperatures. Roost temperatures were recorded for a period of one year (August 2014 - July 2015). Upon completion the *i*-Buttons® were removed and the data were downloaded using ColdChain Thermodynamics software (Fairbridge Technologies, Pretoria). Ambient temperature data were recorded by placing an *i*-Button® in a Stevenson's screen (a shelter used to shield meteorological instruments against direct precipitation and heat radiation, while allowing air to circulate around the instrument) in Kloof throughout the study period. This was placed in a central location among the seven roost sites.

Statistical analyses

We performed Analysis of variance (ANOVA) and Repeated Measures analysis of variance (RMANOVA) to determine whether there was a significant difference in temperatures between the seven roost sites. We used Tukey Post-hoc tests to determine among which sites significant differences in temperatures occurred. We calculated minimum and maximum temperature means for each month by averaging daily minima and maxima over the month. All statistics were performed using the Statistica 7 package (Statsoft Inc., Tulsa, OK, USA).

Results

Seasonal variation in roost temperatures were recorded among the four anthropogenic roosts and three natural roosts within the suburbs of Kloof. Roost temperatures were the lowest in June (winter) and the highest during February (summer) (Fig. 3). Maximum recorded roost temperatures differed significantly between months (RMANOVA, $F_{(11, 20)} = 14.60$, $P < 0.05$, Fig. 3) and the different roosts (RMANOVA, $F_{(6, 6)} = 8.06$, $P < 0.05$, Fig. 4). Maximum roost temperatures (highest recorded 44.5°C) were up to 13°C higher than maximum ambient temperatures (highest recorded 29.7°C) recorded for Kloof during all twelve months (Fig. 4). Minimum roost temperatures differed significantly between months (RMANOVA, $F_{(11, 20)} = 26.64$, $P < 0.05$, Fig 3) and between roosts (RMANOVA, $F_{(6, 6)} = 4.57$, $P < 0.05$, Fig. 4). Minimum roost temperatures were generally higher than the minimum ambient temperatures recorded (Fig. 5b). Maximum roost temperatures within anthropogenic roost sites were significantly higher than natural roost sites (Post-hoc Tukey P< 0.05; Fig 5a). Anthropogenic Roost 2 (AR2) had a significantly higher maximum temperature than all other roosts (Post-hoc Tukey, $P < 0.05$). Anthropogenic Roost 4 (AR4) minimum temperatures were significantly higher than all other minimum temperatures (Post-hoc Tukey, $P < 0.05$). It was also noted that

anthropogenic roost temperatures displayed less variability than those recorded within the natural roost sites (Fig. 5). This was particularly evident between May and August.

Discussion

Nocturnal mammals must select daytime roosts that minimise the effects of inclement weather conditions and predation pressure. Daytime roost characteristics and the selection of roost sites are of particular significance during the breeding season (Scobie et al., 2014). Literature focusing on the roosting habits of large-spotted genets are sparse with no studies focusing on roosting habits within an urban landscape. In a study conducted by Du Plessis (1995), *G. tigrina* was found to use wood cavities for daytime resting areas in the Eastern Cape, South Africa. The possible shortage of natural roost sites within an urban landscape could be overcome through behavioural adaptations, allowing genets to use different types of roost sites (Carvalho et al., 2014).

Large-spotted genets have been reported to use anthropogenic structures such as roof spaces and outbuildings as daytime resting sites (Skinner and Chimimba 2005; Widdows and Downs 2015). Habitat destruction as a result of increasing levels of urbanization has resulted in the possible loss of many natural daylight resting sites and the subsequent increase in the use of anthropogenic structures. Furthermore, the fragmentation of existing habitats within the urban landscape may limit access to natural roost sites and have a negative effect on roost site quality. According to Slauson and Zielinski (2009) alternative roost sites are often used in degraded habitats with limited large trees. However, in a variety of sites that border Metropolitan Open Space System (MOSS) areas and conservancies, genets have been recorded using anthropogenic structures even when natural alternatives are available. Similar results were reported within the suburbs of Connecticut, USA, where female raccoons (*Procyon lotor*)

used anthropogenic den sites for rearing young as opposed to suitable natural roost alternatives (O'Donnell and DeNicola, 2006).

The use of roof spaces as buffers from climatic conditions allows for a reduction in thermoregulatory costs, and the protection from predators facilitates improved juvenile care (Kerth et al., 2001). Anthropogenic roost temperatures were generally warmer than natural roost temperatures throughout the year and may be preferred due to the reduced metabolic costs. Furthermore, small-spotted genets (*G. genetta*) have been observed using anthropogenic structures in the suburbs of Johannesburg, South Africa (Rowe-Rowe, 1992; Skinner and Chimimba, 2005). This indicates a trend towards the use of the urban landscape by *Genetta* species living in close proximity to human populations. Roof spaces have a compartmentalised structure that could create a thermal gradient within these spaces. Areas closer to the geyser tank (hot water boiler) may be warmer than areas towards the edge of roof spaces where cooler air enters into the roost environment. This could provide a range of temperatures for *G. tigrina* to move between, according to an individual's energy requirements.

Large-spotted genets breed in the warm months between September and March. However, pregnant females and the presence of juveniles in roof roosts (Fig. 1) were recorded as early as July within the suburbs of Kloof. A possible reason for this apparent early breeding within urban areas may lie in the temperature of anthropogenic roosts. During the breeding season, roof temperatures peaked at 42°C in January and February (ambient temperatures 24°C) while natural roost temperatures peaked at 27°C. These anthropogenic roost temperatures are not conducive to the rearing of youngsters as both adults and juveniles would be at risk of overheating with potentially fatal consequences. However, the advantages of reduced predation risk and increased presence of invertebrates associated with human dwellings could result in a shift in the breeding season into the cooler winter periods. Furthermore, during the winter period roof temperatures were always higher than natural roost temperatures and so would

buffer juveniles from inclement temperatures. These findings are corroborated by Bronson (2009) who states that food availability and ambient temperature determine energy balance which subsequently influences seasonal breeding. Individuals tend to reproduce when both these factors combine to maximise the probability of reproductive success (Bronson, 1985; Bronson, 2009). Australian magpies (*Gymnorhina tibicen*) in suburban Brisbane, Australia were recorded breeding earlier than rural populations due to limited seasonal variation in anthropogenic resources including availability of nesting material (well-watered gardens) and food resources (insect populations attracted to suburban gardens and provision of food by humans) (Rollinson and Jones, 2002).

The use of anthropogenic structures by mammals is not restricted to large-spotted genets. Japanese badgers (*Meles anakuma*) exploit abandoned buildings as secure shelters (Kaneko et al., 2006). Raccoons, brushtail possums (*Trichosurus vulpecula*) and Virginia opossums (*Didelphis virginianus*) also make use of anthropogenic structures including sewers and chimneys as alternative resting and breeding sites (Prange et al., 2004; Wright et al., 2012; Lowry et al., 2013).

Roost microclimate influences roost selection and a variety of mammals preferentially select warmer roosts (Entwistle et al., 1997). In certain chiropterans, maternity roost temperatures are greater than those experienced in day roosts. These higher temperatures reduce energy expenditure; the energy saved can be used to sustain pregnancy and lactation, and facilitate the growth and development of offspring (Moussy, 2011). The elevated temperatures in roof spaces during winter seem to provide similar thermoregulatory benefits for large-spotted genets. During spring and summer temperatures in roof spaces can get extremely high such that individuals could suffer from heat exhaustion. The early onset of pregnancy of urban genets may be a result of elevated temperatures in roof spaces during the spring and summer periods combined with more readily available, high energy food resources

present within the urban environment. Less variability in maximum and minimum anthropogenic roost temperatures were recorded as opposed to natural roost sites. This was particularly evident during the winter months further supporting a shift in the natural breeding cycle and plasticity in reproductive behaviour.

Another feature involved in roost selection is the availability of foraging opportunities and their distances from the roost (Moussy, 2011). For example, roost selection in bats is often influenced by the feeding ecology of the species. Anthropogenic structures are often home to rodents and invertebrates that are closely associated with humans. Energy expenditure may be reduced when individuals can potentially encounter prey items within the roost sites. The ease of access to food resources could further bias the use of anthropogenic structures as opposed to natural roost sites. In a study conducted by Carvalho et al. (2014), common genets (*Genetta genetta*) preferentially selected roost sites in close proximity to riparian habitats as these sites allowed for the use of food resources concentrated within this habitat. Targeting of synanthropic prey items is not restricted to large-spotted genets and has been recorded in numerous urban carnivores such as red foxes (*Vulpes vulpes*) (Contesse et al., 2004), badgers (*Meles meles*) and coyotes (*Canis latrans*) (Morey et al., 2007). The prey items of urban stone martens (*Martes foina*) were dominated by synanthropic rodents and bird species; and were more important in urban populations as opposed to rural populations (Lanszki, 2003). Anthropogenic roosts sites allow juveniles to move around more freely as they are not visible to avian and terrestrial predators; thus allowing adults to forage for longer periods and leaving the juveniles unsupervised.

Further benefits of anthropogenic structures are the protection from predators, especially large raptors such as the crowned eagle (*Stephanoaetus coronatus*). Two of the three anthropogenic roosts were located within the vicinity (0.5 km and 2.1 km respectively) of an active crowned eagle nest and a further eight nest sites have been identified within this

suburban area. Large-spotted genet remains have been recorded in the nests of crowned eagle in Kloof and the greater Durban area (McPherson et al., 2015). As populations of crowned eagle continue to rise within this urban landscape it is predicted that large-spotted genets may become more vulnerable to predation.

However, there are negative consequences for large-spotted genets utilising anthropogenic structures as resting and breeding sites. The use of anthropogenic structures increases the probability of contact and conflict with humans and domestic pets (Widdows unpublished data). In conclusion, this study indicates the importance of anthropogenic structures for both daylight roosts and breeding for large-spotted genets within urban suburbs of Kloof and described the temperatures associated with these roosts. However, other factors such as predation risk and proximity to food resources are likely to affect roost selection by genets. It is important that these variables are the focus of future research.

Large-spotted genets appear to be balancing the benefits associated with the use of anthropogenic roosts (available food resources and reduced predation risk) with the cost of potential breeding failure due to excessive temperatures during the breeding season. To further reduce the costs associated with anthropogenic roosts, females appear to be breeding earlier to avoid excessive roost temperatures and subsequent breeding failure.

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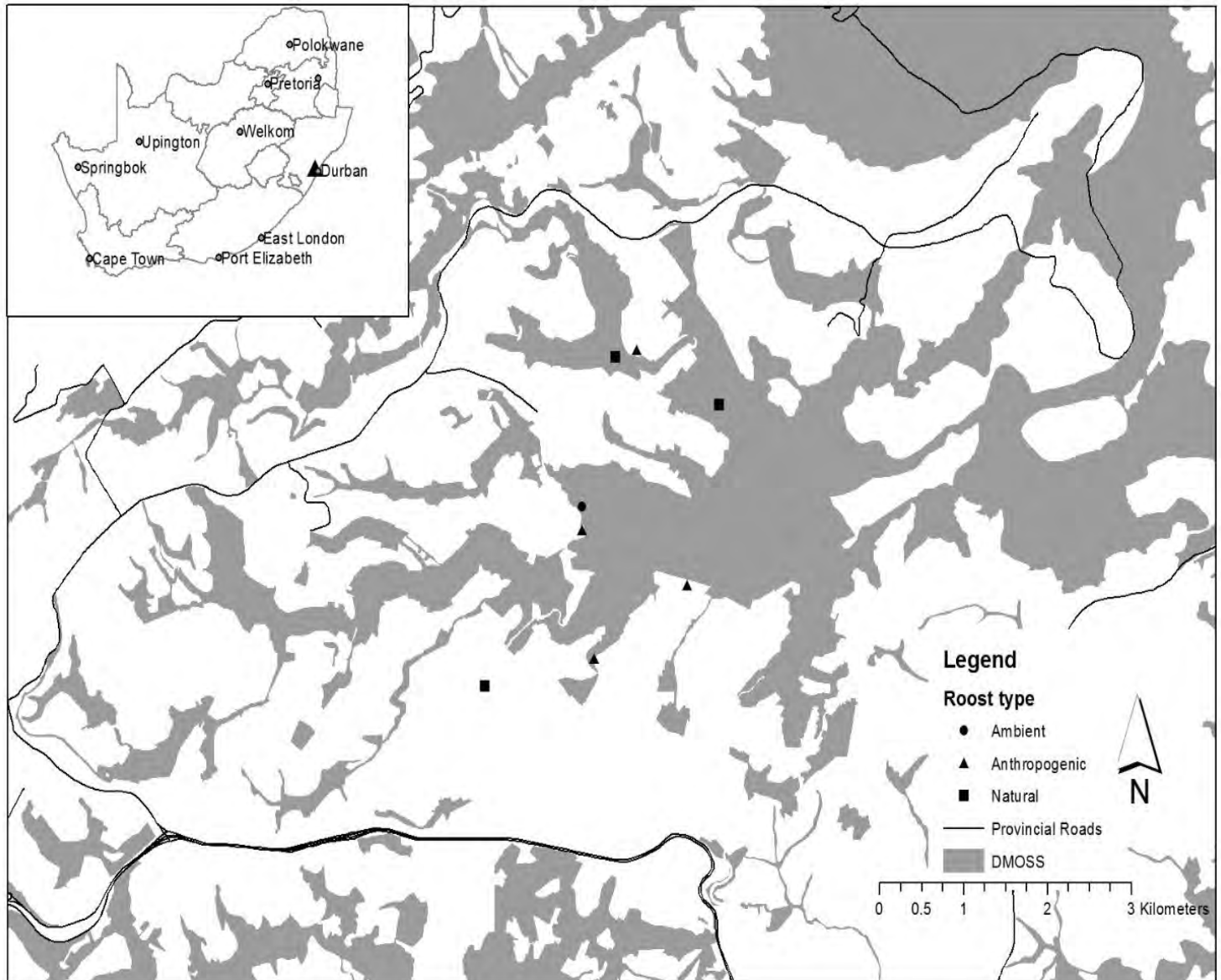
Legends for Figures

Fig. 1. The locations of large-spotted genets (*G. tigrina*) roost sites sampled within the suburbs of Kloof, Durban, South Africa. (DMOSS is Durban Metropolitan Open Space System).

Fig. 2. A female and two juvenile large-spotted genets roosting in a roof in Augusta Country Estate (S -29.789129, E30.783323) Kloof, South Africa.

Fig. 3. Maximum and minimum mean roost and ambient temperatures recorded between August 2014 and July 2015 at large-spotted genet anthropogenic (AR) and natural (NR) roost sites. (Values represent means ± standard deviation).

Fig. 4. Maximum and minimum mean roost and ambient temperatures recorded between August 2014 and July 2015 at large-spotted genet roost sites. (Values represent means ± standard deviation).
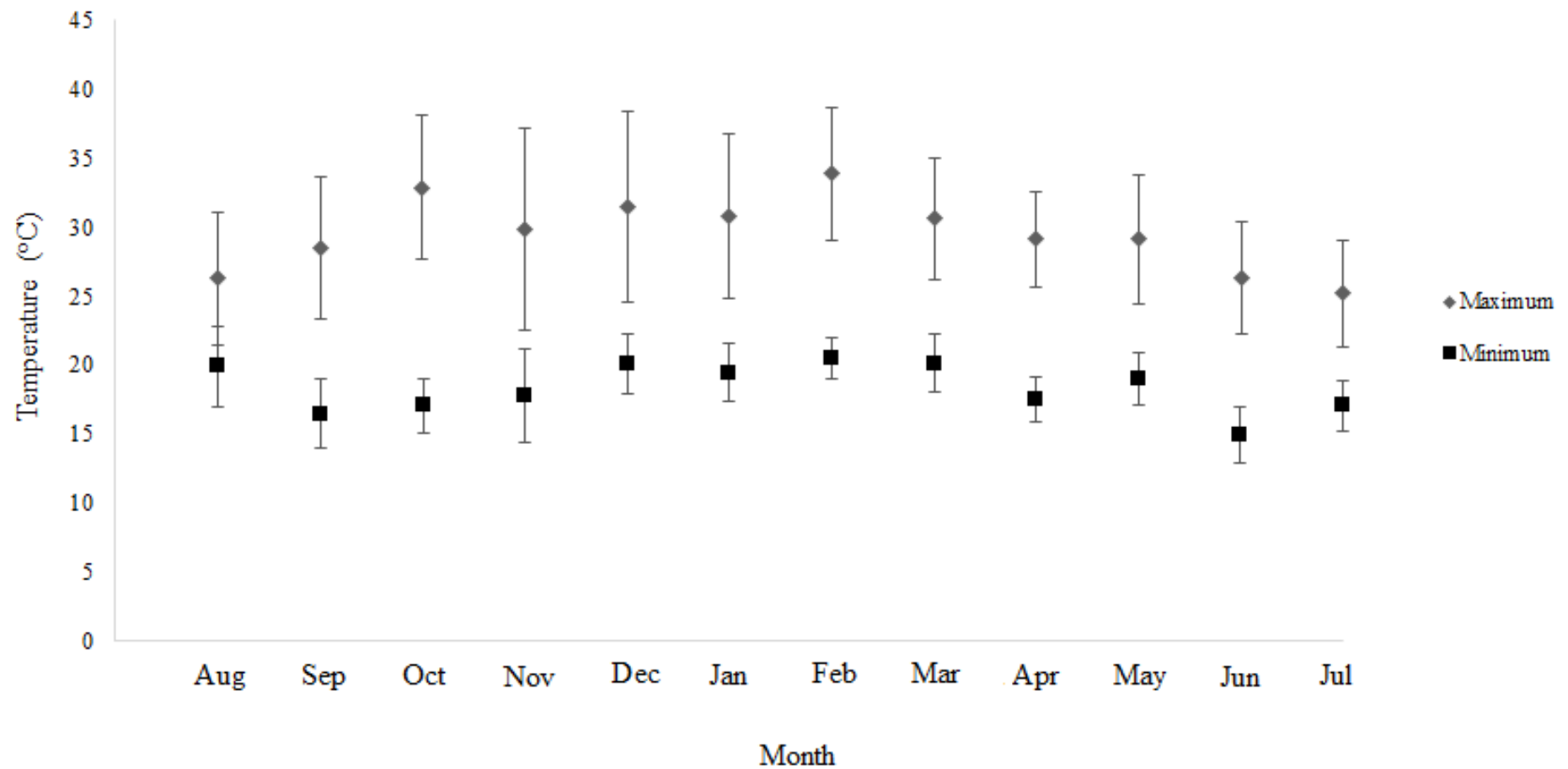
Fig. 5. Monthly maximum (a) and minimum (b) roost temperatures at large-spotted genet anthropogenic and natural roost sites in the suburbs of Kloof, South Africa.

Figures

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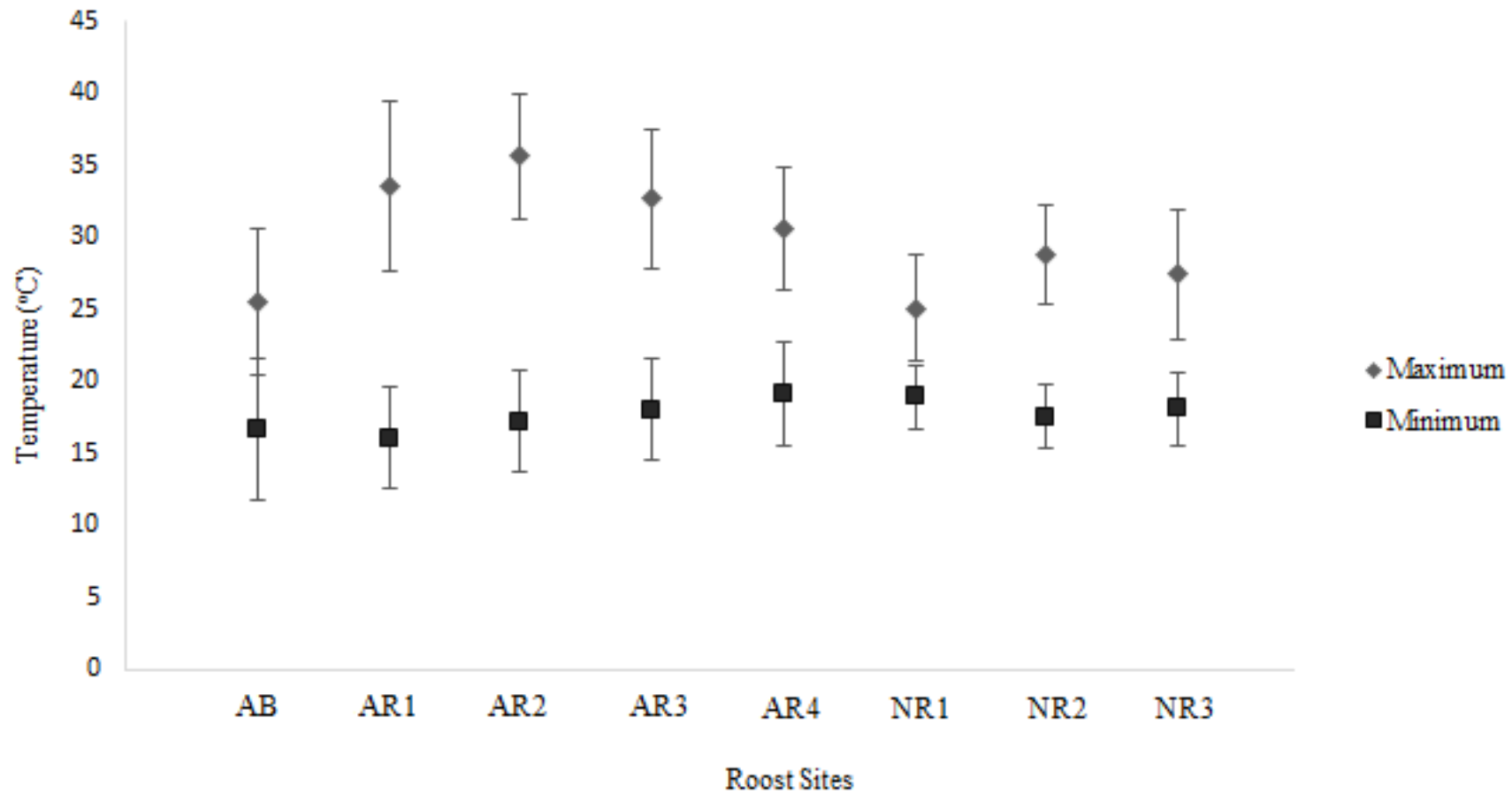
Fig. 2. A female and two juvenile large-spotted genets roosting in a roof in Augusta Country

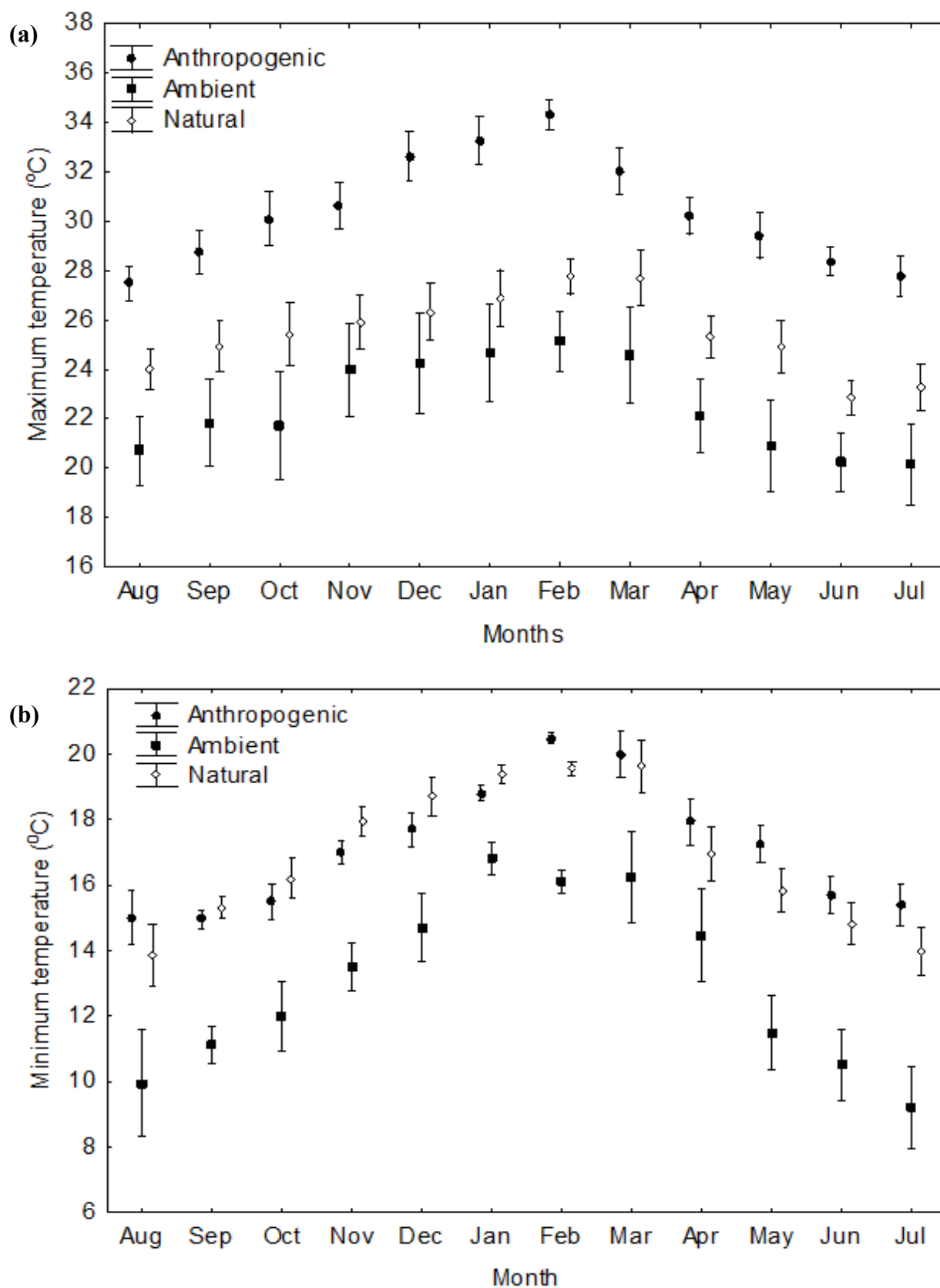Estate (S -29.789129, E30.783323) Kloof, South Africa.

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Fig. 4 Maximum and minimum mean roost and ambient temperatures recorded between August 2014 and July 2015 at large-spotted genet roost sites. (Values represent means ± standard deviation).

Fig. 5. Monthly maximum (a) and minimum (b) roost temperatures at large-spotted genet anthropogenic and natural roost sites in the suburbs of Kloof, South Africa.

CHAPTER 6

Home range and habitat use of large-spotted genets in the urban landscape of Kloof, South Africa

C.D. Widdows, M.B. Pfeiffer and C.T. Downs[*]

School of Life Sciences, University of KwaZulu-Natal, Private Bag X01, Pietermaritzburg, 3209, South Africa

Formatted for African Zoology

[*]Tel: 033 260 5127

Fax: 033 260 5105

[*]Correspondence Email: downs@ukzn.ac.za

Running header: Home range and habitat use of urban genets.

Abstract

Large-spotted genets (*Genetta tigrina*) have a wide distribution across South Africa and have been recorded in urban areas. Given their adaptability and apparent success within the urban environment, knowledge of spatial movements can provide important ecological information. To gain an understanding of their movement ecology within the urban landscape, large-spotted genets were captured and fitted with global positioning system (GPS) mobile transmitters in Kloof, South Africa. Although seven individuals were trapped and five of them equipped with GOS transmitters, detailed data were only obtained for three individuals. Minimum convex polygons (MCPs), 95% and 50% kernel density estimates (KDEs) were calculated for a male

and female large-spotted genet. The 95% kernel density data showed relatively small home range sizes for both individuals, with an area of 48.2 hectares (female) and 17.5 hectares (male). Habitat selection indicated the male large-spotted genet preferred urban residential habitat while the female genet avoided nearby croplands. Both individuals used all the habitats available within their home ranges. Variable habitat used by both individuals in this study confirmed the species' ability to adapt to the urban mosaic. Information on the spatial movement of large-spotted genets provides important information on the species' persistence and success in an urban mosaic environment.

Keywords: Home range, Habitat use, Carnivore, Large-spotted genet.

Introduction

Globally, urbanization has resulted in large scale habitat transformation and fragmentation with further expansion predicted for the next 30 years (Riem *et al*. 2012; Seto *et al*. 2012). Urbanization results in the anthropogenic conversion and altering of natural habitats into commercial, industrial or residential use and generally results in a decline in native biodiversity and ecological function (Bettigole *et al*. 2013). Responses to urbanization vary among species, with some dominating urban landscapes while most avoid these areas (Fischer *et al*. 2012; Newsome *et al*. 2015). Species that have filled this anthropogenic niche generally exhibit significant ecological, behavioural and demographic plasticity (Lowry *et al*. 2013). These adaptations are consistent across carnivore taxa and include reduced dispersal and home ranges (raccoons *Procyon lotor*, Prange *et al*., 2003), increased population densities (coyotes *Canis latrans*, Gehrt *et al*. 2009), and altered diel activity patterns (bobcats *Lynx rufus,* Tigas *et al*. 2002).

Urban areas comprise a mosaic of different green space land use types (residential gardens, golf courses, parks, green belts and conservancies) located within close proximity resulting in the concentration of resources (Baker & Harris 2007). This concentration of urban resources has often resulted in the contraction of home ranges of individuals existing in this anthropogenic matrix in comparison with their rural counterparts (Contesse *et al*. 2004). A reduction in home range within urban landscapes has been recorded in a variety of carnivore species including coyote (Gehrt *et al*. 2009), red fox (*Vulpes vulpes*) (Contesse *et al*. 2004; Iossa *et al*. 2010) and raccoons (Prange *et al*. 2003).

Knowledge of animal home range size, composition and distribution is one of the most fundamental parameters concerning carnivore ecology (Nilsen *et al*. 2005). This provides valuable insight into the influence of environmental factors on individual life histories (Goldingay 2015). A home range can be defined as an area within which an animal moves when performing its normal activities, including foraging, mating, and caring for young (Burt 1943; Powell & Mitchell 2012). The delineation of an individual's home range is vital for conservation as this indicates habitat selection and their relationship with resources and other individuals (Tucker *et al*. 2014). Variation in home range sizes can be attributed to body size, age, sex, foraging habits, food availability, energetic requirements, and competitive interactions between individuals (Kaunda 2001; Wright *et al*. 2012). Therefore, variations in a species' home range in changing environments provides information of the adaptive capacity of the species to the novel environment and subsequently the quality of the habitats with the range (Wright *et al*. 2012). The knowledge of the spatial scales related to carnivore movements is imperative for the implementation of wildlife management and conservation programs (Nilsen *et al*. 2005). This is particularly important for species living within an urban mosaic

landscape due to the greater probability of wildlife conflicts in comparison to rural counterparts.

There have been increasing reports of large-spotted genets (*Genetta tigrina*) within urban landscapes throughout KwaZulu-Natal, South Africa. Large-spotted genets are small nocturnal carnivore and have a broad diet feeding on a wide variety of invertebrates, small rodents, reptiles, birds and wild fruits (Rowe-Rowe 1992; Roberts *et al*. 2007; Widdows & Downs 2015). Large-spotted genets use a variety of anthropogenic resources, including roofs for shelter and anthropogenic food items and have a wide distribution within the urban mosaic landscape (Widdows & Downs 2015; Widdows *et al*. 2015).

An understanding of how carnivores respond to anthropogenic pressures and human-dominated environments is important for the conservation of urban carnivores. Furthermore, this can provide ecological information of movement patterns facilitating species expansion and survival within the urban landscape for the effective implementation of management strategies (Christie *et al*. 2012). The main aim of the study was to determine the home range and habitat use of large-spotted genets within an urban landscape using telemetry. There is indeed a paucity of information on the home range and habitat use of large-spotted genets with no studies focusing on urban areas.

Methods

Study area

The study was conducted in the suburbs of Kloof, KwaZulu-Natal, in the Durban EThekwini Municipality. This area is a mosaic of degrees of urban development interspersed with green areas such as parks, gardens and conservancies (Roberts 1994; eThekwini Municipality 2007).

The mean annual minimum and maximum temperature are 13.9°C and 24°C respectively; the mean annual rainfall is 974 mm (http://en.climate-data.org/location/27097/). Most rainfall occurs in the summer (December-February). Frost is infrequent whereas mist is common and provides additional moisture (Mucina & Rutherford 2006). The natural vegetation in the study area is dominated by KwaZulu-Natal Sandstone Sourveld scattered with low shrubs and patches of riverine forests (Mucina & Rutherford 2006). Most (68%) of this grassland biome has been transformed to forestry plantations, croplands and urban development (Mucina & Rutherford 2006). Some natural vegetation persists within the Durban Metropolitan Open Space System (D'MOSS) which was developed to increase landscape connectivity and improve the function of urban ecosystems (Roberts 1994).

Large-spotted genet captures

The capture, collaring and tracking of large-spotted genets was conducted between March 2014 and June 2015. We captured and collared five individuals, two adult females and three adult males. Two juveniles were released without GPS collars because of their small size. The individuals were trapped using wire mesh cage traps (50 x 50 x 100 cm). Trap sites were chosen in locations where large-spotted genets were regularly seen, determined by public sightings and camera trap evidence.

Cage traps were selected as the capture method because of the simple setup and operation procedure coupled with their success in previous studies with minimal carnivore mortalities (Kamler *et al.* 2003). The cages were designed as a double-door system with a pressure pad trigger plate inside the trap. The cages were camouflaged with indigenous grasses and available branches and the floor of the cage was covered with a hessian cloth to reduce the effects of wire underfoot. Cages were baited with chicken gizzards and some of the blood was

smeared on the cage to reduce human scent. Traps were set in the evening (to avoid the capture of diurnal carnivores such as mongoose species) and were monitored on an hourly basis.

Once a large-spotted genet was trapped, a local veterinarian came on-site to immobilise the individual with a mixture of medetomidine and butorphanol with a top-up of ketamine, based on the animal's weight. Once the individual was fully immobilised, the sex was recorded and morphometric measurements were taken. Measurements included body mass, total length, body length, tail length, and foot dimensions. Large-spotted genets were fitted with GPS-UHF collars (Wireless Wildlife Cc, Pretoria, South Africa) weighing 40 g. Collars recorded the date and time and were programmed to record a GPS fix every 3 hrs between 17h00 and 05h00. Data were stored on the collars and downloaded through a UHF receiver base station that was left in close proximity to the capture site. The base station transferred the data to a remote server through a global system for mobile communication (GSM) where data were saved.

Home range analysis

Data from each collar were downloaded and formatted appropriately in Microsoft Excel, then imported into the open-source statistical software R (R Development Core Team 2012). We conducted a filtering process to confirm the correct UTM, time duplicates and inaccurate speed. The Repeatable Home Range (RHR) package was used to determine time to statistical independence, degree of serial autocorrelation, core delineations and asymptotic functions (Signer & Balkenhol 2015). Spatial manipulations and analyses were performed in ArcGIS 9.3 (ESRI, Redlands, CA, USA) and projected to UTM (WGS 1984 UTM Zone 36S). Home ranges were constructed using both 100% minimum convex polygon (MCP) and 95% kernel density estimates (KDE).

Minimum convex polygons (MCP) were calculated for home ranges as this method has been widely used as a home range estimator. In a study conducted by Goldingay *et al*. (2015),

84% of home range studies identified during the review process made use of MCPs as the primary home range estimator. It is however, acknowledged that there are a number of limitations with this method so kernel density estimates (KDE) were also calculated as they provide more accurate estimates of home range and habitat use (Seaman *et al.* 1999; Börger *et al.* 2006; Laver & Kelly 2008).

The fixed kernel estimate is considered a more robust and least biased estimator of home range due to its better performance with varying sample sizes, shape of utilization distribution, auto-correlated data, and outlier data (Gitzen *et al.* 2006; Jhala *et al.* 2009). Although 95% kernel estimate provides a standard measure of the home range area, we also measured a 50% kernel estimate to provide an indication of core area used (Campioni *et al.* 2013).

Habitat use

We determined the habitat use and availability of land use types within the 100% MCP of large-spotted genets for 3rd-order resource selection using the South African National Land-cover Database to identify habitat types (SANBI 2000).

To determine habitat types used by urban genets, we estimated the potential home range using 100% MCPs to represent the area of habitat available to each individual. The original land use categories present in each home range were reclassified into five habitat types: urban residential, bushland, cultivated land, urban informal and grassland. The area (ha) of each habitat type within the 100% MCP home ranges was determined using ArcGIS 9.3. All locations were overlaid onto the land-use map to provide the observed number of positions within each habitat, using Hawth's Analysis Tools for ArcGIS, version 3.27 (Beyer 2004). Habitat use in proportion to availability, while considering each land use separately, was tested using the Bonferroni *Z*-statistic in Microsoft Excel (Byers *et al.* 1984).

Results

A total of seven genets were captured during the study and morphometric measurements were taken for five of these individuals (Table 1). Two individuals, a male and a female, removed the collars within the first 24 hours after capture. This is likely due to the small head:neck ratio of large-spotted genets. A third GPS collar, attached to a *Genetta maculata*, failed to record any data because of a battery malfunction. However, camera traps positioned within the area were used to collect movement data for this individual as it was easily identified by the presence of the GPS collar (Fig 1). A further two individuals were released without collars as these were both juveniles and were undersized for the collars.

One individual was observed with a metal pellet lodged in the shoulder while two individuals had the last quarter of their tail missing. Furthermore, a male genet was also captured with two broken ribs, highlighting the dangers associated with their urban living.

A female large-spotted genet was captured and collared within a peri-urban landscape in close proximity to the Municipal Road 13 (M13) and the Hillcrest Hospital, Kloof. A total of 196 GPS fixes were recorded over a three-month period. A male large-spotted genet was captured and collared in close proximity to the Kranzkloof Nature Reserve and a total of 97 GPS fixes were recorded over a two-month period. Home ranges were initially calculated using 100% minimum convex polygons (MCP) reflecting the home range of the female 44.59 ha (Fig 2b) and the male 10.41 ha (Fig 2a). The 95% kernel density estimate was 48. 20 ha for the female (Fig 2b) and 17.5 for the male large-spotted genet (Fig 2a). Core-use areas for the female was 27.18 ha and 10.1 ha for the male.

Habitat use

Cultivated lands (sugar cane fields) were avoided by the female large-spotted genet with the other four land use types used in proportion to availability with no statistical significance

recorded (Table 2). Urban residential areas were preferred by the male large-spotted genet and were used more than their availability in the 100% MCP home range (Table 2). Both individuals used the bushland land use type in proportion to availability with no statistical preference recorded.

Discussion

This preliminary study highlights the non-discriminant use of urban land use types by large-spotted genets within this urban landscape. Although the results presented here are limited to only two individuals during a two/three-month period, they provide an insight into large-spotted genet space use within an urban landscape.

The home range size of eight common genets (*Genetta genetta*) within the Coto del Rey National Park, southwestern Spain, were estimated at 541 ha using 95% minimum convex polygons (Palomares & Delibes 1994). The home range of a female *G. genetta* tracked within the Doñ na National Park, Spain, was 141 hectares (Palomares & Delibes 1988). This is significantly larger than the home ranges calculated for urban genets.

The home range of the male large-spotted genet was smaller than that of the female, preferentially using the urban residential habitat. As the home range represents an area traversed by an individual while conducting normal activities of foraging, mating and caring for young, the reduced home range may be explained by the abundance of suitable daytime roost sites and foraging opportunities within the urban landscape. This is particularly true for the male large-spotted genet, as this individual was recorded on a nightly basis at an artificial feeding station within its home range. This individual was observed spending between 1-3 hours at the site feeding and resting during the 12 hr nocturnal active period in close proximity to the feeding station. This artificial feeding station could explain the reduced home range of this individual. Generally, the greater availability of supplementary food reduces the home

range size as the need for extended foraging bouts is reduced (Parker & Nilon 2012). This also explained the preference of the urban residential habitat within Kloof, as there are numerous anthropogenic food resources available to genets within a relatively small area (pers. obs.). Urban badgers (*Meles meles*) in Brighton, United Kingdom, had smaller home ranges than rural counterparts and this reduction is range was attributed to the rich resources available within urban gardens (Davison *et al*. 2009). The apparent relatively small home range sizes of urban large-spotted genets could be an indication of their adaptability to altered ecological circumstances and this has been recorded in a variety of carnivore species (Riem *et al*. 2012). The home ranges of red fox, coyotes and raccoons are smaller in urban environments as opposed to rural habitats (Prange *et al*. 2004; Gehrt *et al*. 2010).

Croplands were used less than expected by the female large-spotted genet. This homogenous habitat may provide limited foraging opportunities in comparison with the urban landscape. Similar results were reported by Palomares & Delibes (1994) as *G. genetta* seldom used croplands within their home range. Both large-spotted genets used a large proportion of available habitats within their home range in the urban mosaic. All three individuals made use of the bushland habitat and natural green areas located within this urban environment. Habitat analysis of urban badgers indicated the use gardens for foraging, while scrub habitats were used primarily for movement within their range (Davison *et al*. 2009). This was evident as the third individual used a band of natural vegetation to move within the urban landscape (Fig. 3).

Two of the individuals trapped during the study period were identified as *Genetta maculata* and form part of the large-spotted genet complex. The overlap in the range of *G. maculata* and *G. tigrina* has been documented within KZN (Gaubert 2003; Gaubert *et al*. 2005). A study conducted by Gaubert *et al*. (2005) identified hybridization between *G. tigrina* and *G. maculata*. Furthermore, *G. maculata* has a varied coloration and spot pattern making

identification, particularly within the hybridizing zone of coastal KZN difficult (Gaubert *et al*. 2005).

Conclusion

The apparent success of large-spotted genets within the urban environment is a consequence of their adaptability to the environment and their variability to changing environmental resources. The reduced home range of the male could be an indication of how large-spotted genets respond to supplementary food resources and anthropogenic structures within the urban environment. Future studies should focus on the movements of *G. tigrina* and *G. maculata* to determine differences in home range sizes as well as potential overlap within these ranges.

Acknowledgements

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Legends for Figures

Fig. 1. Geographic locations of a collared rusty-spotted genet (*Genetta maculata*) within the urban landscape of Kloof, KwaZulu-Natal, South Africa.

Fig. 2. Home range estimates for large-spotted genets (*Genetta tigrina*) showing 50% and 95% kernel density estimates (KDEs) and 100% minimum convex polygons (MCPs) for a male (a) and female (b) respectively.
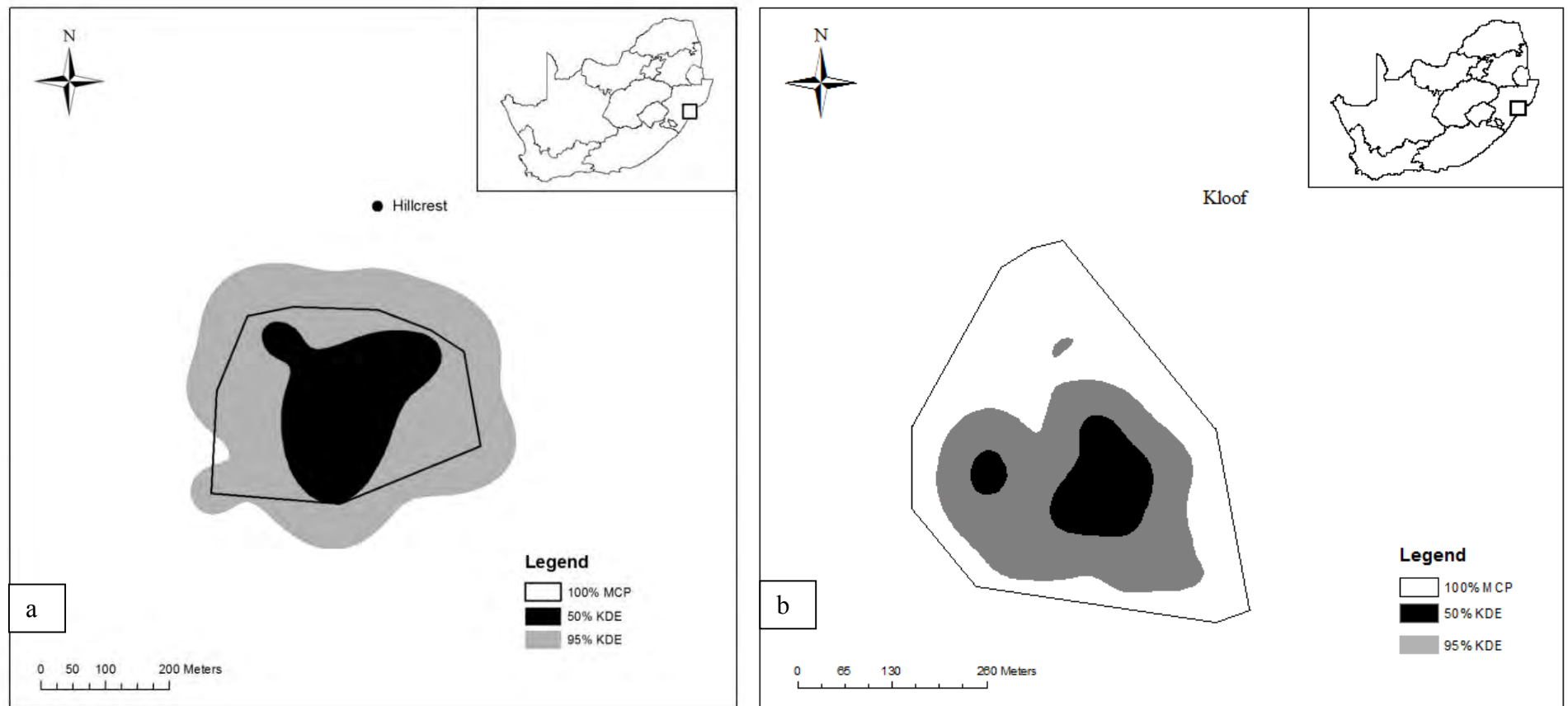
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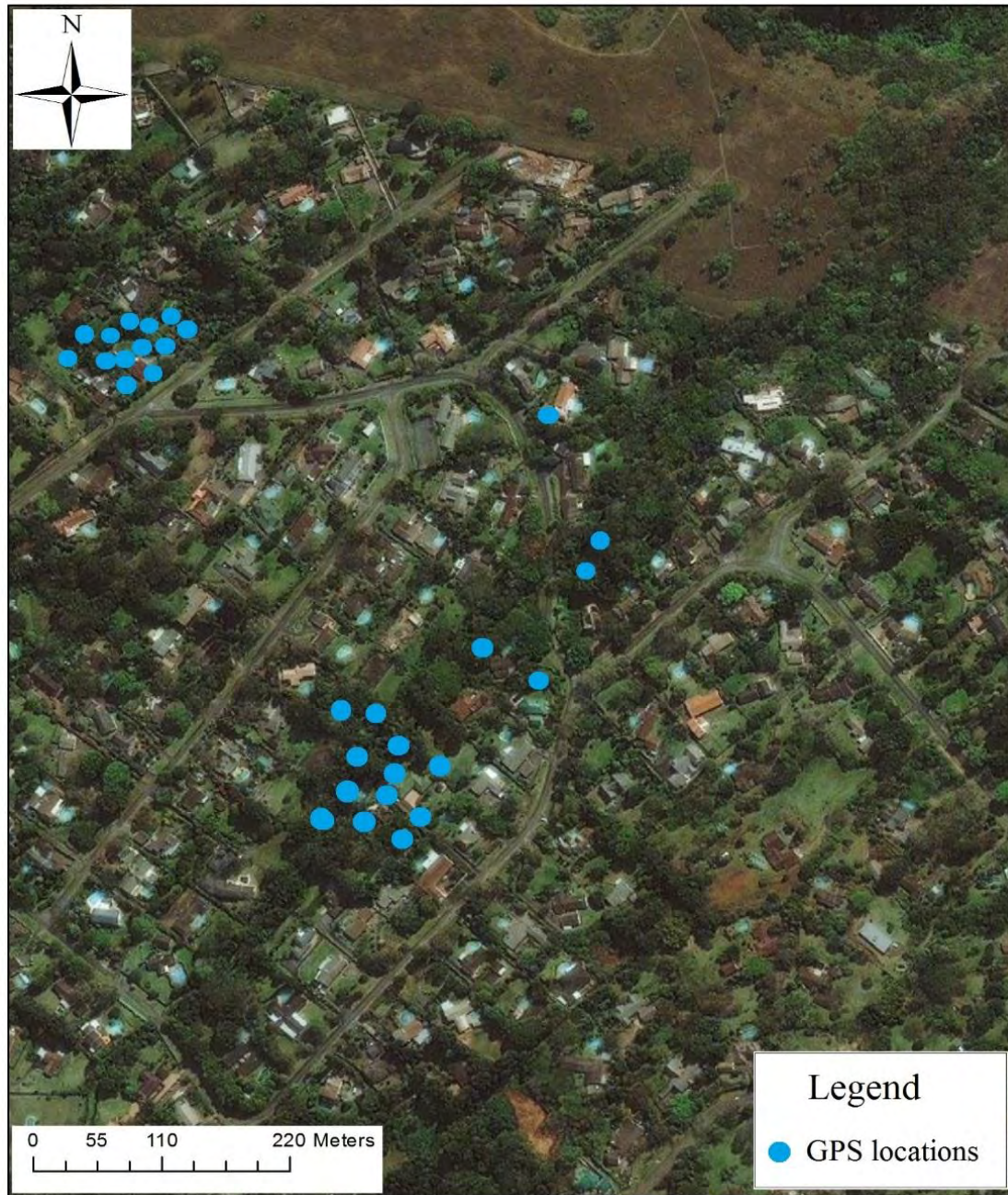
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Tables

Table 1. Morphometric measurements of captured large-spotted genets (*Genetta* sp.) within the suburban Kloof, KwaZulu-Natal.

Species	Sex	Mass (kg)	Total body length (cm)	Tail length (cm)
G. tigrina	Female	2.2	90.5	41.2
G. tigrina	Female	1.82	86.2	38.5
G. maculata	Male	1.96	90.1	43.1
G. tigrina	Male	1.58	70.2	26.9
G. maculata	Male	2.52	89.2	44.5

Table 2. Habitat availability in 100% MCP home ranges based on the reclassified land use map. Bonferroni confidence intervals were used to calculate large spotted genet habitat use.

ID	Habitat type	Area (ha)	P_i	P_{io}	Bonferroni CI	Conclusion
Female	Urban residential	7.20	0.128	0.162	$0.062 < P < 0.204$	Used in proportion
	Bushland	18.37	0.542	0.412	$0.444 < P < 0.722$	Used in proportion
	Cultivated land	4.71	0.028	0.106	$0.005 < P < 0.061*$	Negative selection
	Urban informal	3.17	0.045	0.071	$0.004 < P < 0.087$	Used in proportion
	Grassland	11.31	0.257	0.250	$0.171 < P < 0.369$	Used in proportion
Male	Urban residential	5.01	0.433	0.228	$0.433 < P < 0.259*$	Positive selection
	Bushland	6.67	0.772	0.567	$0.625 < P < 1.181$	Used in proportion
	Cultivated land	-	-	-	-	Not used
	Urban informal	-	-	-	-	Not used
	Grassland	-	-	-	-	Not used

P_i, actual proportion of usage; P_{io}, expected proportion of usage; Bonferroni CI, Bonferroni confidence intervals.

*, a significant difference at $P < 0.05$.

CHAPTER 7

Conclusions

As the human population continues to increase, so too will the conversion of natural landscapes into human-modified environments (McKinney 2002; Magle et al. 2012; Riem et al. 2012). These anthropogenically altered environments come with a new collection of environmental resources and the ability of an animal to successfully exploit these resources will facilitate urban living (Bateman & Fleming 2012). Large-spotted genets are nocturnal omnivores with a wide geographic distribution in Southern Africa that are increasingly reported within urban areas in KwaZulu-Natal (Chapter 2). We investigated the urban ecology of large-spotted genets to determine what factors may facilitate genet populations within an urban mosaic environment.

Through community interviews, we assessed public perception of large-spotted genets, behavioural observations, and wildlife conflict. The majority of respondents expressed positive attitudes towards genets in urban Kloof (Chapter 2); this may be a defining factor influencing their future persistence in urban areas, as human persecution of carnivores is a major cause of carnivore population decline (Poessel et al. 2012; Randa & Yunger 2006). Furthermore, we were able to identify that collisions with vehicles and attacks by domestic animals were the primary causes of genet fatalities (Chapter 2). The application of these findings will enable the implementation of urban management programs.

Environmental aspects influencing the distribution patterns of large-spotted genets are particularly important for populations living in urban areas. We discovered that genet occupancy experienced seasonal influences and was positively associated with winter. The reduction in natural food availability may result in large-spotted genets increasing their foraging bouts and distances in search of alternative food resources in the urban environment. Fringe habitats had a

positive influence on the site occupancy of large-spotted genets and this suggests that the spatial design of this habitat may be structuring the distribution patterns of genets in urban landscapes. Furthermore, bush cover was negatively associated with the detection probability of large-spotted genets (Chapter 3). Wide paths had a negative effect on the site occupancy of large-spotted genets. The avoidance of wide paths could be a strategy to avoid humans and domestic dogs that are more likely to use wider paths as opposed to narrow paths. This study indicated the importance of a variety of landscape factors that influence the distribution and detection probability of large-spotted genets.

We found that the dietary composition of large-spotted genets in an urban environment was very broad, with invertebrates, small mammals, birds, and reptiles forming part of their diet (Chapter 4). Roberts *et al.* (2007), found similar dietary results with the exception of high quantities of cockroaches in the diet of urban genets, which we recorded. This is potentially due to the high abundance of these insects within human-dominated landscapes. Diet varied significantly by season and during the winter months when invertebrates had a low abundance, large-spotted genets shifted to exploit anthropogenic food resources and domestic pet food (Chapter 4). This shows behavioural plasticity and an ability to exploit urban resources when natural resources become scarce. The shift in diet is an adaptation that facilitates the success of genets in the urban environment. Many residents in the area provided supplementary food for large-spotted genets on a regular basis, thus creating artificial feeding stations. Unusual genet behaviour recorded at these artificial feeding stations included diurnal feeding and multiple individuals feeding together with no signs of aggression (Chapter 4). This is unusual behaviour for a nocturnal, solitary carnivore and so indicates high levels of adaptability suited to the urban environment.

Large-spotted genets used anthropogenic structures for both roost sites and rearing young (Chapter 5). These structures provide protection of adults and youngsters from predators and may also provide a high abundance of human-associated alternate prey items. Roost temperatures exceeded 40°C in summer, and individuals that used anthropogenic roosts bred earlier possibly to avoid the excessive heat in the roosts during the summer (Chapter 5).

We found that the home range sizes of urban genets were relatively small ranges for all two individuals that data were obtained. Habitat selection indicated the male large-spotted genet preferred urban residential habitat while the female genet avoided nearby croplands. Both individuals used most of the habitats available within their home ranges. Variable habitat used by both individuals in this study confirmed the species' ability to adapt to the urban mosaic.

According to Lowry et al. (2004), species that are able to exploit the anthropogenic niche available in urban areas exhibit significant ecological and behavioural plasticity; large-spotted genets appear to exhibit this plasticity. The number of public sightings of large-spotted genets within the suburbs of the Ethekwini municipality have steadily increased over the past five years (Figure 1a). This continual increase further supports the ability of large-spotted genets to persist in urban areas. This trend follows our predictive model for successful urban invaders (Figure 1b) (Downs et al. In prep).

Future research should focus on the estimation of the size of urban genet populations within the suburbs of Kloof/Hillcrest. No research has been conducted on large-spotted genet population sizes within an urban landscape. The trend of increasing urban activity within the suburbs of Kloof is not confined to large-spotted genets, as water mongoose (*Atilax paludinosus*), slender mongoose (*Galerella sanguinea*) and porcupine (*Hystrix africaeustralis*) have also been observed within this

urban landscape (Chapter 3). Future research should investigate the factors influencing these species expansion into the suburbs of Kloof /Hillcrest and potential human-wildlife conflict.

This thesis provides valuable insight into the urban ecology of large-spotted genets as well as identifies a variety of factors that influence genet distribution with an urban landscape. The results presented here illustrate the ability of large-spotted genets to successfully exploit a variety of urban resources such as anthropogenic food and shelter.

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Legend for Figures

Figure 1. Frequency of public sightings of large-spotted genets (a) within the eThekwini Municipality, KwaZulu-Natal, South Africa, between March 2012 and August 2015, and (b) as predicted for a successful urban invader.
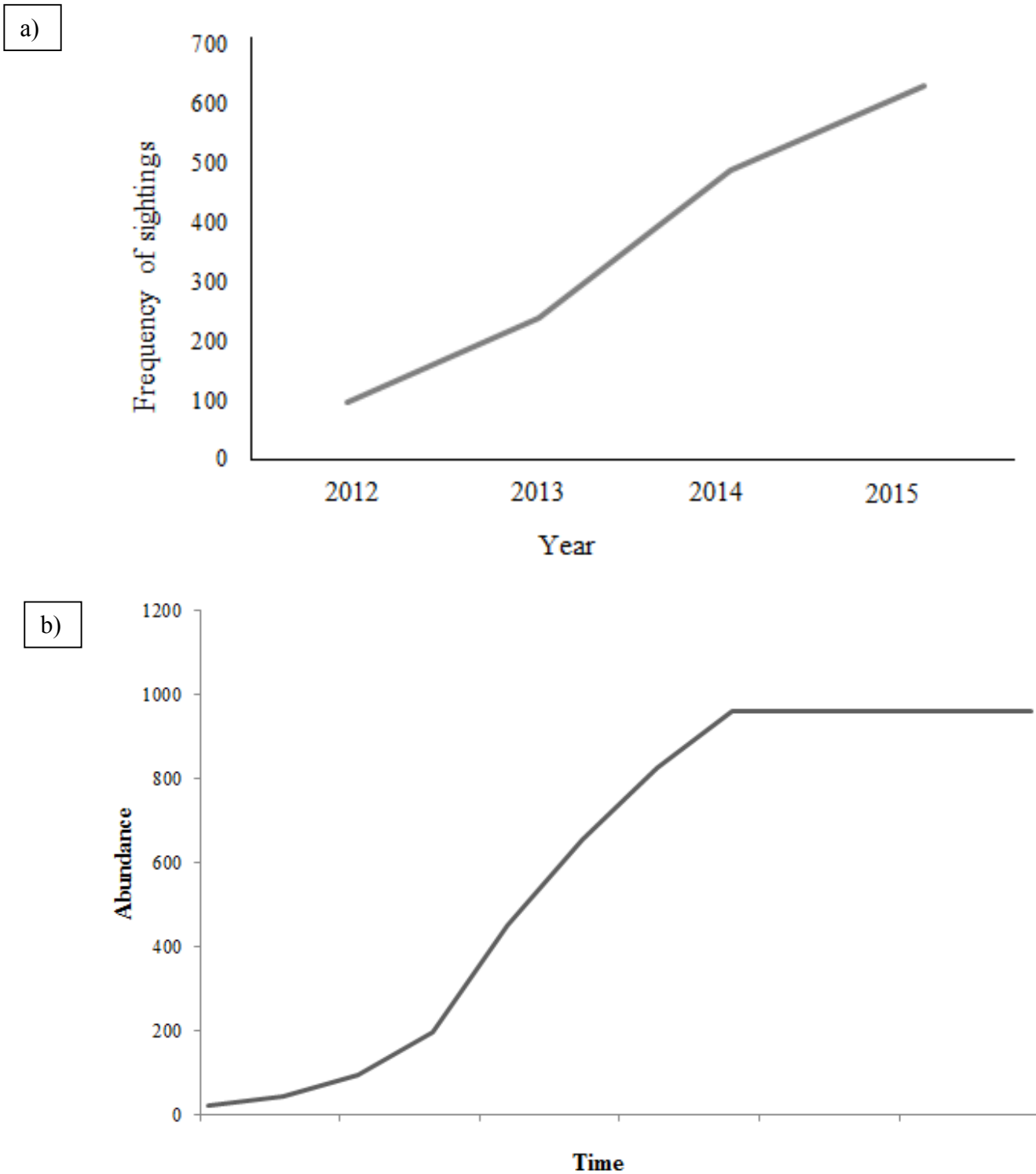
Figures

a)

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