Aspects of the ecology of Trumpeter Hornbill (*Bycanistes bucinator*) across urban-forest mosaics in KwaZulu-Natal, South Africa

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ABSTRACT

There are 57 recognised species of hornbills (Order Bucerotiformes) at present. Of these hornbills, 25 species are either globally threatened or near threatened with extinction. The main factors threatening the extinction of hornbill species are habitat loss, international trade and hunting and anthropogenic climate change. With the current projections indicating that world population will reach 9.3 billion people by 2050, urbanisation is likely to increase and consequently biodiversity loss is likely to escalate in the near future. Thus, understanding how wildlife persists and utilises urban-natural environments are critical for their conservation. This thesis examines aspects of the ecology of the Trumpeter Hornbill (*Bycanistes bucinator*) across the urban-forest mosaics of KwaZulu-Natal (KZN) province, South Africa.

KZN province is unique in that it is part of the Maputaland-Pondoland-Albany hotspot which is one of the recognised earth's biodiversity hotspots. This means that KZN province is biologically rich but is one of the most endangered terrestrial ecoregions. The main land use types in KZN are agricultural, plantation forestry and urban settlements and the province holds one sixth of South Africa's remaining indigenous forests. However, most of these indigenous forests are highly fragmented and isolated due to urbanisation. Despite this, some species of wildlife persist and utilise the urban-forest mosaics of KZN. One such species is the Trumpeter Hornbill. It is the largest obligate frugivore found in South Africa and mainly feeds on fruits a majority of which are *Ficus* spp. The Trumpeter Hornbill has the potential to move between fragmented habitats or forest patches found in an urban environment or agricultural landscapes of KZN. As such, it provides important ecosystem service by transporting seeds of fleshy fruited plants it feeds on across fragmented and isolated forest patches. In order to understand the aspects of the ecology of

Trumpeter Hornbill across the urban-forest mosaics of KZN, a study was designed with the following objectives: 1) to investigate the factors determining the occupancy and detection probability of Trumpeter Hornbills in urban-forest mosaics of KZN using point count data, 2) to understand the home range size and habitat use of Trumpeter Hornbills in an urban-forest mosaic of Eshowe using GPS telemetry data, 3) to understand the movement pattern of Trumpeter Hornbills and assess the fig resources availability in the urban-forest mosaic of Eshowe and 4) to test for site fidelity and estimate the core areas and utilisation distributions using GPS telemetry data collected from the individuals tagged in an urban-forest mosaic of Eshowe.

Results from point count data showed that the average occupancy rate of Trumpeter Hornbills was 0.40 ± 0.09 with a low detection probability of 0.28 ± 0.04. In these urban-forest mosaics of KZN, we found that large trees influenced occupancy positively (sum AIC weight (ω_i) = 79%) while relative human abundance negatively influenced their occupancy (ω_i = 91%). Model selection suggested that housing density had a strong negative influence on detection probability of Trumpeter Hornbills ($\omega_i = 82\%$) and availability of fruiting trees influenced their detection positively ($\omega_i = 29\%$). Results from GPS tracking data indicated that the overall mean monthly home range size was small (mean ± SE), 5.1 ± 1.28 km^2 (95% Minimum Convex Polygon – MCP), 4.6 ± 1.14 km^2 (95% Kernel Density Estimation – KED) and 1.9 ± 0. 46 km^2 (95% Local Convex Hull - LoCoH), with individual variations in monthly and seasonal home range sizes. GPS location data also suggested that Trumpeter Hornbills mainly used the indigenous forest and the urban gardens across the urban-forest mosaic of Eshowe. Further analysis of tracking data revealed that Trumpeter Hornbills tagged (n = 5) showed similar pattern of movement with average daily distances ranging from (mean ± SE) 0.47 ± 0.43 km to 1.06 ± 1.40 km with daily maximum distances of individuals tagged ranging from 2.5 km to 4 km. Only one tagged individual moved to another forest (Entumeni Forest) and covered a maximum daily distance of 7.4 km. Tracking data further established that four of the individuals tagged exhibited site fidelity and that data from two individuals were not statistically independent. The average core area estimated using KDE method (mean ± SE) was 0.62 ± 0.35 km^2 (range: 0.34 km^2 to 1.09 km^2). With LoCoH method, the average core area estimated was 0.07 ± 0.04 km^2 (range: 0.01 km^2 to 0.11 km^2). There was individual variation in the utilisation distribution of the urban-forest environment and the intensity of space use was mainly concentrated in the natural forest and the surrounding urban environment. By sampling the fig resources availability in urban Eshowe, results showed that the most common and abundant figs were *Ficus burkei* (62%) and *Ficus natalensis* (27%) of the seven species identified. The abundance and wide distribution of these fruiting fig trees may be one of the reasons Trumpeter Hornbills persist and use the urban environment of Eshowe.

The results presented show that urban environments characterised by low housing density with relatively low human abundance and at the same time supports healthy natural environments with more large trees and fruiting trees are important for the persistence of Trumpeter hornbills in human-dominated environments. Necessary information with regards to home range size, core areas and habitat use of Trumpeter Hornbills across urban-forest mosaics of Eshowe has also been provided. These results can be used for the present and future conservation and management planning for the Trumpeter Hornbill in the urban-forest mosaics of KZN.

PREFACE

The data described in this thesis were collected in KwaZulu-Natal, Republic of South Africa from August 2013 to March 2015. Experimental work was carried out while registered at the School of Life Sciences, University of KwaZulu-Natal, Pietermaritzburg campus, under the supervision of Prof Colleen T. Downs, and co-supervisors Dr Barry Taylor, Dr Tharmalingam Ramesh, and Prof Mathieu Rouget.

This thesis, submitted for the degree of Doctor of Philosophy in the College of Agriculture, Engineering, and Science, 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.

………………………….
Moses Chibesa
December 2016

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
December 2016

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DECLARATION 1 - PLAGIARISM

I, Moses Chibesa, declare that

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

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

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

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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 (Accepted and in press for Urban Ecosystems**)**

M Chibesa and CT Downs. Factors determining the occupancy of Trumpeter Hornbills in urban-forest mosaics of KwaZulu-Natal, South Africa.

Author contributions:

MC and CTD conceived the paper. MC collected, analysed and wrote the paper. CTD reviewed the manuscript and provided valuable comments

Publication 2 (Accepted and in press for Ostrich - Journal of African Ornithology)

M Chibesa, B Taylor, R Tharmalingam and CT Downs. Home range and habitat use of Trumpeter Hornbills in an urban-forest mosaic, Eshowe, South Africa.

Author contributions:

MC conceived the paper with CTD. MC and BT collected data and RT assisted with analysis. MC wrote the paper. BT, RT and CTD reviewed the manuscript and provided valuable comments

Publication 3 (Formatted for submission to Urban Ecosystems)

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Author contributions:

MC conceived the paper with CTD. MC and BT collected data and RT assisted with analysis. MC wrote the paper. BT, RT and CTD reviewed the manuscript and provided valuable comments

Publication 4 (Formatted for submission to Ostrich - Journal of African Ornithology)

M Chibesa and CT Downs. GPS telemetry of Trumpeter Hornbills: Site fidelity, time to statistical independence, core area and utilisation distribution.

Author contributions:

MC and CTD conceived the paper. MC collected, analysed and wrote the paper. CTD reviewed the manuscript and provided valuable comments

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Moses Chibesa

December 2016

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A Trumpeter Hornbill in a *Ficus lutea* in Eshowe, KwaZulu-Natal, South Africa. (Photo credit M. Chibesa)

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CHAPTER 1

Introduction

1.1 Background

How will hornbills (Order Bucerotiformes) survive in the modern environments characterised by fragmented habitats and carry out the vital role of seed dispersal? In this era of escalating anthropogenic land use change, it is vital to understand the effects of such land use on frugivores as they play a critical role in ecosystem functions. For instance, in some parts of Asia, habitat loss has been considered to be the main cause of hornbill disappearance from many areas of their home range (Pattanavibool and Dearden 2002; Kinnaird and O'Brien 2005; Poonswad et al. 2005). Hunting and exploitation for bushmeat is another cause in the reduction of hornbills in protected areas (Poonswad et al. 2005; Lynam et al. 2006; Trail 2007). In addition, the decline in hornbill species can be attributed to international trade in which African hornbills and their parts are imported into mainly the United States (Trail 2007).

In this chapter, I first review the taxonomy and distribution of Hornbills. Hornbills are only found in the Old World (Africa and Asia). Next, I discuss avian frugivores in general and their role in seed dispersal and later focus on frugivorous hornbills. The disappearance of large frugivores may have serious consequences on seed dispersal and distribution of many tropical forest plants. Then land use change and its impacts on forests and biodiversity is discussed. As world population continues to expand, urbanisation and associated pressures on forest and biodiversity will increase. Thereafter, the study species, Trumpeter Hornbill (*Bycanistes bucinator*) is introduced. The Trumpeter hornbill is a relatively common species along the east coast of South African and it is common in the coastal urban environments of KwaZulu-Natal (KZN). Lastly, the aims, objectives and the outline of the thesis are presented.

1.2 Taxonomy and distribution of hornbills

The taxonomy and biogeography of African and Asian hornbills has been well studied and reviewed (Kemp and Woodcock 1995; Kemp 2001; Viseshakul et al. 2011; Poonswad et al. 2013; Gonzalez et al. 2013). Hornbills belong to the Order Bucerotiformes and consist of two families; Bucorvidae (ground-hornbills) and Bucerotidae (nest-sealing hornbills) (Kemp and Woodcock 1995; Viseshakul et al. 2011; Gonzalez et al. 2013; Poonswad et al. 2013). At present, there are 57 described species of hornbills (15 genera) of which 25 are found in Africa and 32 in Asia (Poonswad et al. 2013). The 25 African hornbills consists of 5 genera (*Bucorvus* – 2 species, *Tockus* – 14 species, *Tropicranus* – 1 species, *Ceratogymna* – 2 species and *Bycanistes* – 6 species) (Poonswad et al. 2013). Their long decurved bill with the casque on top makes it easier to recognise hornbill species in the habitats they are found (Kemp and Woodcock 1995; Pooswad et al. 2013). With the exception of the two species in the Bucorvidae family (*Bucorvus abyssinicus* and *Bucorvus leadbeateri*), the most unique behavioural aspect of hornbills is that of the female sealing herself in the nest cavity and wholly depending on the male for food for most or rest of the nesting period (Poonswad et al. 2013). Hornbills occur throughout sub-Saharan Africa, India, and Southeast Asia to New-Guinea (Figure 1.1) (Kemp and Woodcock 1995; Poonswad et al 2013).

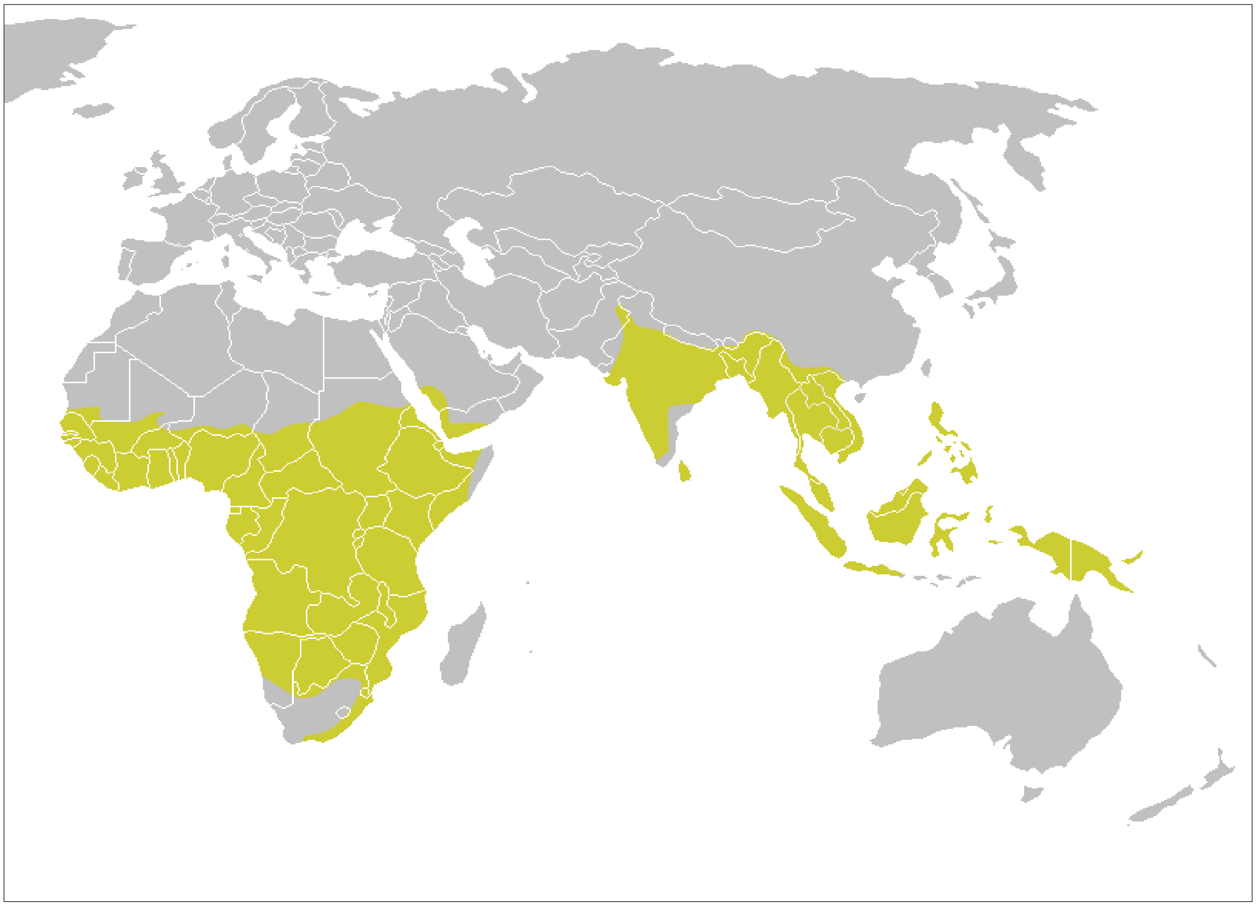

Figure 1.1: Distribution of world hornbills. Hornbills are only found in Africa and Asia.

Source: Poonswad et al. 2013; https://commons.wikimedia.org/wiki/File:Hornbill_distribution_map.png.

1.3 Avian frugivores and their role in seed dispersal

What are frugivores and what is their role in seed dispersal? Animals that eat fruits and play the role of dispersal agents by transporting viable propagules of seeds away from the parent plants are known as frugivores (Howe 1986). In other words, a frugivore can be defined as an animal whose diet is composed of 50% of fleshy fruits (Terborgh 1986). The fruits of most tropical and subtropical tree species are dispersed by frugivorous animals with frugivorous birds being among the main dispersers (Herrera 2002). Seed dispersal is one of the most important mutualistic plant-animal interactions (Howe 1977) with plants relying on animals to disperse their seeds and animals relying on the plants for food in the form of fruits (Bleher 2000). The frugivore communities are influenced by the availability of fruit in an area (Howe and Estabrook 1977; Kitamura 2011) with the frugivores needing to be able to find, eat and subsist either partly or entirely on fruits that are lacking in protein but carbohydrate or lipid rich (Howe and Smallwood 1982).

The vital role that frugivores play in the dispersal of many plant species must be emphasised. For example, in tropical rain forests, birds and mammals disperse more than 80% of woody plants which includes many large seeded plant species (greater than 20 mm in size) that are likely to have less dispersers to rely upon (Howe and Smallwood 1982; Jordano 2000; Tiffney 2004). According to Wang and Smith (2002), understanding the interactions between fruit plants and frugivorous birds has attracted greater attention from evolutionary biologists and field ecologist and are currently the topic of ecological investigations. A review by Fleming and Kress (2011) on the evolutionary history of the mutualistic interaction between angiosperms that produce fleshy fruits and their major consumers (frugivorous birds and mammals) indicates that fleshy fruits eaten by these vertebrates are widely distributed throughout angiosperm phylogeny. However, the dispersal by birds is more common than mammals in all lineages of angiosperm implying that the evolution of bird fruits may have facilitated the evolution of frugivory in primates (Fleming and Kress 2011). As such, frugivores assist in fruit removal, seed rain, seed predation, seed bank dynamics, germination and establishment of dispersed plants (Kollmann 2000). These processes are vital for the determination of composition, spread and persistence of floral populations (Levin et al. 2003; Frohlich and Chase 2007).

In view of this, loss of frugivores and their dispersal services will have a strong negative impact on the ecological and evolutionary dynamics of tropical and subtropical forest communities (Fleming and Kress 2011; Forget et al. 2011; Jordano et al. 2011). Losing large avian frugivorous species with large gape widths will mean plant species with large fruit or seeds that depends on these frugivores for dispersal will be affected (Wheelwright 1985). Many seeds will remain undispersed, falling under the parent plants and largely succumbing to density dependent mortality (Wheelwright 1985; Galetti et al. 2013). By implication, plants that produce large fruits or seeds may be prone to extinction when they lose their natural seed dispersers (Kitamura 2011). It is also important to note that seed dispersal links the end of the reproductive cycle of adult plants with the establishment of their offspring and is generally recognised to have a profound effect on vegetation structure (Wang and Smith 2002).

1.3.1 Frugivorous hornbills and fruiting phenology

Hornbills range in size from small to large (less than 100 g to greater than 3400 g) and majority of the species are the largest avian frugivores in their respective habitats (Poonswad et al 2013). Their vital contribution as long-distance seed dispersers has enabled them to be viewed as important species for sustaining tropical forest ecosystems (Holbrook et al. 2002; Trail 2007). Large frugivores in general are able to handle a wide range of fruit or seed sizes than smaller ones (Wheelwright 1985). As such, large hornbills have played a crucial role in the historical expansion of palaeotropical forests (Viseshakul et al. 2011). In the present environments characterised by fragmented forests that are often separated by a mosaic of farmland, birds have to move long distances to be efficient seed dispersers (Lenz et al. 2011). Being long distance dispersers, hornbills are able to disperse a diverse array of fruits in tropical forests (748 plant species from 252 genera and 79 families) and they are capable of moving many of the seeds far from parent plants (Kitamura 2011). It is for this reason that many ecologists and conservationists consider hornbills to be the main seed dispersal agents for majority of primary forest plants especially large seeded plants (Leighton and Leighton 1984; Becker and Wong 1985; Pannell and Koziol 1987; Kalina 1988; Hamann and Curio 1999; Datta and Rawat 2003; Cordeiro et al. 2004; Kitamura et al. 2004; Sethi and Howe 2009; Kitamura 2011; Lenz et al. 2011; Gonzalez et al. 2013). For instance, African hornbills have been observed to feed on a variety of fruiting trees (Table 1.1) (Kitamura 2011).

In order to determine that hornbills are effective and efficient dispersers of seeds, several benchmarks have been used; such as the size and diversity of fruits ingested, selection of ripe fruit, high fruit consumption, relatively short visitation times and long gut retention times with seeds undamaged after gut passage, fruits swallowed whole with few dropped below parent trees, behavior and movements during and after feeding, and seed deposition at suitable sites for germination (Kitamura 2011; Lenz et al. 2011). The quantity and size of fruits eaten by hornbills vary. A single fruit is sometimes carried by hornbills in the bill tip and transport most fruits in the gular pouch, oesophagus and stomach (Kitamura 2011). The quantity of fruits stored in the expandable gular pouch and oesophagus vary with body size from 100 ml in a 1.2 kg *Anorrhinus* species to 300 ml in a 2.5 kg *Aceros* or *Rhyticero* species that is capable of carrying as much as 500 g of fruit at one moment (Leighton and Leighton 1984; Kinnaird et al. 1996). A study in Uganda by Kalina (1988) showed that *Bycanistes subcylindricus* is able to ingest as many as 200 pea sized fruits or 17 olive-sized fruits for a single delivery to the nest.

Table 1.1: Dietary diversity of African hornbills – A (all seasons), B (breeding season), NB (non-breeding season). * indicates that the complete list of fruits is unavailable and the total number of fruit species is given instead. The information in the table is based on Kitamura (2011).

Hornbill Species	**Study Location**	**Fruit Diversity**			**Season**	**Source**
		Family	**Genus**	**Species**		
Ceratogymna atrata	Cameron	25	41	60	A	(Poulsen *et al*. 2002; Whitney *et al*. 1998)
Bycanistes brevis	Malawi	7	8	15	A	(Dowsett-Lemaire 1988)
	Kenya	10	16	20	A	(Engel 2000)
Bycanistes bucinator	Malawi	6	7	14	A	(Dowsett-Lemaire 1988)
Bycanistes cylindricus	Cameroon	23	35	49	A	(Poulsen *et al*. 2002; Whitney et al. 1998)
Bycanistes fistulator	Cameroon	12	18	23	A	(Poulsen *et al*. 2002)
Bycanistes subcylindricus	Uganda	27	38	46	B	(Kalina 1988)
	Kenya	3	6	8	A	(Flörchinger *et al*. 2010)
Tockus alboterminatus	Kenya	14	16	22	A	(Engel 2000)
Tockus fasciatus	Ivory Coast	12	15	17	NB	(Jensch and Ellenberg 1999)
Tockus fasciatus *Bycanistes fistulator* *Bycanistes albotibialis* *Ceratogymna atrata*	Gabon	22	36	47	A*	(Gautier-Hion *et al*. 1985)
Ceratogymna atrata *Bycanistes cylindricus*	Cameroon	12	N/A	25	B*	(Wang and Smith 2002)
Bycanistes brevis	Tanzania	N/A	N/A	17	B*	(Cordeiro *et al* 2016)

Regarding hornbills feeding behaviour, they mainly feed in the forest canopy and rarely descend to pick fruits from lower vegetation or the ground (Leighton and Leighton 1984; Datta and Rawat 2003; Hadiprakarsa and Kinnaird 2004; Jayasekara et al. 2007; Kitamura et al. 2009). Fruit or seed size, fruit protection, life form (accessibility within the canopy) and fruit ripening to a red, purple or black colour are the important factors for fruit selection by hornbills (Kitamura

2011). Fruits are located by sight and only ripe fruits are selected by their colour (Kitamura et al. 2004). The most preferred fruits plants are the figs as they are high in water, sugar and calcium (Kinnaird and O'Brien 2005). According to Kitamura (2011), the high proportion of hornbill-fruits interaction occurred in trees (77%) followed by lianas (22%) and rarely in shrubs (6%). He further indicated that drupe (35%) followed by berry (38%) and arillate capsular fruit (23%) were the major fruit types eaten by hornbill and black fruits were most common (39%) followed by red (35%) and yellow (19%).

Most phenological studies demonstrate that fruit availability is seasonal (Gordon et al. 1974; Wirminghaus et al. 1999; Kollmann 2000; Wirminghaus et al. 2001; Levin et al. 2003; Bleher 2003; Jordano et al. 2011; Hart et al. 2013; Mulwa et al. 2013) and that there are periods of fruit scarcity and fruit abundance during the different seasons (Terborgh 1986). It is also known that variations in the timing, duration and frequency of fruiting can occur for the different years, habitats and plant species (Opler et al. 1980). The variations in fruit availability result in frugivore communities facing seasonal irregularities in their food resources (Bleher 2000). Some frugivorous birds have been shown to be able to track fruiting plants within forests with hornbills being able to fly distances of up to 290 km in periods of fruit scarcity (Holbrook et al. 2002).

However, there are some tree species that are able to maintain frugivore communities during periods of general fruit scarcity. These tree species are known as 'keystone species' (Leighton and Leighton 1984; Terborgh 1986; Howe 1977; Lambert and Marshall 1991). Studies have identified figs (genus *Ficus*) as being important keystone species (Leighton and Leighton 1984; Terborgh 1986) however, a variety of lipid-rich arillate species have also been identified as playing an important role in the maintenance of the frugivore communities (Leighton and Leighton

1984; Gautier-Hion and Michaloud 1989). The keystone species concept has been defined by Power and Scott Mills (1995) as being ‘a species whose impacts on its community or ecosystem are large and much larger than would be expected from its abundance’. Therefore keystone species are rare, uncommon plant species. (Gautier-Hion and Michaloud 1989). These keystone species are very important not only for maintaining the frugivore communities but also in determining the carrying capacity of these communities (Terborgh 1986).

1.4 Land use change and its impacts on forests and biodiversity

As human population continues to grow, anthropogenic habitats (human modified habitats) are predicted to increase rapidly over the next few decades (Tilman et al. 2001). Agricultural development and intensification, settlements, alien invasion and extraction of natural resources are key land use processes resulting from human activities (Tilman et al. 2001; Rouget and Richardson 2003; Ellis and Ramankutty 2007). As such, the earth’s land cover has changed from mosaic of indigenous ecosystems to an increasing impacted mixture of degraded and fragmented habitats (Ellis 2011). By implication, habitat fragmentation and conversion is the greatest threat to species persistence and conservation of biodiversity (Willig et al. 2007).

Although land use has been generally considered to be a local environmental issue, it is now becoming a force of global importance (Böhning-Gaese 2012). The need to provide food, fibre, water and shelter to an ever growing global population (approximately seven billion people at present) is driving the changes in forests, farmlands, waterways and air (Gibbs et al. 2010; Ellis et al. 2013). Such changes in land use have potentially undermined the capacity of ecosystems to sustain food production, maintain freshwater and forest resources, regulate climate and air quality, and amend infectious diseases (Tilman et al. 2001). The study by Goldewijk (2013) showed that global primary forests have declined 20%, natural grasslands and savannas have declined nearly 40%, whereas croplands have increased 390% and pasture 460% in the last three centuries. These fragmented habitats resulting from anthropogenic land use changes present a serious challenge to the survival, reproduction and ecology of many species of frugivorous birds and other taxa of wildlife. Therefore, understanding how frugivores adapt to fragmented habitats and carry out the vital function of seed dispersal has implications for much of Africa's flora.

According to FAO (2010), forests and other wooded land make up 31 % of the world land cover. Africa has the world's largest proportion of dry forest ecosystem and accounts for 70 to 80% of forested areas (Murphy and Lugo 1986). The term dry forest is used to mean all the deciduous or seasonal forests between the tropical forests and woodlands to the north and south of the equator (Bodart et al. 2013). These dry forests provide numerous benefits to rural communities and society at large (Shackleton et al. 2007). However, they are under threat as Sub-Saharan Africa has been subjected to a series of major disturbances (both natural and man-made) in the previous two and a half decades (Brink and Eva 2009). Factors such as drought, civil disturbances leading to migration, large population increase and globalisation have implications for land use requirements with subsequent impacts on natural vegetation cover, biodiversity, and socio-economic stability (Brink and Eva 2009).

The decline in tropical forests has been attributed to so many factors. The proximate causes are infrastructure extension, agricultural expansion and wood extraction (Geist and Lambin 2002). The underlying causes are multiple and ranges from demographic factors, economic factors, technological factors, policy and institutional factors and cultural factors (Geist and Lambin 2002).

Butchart et al. (2010), reported that most indicators of the state of biodiversity (that includes species' population trends, extinction risk, habitat extent and condition, and community composition) showed declines, whereas indicators of pressures on biodiversity (including resource consumption, invasive alien species, nitrogen pollution, overexploitation, and climate change impacts) showed increases.

These pressures and the proximate causes outlined above have resulted in significant land cover and land use changes in Africa. In the period 1990 and 2000, 3.3 Million hectares (Mha) of dense tree cover, 5.8 Mha of open tree cover and 8.9 Mha of other wooded land were lost and an addition of 3.9 Mha dense tree cover was converted to open tree cover (Bodart et al. 2013). For the same period of time, FAO (2010) reported that 34 Mha of forest was lost in Africa. The demand for agricultural land seems to be the main causal effect of forest decline in Africa. The study by Brink and Eva (2009) revealed that agricultural land increased by 57% during the period 1975 and 2000 at the expense of natural vegetation which had declined by 21% over the same period. Other factors contributing to forest decline in Africa are deforestation and degradation (FAO 2010). A recent study by Céline et al. (2013) reported that between 1990 and 2000 the annual rate of net deforestation and degradation was estimated to be 0.09% and 0.05% respectively. They further stated that between 2000 and 2005, the annual rate of net deforestation and degradation had increased to 0.17% and 0.09% respectively. The direct causes and drivers of this deforestation are population density, small-scale agriculture, fuel wood collection and forest accessibility (Céline et al. 2013).

What is the case like in South Africa which is the study country for this project? South Africa contains the fifth highest number of plant species in the world and the Cape floristic region is recognised as one of the six floral kingdoms of the world with 8200 plant species (Reyers et al. 2001). There are eight biomes in South Africa of which the forest biome is the smallest (Rutherford and Westfall 1986; Mucina and Geldenhuys 2006). KwaZulu-Natal province contains one sixth of South Africa's forests (Low and Rebelo 1996; Mucina and Geldenhuys 2006). Historically, these forest are fragmented due to fires caused by lightning and humans during the late Holocene (Lawes 1990). Another factor responsible for fragmentation of forests is population expansion which has resulted in increased demand for agricultural land and settlements and increased presures on forests resources for provisonning of fuel wood and construction timber (Reyers et al. 2001; Rouget et al. 2003). In addition, exploitation of timber in the past, particularly of straight stemmed *Podocarpus* trees is considered to have negatively affected forest fragments of this afromontane matrix (Wirminghaus et al. 1999; Lawes et al. 2007).

1.5 Trumpeter Hornbill

The Trumpeter Hornbill is the largest obligate frugivore in South Africa with an average body mass of 565 g to 720 g for females and males respectively (Kemp and Woodcock 1995). Males have a larger casque though sexes are alike with regards to the black and white plumage coloration (Kemp 2005). It is found in Sub-Equatorial African which includes north and south-east Angola, southern Democratic Republic of Congo (DRC), Kenya, Zambia, northern Namibia and Botswana and Zimbabwe (Figure 1.2) (Poonswad et al. 2013). In South Africa, it is distributed in the south and east coasts from Alexandria and Knysna Forests, Eastern Cape, and the coastal lowland and montane forests of KZN (Kemp and Woodcock 1995; Poonswad et al. 2013). It feeds mainly on fruits (89%) and is able to swallow small to large sized fruits owing to its large gape width (Kemp and Woodcock 1995). In South Africa, the Trumpeter Hornbill has been observed to feed on the following forest fruit trees; *Ficus* species, *Trichilia* species, *Ekebergia* species, *Rauvofia caffra*, *Berchemia* species, *Xanthocercis zambesiaca*, *Afzelia quanzensis*, *Rhoicissus* species, *Antidesma* species *Monanthotaxis caffra*, *Pterocarpus* species and *Strychnos* species (Kemp and Woodcock 1995). It feeds on a diversity of fruits from at least 14 genera (Poonswad et al. 2013). It has also been noticed to feed on cultivated fruits (*Psidium guajava*, *Litchi chinensis*, *Mangifera indica* and *Carica papaya*) and occasionally on nectar rich flowers including Weeping Boer-bean (*Schotia brachpetala*) (Kemp and Woodcock 1995). Small fruits especially figs form the bulk of its diet (Kemp and Woodcock 1995). It also feeds on animal diet which includes woodlice, millipedes, caterpillars, spiders, birds' eggs and nestlings and crabs (Poonswad et al. 2013).

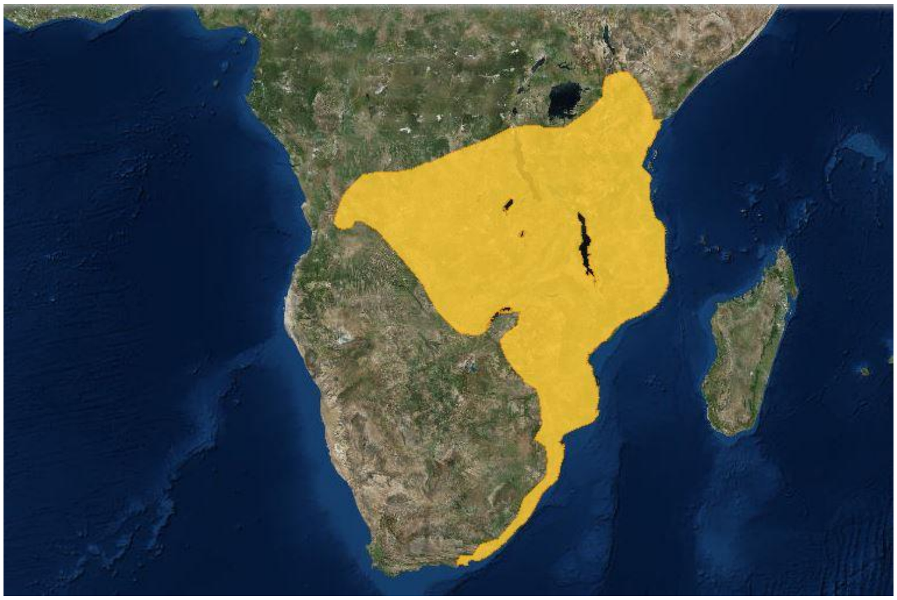

Figure 1.2: Map showing the distribution of the Trumpeter Hornbill in Africa (*Source*: IUCN 2012).

The Trumpeter Hornbill can fly at least 10 km across the savanna between forest patches in search of fruiting trees but it is mostly resident (Poonswad et al 2013). The breeding period in South Africa is from October to January and nesting lasts for at least 94 days (Kemp and Woodcock 1995). During the breeding period, the female encloses itself in the nesting cavity (natural cavity at 2 -13 m up in a tree or rock faces) and solely depends on the male for feeding (Poonswad et al. 2013). Juveniles remain with both parents for approximately 6 months (Kemp and Woodcock 1995). Little is known about its breeding but it is presumed to be monogamous and moves in pairs during the breeding season and forms large flocks consisting of juveniles and adults during the non-breeding season (Kemp and Woodcock 1995). Trumpeter Hornbill is currently classified as a least concern species but is threatened by habitat lose, international trade and possibly hunting (IUCN 2012; Trail 2007).

1.6 Problem statement and significance of the study

Despite the existence and persistence of some avian species in the urban environment of South Africa, there is little urban ecological research conducted in the country (Cilliers and Siebert 2012). Urbanisation transforms and degrades natural habitats forcing animals to live in close proximity to humans (Marzluff et al. 2001; McKinney 2002; Chace and Walsh 2006; Bonier et al. 2007; McKinney 2008). In such degraded environments, many species withdraw into reduced ranges in response to spread of urban environments and anthropogenic climate change (Péron and Altwegg 2015). However, the Trumpeter Hornbill still persist and utilises the urban-forest mosaic of KZN. It is a relatively common forest species along the east coast of South African.

Consequently, understanding how the Trumpeter Hornbill persists and utilises the urban-forest mosaics in KZN is important for its conservation and management. For instance, what factors influence its occupancy and distribution in human dominated environments? What is the home range size of Trumpeter Hornbills in an urban-forest environment? What food resources are key in sustaining the Trumpeter Hornbill in urban environments? Addressing questions such as these will help us understand how the Trumpeter Hornbill persists and utilises the urban-forest environment. As such, this study is important and was designed to contribute to our understanding on the aspects of the ecology of the Trumpeter Hornbill across the urban-forest environment in KZN. We are optimistic that the results presented in this study will be of great help in the management and conservation of this largest obligate frugivore in human dominated environments of KZN.

1.7 Aim and objectives

The main aim of the study was to understand the aspects of the ecology of Trumpeter Hornbill across the urban-forest mosaics of KZN, South Africa. The objectives of the study were: 1) to investigate the factors determining the occupancy and detection probability of Trumpeter Hornbills in urban-forest mosaics of KZN using point count data, 2) to understand the home range size and habitat use of Trumpeter Hornbills in an urban-forest mosaic of Eshowe using GPS telemetry data, 3) to understand the movement pattern of Trumpeter Hornbills and assess the fig resources availability in the urban-forest mosaic of Eshowe and 4) to test for site fidelity and estimate the core areas and utilisation distributions using GPS telemetry data collected from the individuals tagged in an urban-forest mosaic of Eshowe.

1.8 Thesis outline

The thesis consists of six chapters and four of the chapters (2 to 5) are presented as data chapters for submission and ultimately publication in relevant peer reviewed journals. Thus, some repetitions in the chapters was unavoidable especially in the respective method section as the datasets collected for Trumpeter Hornbills in the urban-forest environments of KZN were used to address different questions and objectives. The chapters are outlined as follows:

Chapter 2. Factors determining the occupancy of Trumpeter Hornbills in urban-forest mosaics of KwaZulu-Natal, South Africa.

Chapter 3. Home range and habitat use of Trumpeter Hornbills in an urban-forest mosaic, Eshowe, South Africa.

Chapter 4. Movement pattern of Trumpeter Hornbills and fig resources availability in an urban-forest mosaic, Eshowe, South Africa.

Chapter 5. GPS telemetry of Trumpeter Hornbill: Site fidelity, time to statistical independence, core area and utilisation distribution.

Chapter 6. Conclusion.

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CHAPTER 2

Factors determining the occupancy of Trumpeter Hornbills in urban-forest mosaics of KwaZulu-Natal, South Africa

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Running header: Trumpeter Hornbills in urban – forest mosaics

Abstract

Understanding the factors determining the occupancy and detection probability of birds in human dominated environments is important for their conservation. In this study we investigated various environmental variables believed to influence the site occupancy and detection probability of Trumpeter hornbill (*Bycanistes bucinator*) in urban-forest mosaics of KwaZulu-Natal, South Africa. Presence/absence data were collected from a total of 50 point count stations established between September 2014 and March 2015 in urban-forest mosaics of Durban, Eshowe and Mtunzini. Average occupancy rate of Trumpeter Hornbill was 0.40 ± 0.09 with a low detection probability of 0.28 ± 0.04. For Trumpeter Hornbills, large trees influenced their occupancy positively (sum AIC weight (ω_i) = 79%) while relative human abundance negatively influenced their occupancy (ω_i = 91%). Model selection suggested that housing density had a strong negative influence on detection probability of Trumpeter Hornbills ($\omega_i = 82\%$) and availability of fruiting trees influenced their detection positively ($\omega_i = 29\%$). With continued changing land use in KwaZulu-Natal, these finding are important for conservation of Trumpeter Hornbills as we provide insight into landscape variables or features that influence Trumpeter Hornbill's occupancy and detection in areas of urban-forest mosaics.

Keywords: Point count, Trumpeter Hornbill, Detection probability, Occupancy, Urban-forest mosaic, Land use, Urbanisation

2.1 Introduction

Some of the leading causes of biodiversity lose are climate change, habitat fragmentation due to land use change and illegal international trade in flora and fauna species (Trail 2007; Vačkář et al. 2012; WWF 2016). The current Living Planet Report published by World Wide Fund for Nature in collaboration with Global Footprint Network and Zoological Society of London indicates that global vertebrate population may decline by 67% in the year 2020 as a result of human exploitation of natural resources (WWF 2016). As the world population continues to grow and is projected to reach 9 billion people by 2050 (UN 2015), natural landscapes are greatly being transformed by human encroachment and this has resulted in huge pressure being exerted on the environment (Foley et al. 2005; Péron and Altwegg 2015). These human induced land use patterns result in large-scale transformation of the environment as natural habitats are being converted to agricultural land, settlements, plantation forestry and livestock farming for the sole purpose of providing food, fibre, water and shelter for the growing global population (Foley et al. 2005). Living in these transformed environments, some birds will be favoured at the expense of others as a result of these land use changes (Hockey et al. 2011). For instance, large frugivorous birds such as, the Trumpeter Hornbills (*Bycanistes bucinator*), that persist in anthropogenic environments have the potential to move within fragmented landscapes and able to fly between forest patches (Lenz et al. 2011; Lenz et al. 2015). In addition, the disappearance of indigenous forests has resulted in some forest associated species, for example the Red-necked Spurfowl (*Pternistis afer*), to utilise commercial plantation forests in areas where indigenous forest patches covering a small part of the landscape have been extensively fragmented (Ramesh and Downs 2014). For some species such as, the Crested Guinea-fowl (*Guttera edouardi*), natural forests are important for their survival in landscapes modified for agroforestry (Maseko et al. 2016). These shifts in habitat use by many species of birds is not only a function of land use change but also climate change and that both factors may be acting simultaneously in influencing the dynamic range shifts by South African birds (Hockey et al. 2011). In South Africa, there are few studies on urban ecology (Cilliers and Siebert 2012). As the number of people living in urban areas is expected to increase in many areas globally (McPhearson et al. 2016; UN 2014), understanding the factors influencing the distribution and occupancy of wildlife species that persist and utilise the urban-forest environment is necessary for their management and conservation. This kind of information is lacking for the Trumpeter Hornbill in the urban environments of KwaZulu-Natal (KZN).

Hornbills and parrots are among the world's most threatened group of birds (Marsden and Pilgrim 2003). Among the frugivorous birds of Africa and Asia, hornbills belong to the major seed dispersers of majority of fruiting trees (Kemp and Woodcock 1995; Kinnaird and O'Brien 2007; Kitamura 2011; Poonswad et al. 2013). The Trumpeter Hornbill is the largest obligate frugivore in South Africa and it is relatively common along the east coast of the country (Kemp and Woodcock 1995). Although the Trumpeter Hornbill is considered as "Least conservation concern" by International Union for Conservation of Nature (IUCN 2012), the species is threatened by habitat lose, international trade and possibly hunting (Trail 2007). The impacts of land use change on this forest dependent bird are relatively poorly known. Studies done on this species recently focused on its movement and seed dispersal patterns in fragmented landscapes dominated by agricultural activities in KZN (Mueller et al. 2014; Lenz et al. 2011, 2015). Other studies have highlighted on the aspects of its general biology, ecology, taxonomy and foraging behaviour (Kemp and Woodcock 1995; Viseshakul et al. 2011; Poonswad et al. 2013; Gonzalez et al. 2013). To our knowledge, little is known about the factors influencing the occupancy of Trumpeter Hornbill in urban-forest mosaics of KZN. We employed point count method to collect data on important environmental variables we predicted would influence their occupancy and detection probability in KZN urban-forest mosaics.

The study of bird abundances is commonly achieved by point counts sampling method (Marsden 1999; Diefenbach et al. 2003; Royle and Nichols 2003;Mackenzie and Royle 2005). At a slightly larger spatial scale, the use of a grid of points (spatial replications) without repeated visits or with fewer repeated visits (temporal replications) to study units is another method used to study avian community (Purcell et al. 2005; Sliwinski et al. 2015). Species presence or absence in a particular environment can be used as a surrogate for population size and abundance when monitoring populations (Mackenzie and Royle 2005). Point count survey is considered as a better method for surveying birds and in determining abundance, occupancy and habitat use (Ralph et al. 1995; Royle and Nichols 2003; Diefenbach et al. 2003; MacKenzie and Royle 2005; Purcell et al. 2005; MacKenzie et al. 2006). The method is cost-effective and its use for systematic detection and non-detection survey provides better assessment of the status of a species by detecting changes in their occupancy and probability estimates as a function of covariates (MacKenzie et al. 2006).

Urbanisation transforms and degrades natural habitats forcing animals to live in close proximity to humans (Marzluff et al. 2001; Chace and Walsh 2006; Bonier et al. 2007; McKinney 2002, 2008). In such degraded environments, many species withdraw into reduced ranges in response to spread of urban environments and anthropogenic climate change (Péron and Altwegg 2015). However, the Trumpeter Hornbill still persist and utilises the urban-forest mosaics of KZN. Little is known on how urban landscapes dominated by human activities influence the occupancy and distribution of this largest obligate frugivore. Here we estimated site occupancy and detection probabilities using presence/absence modelling framework (MacKenzie et al. 2002). We used point count data to evaluate Trumpeter Hornbill occupancy as a function of various land use covariates predicted to influence its occupancy and detection probability in the urban-forest mosaics of KZN. Our main objective was to examine the response of Trumpeter Hornbill to varying land use patterns and establish reliable estimates of occupancy and detection probabilities. Based on the species diet mainly consisting of fruits (Kemp and Woodcock 1995), we predicted that the presence of large trees and fruiting trees would positively influence the occupancy and detection probability of Trumpeter Hornbills in an urban-forest mosaic. Sampling points with large trees and with fruiting trees will be preferred as they provide better refuge and foraging opportunities in an urban-forest environment. We further predicted that human abundance and housing density would negatively influence the occupancy and detection probability of the study species.

2.2 Methods

2.2.1 Study area

The study was conducted in urban-forest mosaics of KZN, South Africa. This province is situated on the east coast of South Africa and supports one sixth of the remaining South Africa's indigenous forest which is the smallest biome represented in the country (Eeley et al. 1999; Mucina and Rutherford 2006). The two major forest types, Afromontane forest and Indian Ocean coastal belt forest, which differ in species composition are found in KZN (Mucina and Rutherford 2006). However, these forests have been severely altered by anthropogenic changes and are highly fragmented (Eeley et al. 1999). The urban environment of KZN is dominated by anthropogenic structures (for example, buildings and roads) and natural vegetation is continuously being converted to agricultural land and patches of plantation forests. Fragmented indigenous forests (protected or not protected) that are remaining in the province are unique as they support a significant proportion of the country's diverse flora and fauna species (Eeley et al. 1999). Three towns in the province were selected for this study. These include Kloof-Durban (Site A), Eshowe (Site B) and Mtunzini (Site C) (Figure 2.1). The choice of these towns was based on the fact that each one of them has one or more protected areas (Forests or Nature Reserves) surrounded by human settlements and resultant anthropogenic structures. Such urban or suburban areas provide a perfect scenario for studying the factors influencing the distribution pattern and occupancy of avian species that persist and utilise an urban-forest mosaic.

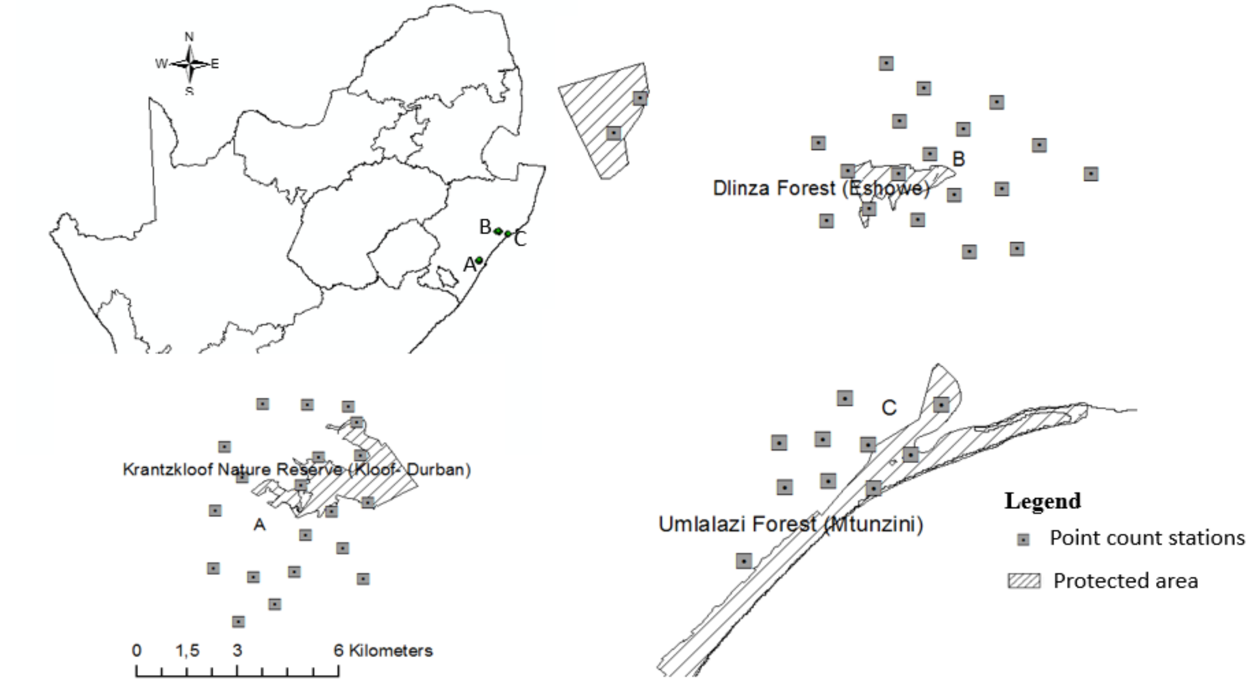

Figure 2.1: Map of the study area with point count stations in urban-forest mosaics of Kloof (20 points), Eshowe (20 points) and Mtunzini (10 points).

The climate of KZN is generally described as warm and temperate and most rainfall occurs in summer (Mucina and Rutherford 2006). Summary information on climate and selected protected areas for the three towns considered in this study is presented together with information on altitude, coordinates and number of point count stations established in each site (Table 2.1).

Table 2. 1: Summary information on climate, altitude, coordinates, protected areas and number of point count stations established in each study site.

Description	**Eshowe**	**Durban(Kloof/Hillcrest)**	**Mtunzini**
Climate or physical component			
Average Annual Rainfall (mm)	1119	974	1104
Average Annual Temp (°C)	19.0	20.9	21.2
Hottest Month (Ave Temp(°C))	February (22.1°C)	February (24.5°C)	January (24.7°C)
Coldest Month (Ave Temp (°C))	June (15.7°C)	July (16.8°C)	July (17.1°C)
Altitude (m ASL)	539	560	92
Co-ordinates	S 28° 53′ 11″ E 31° 28′ 11″	S 29° 47′ 0.24″ E 30° 49′ 59.88″	S 28° 57′ 0″ E 31° 45′ 0″
No. of Point Count Stations Established	20	20	10
Protected Area/Forest			
Name	Dlinza Forest	Kloof Nature Reserve	Umlalazi Forest
Type of Forest	Coastal scarp forest with few glades of grassland	Coastal Forest and Grassland	Dunes and thick coastal forest
Size (ha)	250	600	1028

Source: (http://en.climatedata.org/location/12807/; http://en.climatedata.org/location/772733/; http://en.climatedata.org/location/511/; Mucina and Rutherford 2006).

2.2.2 Data collection and analysis

We established 50 point count stations to record the presence and absence of Trumpeter Hornbills in the three study areas in KZN (Figure 2.1). The points were established systematically by selecting the first point at random and setting the remaining points in relation to the first point with inter point distance of approximately 1 km. The point count stations were established using a hand held GPSMAP 62sc (Garmin International, Kansas, USA). To reliably separate out occupancy from detection (i.e. where the species is versus where the species is found), repeated surveys are required. In view of this, each point count station and survey occasion was treated as independent and presence (1) and absence (0) data were collected by temporal replication by visiting the same point more than once. To avoid heterogeneity in detection probabilities resulting from multiple observers, presence/absence data were collected by a single observer. According to the law of diminishing returns, the number of visits suggested for studies using point counts is between two and five mostly based on forest bird studies (Ralph et al. 1995; Grant et al. 2004; Field et al. 2005; Koper et al. 2009). However, Sliwinski et al. (2015) argue that unless the species or all species in the community have detection probabilities of greater than 0.7, repeated visits of between two to five times may be insufficient sampling effort for detecting species or communities at single points with 90% confidence. They recommended at least seven visits to the same count location to be confident that the species are truly absent if not detected. In view of this, sampling points were each surveyed 10 times between September 2014 and March 2015. Data were collected from 6h00 to 11h00 and 20 min was spent at each sampling point. At each point, important site-specific covariates were also collected within a radius of 30 m. Each point was assessed with regards to the number of fruiting trees available, number of large trees, human abundance and elevation. Fruiting trees were defined as any tree bearing fruits (indigenous, alien or cultivated), and large trees were defined as any tree with diameter at breast height (DBH) of greater than 50 cm and were counted. The number of humans and vehicles counted/10 sampling occasions at each point count station was considered as relative abundance index (RAI) of human activity. By considering the average daily distance covered by Trumpeter Hornbill (about 1 km) in an urban-forest mosaic (Chibesa et al. 2017), we extracted the housing density at each point count station within a 1 km square grid using ArcGIS 9.3.1 (ESRI, Redlands, CA, USA) from the 2005/6 housing mapping map for the eastern region of the country (GeoterraImage 2010). All covariates were standardised to z scores (Cooch and White 2005). Many factors could influence the occupancy and distribution of Trumpeter Hornbills in the environments they are found. In this study, we only considered those factors we thought would influence the occupancy and detection probability of the study species in an urban-forest environment (Table 2.2).

Table 2.2: Covariates used in this study.

Abbreviation	**Explanation**
FAV	Number of fruiting trees available at each sampling point
LTREES	Number of large trees available at each sampling point
HA	Relative abundance index of human
HD	Housing density
ELEV	Elevation

We used a single-season occupancy model to estimate the occupancy (ψ or psi) and detection probability (p) of Trumpeter Hornbill (MacKenzie et al. 2006). For each point count station, we generated detection history of Trumpeter Hornbill consisting of '1' meaning species detected during the sampling occasion and '0' indicating species not detected (Otis et al. 1978). PRESENCE 11.6 (Hines 2006) was used to model site occupancy and detection probability with its covariates. A global model that contained all potential covariates for occupancy was calculated. We then allowed detection probability (p) to vary by all covariates. A two-step procedure was followed, with detection probability (p) modelled first, then occupancy (ψ). Next we allowed the potential covariates for occupancy to vary singly or in combination, whereas detection was maintained either in the global model or kept constant (that is, ψ(covariate), p(covariate) or ψ(covariate),p(.)). For model selection, calculation of model weights and averaging of parameters, we followed the framework of Burnham and Anderson (2002). Using 10,000 parametric bootstrap in the final model, we tested model fit by estimating mean dispersion parameter ($\hat{c}$ or c-hat) (White and Burnham 1999). Models with $\hat{c}$ values of ~ 1 were better descriptors of data and models with $\hat{c} > 1$ indicate that there was more dispersion or variation in observed data than anticipated (Burnham and Anderson 2002). Akaike's information criterion (AIC$\leq$ 2) was used to rank the models (Burnham and Anderson 2002; Hines 2006). Occupancy and detection probability parameters were estimated from the best model that had the lowest AIC and ΔAIC values and high value of Akaike weights (AICwgt or ω_i). The variable strength on occupancy and detection probability was determined by calculating the Akaike weights. To determine the relative importance of each covariate on Trumpeter Hornbill occupancy and detection, model weights were summed over all models containing the particular covariate of interest.

2.3 Results

The estimated site occupancy and detection probability of Trumpeter Hornbill based on the model with all parameters held constant (i.e. psi(.),p(.)) was 0.40 ± 0.07 and 0.28 ± 0.03 respectively. The difference between naive occupancy (0.38) and estimated site occupancy was minimal (Table 2.3). Four of the variables considered were substantially associated with Trumpeter Hornbill occurrence (High sum of AIC weight, Table 2.3). A test of goodness of fit for the global model suggested no lack of fit ($\hat{c} = 1.1$) and the best occurrence model ($\Delta AIC_c = 0$) was ψ(HA+LTREES),*p*(HD) (Table 2.3, highest AIC weight = 0.42) indicated that the variables, number of large trees influenced occupancy positively (β = 0.86 ± 0.45, Table 2.4) and relative human abundance influenced occupancy negatively (β = -1.21 ± 0.68, Table 2.4; Figure 2.2b). In the same model, the detection probability of Trumpeter Hornbill was 0.27 ± 0.04 and it was negatively influenced by housing density (β = -0.39 ± 0.15, Table 2.4). Of the two top ranked models ($\Delta AIC_c \leq 2$, Table 2.3), occupancy for both models was positively influenced by the presence of large trees while relative human abundance influenced occupancy negatively and detection was negatively influenced by housing density (Figure 2.2a, b, 2.3b). In the second ranked model, detection was positively influenced by fruiting trees availability (Figure 2.3a).

The overall summed model weights for the four variables in the top two models with respect to Trumpeter Hornbill occupancy were human abundance (91%) and number of large trees (79%). The influence of elevation on occupancy was negligible (5%). The variables that best predicted Trumpeter Hornbill detection probability across all models were housing density ($\omega_i = 0.82; negatively$) and fruit availability ($\omega_i = 0.29; positively$). The average model occupancy

(0.40 ± 0.09) and detection probability (0.28 ± 0.04) were chosen as final estimates. This corresponded to a difference of 5.2% from naive occupancy.

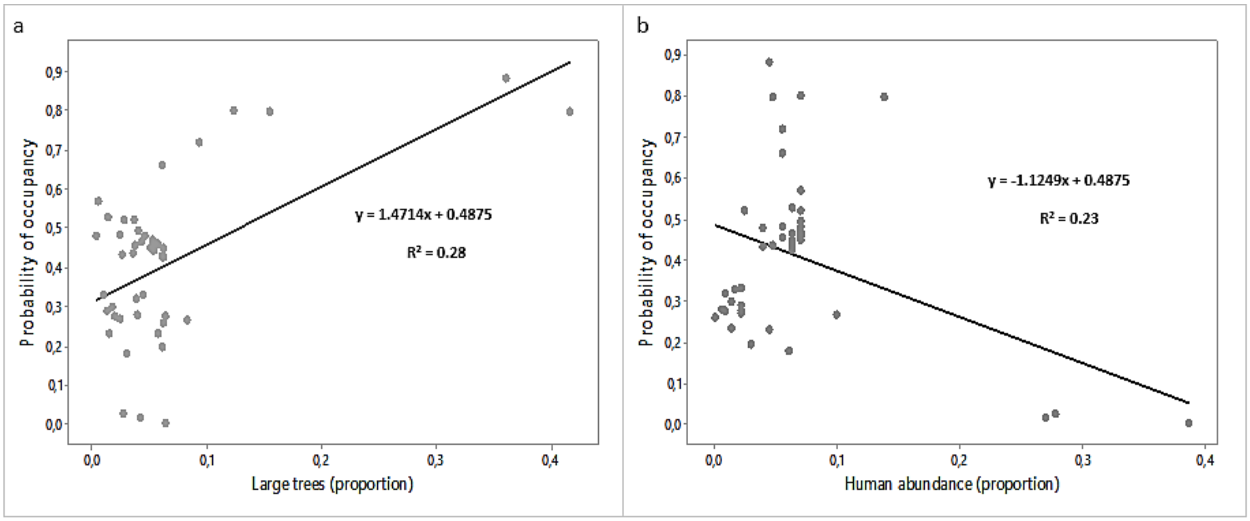

Figure 2.2: Relationship of large trees abundance (a) and relative human abundance (b) with occupancy probability of Trumpeter Hornbill based on top models in urban-forest mosaics of KZN, South Africa.

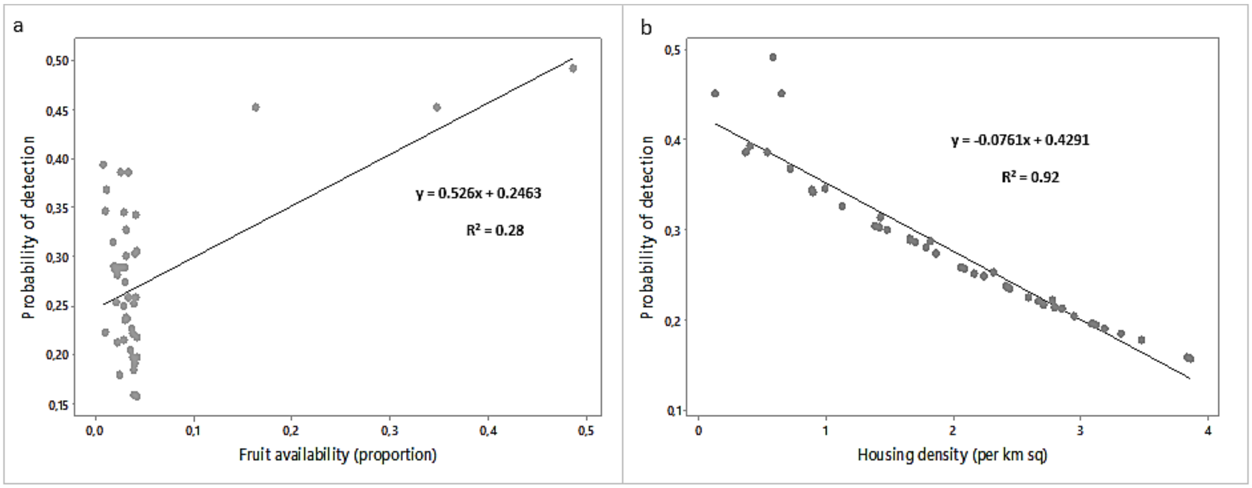

Figure 2.3: Relationship of fruit availability (a) and housing density (b) with detection probability of Trumpeter Hornbill based on top models in urban-forest mosaic of KZN, South Africa.

Table 2.3: Summary of AIC$_c$ model selection and parameter estimates of site occupancy and detection for Trumpeter Hornbill (*Bycanistes bucinator*) in the study sites.

Model	AIC	ΔAIC	AIC wgt	Model likelihood	No. Par.	2LL	ψ ± SE	*p* ± *SE*
psi(HA+LTREES),*p*(HD)	289.00	0.00	0.4171	1.0000	5	279.00	0.40± 0.10	0.27± 0.04
psi(HA+LTREES),*p*(HD+FAV)	290.74	1.74	0.1747	0.4190	6	278.74	0.40± 0.10	0.27± 0.05
psi(HA),*p*(HD)	292.17	3.17	0.0855	0.2049	4	284.17	0.40± 0.09	0.27± 0.04
psi(HA+LTREES),*p*(FAV)	292.67	3.67	0.0666	0.1596	5	282.67	0.40± 0.10	0.27± 0.04
psi(LTREES),*p*(HD)	293.57	4.57	0.0424	0.1018	4	285.57	0.40± 0.09	0.27± 0.04
psi(HA+LTREES),*p*(.)	293.86	4.86	0.0367	0.0880	4	285.86	0.39± 0.10	0.28± 0.03
psi(HA+FAV),*p*(HD)	294.00	5.00	0.0342	0.0821	5	284.00	0.40± 0.10	0.27± 0.04
psi(HA+LTREES),*p*(LTREES+FAV)	294.14	5.14	0.0319	0.0765	6	282.14	0.40± 0.10	0.27± 0.05
psi(HA+ELEV),*p*(HD)	294.16	5.16	0.0316	0.0758	5	284.16	0.40± 0.11	0.27± 0.04
psi(ELEV+LTREES),*p*(HD)	295.57	6.57	0.0156	0.0374	5	285.57	0.40± 0.11	0.27± 0.04
psi(HA+FAV),*p*(HD+FAV)	295.77	6.77	0.0141	0.0339	6	283.77	0.40± 0.11	0.27± 0.05
psi(HA),*p*(HA)	295.93	6.93	0.0130	0.0313	4	287.93	0.40± 0.09	0.32± 0.06
psi(FAV),*p*(HD)	296.19	7.19	0.0115	0.0275	4	288.19	0.40± 0.09	0.27± 0.04
psi(HA),*p*(.)	297.39	8.39	0.0063	0.0151	3	291.39	0.40± 0.09	0.28± 0.03
psi(.),*p*(FAV)	298.39	9.39	0.0038	0.0091	3	292.39	0.40± 0.07	0.28± 0.04
psi(.),*p*(HA)	298.55	9.55	0.0035	0.0084	3	292.55	0.40± 0.07	0.33± 0.06
psi(LTREES),*p*(.)	298.62	9.62	0.0034	0.0081	3	292.62	0.39± 0.09	0.29± 0.03
psi(HA+FAV),*p*(LTREES+FAV)	299.34	10.34	0.0024	0.0057	6	287.34	0.40± 0.10	0.27± 0.05
psi(.),p(.)	299.55	10.55	0.0021	0.0051	2	295.55	0.40± 0.07	0.28± 0.03
psi(LTREES),*p*(LTREES)	299.67	10.67	0.0020	0.0048	4	291.67	0.39± 0.09	0.28± 0.04
psi(FAV),*p*(FAV)	300.29	11.29	0.0015	0.0035	4	292.29	0.40± 0.09	0.29± 0.04

Delta Akaike information criterion (ΔAIC), twice the log likelihood (2LL), number of parameters (No.Par), estimated occupancy (*ψ*), estimated detection probability (*p*) is presented for each model, HD Housing density, FAV number of fruiting trees available, LTREES number of large trees, HA relative abundance of human, ELEV elevation.

Table 2.4: Untransformed parameter estimates for explanatory variables from the best occupancy and detection probability models for Trumpeter Hornbill (*Bycanistes bucinator*) in the study sites.

Model	**Site occupancy**			**Site detection probability**		
	Covariates	Estimates	Standard error	Covariates	Estimates	Standard error
Model 1	Intercept	-0.53	0.38	Intercept	-0.99	0.18
	HA	-1.21	0.68	HD	-0.39	0.15
	LTREES	0.86	0.45			
Model 2	Intercept	-0.53	0.38	Intercept	-0.99	0.18
	HA	-1.22	0.69	HD	-0.35	0.18
	LTREES	0.86	0.45	FAV	0.10	0.19

2.4 Discussion

Our study indicated the importance of various environmental factors that influence the occupancy and detection probability of Trumpeter Hornbills in an urban-forest mosaic. In such modified landscapes dominated by human activities the importance of these covariates is of relevance to Trumpeter Hornbills conservation and for the formulation of management strategies for the persistence of forest dependent species. Trumpeter Hornbills were only detected at 19 of the 50 point count stations (naive occupancy of 0.38). Often Trumpeter Hornbills were only detected in one or two of the repeated surveys, clearly indicating that detection probabilities are less than 1. There conceivably may be a number of points where the Trumpeter Hornbills were indeed present but simply never detected during the survey. Such low detection probabilities observed could be possibly attributed to their movement and flocking patterns as the presence/absence data were collected during the time that encompassed their breeding season. In the breeding period, majority of the females are sealed in their nests and only small groups of 3 to 5 individuals are observed as opposed to non-breeding period when large flocks of up to 100 individuals are observed often at a fruiting tree (Kemp and Woodcock 1995; pers. obs.). In addition, the abundance of large frugivorous hornbills is known to be associated with food availability and some species are also negatively related to habitat disturbance due to lower availability of food resources (Anggraini et al. 2000). This agrees with what we found as the detection probability of Trumpeter Hornbills in the urban-forest mosaics of KZN were positively influenced by availability of fruiting trees and negatively influenced by housing density. Both indigenous, alien and cultivated fruits were available in various urban gardens of KZN thus providing food resources for the Trumpeter Hornbills all year round as they do not all fruit at the same time (Bleher et al 2003). Areas with high housing density tend to have fewer large trees and fruiting trees as most of the natural habitat is cleared for housing development and other anthropogenic structures such as access roads. Although a variety of cultivated fruits and isolated keystone species such as figs which are presumably preferred by frugivorous hornbills (Lambert and Marshall 1991; Kemp and Woodcock 1995; Kitamura 2011; Winarni and Jones 2012) may be found in high housing density areas, hornbills tend to avoid such landscapes as they have fewer or no larger trees in close proximity to a fruiting tree which are important for perching and providing cover when hornbills are disturb from the fruiting tree (pers. obs.). Another possible explanation for the low detection probabilities observed may be attributed to the scarcity of ripe fruits and fruiting trees during the period when presence and absence data were collected. Trumpeter Hornbills selectively feed on ripe fruits

(Kemp and Woodcock 1995) and the peak periods of fruiting trees and ripe fruits in KZN have been reported to be during the end of August to early September and highest peak being between March and May (Bleher et al. 2003). It is highly likely that detection probabilities would have been higher than what we found during these periods of high fruit availability which is also a non-breeding season of the study species when large flocks are observed.

We also found that the pattern of occupancy by Trumpeter Hornbills in urban-forest mosaics of KZN were positively influenced by the presence of large trees and negatively affected by relative human abundance. Large trees provide suitable opportunities for nesting (Kemp and Woodcock 1995; Kinnaird and O'Brien 2007; Poonswad et al. 2005, 2013), although there is little evidence of Trumpeter Hornbills nesting in urban areas in the absence of a nearby protected natural forest, and its large trees within the canopy which may produce the larger fruit crops (Kinnaird and O'Brien 2007). The presence of protected natural forest in urban environments act as roosting, foraging and nesting sites for Trumpeter Hornbills. However, during periods of food resources scarcity, urban environments that maintain a health state of vegetation cover (low human abundance and low housing density) presents a suitable alternative for foraging opportunities and possibly nesting sites for Trumpeter Hornbills. The availability of large trees and the presence of a variety of fruiting trees attract Trumpeter Hornbills to such less modified urban settlements. The negative consequence is that the Trumpeter Hornbills might be dispersing the seeds of alien plants to natural forests by consuming fruits of alien plants from urban environments and transporting them to natural forests within KZN. As such, advising and encouraging people living in urban environment where the Trumpeter Hornbills are a come sighting to plant indigenous fruiting trees in their gardens will be a positive move in trying to halt the proliferation of alien plants in natural forests resulting from alien seeds possibly dispersed by Trumpeter Hornbills from urban gardens. The influence of elevation on occupancy was minimal possibly due to the fact that the difference in elevation for the three sites considered in this study was not significant.

Our occupancy modelling indicated a clear understanding of the factors determining the occupancy and detection probabilities of Trumpeter Hornbills in urban-forest mosaics of KZN. Four important environment covariates influenced occupancy and detection probabilities. Our results indicated that the distribution and occupancy of Trumpeter Hornbills is strongly influenced by the availability of large trees and relative human abundance and that detection is a function of fruit availability and housing density. However, we believe that there could be other factors that might influence the occupancy and detection probabilities of the study species that were not included in this study. Our findings have important conservation implications for managing the Trumpeter Hornbills in urban-forest mosaics of KZN. We provide insight into landscape variables or features that influence Trumpeter Hornbill's occupancy and detection in an urban-forest mosaic. However, further research is required to determine whether this is typical throughout its range and how this varies with season.

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CHAPTER 3

Home range and habitat use of Trumpeter Hornbills in an urban-forest mosaic, Eshowe, South Africa

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Running header: Trumpeter Hornbills in an urban – forest mosaic

Abstract

Despite the negative impacts of urbanisation, some species adapt to pressures of habitat loss and fragmentation. Trumpeter Hornbills *Bycanistes bucinator* are a large avian forest frugivore that uses urban environments in South Africa. Consequently, we used GPS/UHF transmitters to study their home range size, movement and habitat use in an urban-forest mosaic in Eshowe, South Africa from March to October 2014. We estimated the home range size using three methods: Minimum Convex Polygon (MCP), Kernel Density Estimation (KDE) and Local Convex Hull (LoCoH). Our results showed that overall monthly home range size was 5.1 ± 1.28 km^2 (mean ±

SE; 95% MCP), 4.6 ± 1.14 km^2 (95% KDE) and 1.9 ± 0. 46 km^2 (95% LoCoH). However, individual home range sizes varied monthly and seasonally. We found that all individuals tagged used mostly the indigenous forest and frequently utilised urban residential areas (gardens) with little or no use of cultivated land. Observed individual variations in monthly and seasonal home ranges could be a response to variations in availability of key fruit resources in the urban residential and indigenous forest mosaic. This study supports the use of more than one method of home range estimation for insight regarding home range and habitat use in urban-forest mosaics for this large forest frugivore.

Keywords: GPS telemetry, *Bycanistes bucinator,* home range, forest, urban, habitat use, urbanisation, Minimum Convex Polygon (MCP), Kernel Density Estimation (KDE), Local Convex Hull (LoCoH)

3.1 Introduction

Hornbills are highly mobile species and some, for example; Black-casqued Hornbill *Ceratogymna atrata* and White-thighed Hornbill *Bycanistes albotibialis* in the tropical rain forests of Cameroon, have been observed to travel longer distances and are able to cover large areas (Chaser et al. 2014). Hornbills of the genus *Ceratogymna* can undertake large scale movements of up to 290 km (Holbrook et al. 2002) and have been shown to track fruit resources (Whitney and Smith 1998). Studies done in South Africa on the Trumpeter Hornbill *Bycanistes bucinator* shows that it covers much smaller distances than those recorded for *Ceratogymna* hornbills (Lenz et al. 2011; Muller et al. 2014). As such, conservation of highly mobile species presents a challenge owing to the fact that actions implemented at one site are affected by the conditions and actions implemented on other sites that are geographically distant, but still utilised by the species (Runge et al. 2014). For effective management and conservation of highly mobile species in anthropogenic landscapes, there is a need to understand the linkages between sites, habitat use and the scale of movements. In addition, knowledge about the home range sizes of species that have adapted to utilise these human dominated environments is required.

Urbanisation results in habitat fragmentation and dramatically alters the composition and diversity of biotic communities (Bonier et al. 2007). As a result, habitat fragmentation leads to loss of habitat, reduced patch size and an increase in distance between patches (Andren 1994). The presence of species in these habitat patches that dominate the urban environment may be a function of patch size and isolation and also the existence of neighbouring habitats (Andren 1994). In view of this, urban birds that persist and utilise these urban environments dominated by anthropogenic structures and fragmented habitats are considered to have greater environmental tolerance than rural congeners as they exhibit greater behavioural, physiological and ecological flexibility (Bonier et al. 2007). Bonier et al (2007) further showed that urban birds have wider elevation and latitudinal ranges than rural congeners. Ultimately, understanding the space use of avian species in urban environments is important for their conservation.

Throughout their geographical distribution, hornbills have been recognised to play an important ecological role in the different ecosystems they inhabit. Studies in both Asia and Africa have identified large frugivorous hornbills as key species for sustaining tropical forests because of their crucial role in long distance seed dispersal of many fruit bearing plant species (Howe 1984; Bleher and Böhning-Gaese 2001; Holbrook 2002; Kitamura 2011; Lenz et al. 2011; Jordano et al. 2011; Chasar et al. 2014, Naniwadekar et al. 2015) and their ability to facilitate functional connectivity of fragmented landscapes (Mueller et al. 2014). Of the 57 species of hornbills, 25 are either globally threatened or near threatened with global extinction (Poonswad et al. 2013). The major threats identified are habitat loss, hunting, international trade and climate change (Trail 2007; Poonswad et al. 2013; WWF 2014). As such, conservation efforts directed at mitigating these threats are important as the disappearance of hornbill species could negatively affect the future of African and Asian tropical forests (Trail 2007; Kitamura 2011).

The Trumpeter Hornbill is the largest obligate frugivore in South Africa with an average body mass of 565 g (range of 452 – 670 g) to 720 g (range of 607 – 941 g) for females and males respectively (Kemp and Woodcock 1995; Poonswad et al. 2013). Males have a larger casque, although sexes are alike in their black and white plumage coloration (Kemp 2005). In South Africa, the species is distributed in the south and east coasts from Alexandria and Knysna Forests, Eastern Cape, the coastal lowland and montane forests of KwaZulu-Natal (KZN) and the north-east encompassing the Kruger National Park (Kemp 2005). It feeds mainly on fruits (89%) and is able to swallow large sized fruits owing to its large gape width (Kemp and Woodcock 1995; Kemp 2005; Poonswad et al. 2013). However, small fruits, especially figs (*Ficus* spp.), form the bulk of its diet (Kemp and Woodcock 1995; Kemp 2005; Poonswad et al. 2013). It also feeds on woodlice, millipedes, caterpillars, spiders, bird eggs and nestlings and crabs opportunistically (Kemp 2005). The breeding period in South Africa is from October to January and nesting lasts for at least 94 days (Kemp 2005). During breeding, the female encloses itself in the nesting cavity, and is dependent on the male for feeding (Kemp 2005). Juveniles remain with both parents for approximately 6 months (Kemp and Woodcock 1995; Kemp 2005). Little is known about its breeding but it is thought to be monogamous, moving in pairs during the breeding season and forming large flocks consisting of juveniles and adults during the non-breeding season (Kemp 2005). Although the global population has not been estimated, the Trumpeter Hornbill is reported to be locally common and thinly distributed across a wide range (de Hoyo et al. 2001). As such, its conservation status is categorised as Least Concern (LC) based on the current trends of its population which is thought to be stable in the absence of evidence for any decline and significant threats (IUCN 2012).

In this study, we seek to understand the home range size and habitat use of the Trumpeter Hornbill in an urban-forest mosaic. Burt (1943) defined the home range as "the area traversed by an individual during its normal activities of food gathering, mating and caring for its young and does not include occasional sallies". Recently, Powell and Mitchell (2012) defined the home range as "the cognitive map of an animal's environment that it chooses to keep updated" and advises that occasional sallies must be inspected based on what is known about each individual animal and about the species before they are excluded in the estimation of the home range size. The size of the home range usually varies with respect to sex, age and season (Burt 1943; Powell and Mitchell 2012).

Studies that have made use of radiotracking technology e.g. Global Positioning System (GPS) transmitters, have enhanced our understanding of movement ecology, behaviour and home range and habitat use by wild animals in space and time (Cagnacci et al. 2010). As urbanisation continues to expand rapidly (McHale et al. 2013; UN 2014), understanding how a species copes with and adapts to urban pressures is critical for their management and conservation. In South Africa, a few studies have been conducted on Trumpeter Hornbills using telemetry, with the focus on seed dispersal, frugivory, movement behaviour and functional connectivity in fragmented landscapes largely dominated by agricultural activities (Lenz et al. 2011, 2015; Mueller et al. 2014). Here, we used GPS/UHF transmitters to study the diurnal foraging movements of Trumpeter Hornbills in the urban-forest mosaic of Eshowe, KZN. Our aim was to understand their home range, habitat use and movement in this urban-forest mosaic. The objectives were; 1) to estimate their mean distances moved monthly and seasonally, 2) to identify their key habitats used in the urban-forest mosaic, and 3) to estimate their monthly and seasonal home range size. Since Trumpeter Hornbills feed mainly on fruits (Kemp and Woodcock 1995), and the availability of fruiting trees vary in space and time (Wirminghaus et al. 2001), we predicted that there will be monthly and seasonal variation in home range and distances covered by individuals.

3.2 Methods

3.2.1 Study area

The study was conducted in the municipality of Eshowe (28.89444° S and 31.44889° E) in KZN, South Africa (Figure 3.1), which includes the indigenous protected Dlinza Forest (250 ha, protected since 1947; O'Reagain 2001). Some of the tree species found in the forest include: Giant Umzimbeet *Millettia sutherlandii* (which dominates), Wild Plum *Harpephyllum caffrum*, Fluted Milkwood *Chrysophyllum viridifolium*, Natal Forest Cabbage *Cussonia sphaerocephala*, Forest Iron Plum *Drypetes gerrardii*, Natal Milk Plum *Englerophytum natalensis* and Common Wild Fig *Ficus burkei*.

3.2.2 Capturing, tagging and tracking

In March 2014 we used standard mist nets placed under a fruiting tree (*Ficus lutea*) to capture Trumpeter Hornbills in Eshowe. The birds were weighed, measured and ringed before being released at the capture location. GPS/UHF transmitters (Wireless Wildlife, Potchefstroom, South Africa) weighing 28 g each were used and we adopted the criterion that the weight of the transmitter should not exceed 3-5% of the body mass of the bird (Kenward 2001). Of the nine hornbills captured, five attained this criterion and were fitted with transmitters. The transmitters were attached as a 'backpack' using Teflon straps looped under the wings and with predetermined breakage points to facilitate their dropping off at a later stage.

The transmitters were programmed to record location data every 4 hours starting from 6h00 and ending at 18h00 resulting in 4 GPS fixes per day for up to a year. At night, transmitters switched to sleep mode to preserve battery life. Data was stored on board memory within the transmitter and downloaded when the tagged individuals were within a radius of 10 km of the solar-powered base station. Tracking of each individual started on the first day it was captured, tagged and released. We only managed to download data for the period from March to October 2014. Thereafter, the strength of the batteries became too weak to download further location data. We were however able to check the movement pattern of the birds until May 2015 when the batteries completely ceased operating.

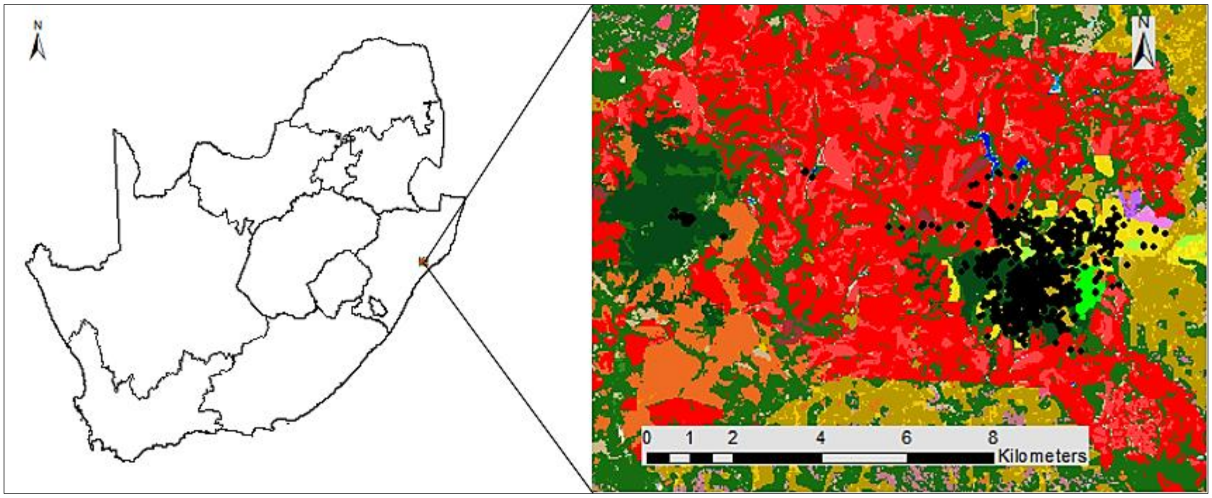

Figure 3.1: Location of the study area (Eshowe, KwaZulu-Natal, South Africa) showing the main land use types (dark green- indigenous forest, light green -thickets, red- cultivated cane commercial, pink- cultivated subsistence, yellow- urban, blue water body, mustard- communal villages and lands) that characterise the area and the distribution of GPS fixes (black dots) for the five Trumpeter Hornbills that were tagged.

3.2.3 Data analysis

GPS fixes for the five individuals tagged were imported into ArcGIS 9.3.1 (Geographic Information System; Environmental Systems Research Institute, Redland, California) and projected into Universal Transverse Mercator (UTM) projection, WGS 1984, UTM Zone 35s. Duplicate GPS fixes were removed using the spatial analysis tool in ArcGIS and each layer was then overlaid onto the 2014 land cover map for KZN (Ezemvelo KZN Wildlife 2014).

We assessed the habitat use and preference by qualitative description based on the number of GPS fixes falling in each habitat type. Thomas and Taylor (2006) states that the term use may mean the number or weight of food consumed or the time spent, distance travelled or the number of locations in a habitat type. First, we estimated the 100% Minimum Convex Polygon (MCP) for each individual using Home Range Tools (HRT) (Rodgers et al. 2007). This was done in order to define the boundaries within which the GPS fixes were found for each individual. Secondly, we identified six habitat types based on the GPS fixes falling within the 100% MCP for each individual. Finally, we calculated the proportion of GPS fixes falling in each habitat type for each individual and used this as a proximate measure of habitat use.

Home range estimation is the most useful way of analysing radio-tracking data (Signer and Balkenhol 2015) and can be achieved with the use of different methods. The various methods available have different limitations and advantages, and there are several software packages. (Signer and Balkenhol 2015). MCP and KDE are the oldest and widely used methods (Laver and Kelly 2008; Kie et al. 2010; Cumming and Cornélis 2012) whereas the LoCoH is a more recent nonparametric kernel method (Getz et al. 2007) that generalises the MCP method and is considered to be more appropriate than parametric kernel methods for constructing home ranges and utilisation distributions because of its ability to identify hard boundaries (Getz et al. 2007). Thus, the choice of which method and software to use still presents a challenge to researchers.

We used the reproducible home range (rhr) package (Signer and Balkenhol 2015) in programme R, version 3.2.3 (R Development Core Team 2015) to comprehensively estimate the monthly and seasonal home range size for the four individuals that gave us enough data for eight months. Data from TH 3 was insufficient (35 GPS fixes for 10 days) for home range estimation and was excluded from the analysis. We estimated the home range using the three methods: MCP, KDE and LoCoH (Worton 1989; Getz et al. 2007). For the KDE method, we used the least-squares cross-validation (LSCV) as a default bandwidth selection method since it performs better with distribution types where tight clumps are identified (Gitzen et al. 2006; Signer and Balkenhol 2015), which was the case with our data set. Seasonal home ranges of each individual were estimated for autumn (1 March – 31 May), winter (1 June – 31 August), and spring (1 September – 30 November).

Comparisons of monthly and seasonal distances covered by the tagged individuals were conducted using the Analysis of Variance (ANOVA). Where significant differences were detected, a Tukey post-hoc test was performed. Since the home range estimates were not normally distributed, Kruskal-Wallis ANOVA was performed to see if there was a difference in the monthly and seasonal home range sizes and the three estimation methods used. These statistical analyses were performed using STATISTICA 10 (Stat-soft Inc., Tulsa, USA).

3.3 Results

3.3.1 Tagged individuals

Five individuals were tagged (1 male, 4 females), with the average body mass of these being 671.2 g (range 600 - 811 g) (Table 3.1). After filtering the data, a total of 3461 GPS fixes were obtained (range 35 - 860 GPS fixes per individual) with a sampling duration of between 10 - 226 days (Table 3.1). We could not determine whether TH 3 died or the transmitter dropped-off as all the GPS fixes after the 10th day from when the individual was tagged were clustered at one point. Several attempts to find the transmitter and/ the bird failed.

2.3.2 Habitat use

The 100% MCP estimated for each individual were 4.9 km^2 (TH 1), 12.8 km^2 (TH 2), 12.8 km^2 (TH 3), 5.0 km^2 (TH 4) and 4.8 km^2 (TH 5) (Figure S2).We identified six habitat types that were utilised by Trumpeter Hornbills within the overall 100% MCP home range of each individual (Figure 3.2). Based on the proportion of GPS fixes falling in each habitat type, the indigenous forest and the urban residential gardens were the most frequently used by all individuals, while cultivated areas, especially sugarcane *Saccharum* sp. seemed to be avoided (Figure 3.1, Figure S2). There was however individual variation. Tagged individuals also used thickets (dense bush) and occasionally commercial or industrial areas by tracking fruiting trees distributed in these habitats (Figure 3.2).

Table 3.1: Details (sex, start date, end date, duration in days, number of GPS fixes, mass and ring numbers) of Trumpeter Hornbills (TH) tagged with GPS/UHF transmitters in Eshowe, KZN, South Africa.

Individual ID	Sex	Start Date	End Date	No. of Days	No. of GPS fixes	Mass (g)	Ring No.
TH 1	F	07/03/2014	18/10/2014	225	859	670	799125
TH 2	F	07/03/2014	19/10/2014	226	857	665	878788
TH 3	F	07/03/2014	16/03/2014	10	35	600	885271
TH 4	F	10/03/2014	19/10/2014	223	850	610	885272
TH 5	M	10/03/2014	17/10/2014	221	860	811	885274

3.3.3 Home range

Home range size estimates were possible for four Trumpeter Hornbills (TH 1, TH 2, TH 4 and TH 5). There was variation in the individual home ranges estimated using the three different methods (Figure 3.3a) There was no significant difference among the three home range estimators on the overall home ranges of trumpeter hornbills (Kruskal-Wallis ANOVA, $H_{(2, 12)} = 5.8$, $p = 0.055$, Figure 3.3b and Figure S1a). However, home ranges estimated using the LoCoH methods were smaller for all the individuals compared to the other two methods. Overall mean (± SE) home ranges using the different estimators were 5.1 ± 1.28 km^2 (95% MCP), 4.6 ± 1.14 km^2 (95% KDE) and 1.9 ± 0.46 km^2 (95% LoCoH) (Figure 3.3b).

In contrast, there was significant difference among the three home range estimators for monthly home ranges of Trumpeter Hornbills (Kruskal-Wallis ANOVA, $H_{(7, 32)} = 17.5$, $p = 0.014$, Figure 3.4d, Figure S1b). Individual monthly home range sizes varied with individuals exploring a greater area during the months of March, April, September and October (Figure 3.4). When home range estimators for Trumpeter Hornbills were compared between seasons they differed significantly (Kruskal-Wallis ANOVA, $H_{(2, 32)} = 10.2$, $P < 0.006$, Figure 3.5). Generally individuals used greater areas during autumn and spring compared to winter season.

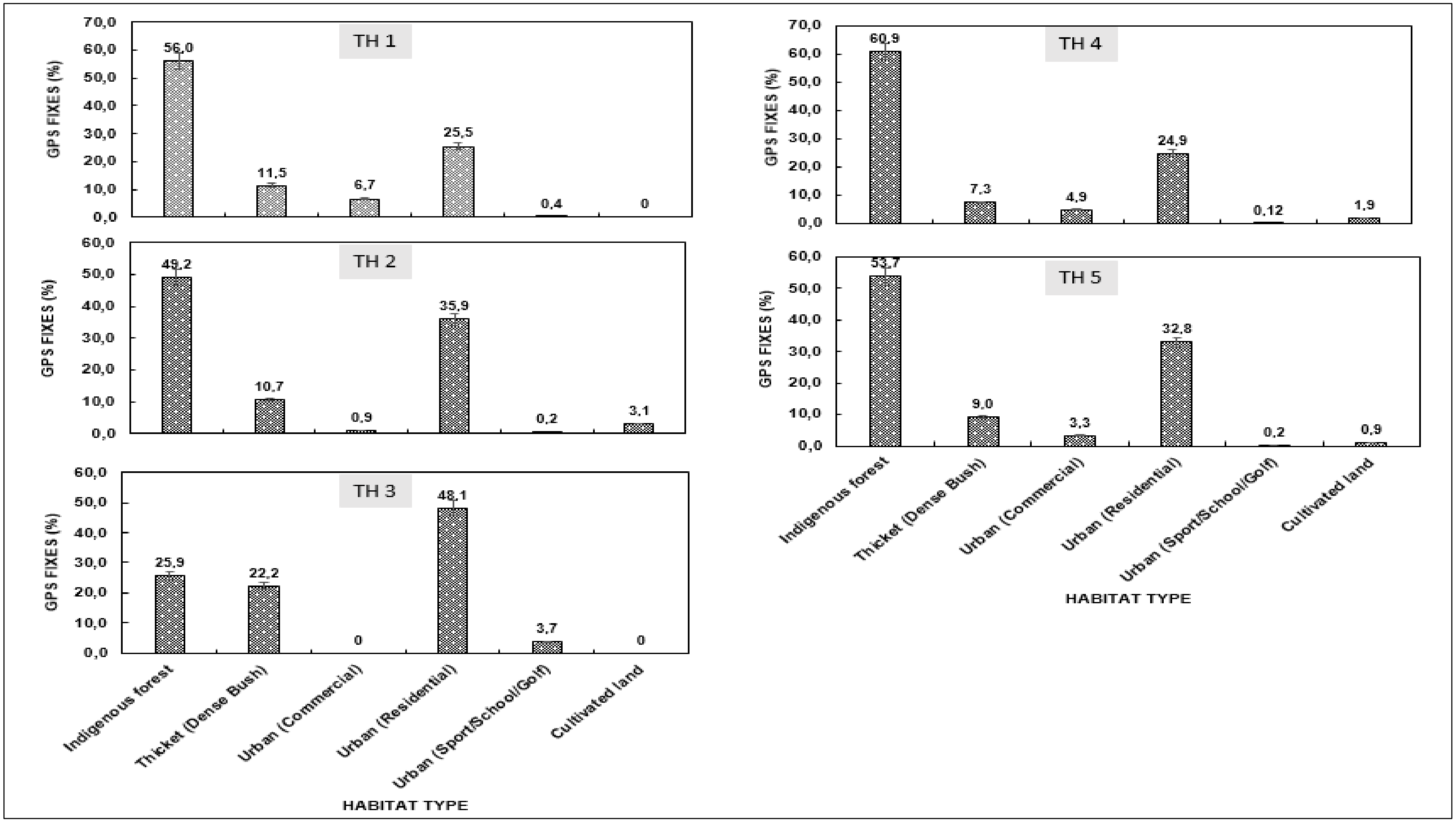

Figure 3.2: Comparison of habitat use by five individuals of Trumpeter Hornbills (TH) tagged in an urban-forest mosaic of Eshowe, South Africa, based on the proportion of GPS fixes falling in each of the six habitat types identified.

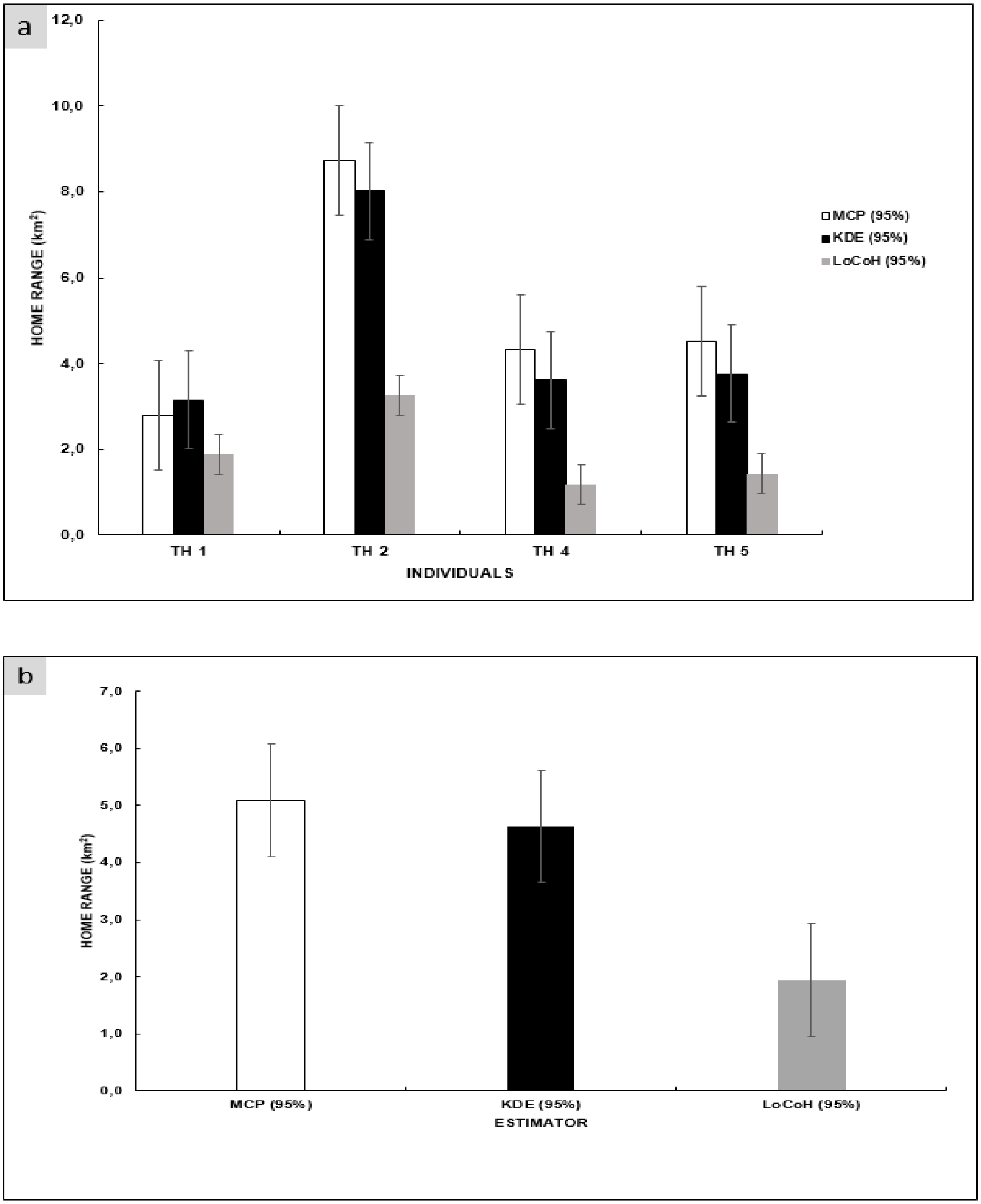

Figure 3.3: Trumpeter Hornbill (TH) mean (+ SE) overall home range size estimated using 95 %MCP, 95% KDE and 95% LoCoH methods a. for the four individuals with sufficient GPS fixes, and b. overall mean (+ SE) monthly home range size estimated using the various methods.

3.3.4 Monthly and seasonal distances moved

All birds tagged within the Dlinza Forest – Eshowe urban mosaic stayed in the area for the entire study period, except for TH 3 that moved to Entumeni Forest (about 8 km from Eshowe). Mean monthly distance moved by individuals differed significantly (ANOVA $F_{7, 3453} = 44.05$, $P < 0.001$). The overall mean monthly distance was greatest in the month of October (1.0 ± 0.07 km) (Figure 3.6a and Table S1). Although there was variation in the mean monthly distance travelled by each individual, the movement distribution was similar for all individuals. Individuals covered slightly longer distances in the months of March, April, September and October, and moved less during the months of May to August (Figure 3.6a). For instance, TH 2 covered a daily maximum distance of 5.9 km in the month of September (Table S1). There was a significant difference in the overall distance covered by Trumpeter Hornbills per season (ANOVA $F_{2, 3458} = 104.76$, $P < 0.001$). Overall, individuals covered longer distances in spring (mean ± SE = 0.8 ± 0.04 km) and autumn (0.6 ± 0.04 km) as opposed to winter (0.4 ± 0.02 km). Individuals showed similar distribution in their movement, with each individual travelling little in the winter season (Figure 3.6b and Table S1).

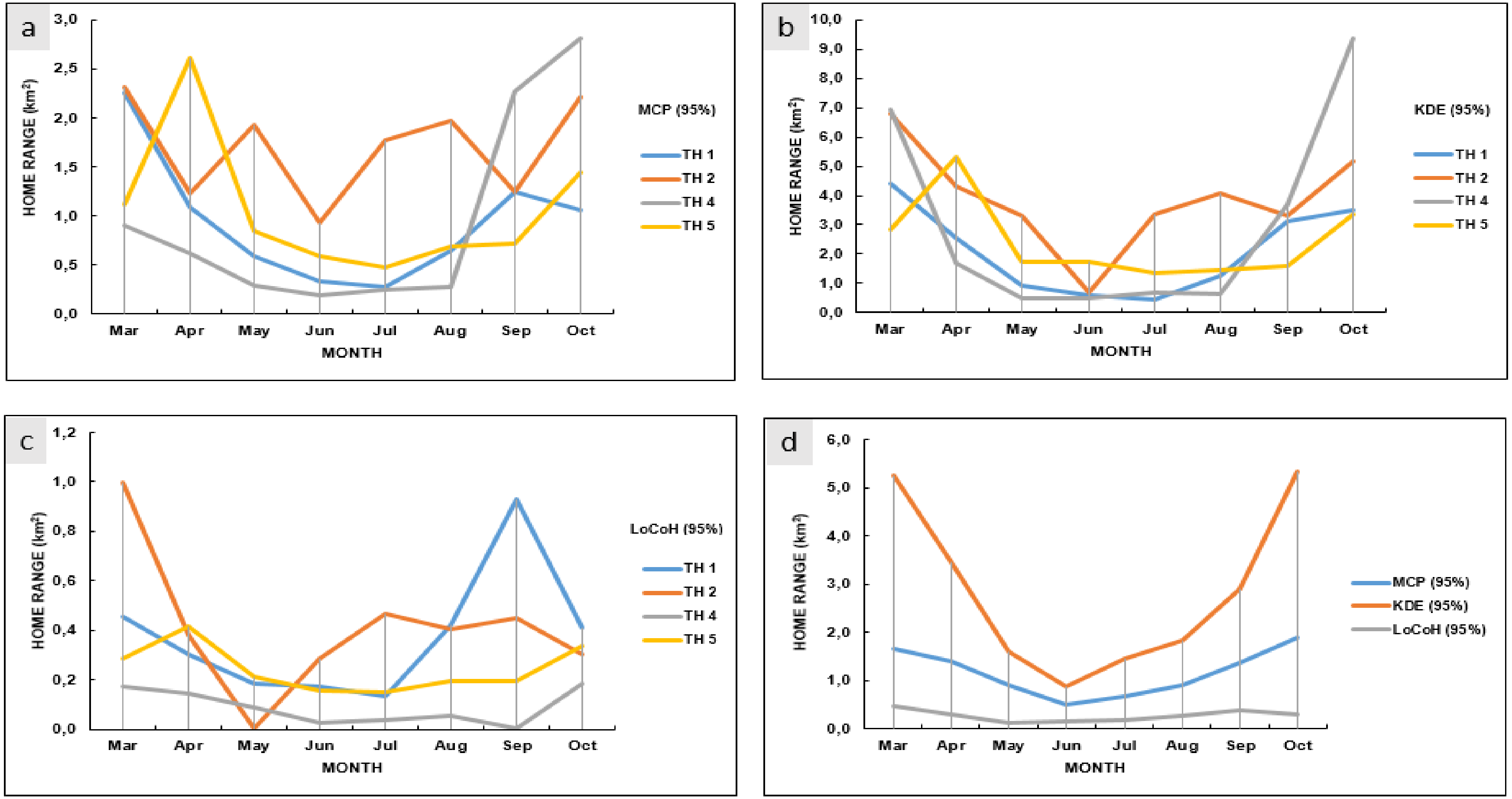

Figure 3.4: Variation in individual monthly home range of Trumpeter Hornbills estimated using three different methods; 95% MCP (a), 95% KDE (b) and 95% LoCoH (c).

The 95% KDE home range sizes were larger than the 95% MCP and the 95% LoCoH home range sizes were the smallest for each month (d).

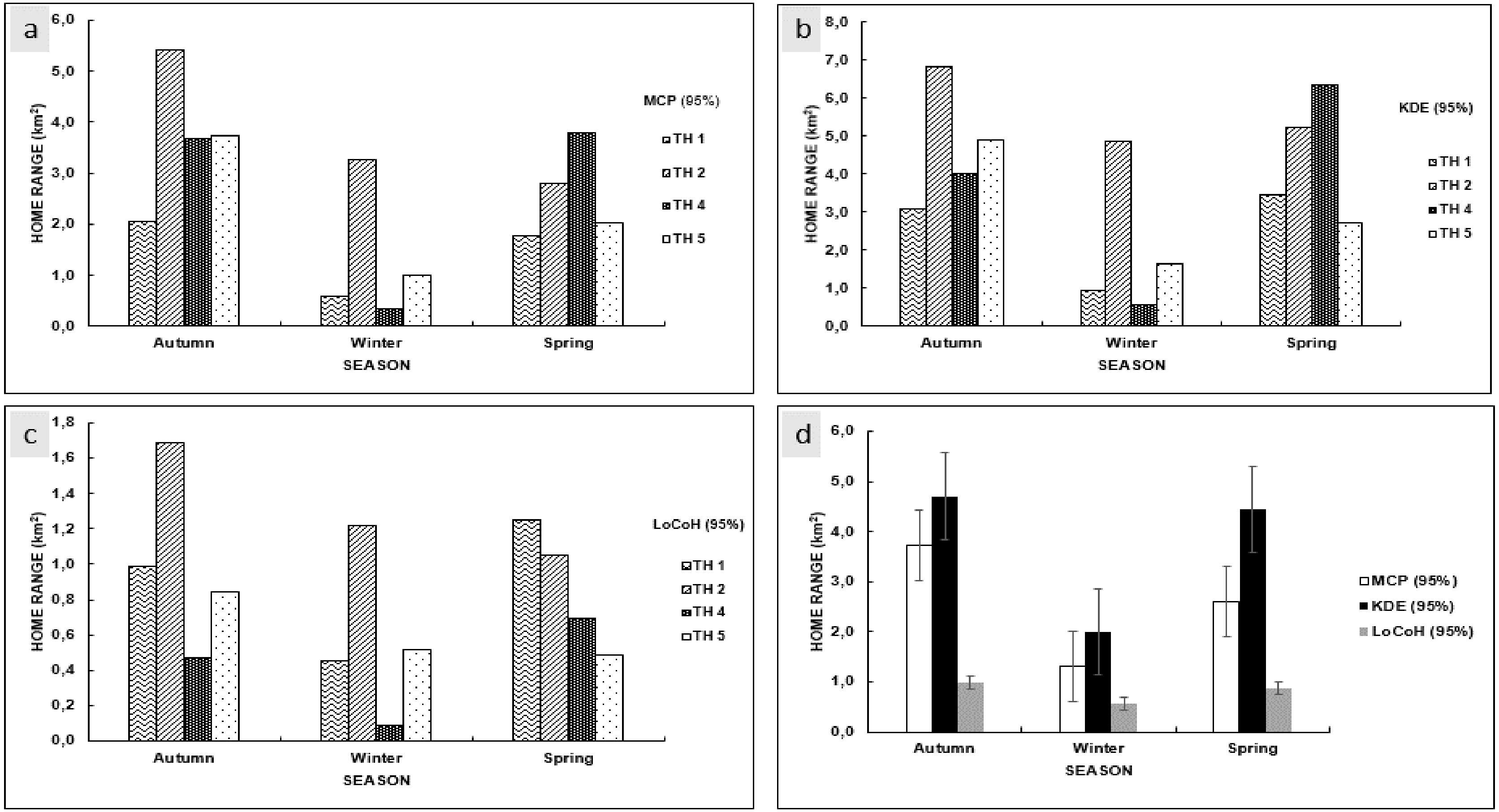

Figure 3.5: Seasonal variation in individual home range size estimated using the three different methods a. 95% MCP, b., 95% KDE, c. 95% LoCoH and d. overall mean (± SE) seasonal home range using the respective methods for Trumpeter Hornbills (TH).

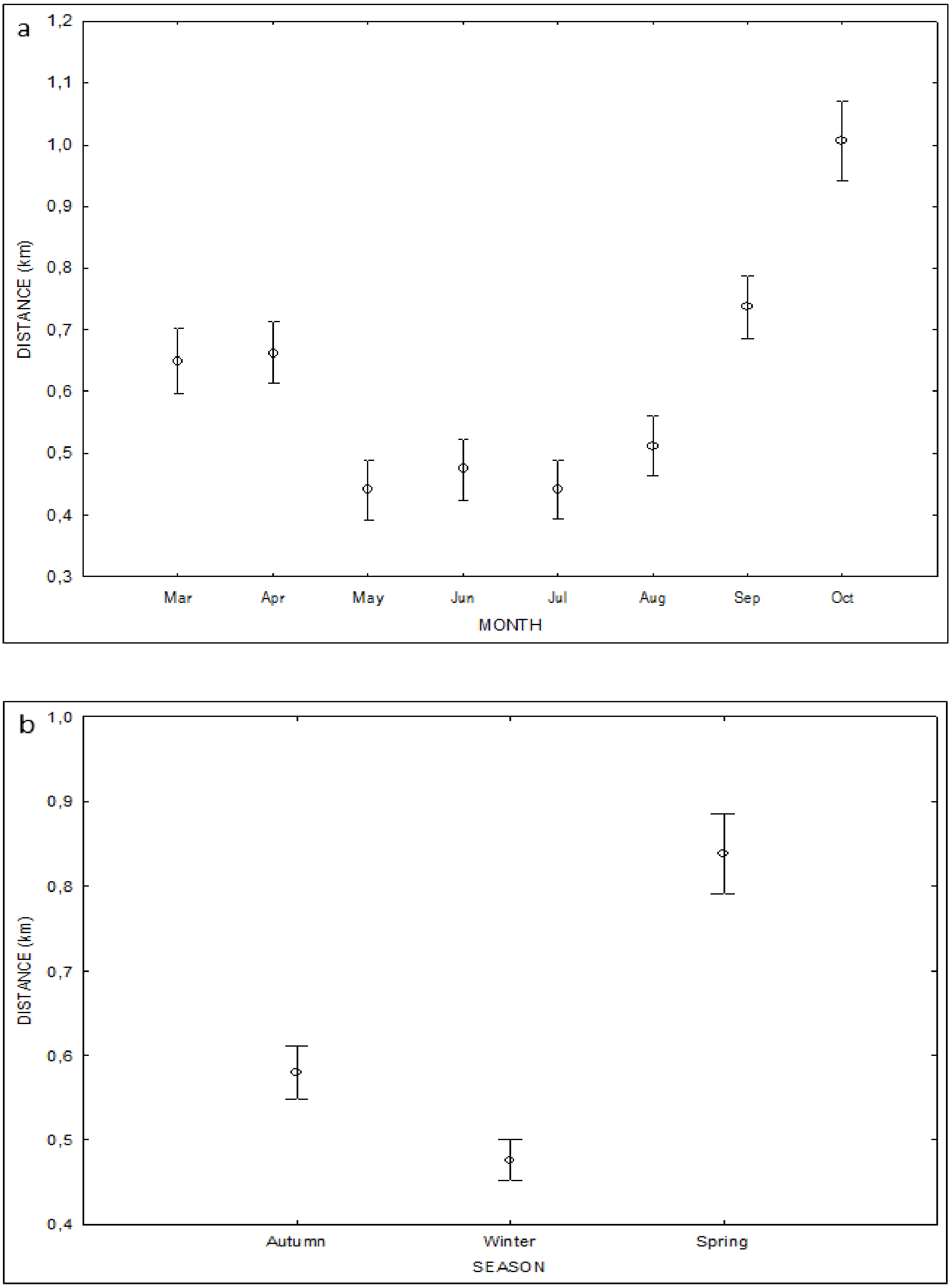

Figure 3.6: Variation in a. mean monthly distances covered, and b. mean seasonal distance covered by Trumpeter Hornbills in the urban-forest mosaic of Eshowe, South Africa.

3.4 Discussion

To our knowledge, we present the first results of home range and habitat use of Trumpeter Hornbill in an urban-forest environment of Eshowe, South Africa. In order to understand the habitat use by each of the tagged Trumpeter Hornbills, we first determined the total area used by each individual using the 100% MCP. Although the MCP can overestimate home range size by including areas that may not be used by an individual on a regular basis (Burgman and Fox 2003), it is the simplest home range estimation technique that gives an approximation of the total area used by an animal. In addition, the MCP makes no assumptions regarding the statistical independence of radio-fixes (De Solla et al. 1999). Using the 100% MCP estimation our results showed that the total area covered by each individual varied (from 4.8 km^2-TH 5 to 12.8 km^2-TH 3) and that all tagged individuals used mostly the indigenous Dlinza Forest and urban residential areas. These two habitats appear to be key for Trumpeter Hornbills in this urban-forest mosaic.

According to Akçakaya (2000) and Thomas and Taylor (2006), estimation of home range size is an important first step that allows us to understand the mechanisms and spatial relationships that affect habitat choices and responses of animals to environmental changes. We estimated the home ranges of Trumpeter Hornbill individuals using three different estimation methods: MCP, KDE and LoCoH, and determined that all are affected by the species, its biology, and habitat availability. Our results showed that the three methods employed produced different home range size estimations. Home range sizes estimated using the LoCoH method were markedly smaller than MCP and KDE methods. This marked difference can be attributed to the ability of the LoCoH estimation method to describe the perimeter of space used by Trumpeter Hornbills more accurately than MCP and KDE methods. These two methods include areas that are not utilised by an individual whereas the LoCoH method does not include areas that are not utilised by an individual within the boundary of its home range (Getz et al. 2007). We found relatively small differences between the MCP and

KDE home range estimates although the mean estimates for overall data was larger for MCP method than KDE method. However, mean monthly and season home range estimates were larger for KDE method than MCP method. These small and inconclusive differences between MCP and KDE methods can be attribute to the sensitivity of home range estimates to varying sample sizes, time scale, seasonal and behavioural variations and other limitations associated with these methods (See Boulanger and White 1990; Girard et al. 2002; Nilsen et al. 2008). In view of this, we agree with previous work that support the use of more than one home range estimation methods (see Biebouw 2009; Boyle et al. 2009; Pebsworth et al. 2012; Reinecke et al. 2014). Despite this, it is clear there is much individual variation. These variations in individual monthly or seasonal home range sizes may be related to changes in resource availability and dietary shifts. Furthermore, the onset of the reproductive season could further influence variation in individual monthly or seasonal home ranges, although our study did not incorporate the breeding season and we did not quantify resource availability.

Leighton and Leighton (1984) categorised movements in response to resource scarcity as true migratory movements, nomadic movements or range expansion movements. Trumpeter Hornbills may undertake long distance movement in search of fruit resources during periods of scarcity. However, all tagged individuals stayed in the Dlinza Forest – Eshowe urban mosaic for the entire study period, except for TH 3 that moved to Entumeni Forest (about 8 km from Eshowe), although there was monthly and seasonal individual variation in movements. Previously, Trumpeter Hornbills have covered much longer dispersal distances in fragmented habitat landscapes (14.5 km; Lenz et al. 2011) than the daily maximum of 5.9 km we found in the forest urban-mosaic that TH 2 covered. Previous studies indicate that hornbill species move in response to fruit availability and thus play an important role in seed dispersal (Holbrook et al. 2002; Lenz et al. 2011; Mulwa et al. 2013; Mueller et al. 2014; Naniwadekar et al. 2015). It is also important to note that other characteristics may influence bird movements in fragmented or urban landscapes. Traits such as dietary specialisation, foraging behaviour, body size, reproductive cycles, survival strategies and habitat affinity (in our case the Trumpeter Hornbill is a forest dependent species) may influence bird movements in human-dominated or fragmented habitats (Lenz et al. 2011; Neuschulz et al. 2013; Chasar et al. 2014; Mueller et al. 2014). In addition, the configuration of a particular landscape in terms of interpatch distance, structural connectivity and fragment size may also influence bird movements (Díaz Vélez et al. 2015).

Effective management and conservation of a species depends on understanding the species pattern or form of habitat use and home range size. This information is limited or lacking for Trumpeter Hornbills especially in the urban-forest mosaic. Use of different telemetry techniques and associated data analyses for home range size, behaviour patterns, habitat use, movement strategies and resource selection for various species have been used to develop a greater understanding and aid in the management of various species (e.g. Chaser et al. 2014; Lenz et al. 2011). This study contributes to the understanding of the movement, habitat use and home range size of Trumpeter Hornbills in an urban-forest mosaic. We highlight the importance of indigenous forest and urban residential gardens for the persistence of Trumpeter Hornbills in human-modified landscapes. Since we did not quantify the fruit abundance and distribution in this study, we recommend that spatial and temporal distribution of fruiting trees (especially plants of the genus *Ficus*) as key food resources to be investigated and compare the data with the movement pattern of Trumpeter Hornbills in an urban-forest mosaic.

3.5 Acknowledgements

We thank the National Research Foundation (ZA), the University of KwaZulu-Natal (South Africa) and the Copperbelt University (Zambia) for the financial assistance provided to the first author. We thank EKZN Wildlife for granting us permission to conduct research in their nature reserves. We thank all nature conservancies in KZN for assisting with information on Trumpeter Hornbill sightings. Last but not the least, thanks to all graduate and post-graduate students that assisted with data collection. We had ethical clearance from the University of KwaZulu-Natal to catch and fit transmitters on the hornbills.

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3.7 Supplementary Material

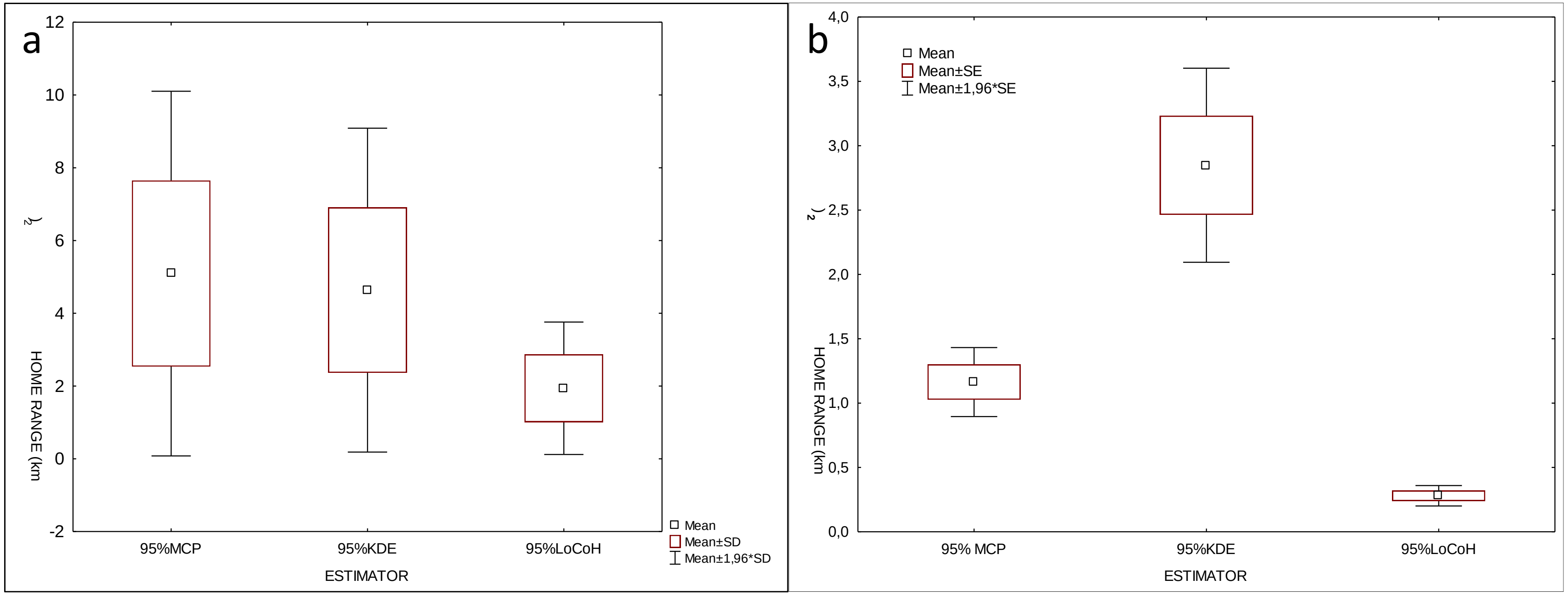

Figure S1: Comparison of three different home range estimation methods (95% MCP, KDE and LoCoH) of overall home ranges (a) and monthly home ranges (b) of Trumpeter Hornbills (n = 4) in the urban-forest mosaic of Eshowe, South Africa.

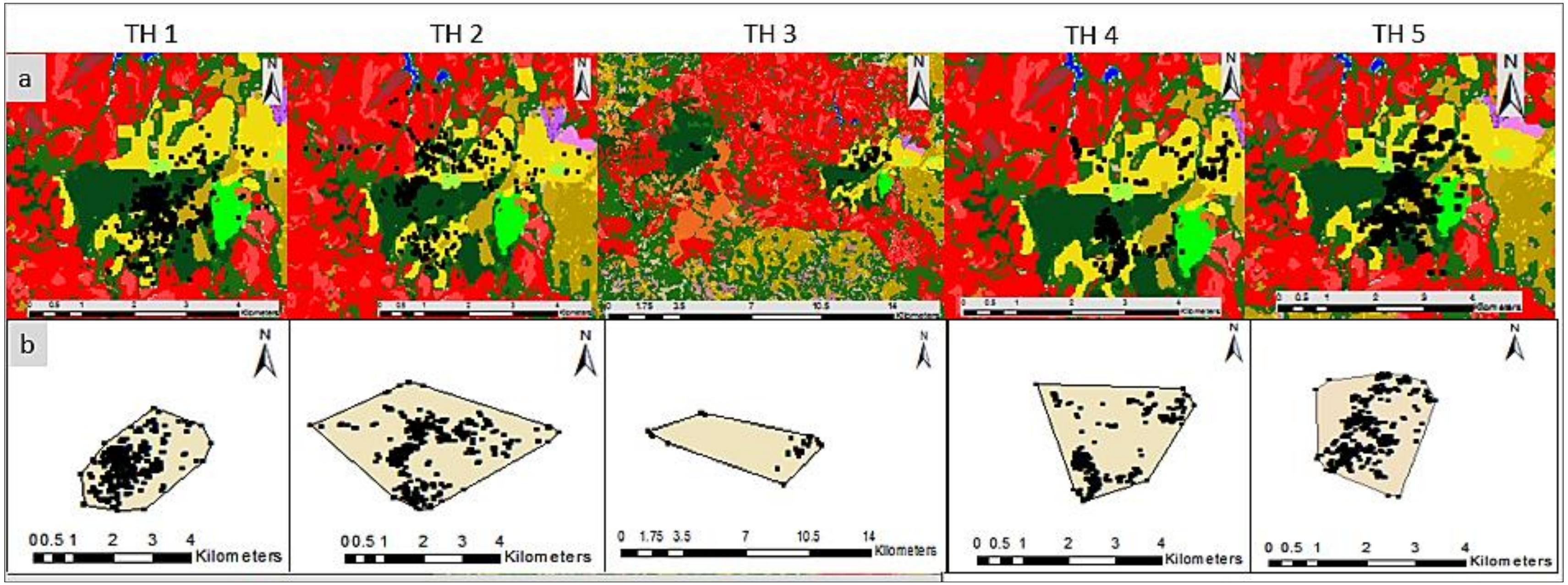

Figure S2: GPS fixes digitised on the land use map showing different habitats (dark green- indigenous forest, light green -thickets, red- cultivated cane commercial, pink cultivated subsistence, yellow- urban, blue water body, mustard- communal villages and lands) visited by each individual tagged (a) and the 100% MCP home range size to show the total area covered by each individual for the entire tracking period (b). The 100% MCP estimated for each individual were 4.9 km^2 (TH 1), 12.1 km^2 (TH 2), 12.8 km^2 (TH 3), 5.0 km^2 (TH 4) and 4.8 km^2 (TH 5). Six habitat types used were identified based on the proportion of GPS fixes falling in each habitat within the 100% MCP home range size for each individual (see Figures 2).

Table S1: Summary statistics of the monthly distance moved by each of the five Trumpeter Hornbills tagged in the urban-forest mosaic of Eshowe.

Individual ID	Month	Mean (km)	No. of GPS fixes	Sum (km)	Std.Dev(km)	Std. Err (km)	Min (km)	Max (km)
1	Mar	0.370	97	35.859	0.394	0.040	0.003	1.287
	Apr	0.617	113	69.716	0.493	0.046	0.002	1.681
	May	0.345	120	41.381	0.277	0.025	0.005	1.398
	Jun	0.301	117	35.259	0.234	0.022	0.003	1.082
	Jul	0.257	120	30.813	0.187	0.017	0.001	0.699
	Aug	0.377	117	44.120	0.335	0.031	0.004	1.450
	Sep	0.749	112	83.942	0.440	0.042	0.003	1.733
	Oct	0.772	63	48.616	0.474	0.060	0.012	2.013
2	Mar	0.744	95	70.675	0.744	0.076	0.008	2.478
	Apr	0.718	114	81.852	0.725	0.068	0.004	2.751
	May	0.599	116	69.513	0.593	0.055	0.000	2.352
	Jun	0.729	115	83.812	0.782	0.073	0.002	3.920
	Jul	0.592	121	71.644	0.490	0.045	0.002	1.881
	Aug	0.840	115	96.544	0.661	0.062	0.001	2.293
	Sep	1.049	105	110.109	0.916	0.089	0.004	5.940
	Oct	1.025	76	77.899	0.580	0.066	0.005	2.583
3	Mar	0.913	35	31.947	0.950	0.161	0.010	3.268
4	Mar	0.745	77	57.372	0.663	0.076	0.011	2.050
	Apr	0.608	116	70.557	0.559	0.052	0.004	2.388
	May	0.246	118	29.021	0.271	0.025	0.002	0.993
	Jun	0.290	117	33.958	0.288	0.027	0.003	0.991
	Jul	0.395	122	48.140	0.316	0.029	0.003	1.040
	Aug	0.368	120	44.120	0.282	0.026	0.002	0.858
	Sep	0.590	113	66.694	0.585	0.055	0.003	2.421
	Oct	1.272	67	85.237	0.913	0.112	0.009	2.679
5	Mar	0.668	86	57.427	0.464	0.050	0.002	1.680
	Apr	0.705	113	79.718	0.578	0.054	0.003	1.685
	May	0.578	119	68.729	0.453	0.042	0.003	1.443
	Jun	0.580	116	67.337	0.475	0.044	0.001	1.541
	Jul	0.522	121	63.144	0.401	0.036	0.001	1.530
	Aug	0.477	121	57.735	0.372	0.034	0.003	1.322
	Sep	0.588	119	69.995	0.339	0.031	0.010	1.558
	Oct	0.935	65	60.756	0.442	0.055	0.007	1.637

CHAPTER 4

Movement pattern of Trumpeter Hornbills and fig resources availability in an urban-forest mosaic, Eshowe, South Africa

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Formatted for Urban Ecosystems

Abstract

Understanding how wildlife persists and responds to urban life is critical to biodiversity conservation and urban planning. We investigated the movement pattern of trumpeter hornbills (*Bycanistes bucinator*) and fig resources availability in an urban-forest mosaic, Eshowe, South Africa. We used GPS/UHF transmitters to record the diurnal movements of trumpeter hornbills. Five individuals were attached with transmitters (4 females and 1 male) and we monitored their movement for a period of 10 to 226 days. Daily maximum distances of individuals tagged ranged from 2.5 km to 4 km. Only one tagged individual moved to another forest (Entumeni Forest) and covered a maximum daily distance of 7.4 km. Daily distances covered by the individuals ranged from (mean ± SE) 0.47 ± 0.43 km to 1.06 ± 1.40 km. We identified seven species of figs that are found in the urban environment of Eshowe from 138 trees (stems) encountered or sampled. Most common and abundant figs were *Ficus burkei*

(62%) and *Ficus natalensis* (27%) and the abundance and wide distribution of these fruiting trees may be one of the reasons Trumpeter Hornbills persist and use the urban environment of Eshowe.

Keywords: Trumpeter Hornbill. Movement patterns. Urbanisation. Fig resources availability

4.1 Introduction

Despite the many challenges posed by urbanisation to wildlife, some species of wildlife use and persist in urban environments successfully. With reference to vertebrates in urban areas, avian species generally have a greater environmental tolerance except for ground nesting birds (Bonier et al. 2007). Furthermore, urban parks or green spaces act as a refuge for native species in areas which are densely populated (Fernández-Juricic et al. 2001). The composition and structure of vegetation in urban areas determines the presence and absence of avian species. Consequently urban areas that retain native vegetation generally retain more native species (Chace and Walsh 2006). However, the survival of birds in urban areas is largely controlled by the availability of food supply, variation in predator assemblages, and risk of collision with anthropogenic structures (Chace and Walsh 2006). The proliferation of certain avian species in urban areas is generally an indication of species-specific adaptation to urban resources, levels of nest predation and parasitism, reduced migratory behaviour and enhanced divergence from the ancestral populations (Chace and Walsh 2006; Bonier et al. 2007; Cilliers and Siebert 2012; Evans et al. 2011, 2012).

In the global forest resources assessment report of 2010, forests and other wooded land make up only 31% of the world land cover with Africa and South America continuously having the largest net loss of forest (FAO 2010). In Africa, the diversity of forest ranges from the dry forests of the Sahel and eastern, southern and northern Africa to the tropical forests of western and central Africa which are humid (FAO 2012; Bodart et al. 2013). With regards to South Africa, the forest biome is the smallest of the eight biomes that exist and KwaZulu-Natal (KZN) Province contains one sixth of the total forest (Mucina and Rutherford 2006). The disappearance of forests and extinction of species is strongly driven by human population growth whose economic activities have increased thus exerting pressure on natural resources (FAO 2010, 2012; Pimm et al. 2014). Other factors that are contributing to deforestation and degradation of the African forest include: 1) poverty, 2) lack of secure land tenure patterns, 3) inadequate recognition within national laws and jurisdiction of the rights and needs of forest-dependent indigenous and local communities, 4) inadequate cross-sectoral policies, 5) undervaluation of forest products and ecosystem services, 6) lack of participation, 7) lack of good governance, 8) absence of a supportive economic climate that facilitates sustainable forest management, 9) illegal trade, 10) lack of capacity, 11) lack of enabling environment at both the national and international levels, and 12) having national policies that distort markets and encourage the conversion of forest land to other uses (FAO 2012).

In view of the above considerations, understanding landscape use by avian frugivores, especially large frugivores which are key dispersers of many plant species of tropical forests in Africa and Asia is vital to the conservation of these forests. A good example of large frugivores occurring in Africa are hornbills (Aves: Bucerotidae) which are among the principal fruit-eaters and are important for long distance seed dispersal (Nathan 2006: Kitamura 2011; Gonzalez et al. 2013). There are 57 species of hornbills that occur throughout sub-Saharan Africa and tropical Asia (Viseshakul et al. 2011; Kitamura 2011; Poonswad et al. 2013;) of which 25 species are found in Africa (Poonswad et al. 2013). Despite being threatened by hunting, habitat loss and international trade, African hornbills are important species because of their vital ecosystem service of contributing to long-distance seed dispersal (Holbrook et al. 2002; Trail 2007). In Cameroon, hornbills cover monthly distances ranging from 23 km to 150 km, with one individual reported to have moved up to 290 km in less than three months (Holbrook et al. 2002). In fragmented landscapes of South Africa, the trumpeter hornbill (*Bycanistes bucinator*) is reported to cover a potential dispersal distance of up to 14.5 km (Lenz et al. 2011). Poonswad et al. (2013) indicated that trumpeter hornbills sometimes fly at least 10 km across the savanna between patches of forests.

In South Africa, a few studies have been conducted on trumpeter hornbills using telemetry and the focus of these studies were on seed dispersal, frugivory, movement behaviour and functional connectivity in fragmented landscapes largely dominated by agricultural activities (Mueller et al. 2014; Lenz 2011 2014; Lenz et al. 2015). We used GPS/UHF transmitters to record the diurnal foraging movements of trumpeter hornbill in an urban-forest mosaic of Eshowe, KZN. Our aim was to understand the movement pattern of trumpeter hornbills and to investigate the availability and distribution of figs as key resources maintaining trumpeter hornbill in the urban environment. As trumpeter hornbills feed mainly on figs (Kemp 2005; Poonswad et al. 2013), the abundance and spatial distribution of fruiting trees in urban areas, especially figs, may have a profound effect on the movement pattern and distribution of the species in the urban environment they are found.

4.2 Methods

4.2.1 Study area

The study was conducted in the municipality of Eshowe (28.89444° S and 31.44889° E) in KZN, South Africa (Fig. 4.1). Within the town is an indigenous protected forest (Dlinza Forest

- 250 ha) established in 1947 and is known for its aerial boardwalk. The forest is rich in biodiversity ranging from mammals, birds, insects, frogs and reptiles (O'Reagain 2001). Some of the tree species found in the forest include: giant umzimbeet (*Millettia sutherlandii*) which is a dominant tree, wild-plum (*Harpephyllum caffrum*), fluted-milkwood (*Chrysophyllum viridifolium*), natal forest cabbage (*Cussonia sphaerocephala*), forest ironplum (*Drypetes gerrardii*), natal milkplum (*Englerophytum natalensis*) and common wild fig (*Ficus burkei*) whose fruits are enjoyed by trumpeter hornbill. Some of the bird life found in the area include the endangered spotted ground thrush (*Zoothera guttata*), crowned eagle (*Stephanoaetus coronatus*), narina trogon (*Apaloderma narina*), purple-crested turaco (*Tauraco porphyreolophus*), red-eyed dove (*Streptopelia semitorquata*), black-collard barbet (*Lybius torquata*), crowned hornbill (*Tockus alboterminatus*) and trumpeter hornbill (*Bycanistes bucinator*) (O'Reagain 2001).

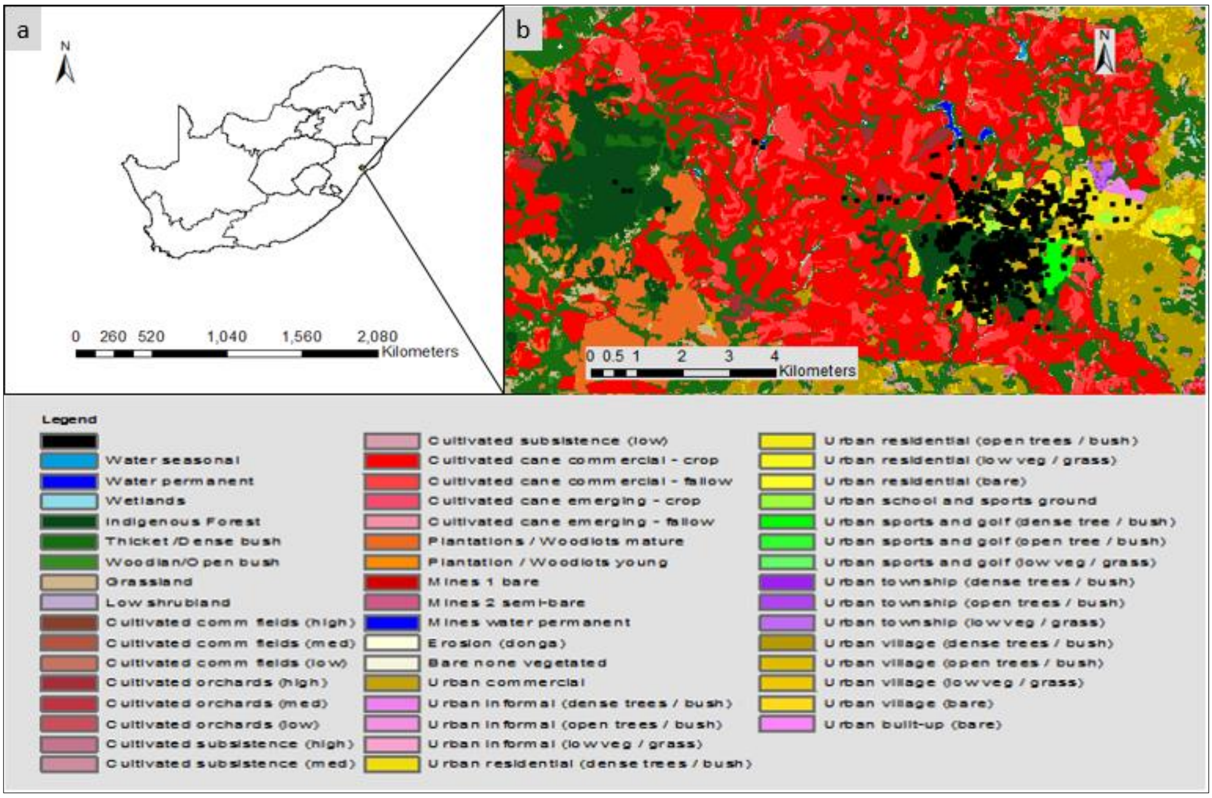

Fig. 4.1: Location of the study area (a) Eshowe, in KZN, South Africa and (b) the main land use types that characterise the study area and the surrounding areas. (Black dots are the GPS fixes of the tagged individuals in the urban-forest mosaic of Eshowe).

4.2.2 Study species

The trumpeter hornbill is the largest obligate frugivore in South Africa with an average body mass of 565 g to 720 g for females and males respectively (Kemp and Woodcock 1995). Males have a larger casque though sexes are alike with regards to the black and white plumage coloration (Kemp and woodcock 1995; Poonswad et al. 2013). In South Africa, it is distributed in the south and east coasts from Alexandria and Knysna Forests, Eastern Cape, and the coastal lowland and montane forests of KZN (Kemp and Woodcock 1995). It feeds mainly on fruits (89%) and is able to swallow small to large sized fruits owing to its large gape width (Kemp and Woodcock 1995; Poonswad et al. 2013). Small fruits especially figs form the bulk of its diet (Kemp and Woodcock 1995). It also feeds on animal diet which includes woodlice, millipedes, caterpillars, spiders, birds' eggs and nestlings and crabs (Kemp and Woodcock 1995; Poonswad et al 2013). The breeding period in South Africa is from October to January and nesting lasts for at least 94 days (Kemp and Woodcock 1995). During the breeding period, the female encloses itself in the nesting cavity and solely depends on the male for feeding. Juveniles remain with both parents for approximately 6 months (Kemp and Woodcock 1995). Little is known about its breeding but it is presumed to be monogamous and moves in pairs during the breeding season and forms large flocks consisting of juveniles and adults during the non-breeding season (Kemp and Woodcock 1995).

4.2.3 Capturing, tagging and tracking

We used standard mist nets placed under a fruiting tree (*Ficus lutea*) to capture the trumpeter hornbills in Eshowe (Appendix A). The capturing was done in March 2014. Birds were weighed, measured and ringed before being released at the same point where they were captured. Transmitters weighed 28 g each and were within the accepted threshold of less than 3 - 5% of the body mass of the species (Kenward 2001). Only five individuals of the nine captured attained this criterion and were fitted with the GPS/UHF transmitters (Wireless Wildlife, Potchefstroom, South Africa). Transmitters were attached as a backpack using 0.25" natural tubular Teflon® tape (Bally Ribbon Mills, Bally, PA) straps looped under the wing with predetermined breakage points to facilitate the dropping-off at a later stage (Appendix A).

Transmitters were programmed to record location points after every 4 h from 6h00 to 18h00 resulting in 4 GPS fixes per day for up to a year. At night, transmitters switched to sleep mode to preserve battery life. Data were stored in the transmitter and downloaded using a base station when tagged individuals were within a radius of 10 km from the base station location (Appendix A). Tracking of each individual started on the very day it was captured, tagged and released. We only managed to download data for the period March to October 2014. Thereafter, the strength of the batteries became too weak for downloading further location data but were strong enough for monitoring the activity pattern of the birds until May 2015 when the batteries ceased operating.

4.2.4 Fig sampling

Based on the prior knowledge that large bodied hornbills feeds mainly on fruits with trumpeter hornbill having a strong preference for *Ficus* species (Kemp and Woodcock; Poonswad et al. 2013; Naniwadekar et al. 2015), we collected data on fig tree distribution in urban Eshowe using systematic random sampling between May and June 2014. A starting sampling point was selected at random and the observer walked systematically through the streets and the location of every fig observed was recorded using a hand held GPSMAP 62sc (Garmin International, Kansas, USA). For every fig observed, additional information on whether the tree was fruiting or not and whether the fruiting tree was ripe or not was collected. Identification of figs was done using Burrows and Burrows (2003) and Boon and Pooley (2010).

4.2.5 Data analysis

GPS fixes were filtered using excel 2010 and imported into ArcGIS 9.3.1 (Geographic Information System; Environmental Systems Research Institute, Redland, California) as layers in a Universal Transverse Mercator (UTM) projection, WGS 1984, UTM Zone 35s. We overlaid each layer on the 2014 land cover map for KZN (Ezemvelo KZN Wildlife 2014) and generated movement paths for each individual using the Home Range Tools (HRT) software in ArcGIS (Rodgers et al. 2007). Comparison of distances covered per day for the individuals tagged was conducted using the Analysis of Variance (ANOVA). A Turkey's post-hoc test was performed to determine which distances were significantly different and summary statistics on average distance covered daily by each individual tagged were calculated using Minitab 17 (Minitab Inc., Pennsylvania, USA).

The fig positional data collected were also analysed using ArcGIS 9.3.1 by digitising the location of each fig on the 2014 land cover map for KZN. This enabled us to determine the spatial distribution of the figs sampled.

4.3 Results

4.3.1 Species captured

A total of nine trumpeter hornbills were captured and ringed. Individual body masses ranged from 510 g to 811 g. Only five individuals met the body mass to transmitter mass criterion to be fitted with the transmitters. Only one male was captured and tagged with a body mass of 811 g. Body masses of the other four females tagged ranged from 600 g to 670 g (Table 4.1).

4.3.2 Average distance moved

Individuals fitted with transmitters gave data from a minimum of 10 days to a maximum of 226 days yielding a total of 3461 GPS fixes (Table 4.2). Mean daily distances covered by each individual ranged from 0.47 km (TH 1) to 1.06 km (TH 3). Daily mean distance moved by each of the five individuals tagged varied (Fig. 4.2). Daily distances moved were significantly different (ANOVA $F_{4,\ 3456}$ = 37. 71, $p < 0.05$) and a Tukey post-hoc showed which were significantly different (Table 4.2).

Table 4.1: Details of the trumpeter hornbill (TH) captured in Eshowe for this study.

Capture Date	**Individuals ID**	**Mass (g)**	**Ring No.**	**Sex**	**Transmitter fitted**
07/03/2014	TH	588	878787	F	No
07/03/2014	TH 1	670	799125	F	Yes
07/03/2014	TH 2	665	878788	F	Yes
07/03/2014	TH	591	878789	F	No
07/03/2014	TH 3	600	885271	F	Yes
10/03/2014	TH 4	610	885272	F	Yes
10/03/2014	TH	510	799124	F	No
10/03/2014	TH	585	885273	F	No
10/03/2014	TH 5	811	885274	M	Yes

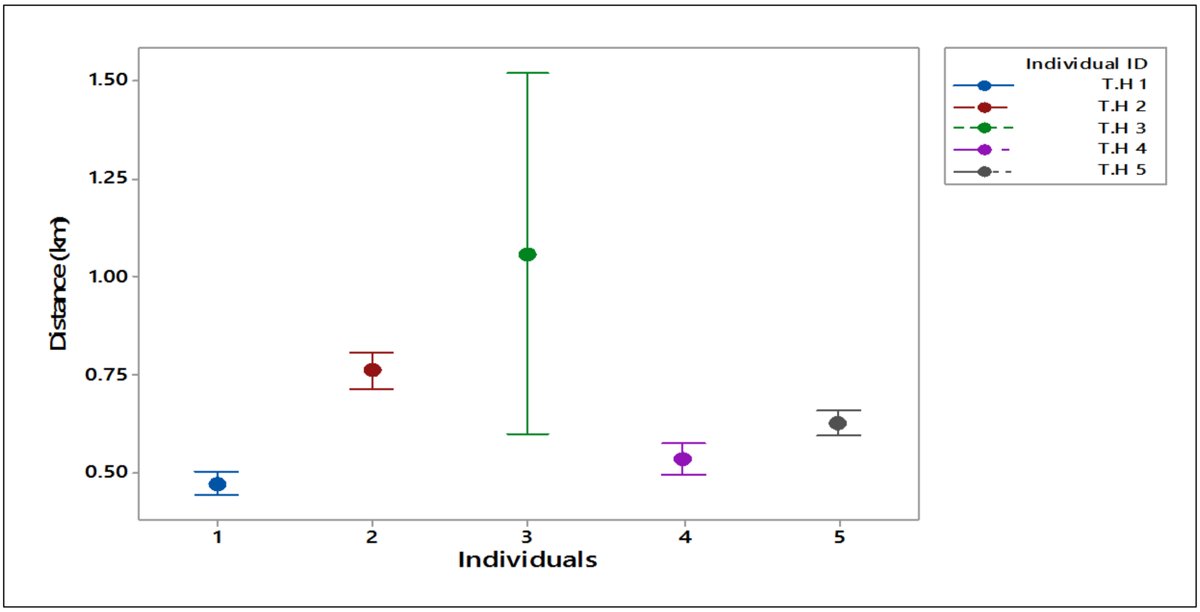

Fig. 4.2: Interval plot of the daily mean (± SE) distance travelled for the five individual trumpeter hornbills tracked in the urban-forest mosaic of Eshowe.

Table 4.2: Results of the average daily distance travelled and the multiple comparison analysis to show which individuals of trumpeter hornbill differed significantly for the five individuals tagged. The means of the individuals that do not share the same letter are significantly different.

Individual ID	Start Date	End Date	No. of Days	No. of GPS fixes	Mean ±SD (km)	Min (km)	Max (km)	Grouping
T H 3	07/03/2014	16/03/2014	10	35	1.058 ± 1.401	0.0100	7.390	A
T H 2	07/03/2014	19/10/2014	226	857	0.759 ± 0.669	0.0003	3.928	B
T H 5	10/03/2014	17/10/2014	221	860	0.627 ± 0.477	0.0008	2.566	C
T H 4	10/03/2014	19/10/2014	223	850	0.534 ± 0.601	0.0016	2.834	D
T H 1	07/03/2014	18/10/2014	225	859	0.471 ± 0.431	0.0006	2.519	D

4.3.3 Movement patterns

Four of the trumpeter hornbills tagged (TH 1, TH 2, TH 4 and TH 5) showed similar patterns of movements by concentrating their daily movements and use of space within Dlinza forest and the nearby urban environment (Fig. 4.3). Occasionally, tagged individuals visited riparian woodland and forest patches on sugar cane farms surrounding the urban Eshowe. Only one individual (TH 3) moved to another nearby protected forest reserve (Entumeni Forest) approximately 8 km from Dlinza Forest (Fig. 4.3). TH 3 only gave 10 days of data for analysis. After the 10th day following tagging, all its GPS points fell at one place. It is not clear whether TH 3 died or the transmitter dropped off as all efforts to find the individual or locate the transmitter failed. The other four individuals remained in the Dlinza Forest-urban mosaic for the duration of the study.

4.3.4 Fig species in urban Eshowe

During the sampling period, a total of 138 figs consisting of 7 species were encountered in the study area. The most abundant were *Ficus burkei* (62% - 85 stems) followed by *F. natalensis* (20% - 27 stems). The least common fig species in urban Eshowe were *F. craterostoma*, *F. polita* and *F. sycomorus* of which only one stem was recorded for each species. The remaining two species (*F. lutea* and *F. sur*) each represented 11% (15 stems) and 6 % (8 stems) of the total figs recorded. 72% of the figs recorded were without fruit and 28% were fruiting at the time of the surveys. Of the fruiting trees, 54% were ripe and 46% were not ripe (Fig. 4.4). The fig trees that were recorded were widely and randomly distributed in the urban environment of Eshowe (Fig. 4.5). Trumpeter hornbills frequently visited and fed on ripe figs and were mostly seen feeding on *F. natalensis*, *F. burkei* and *F. lutea*. Whenever one of these fig species had ripe fruits, trumpeter hornbills were observed feeding on them until all the ripe fruits were finished (pers. obs.).

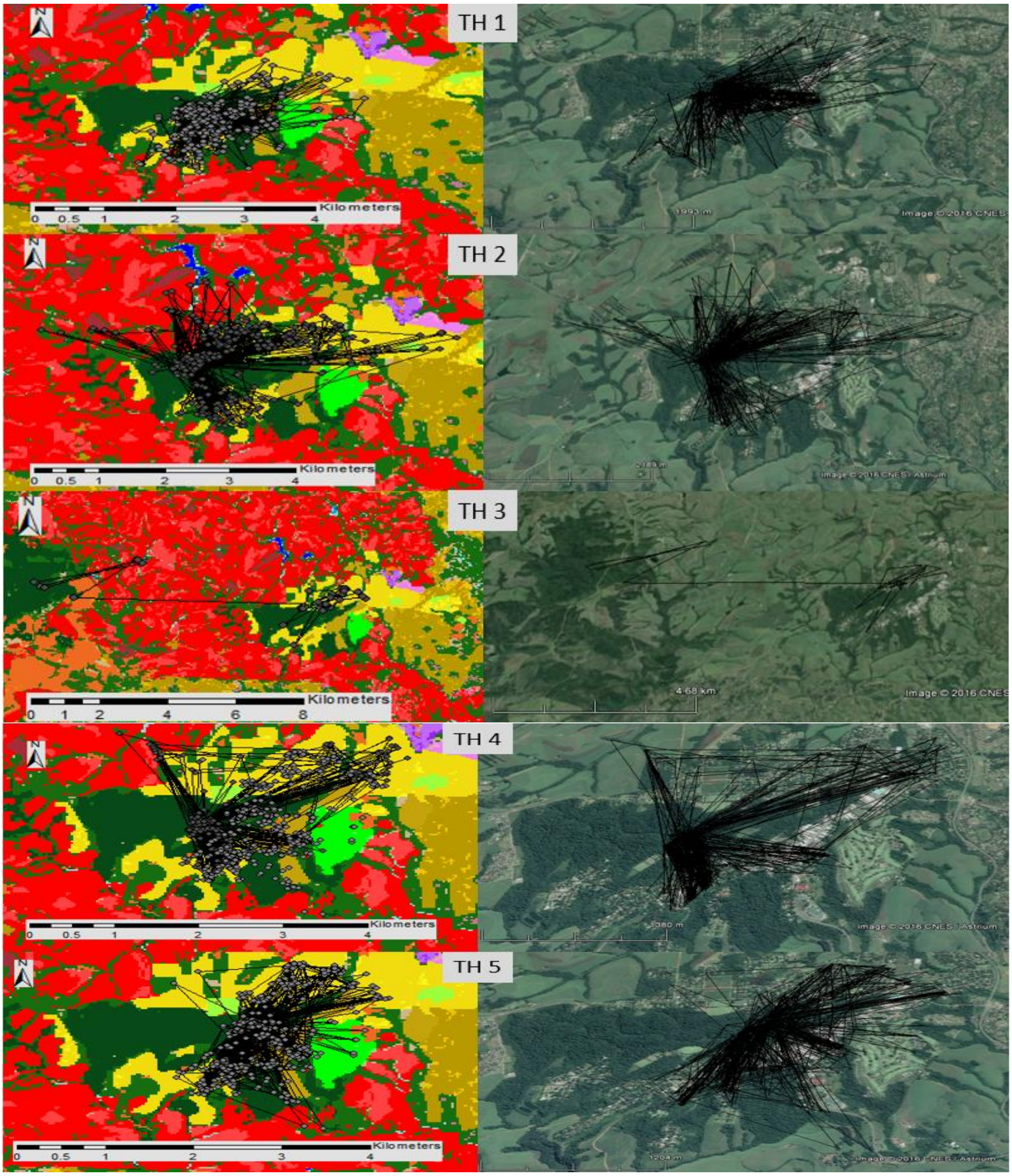

Fig. 4.3: Movement patterns of five trumpeter hornbills in the urban-forest mosaic of Eshowe. Movement patterns were plotted on both the land cover map and Google earth map to show the areas of urban environment used by the trumpeter hornbills.

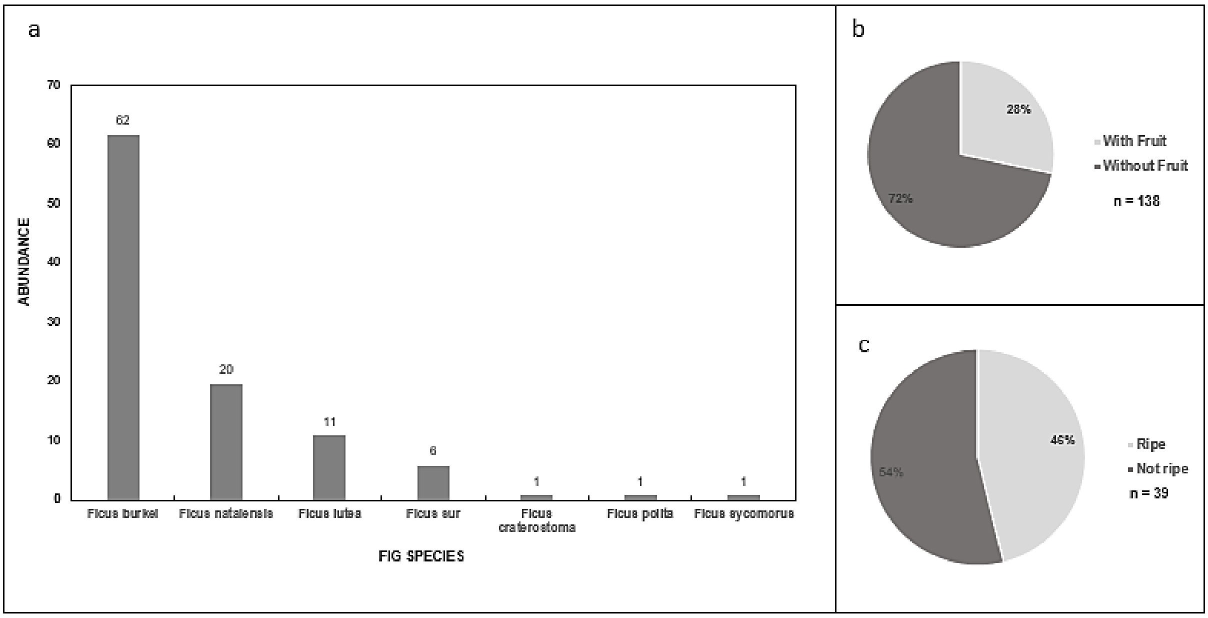

Fig. 4.4: *Ficus* species in urban Eshowe where (a) is the abundance, (b) shows the proportion of figs with or without fruits, and (c) the proportion of ripe and unripe figs.

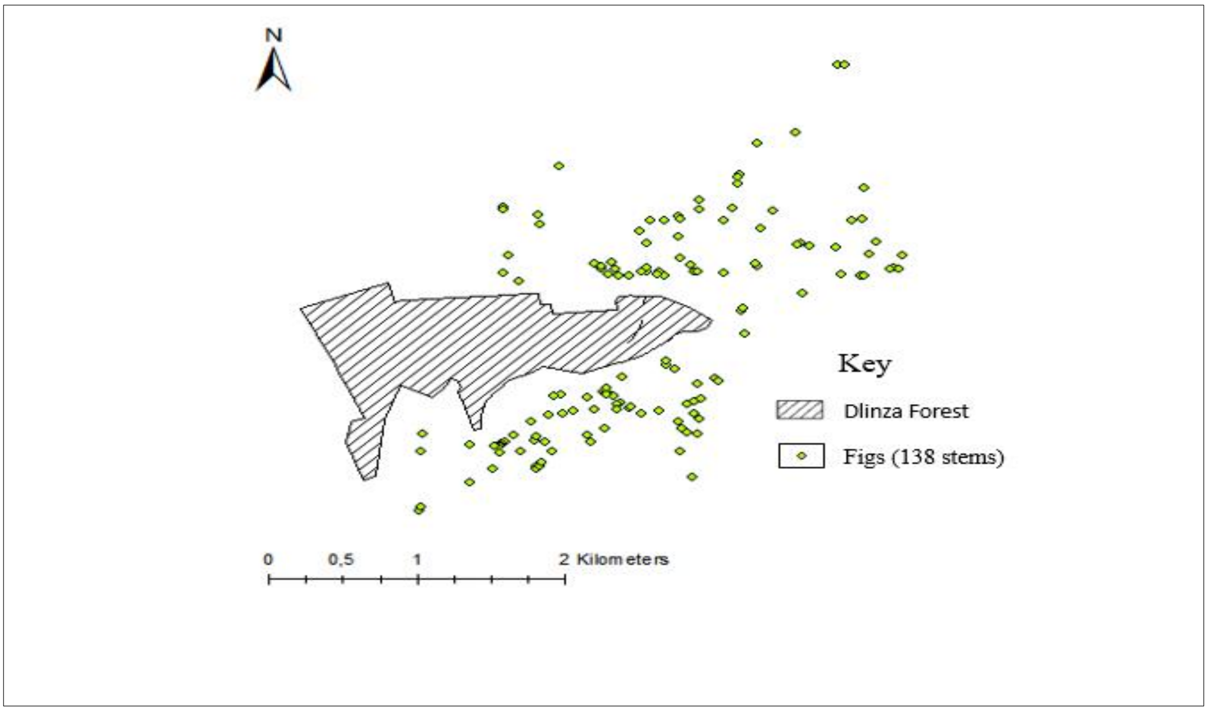

Fig. 4.5: Distribution of *Ficus* species sampled in the urban environment of Eshowe. A total of 138 trees consisting of 7 species were found.

4.4 Discussion

Enhancing our understanding with regards to the persistence and use of urban landscape mosaics by large avian frugivores and other urban exploiters is important for planning for their conservation. Our data showed that the movement patterns of trumpeter hornbills in the urban-forest mosaic of Eshowe was similar as individuals tagged utilised the same space with minimal variations. The possible explanation for this observation may be that individuals tagged were most likely tracking the same food resources in space and time. Furthermore, trumpeter hornbills foraging distances were relatively short in this urban-forest mosaic. Four of the individuals tagged had maximum foraging distances ranging from 2.5 km to 4 km within the urban-forest mosaic of Eshowe. They did not remain in the forest but frequently visited suburban gardens. Only one individual moved to another forest and covered a maximum distance of 7.4 km. The existence of an intact protected indigenous forest (Dlinza forest) within Eshowe town acted as a core area for feeding and roosting and is very important for conservation of trumpeter hornbills and other forest dependent species in an urban environment of Eshowe. As such, there was repeated daily movement of trumpeter hornbills from the forest to the urban areas and vice-versa tracking available food resources. With reference to other fragmented landscapes in South Africa, large frugivores have been observed to cover longer foraging distances. For instance, in a study by Lenz et al. (2011) in a fragmented landscape near Oribi Gorge Nature Reserve, they reported a potential dispersal distance of up to 14.5 km for trumpeter hornbills. For other hornbill species in Africa and Asian continent, general daily movement distances are reported to be around 10 km with some individuals able to travel up to 30 km daily when crossing over non-forest habitats to offshore Islands (Kemp and Woodcock 1995; Kinnaird et al. 1996; Holbrook et al. 2002; Kitamura 2011; Horvitz et al. 2014).

By understanding the movement patterns of large frugivores, identification of key habitats and feeding points that exist in urban environments and other fragmented landscapes can be highlighted (Mueller et al. 2014). In this study, we found tagged individuals repeatedly moved between the indigenous forest and urban residential areas. However, the movement pattern and the presence or absence of avian frugivores and other species in urban environments is greatly influenced by a range of anthropogenic and non-anthropogenic factors. For example, it has been previously reported that factors such as interpatch distance, structural connectivity, settlement structures, and fragment size which affect landscape configuration can influence the movement of birds in urban areas and other fragmented habitats (Díaz Vélez et al. 2015).

In particular availability of key food resources in an urban environment influences the presence and absences of urban exploiters (Díaz Vélez et al. 2015), so there is a need to have an understanding of the abundance and distribution of key food resources here. In this study, we examined the spatial distribution of figs as it has been previously shown that they are a major component of trumpeter hornbills' diet (Kemp and Woodcock 1995; Kitamura, 2011; Poonswad et al. 2013).We identified seven species of fig in the urban environment of Eshowe with *F. burkei* being the most abundant species. Trumpeter hornbills were frequently seen feeding on *F. burke*i and *F. natalensis*. However, medium sized fruits of *Ficus lutea* were also fed on as observed at the time of capture and tagging when there were relatively few other fig species with ripe fruits (pers. obs.). Their feeding on a range of *Ficus* species is similar to those reported in previous studies (Kemp and Woodcock 1995; Kitamura 2011; Poonswad et al. 2013). Trumpeter hornbills were seen to congregate in small flocks and feed on fig trees with ripe fruits and rarely seen consuming other fruits (both alien and native) found in the study area (pers. obs.). *Ficus* species are important as key food resources as they fruit at different times throughout the year thus sustaining many frugivores that depend on them all year round (Gautier-Hion and Michaloud 1989; Lambert 1989).

In addition other native tree species present in the urban-forest mosaic of Eshowe may have influenced the movements of trumpeter hornbills by providing fruits at different times of the year. Important native species that have been recorded as food resources and occur in Dlinza Forest include: *Drypetes gerrardii, Rauvolfia caffra, Celtis africana, Rhoicissus tomentosa, Celtis durandii, Syzygium gerrardii, Trichilia dregeana, Harpephyllum caffrum, Englerophytum natalensis, F. burkei and Protorhus longifolia* (Chittenden unpublished data). Trumpeter hornbills also fed on cultivated or invasive fruits such as papayas (*Carica papaya*), mangoes (*Mangifera indica*), guavas (*Psidium guajava*), Indian laurel (*Litsea glutinosa*) and lychees (*Litchi chinensis*) as observed in other studies (pers. obs., Kemp and Woodcock 1995; Poonswad et al. 2013).

In summary, this study is the first attempt to map the distribution of figs and movement pattern of trumpeter hornbills in an urban-forest mosaic. It is possible that the distribution of key food resources such as figs may have influenced the movement pattern of trumpeter hornbills although our data was only for a short period. In view of this, we recommend a long term study that looks at the fruiting phenology across urban-forest gradient of Eshowe and relate such data to long term movement pattern of trumpeter hornbills over the same period.

4.5 Acknowledgements

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4.7 APPENDIX

Appendix A –Trumpeter hornbills (n = 9) were captured at (a) a residential area of Eshowe, South Africa where a free-standing net was place below the fruiting *Ficus lutea*, and (b) fitted with a GPS/UHF transmitter as a backpack, and (c) tracked and monitored using a base station.

CHAPTER 5

GPS telemetry of the Trumpeter Hornbill: Site fidelity, time to statistical independence, core area and utilisation distribution

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Abstract

The study of wildlife ecology at a fine spatial-temporal scale has been advanced with the latest developments in tracking technologies such as the application of Global Positioning System (GPS) data loggers. In this study, we use data collected from Trumpeter Hornbills *Bycanistes bucinator* (n = 4) that were tagged with GPS/UHF transmitters in the urban-forest environment of Eshowe to investigate the aspects of site fidelity and time to statistical independence (TSI) and to estimate the core areas and utilisation distributions (UDs) of the individuals tagged. We analysed the data using reproducible home range (rhr) software package in R and the KDE and LoCoH methods were implemented to estimate the core areas and to construct UDs. Our results showed that all the individuals tagged exhibited site fidelity and that data from two individuals were not statistically independent. The mean core area estimated using KDE method (mean ± SE) was 0.62 ± 0.35 km^2 (range: 0.34 km^2 to 1.09 km^2). With LoCoH method, the mean core area estimated was 0.07 ± 0.04 km^2 (range: 0.01 km^2 to 0.11 km^2). There was individual variation in the utilisation distribution of the urban-forest environment and the intensity of space use was mainly concentrated in the natural forest and the surrounding urban environment. The site fidelity exhibited by Trumpeter Hornbills may indicate the productivity of the area (urban-forest mosaic of Eshowe) and the spatiotemporal variability of food resources that sustain the Trumpeter Hornbills in this environment. The small core areas observed may be explained by the abundance of food or suitable nesting and roosting sites in the urban-forest mosaic of Eshowe.

Keywords*:* Site fidelity, Time to statistical independence (TSI), Core area, Utilisation distribution (UD), Kernel density estimation (KDE), Local convex hull (LoCoH), GPS telemetry, Trumpeter Hornbill

5.1 Introduction

The study of wildlife ecology at a fine spatial-temporal scale has been advanced with the latest developments in tracking technologies such as the application of Global Positioning System (GPS) data loggers (Kenneth et al. 2006). Tracking animals using GPS telemetry offers many advantages compared to other tracking methods such as ARGOS satellite telemetry, VHF or UHF radio telemetry and light-based geolocation (Cagnacci et al. 2010; Tomkiewicz et al. 2010; Kennedy et al. 2015). Notable advantages of using GPS telemetry include the ability to track the movements of tagged individuals continuously for a long period of time even in challenging climatic and topographic conditions (Arthur and Schwartz 1999), collecting bias-free locations in real time without human observers (Hebblewhite and Haydon 2010), its high spatial accuracy and capacity to record and store large numbers of observations (Cagnacci et al. 2010). Despite the advantages, GPS data loggers are still comparatively expensive and designing studies that requires tagging many individuals are practically unattainable and many studies that have employed GPS data loggers in the past had the challenge of tagging many individuals may be due to the expensiveness of using GPS telemetry or the difficulties associated with trapping some wildlife species (Cagnacci et al. 2010; Tomkiewicz et al. 2010). However, even the tagging of one individual for a continuous long period of time has revealed interesting results previously unknown to biologists and ecologists as a result of using GPS telemetry (Kays et al. 2015).

The implementation of site fidelity and time to statistical independence (TSI) as preliminary steps to home range size estimation is still a subject of discussion by ecologists on whether to implement these steps or not (Munger 1984; Swihart and Slade 1985; De Solla et al. 1999; Fieberg 2007; Fleming et al. 2015). Review of literature showed that there is variation in home range studies with regards to the implementation of site fidelity and TSI before home ranges are estimated (Laver and Kelly 2008). Site fidelity exists when the observed area an animal uses is smaller than the area used if an individual's movement was random (Munger 1984). It is assumed that if an animal reveals site fidelity, then it has a home range (Spencer et al. 1990). TSI test ensures that there is temporal independence in animal relocations or simply that there is no autocorrelation in the adjacent observations (Swihart and Slade 1985; Fleming et al. 2015). TSI determines the critical time interval after which two subsequent relocations are statistically independent and this information is important for accurate interpretation of home range studies (De Solla et al. 1999). However, testing for TSI as a prerequisite for home range estimation has been a subject of debate in the past (De Solla et al. 1999; Fieberg 2007) but it is one of the important steps suggested by Laver and Kelly (2008) as a preliminary step before home range estimation. According to De Solla et al. (1999), they indicated that kernel density estimation (KDE) does not require serial independence of observations and recommended that researchers should maximise the number of observations using constant time intervals to increase the accuracy of their estimates. Study duration and the within sampling rate determines the degree of autocorrelation in relocation data (De Solla et al. 1999).

In the past, studies that analysed space use have treated utilisation distributions (UDs) as an intermediate step in the estimation of home range boundary or core areas (Lichti and Swihart 2011). UD is a bivariate probability density function that tends to map the variation in the intensity of use of space by an individual and assumes that the pattern of space use is stable over the time period being analysed (Lichti and Swihart 2011). Core area is the region within an animal’s home range where it concentrates its activities and may contain important habitat features such as food resources and nesting or sleeping sites (Ramos-Fernandez et al. 2013). Two methods are commonly used to construct UDs of animals in ecological studies from location data. The KDE method is the oldest and widely applied method (Worton 1989; Worton 1995; Gitzen and Millspaugh 2003; Gitzen et al. 2006) whereas the local convex hull (LoCoH) is a recently developed method (Getz et al. 2007). Each method has its own advantages and disadvantages and the choice of which method to use to estimate UDs should be based on the research questions to be addressed (see Seaman et al. 1999; Gitzen et al. 2006; Getz et al. 2007; Lichti and Swihart 2011).

In this study, we used location data collected from four Trumpeter Hornbills (*Bycanistes bucinator*) that were tagged in an urban-forest environment of Eshowe, South Africa to test for site fidelity and TSI as preliminary steps for home range analysis following the suggestions by Laver and Kelly (2008). In addition, we estimated the core area and utilisation distribution for each individual tagged using KDE and LoCoH methods with the objective of determining the size of the area that is intensively used by each individual tagged in an urban-forest environment of Eshowe.

5.2 Methods

5.2.1 Study area

The study was conducted in the municipality of Eshowe (28.89444° S and 31.44889° E) in KwaZulu-Natal (KZN) Province, South Africa (Figure 5.1), which includes the indigenous protected Dlinza Forest (250 ha, protected since 1947) (O'Reagain, 2001). The forest is rich in biodiversity ranging from mammals, birds, insects, frogs and reptiles (O'Reagain, 2001). Eshowe town receives a great deal of rainfall (mean annual rainfall - 1119 mm) and its climate is warm and temperate (http://en.climatedata.org/location/12807/). On average, temperature is highest in February (around 21°C) and lowest in June (around 15°C). The main land use types in the area are settlements and agriculture, mainly sugar cane farming (Figure 5.1).

5.2.2 Study species

The Trumpeter Hornbill is a relatively common forest species along the east coast of South Africa and it is the largest obligate frugivore (Kemp and Woodcock 1995; Poonswad et al. 2013). It is categorised as least concern (IUCN 2012) but the species is threatened by habitat lose, international trade and hunting (Trail 2007). The Trumpeter Hornbill is a medium-sized bird (average weight: 550 g for females and 720 g for males) and 89% of its diet constitute of fruits (Kemp and Woodcock 1995). In South Africa, breeding is from October to January and nesting lasts for ~ 94 days (Kemp and Woodcock 1995). Females remain sealed in the nest the entire nesting period (mostly tree holes) and depend on the males for feeding (Kemp and Woodcock 1995) .

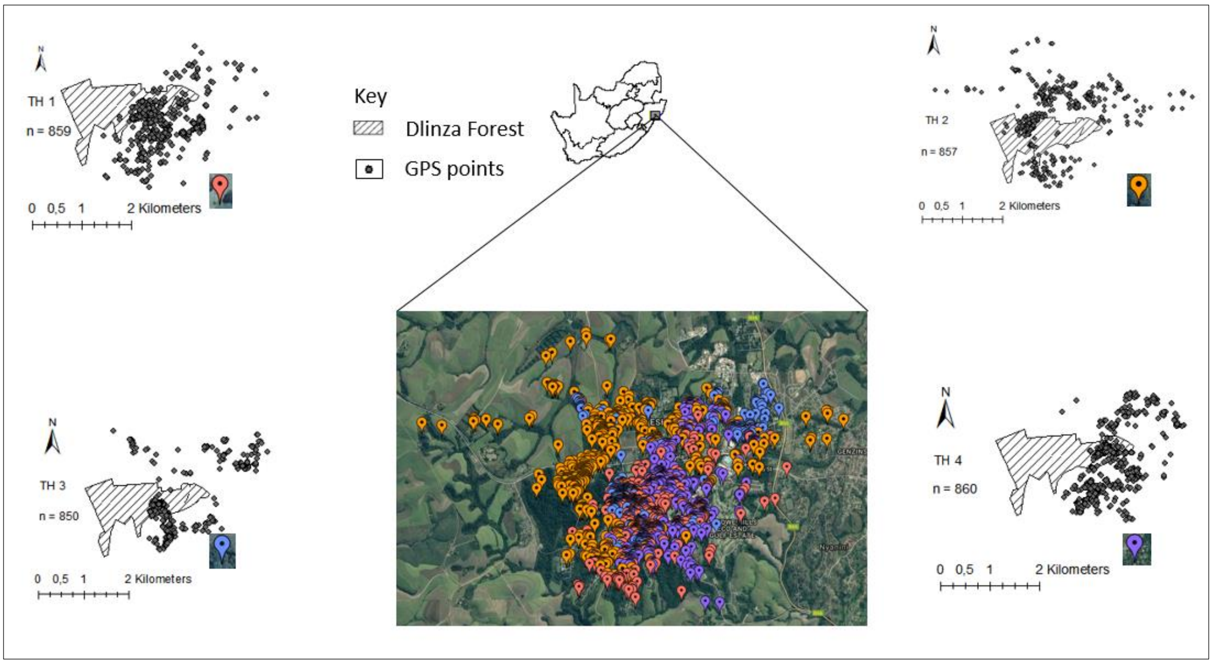

Figure 5.1: Study area showing the dstribution of the GPS points of the four Trumpeter Hornbills (TH) tagged in an urban-forest environment of Eshowe, South Africa.

5.2.3 Data collection and analysis

We used standard mist nets placed under a fruiting tree (*Ficus lutea*) to capture the Trumpeter Hornbills in urban Eshowe. The capturing was done in March 2014. Birds were weighed, measured and ringed before being released at the same point where they were captured (Chapter 3). Transmitters weighed 28 g each and were within the accepted threshold of less than 3 - 5% of the body mass of the species criterion (Kenward 2001). Five individuals of the nine captured attained this criterion and were fitted with the GPS/UHF transmitters (Wireless Wildlife, Potchefstroom, South Africa) (Chapter 3). Transmitters were attached as a backpack using 0.25" natural tubular Teflon® tape (Bally Ribbon Mills, Bally, PA) straps looped under the wing with predetermined breakage points to facilitate the dropping-off at a later stage. Transmitters were programmed to record location points after every 4 h from 6h00 to 18h00 resulting in 4 GPS fixes per day. At night, transmitters switched to sleep mode to preserve battery life. Data was stored on-board memory within the transmitter and downloaded when the tagged individuals were within a radius of 10 km of the solar-powered base station. Tracking of each individual started on the first day it was captured, tagged and released. We only managed to download data for the period from March to October 2014 (Chapter 3).

Data sorting, filtering, conversion of GPS points to Universal Transverse Mercator (UTM) and analysis was done using reproducible home range (rhr) package (Signer and Balkenhol 2015) in programme R, version 3.2.3 (R Development Core Team 2015). Using rhr, we implemented the following:

1) **Site fidelity** – Two metrics, Linearity index (LI) and the mean squared distance (MSD) from the centre of activity are compared to true and simulated trajectories. If the area used by an individual moving at random is greater than the observed area used, then site fidelity exists (Munger 1984). If there is the absence of site fidelity, home range analysis may not be the best analytical method. The calculation of site fidelity was based on the methods by Spencer et al. (1990). The analysis was conducted at $\alpha = 0.05$ with 100 bootstrap replications. A histogram for LI and MSD is produced and shows the critical threshold (red dashed lines) and the observed value (red solid line). Site fidelity is present if the solid red line is below the interval indicated by the red dashed lines on the plot.
2) **Time to Statistical Independence (TSI**) – This is the time interval that is required until two consecutive location points or observations are statistically independent (Swihart and Slade 1985). Swihart and Slade (1985) method used to implement TSI showed how to determine the Schoener statistic (Schoener 1981) which is used as a critical value. A consecutive sampling regime was used since our relocations were separated by equal time interval (4 h or 14400 s). TSI analysis produces a plot with two panels. The upper panel

shows the value of the test statistics (Schoener statistic) and the lower panel indicates the number of relocations used. On the plot, the solid black line indicates the value of the test statistic and the grey line indicates the critical value needed to be reached to have temporal independence of consecutive relocations. TSI is reached when a red dot and dashed line is shown.

3) **Core area and utilisation distributions (UDs) estimation** – To know the size of area within the home range that are used more intensely than others for each individual tagged, we estimated the core area and UDs using the LoCoH (Getz et al. 2007) and KDE (Worton, 1989; Lichti and Swihart 2011). In both the KDE and LoCoH methods, the shape of UD is determined by the tuning parameter (Seaman and Powell 1996; Seaman et al. 1999; Gitzen et al. 2006; Getz et al. 2007). For KDE method, the tuning parameter is known as the bandwidth or smoothing parameter (*h*) (Gitzen et al. 2006) while the LoCoH method constructs a convex hull around each point and the point's nearest neighbours (*n*) which is determined by one of many potential rules (Getz et al. 2007). Both *h* and *n* play similar role but large values of *h* or *n* generate broader and more even UD surfaces that reduce variation among datasets while smaller values enables the estimator to better fit a specific dataset thus increasing the resolution of the peaks and valleys (Fieberg 2007; Lichti and Swihart 2011). The commonly used isopleths for determining UD are 50% (core area), 95% and 100% but the isopleth range can be between 0% and 100% (Lichti and Swihart 2011).

5.3 Results

5.3.1 Site fidelity and TSI

Our results showed that the four tagged Trumpeter Hornbills used a range of habitats in the landscape (Figure 5.2) and each exhibited site fidelity in the urban-forest environment of Eshowe (Figure 5.3). All the individuals tagged remained in the area for the entire study period. Results for TSI analysis indicated that there was temporal independence in the location points of two individuals (TH 1 and TH 2) and there was lack of independence in the successive location points for the other two individuals (TH 3 and TH 4) (Figure 5.4).

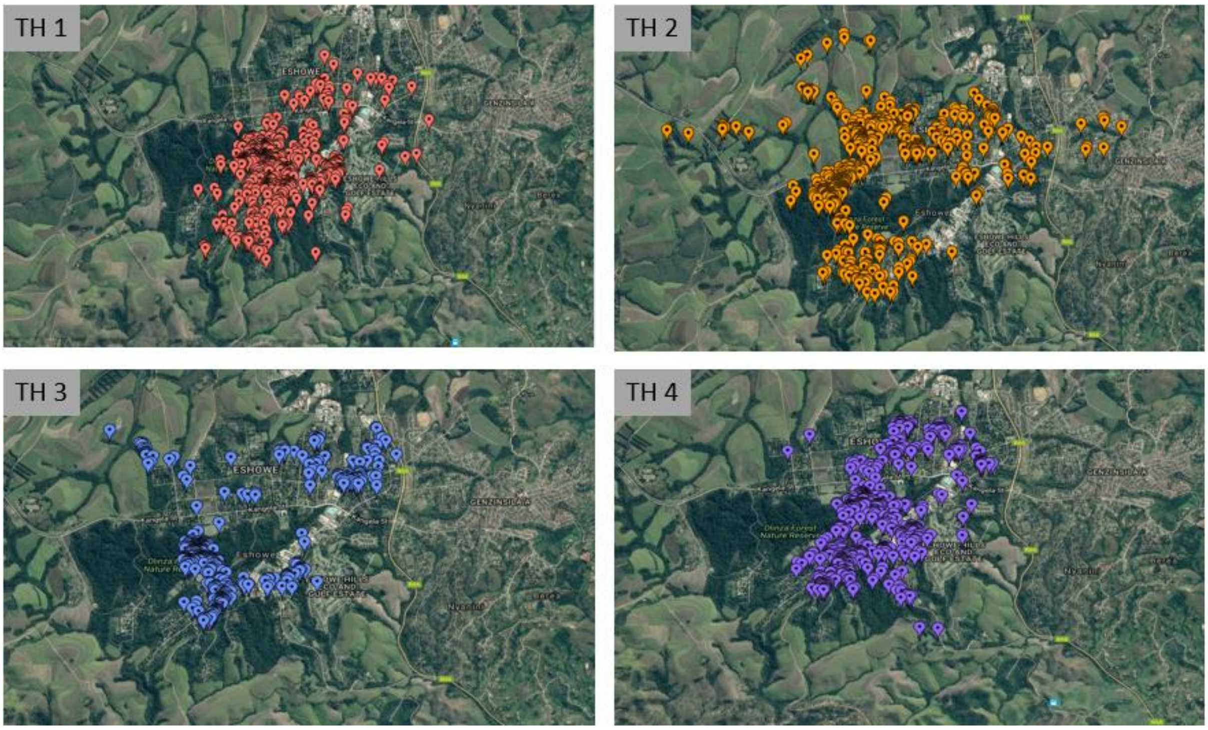

Figure 5.2: Distribution of GPS point of the four Trumpeter Hornbills tagged in an urban-forest mosaic of Eshowe showing the variation of space use.

5.3.2 Core area and UD

With the KDE method, core area estimates ranged from 0.34 km^2 to 1.09 km^2 and mean core area was 0.62 ± 0.35 km^2 (Table 5.1, Figure 5.5). Using the LoCoH method, estimated core areas ranged from 0.01 km^2 to 0.11 km^2 with a mean core area of 0.07 ± 0.04 km^2 (Table 5.1, Figure 5.6). LoCoH utilisation distribution showed variations among individuals in the general use of space and those areas that were intensely used by each individual.

Table 5.1: Summary information on four Trumpeter Hornbills tagged in an urban-forest environment of Eshowe with core areas estimated using the KDE and LoCoH methods. The smoothing parameter used for KDE was least square cross validation **(h_{LSCV})** and the default tuning parameter for LoCoH was k.

Individual ID	**Start Date**	**End Date**	**sex**	**No. of Days**	**No. of GPS fixes**	**Mass (g)**	**Core Area (Km^2)**	
							KDE (h_{LSCV})	**LoCoH (k)**
T H 1	07/03/2014	18/10/2014	F	225	859	670	0.39(131.6)	0.09(29)
T H 2	07/03/2014	19/10/2014	F	226	857	665	1.09(227.6)	0.11(29)
T H 3	10/03/2014	19/10/2014	F	223	850	610	0.34(177.0)	0.01(29)
T H 4	10/03/2014	17/10/2014	M	221	860	811	0.65(163.0)	0.08 (29)
Mean ± SD				223.6 ± 2.22	856.5 ± 4.51	689.0 ±85.76	0.62 ±0.35	0.07 ±0.04

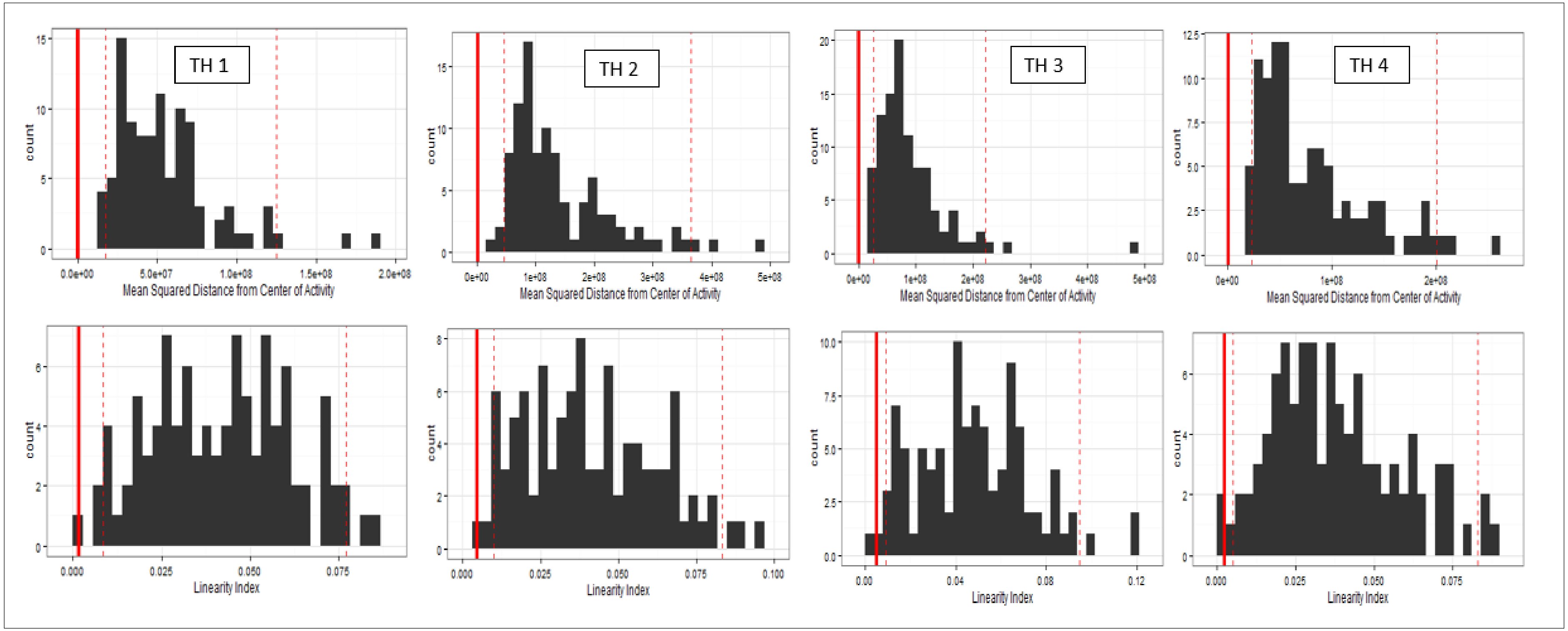

Figure 5.3: Site fidelity test results of Trumpeter Hornbills (TH) tagged in an urban-forest environment of Eshowe, KZN. The results showed that all four individuals tagged exhibited site fidelity for the entire period of the study (March to October 2014) as the observed value (solid red line) was below the critical threshold (red dashed lines) for both linearity index (LI) and mean square distance (MSD) metrics.

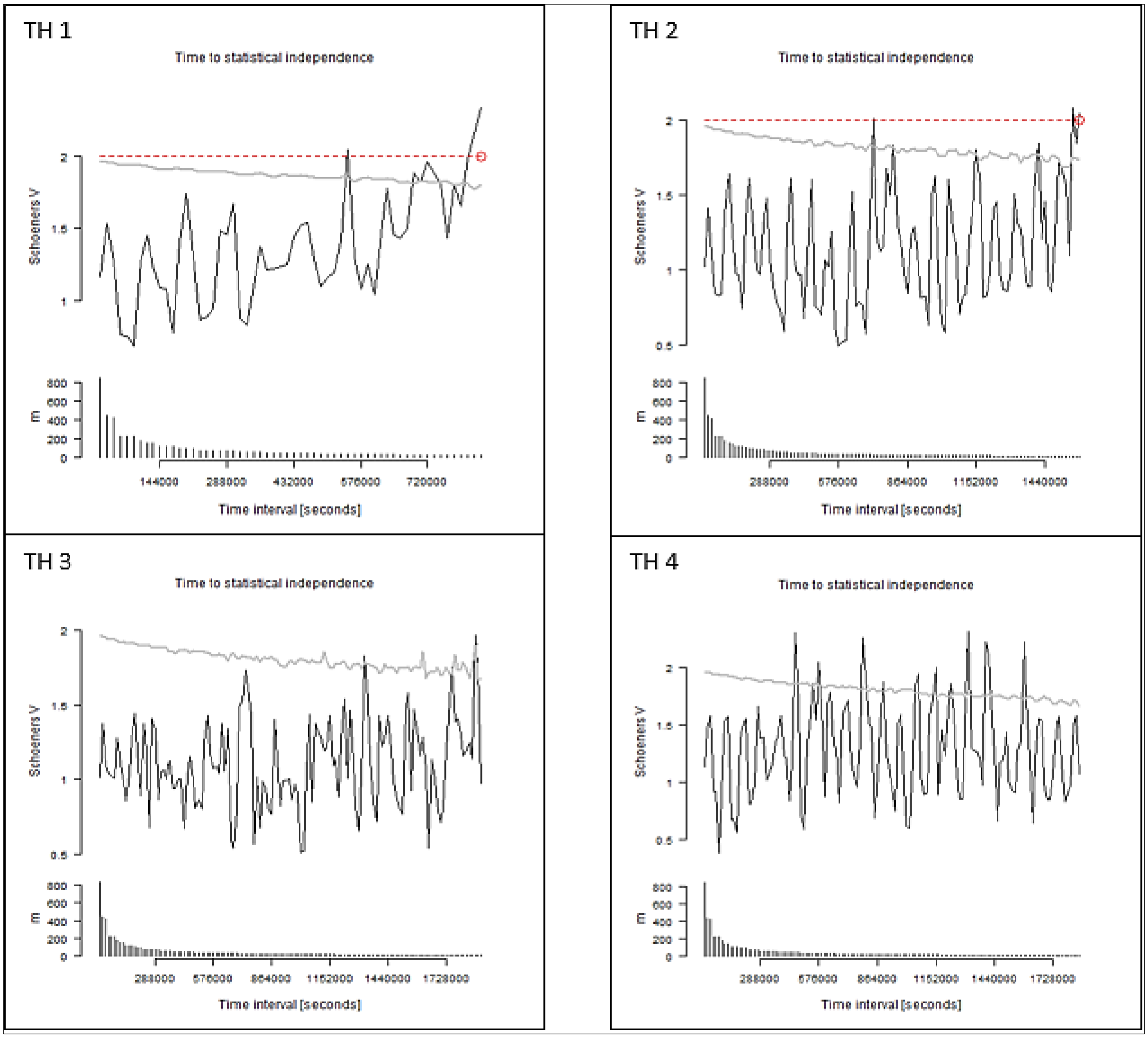

Figure 5.4: TSI results of Trumpeter Hornbills (TH) tagged in an urban-forest environment of Eshowe, KZN. TH 1 and TH 2 had independent datasets (red dot and dashed line shown) whereas for TH 3 and TH 4 the datasets were not independent.

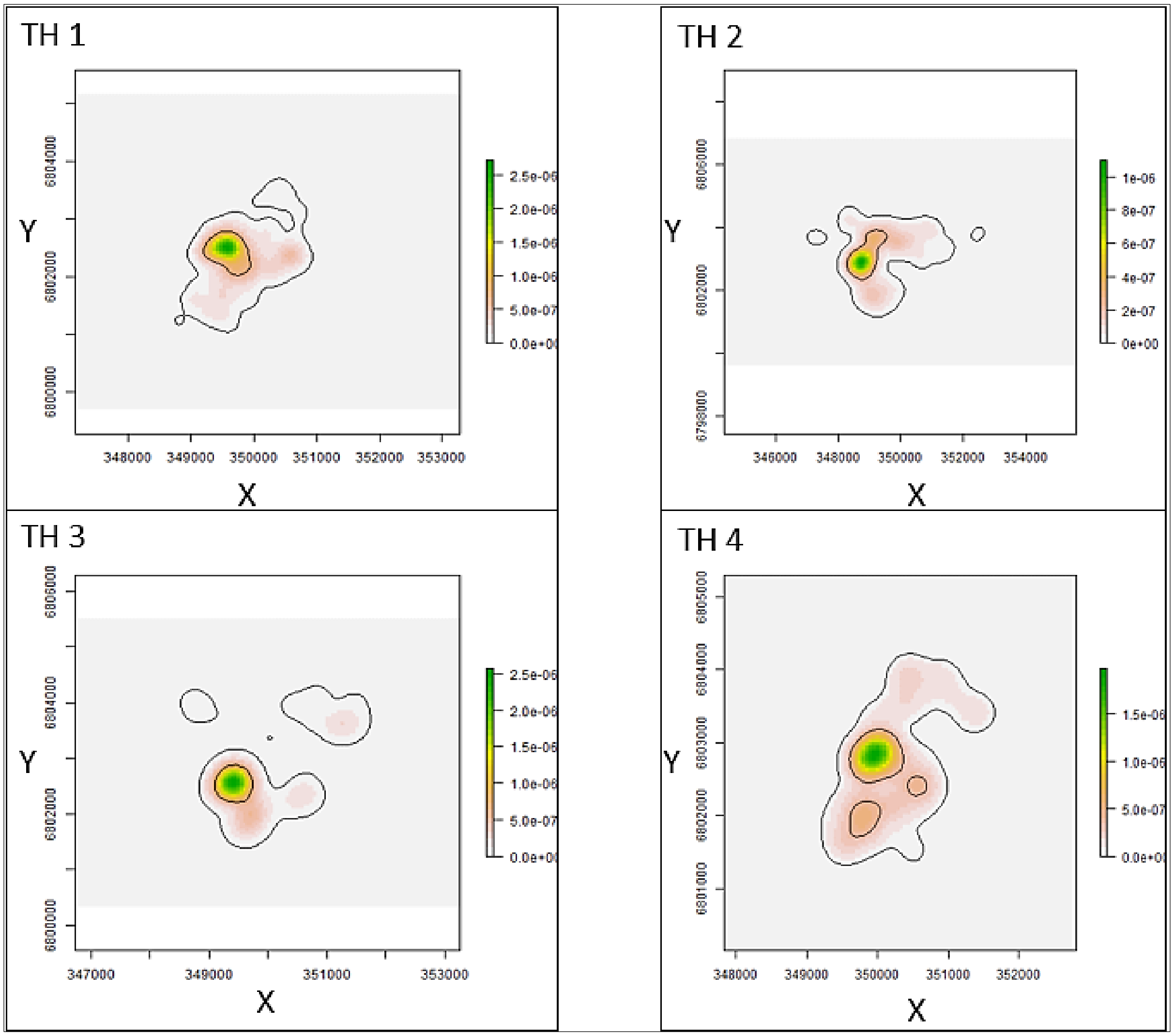

Figure 5.5: KDE estimation of core area and UD of the four Trumpeter Hornbills (TH) tagged in an urban-forest environment of Eshowe, KZN. The outer lines are 95% isopleths and the inner lines are 50% isopleths (core areas) for each individual. The 95% isopleth areas were 3.16 km^2, 8.02 km^2, 3.61 km^2 and 3.76 km^2 for TH 1, TH 2, TH 3 and TH 4 respectively.

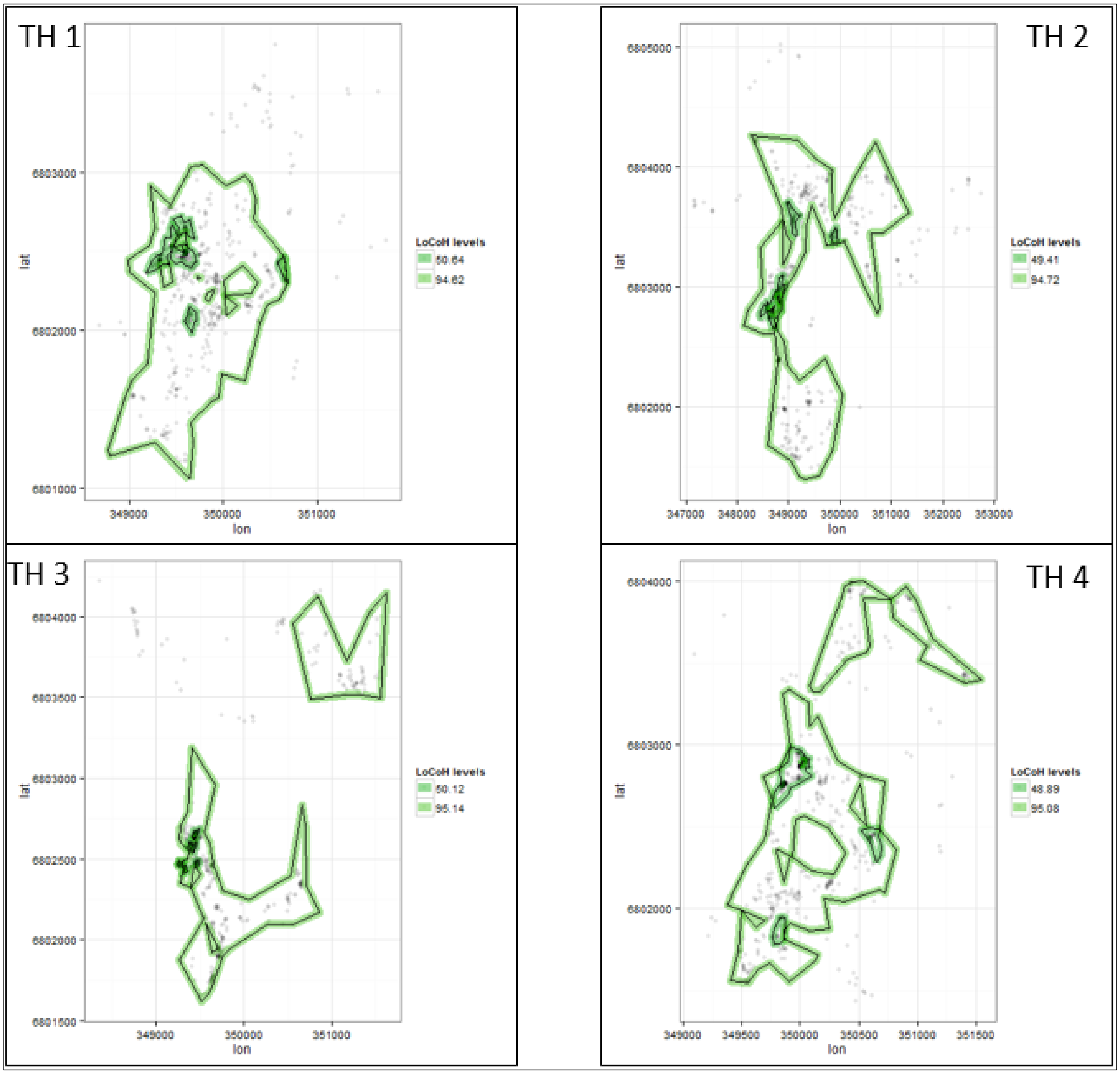

Figure 5.6: LoCoH estimation of core area and UD of the four Trumpeter Hornbills (TH) in an urban-forest environment of Eshowe, KZN**.** The 95% isopleth areas were 1.89 km^2, 3.26 km^2, 1.17 km^2 and 1.44 km^2 for TH 1, TH 2, TH 3 and TH 4 respectively.

5.4 Discussion

The tendency of individuals to return to the same area repeatedly or remain in an area for an extended period of time is referred to as site fidelity (McSweeney et al. 2007). Our results showed a degree of site fidelity by the four individual tagged Trumpeter Hornbills as they remained in the area for the entire period of the study. The observed pattern of site fidelity may indicate the productivity of the area (urban-forest mosaic of Eshowe) and the spatiotemporal variability of food resources that sustain the Trumpeter Hornbills in this environment. Another possible explanation for the site fidelity observed could be that Trumpeter Hornbills are using memory-based movements to return to previously visited sites (Janson 2000; Janson and Byrne 2007). Evidence indicates that home range can be expected if animals move randomly in the environment and keep updated records of the fruiting status of preferred spots (Van Moorter et al. 2009). However, with the changing landscapes dominated by human activities that results in changes in the local abundance of resources, then a combination of random exploration with memory-based processes could lead to shifts in the size and location of core areas or home ranges (Börger et al. 2008). As such, by exhibiting site fidelity to the urban-forest mosaic of Eshowe, it is possible that Trumpeter Hornbills are using memory-based processes to reinforce the use of a known area and at the same time using random explorations to find new sources of fruits as the environment keeps on changing. Having said that, a long term tracking of Trumpeter Hornbills (say more than fives) in the urban forest-urban environment of Eshowe is needed in order to confidently say that a combination of memory-based processes and random exploration are some of the factors helping Trumpeter Hornbills to persist in the urban-forest mosaic in addition to the existence of an intact natural forest.

With regards to TSI, two of the individuals tagged had their locations not statistically independent. Because the time interval between successive observations was long (4h), we expected independence of successive observations to be achieved for the four individuals tagged. However, the fact that animals move at random, strongly autocorrelated data sets are expected for some individuals especially when data are collected using radio telemetry (De Solla et al. 1999). As such, when estimating the core areas and UDs, we decided not to remove the autocorrelated fixes as eliminating autocorrelated locations from the data reduces the sample size and may limit the biological significance of the analysis (De Solla et al. 1999), especially as Trumpeter Hornbills usually return daily to a roosting/ nesting cavity (pers. obs.). Furthermore, the lack of spatial independence with the two individuals tagged does not violate assumptions of home range analysis or core area and UDs estimation since our primary goal was not to estimate time partitioning within the home range ((De Solla et al. 1999; Haines et al. 2006).

Using KDE and LoCoH estimation methods, the core areas and UDs estimated varied in size and shape for the individuals tagged. Because the two methods differ in constructing the UDs, we were able to get more insight on the space use by Trumpeter Hornbill in the urban-forest mosaic of Eshowe. With the KDE method, the core areas seem to coincide in the same general location whereas the LoCoH core areas are distributed within the individual's home range. The LoCoH method is considered to be superior over the KDE method when used to construct UDs because of its ability to identify hard boundaries, excluding unused areas and convergence to the true distribution as sample size increases (Getz et al. 2007). Our results showed small core areas used by Trumpeter Hornbills in an urban-forest mosaic of Eshowe. As the home range represents an area traversed by an individual in its normal activities of food gathering, mating and caring for its young (Burt 1943), small core areas observed may be explained by the abundance of food or suitable nesting and roosting sites in the urban-forest mosaic of Eshowe. The intensity of space use or activities was mainly concentrated in the main forest and the surrounding urban areas (Figure 5.2). However, some individuals (TH 2 and TH 3) made occasional sallies to forest patches or riparian woodland in the agricultural landscape.

In fragmented landscape such as the urban-forest mosaic of Eshowe, countryside riparian woodlands and forest patches in agricultural matrix provide critical habitat and connectivity for large frugivorous and forest birds like Trumpeter Hornbill. By tracking fruits in these fragmented landscapes, the Trumpeter hornbill acts as a mobile link by moving seeds across fragmented habitats (Lenz et al 2011) and in turn facilitate functional connectivity of isolated forest patches (Mueller et al. 2014). Although the short-term data are insufficient for inferring space use patterns in a given population or group, our results are important for management and conservation of Trumpeter Hornbill as we provide the first insight on the core areas and utilisation distributions of urban-forest mosaic of Eshowe. In view of this, we recommend a long-term telemetry study on Trumpeter Hornbills in an urban-forest environment that will enable us understand the variation of core areas and home ranges from year to year as food resources vary in space and time.

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CHAPTER 6

Conclusion

6.1 Introduction

This chapter discusses and summarises the main research findings in relation to the aim and objectives of the study. Based on the findings of this research, overall management and conservation recommendations are suggested and options for future research are presented.

Globally, urbanisation is spreading rapidly and this presents enormous challenges for biodiversity conservation (Marzluff et al. 2001; Marzluff and Ewing 2001; McKinney 2002; McKinney 2008). Urbanisation contributes to the loss of world's biodiversity and homogenisation of its biota (Aronson et al. 2014). With reference to global biodiversity loss, the number of mammals, fish, amphibians, reptiles and birds have declined by half since 1970 due to habitat loss and degradation, hunting and climate change as the major threats facing the world's biodiversity (WWF 2014). At present, urbanisation is taking place at a rapid rate in most parts of the developing world with the fastest growth being experienced in Sub-Saharan Africa (McHale et al. 2013; UN 2014; WWF 2014). With continued increase in the global population living in urban areas or cities (Grimm et al. 2008), natural habitats are facing enormous anthropogenic pressures and this has serious implications for the goods and services that urban ecosystems can provide (Gaston et al. 2013). Catterall (2009) indicated that human population explosion and contamination of air, water and soil are the many features shared by urban areas. Therefore, studies that analyse biodiversity in urban-forest mosaics are important for understanding how certain species of wildlife adapt and persist in urban environments (McHale et al. 2013).

The smallest biome in South African is the indigenous forest and KwaZulu-Natal (KZN) Province holds one sixth of the remaining indigenous forest (Mucina and Rutherford 2006). KZN is a unique province in that it supports both the Afromontane forest (that is, montane and mist belt) and Indian Ocean coastal belt forest (that is, dune forest, swamp forest, sand forest, riverine forest, coastal lowland forest and coast scarp forest) which are the major forest types of southern African subcontinent (Eeley et al. 1999). In addition, KZN is part of the Maputaland-Pondoland-Albany biodiversity hotspot which is recognised as one of earth's biologically rich and most endangered terrestrial ecoregions (Steenkamp et al. 2004). In the past, KZN Province and South Africa in general has been subjected to anthropogenic conversion of natural habitats to other land uses in addition to the impacts of climate change on the distribution of indigenous forest (Eeley et al. 1999; Wethered and Lawes 2003, 2005; Steenkamp et al. 2004; Mucina and Rutherford 2006). The current and possibly future biodiversity loss is and will mainly be due to land use change (Newbold et al. 2013) with conversion of natural habitats for agricultural land having the greatest impact (Green et al. 2005). Consequently, with the world population projected to grow in years to come (UN 2015) and urbanisation expected to increase (UN 2014), studies that seek to understand how wildlife adapt and persist in human dominated habitats are important for wildlife management and conservation. In South Africa, there is already recognition of the fact that there is little urban ecological research done at the moment and that more needs to be done (Cilliers and Siebert 2012). To contribute to this knowledge gap, we designed a study to look at the aspects of the ecology of Trumpeter Hornbill *Bycanistes bucinator* across the urban-forest mosaics in KZN. The Trumpeter Hornbill is the largest obligate frugivore in South Africa (Kemp and Woodcock 1995). Like many other large frugivorous hornbills, the Trumpeter Hornbill has been recognised to play an important role of long distance seed dispersal by feeding on fruits of many tropical plants (Kemp and Woodcock 1995; Kitamura 2011; Poonswad et al. 2013). However, like many other hornbill species, the Trumpeter Hornbill is threatened by anthropogenic factors that include habitat loss, international trade and hunting and possibly climate change (Jetz et al. 2007; Trail 2007; Williams et al. 2014). As the Trumpeter Hornbill is common along the coastal urban environments of KZN, understanding how it persists and survives will help in the management and conservation of the species in anthropogenic habitats. In this study, we used GPS telemetry and point count methods to collect data across the urban-forest mosaics of KZN. Study locations included the towns of Eshowe, Mtunzini and Durban.

6.2 Research findings

We formulated four separate research objectives in order to understand the aspects of the ecology of Trumpeter Hornbill across the urban-forest mosaics of KZN.

The first objective was to use point count data to investigate the factors determining the occupancy and detection probability of Trumpeter Hornbill across-urban forest mosaics of KZN. Point count is the most common and widely used method to sample birds (Marsden 1999; Diefenbach et al. 2003; Royle and Nichols 2003; Mackenzie and Royle 2005). The mean occupancy rate of Trumpeter Hornbills was 0.40 ± 0.09 with a low detection probability of 0.28 ± 0.04 (Chapter 2). In these urban-forest mosaics of KZN, we found that large trees influenced occupancy positively (sum AIC weight (ω_i) = 79%) while relative human abundance negatively influenced their occupancy (ω_i = 91%). Model selection suggested that housing density had a strong negative influence on detection probability of Trumpeter Hornbills ($\omega_i = 82\%$) and availability of fruiting trees influenced their detection positively ($\omega_i = 29\%$) (Chapter 2).

The second objective was to use data collected from tracking five individuals of Trumpeter Hornbills across the urban-forest mosaic of Eshowe to investigate their home range size and habitat use. This study was the first to provide information on the monthly and seasonal home range size and general habitat use across the urban-forest mosaic of Eshowe. GSP tracking data indicated that the overall mean monthly home range size was small (mean ± SE), 5.1 ± 1.28 km^2 (95% MCP), 4.6 ± 1.14 km^2 (95% KDE) and 1.9 ± 0. 46 km^2 (95% LoCoH), with individual variations in monthly and seasonal home range sizes (Chapter 3). GPS location data also suggested that Trumpeter Hornbills mainly used the indigenous forest and the urban gardens across the urban-forest mosaic of Eshowe (Chapter 3). The estimated overall home size for the Trumpeter Hornbill across the urban-forest mosaic of Eshowe is an important first step that will enable us to further understand the mechanisms and spatial relationships that affect habitat choices and responses of animals to environmental changes (Akçakaya 2000; Thomas and Taylor 2006)

The third objective of the research was to examine the movement pattern of Trumpeter Hornbills using telemetry data and at the same time assess the availability of fig resources in the urban environment of Eshowe. Trumpeter Hornbills feed mainly on fruits (89%) and small fruits especially figs form the bulk of its diet (Kemp and Woodcock 1995; Poonswad et al. 2013). Trumpeter Hornbills tagged showed similar patterns of movement with mean daily distances ranging from (mean ± SE) 0.47 ± 0.43 km to 1.06 ± 1.40 km (Chapter 4). We identified seven species of figs that are found in the urban environment of Eshowe from 138 trees (stems) encountered or sampled. Most common and abundant figs were *Ficus burkei* (62%) and *Ficus* *natalensis* (27%) of the seven species identified and the abundance and wide distribution of these fruiting fig trees may be one of the reasons Trumpeter Hornbills persist and utilises the urban environment of Eshowe (Chapter 4)

The fourth objective was to test for site fidelity and time to statistical independence and at the same time estimate core areas and utilisation distributions using tracking data from four individuals tagged in the urban-forest environment of Eshowe. It is assumed that if an animal exhibits site fidelity, then it has a home range (Spencer et al. 1990). Using tracking data, we established that the four individuals tagged exhibited site fidelity and that data from two individuals were not statistically independent (Chapter 5). The average core area estimated using KDE method (mean ± SE) was 0.62 ± 0.35 km^2 (range: 0.34 km^2 to 1.09 km^2). With LoCoH method, the mean core area estimated was 0.07 ± 0.04 km^2 (range: 0.01 km^2 to 0.11 km^2) (Chapter 5). There was individual variation in the utilisation distribution of the urban-forest environment and the intensity of space use was mainly concentrated in the natural forest and the surrounding urban environment (Chapter 5).

6.3 Discussion and recommendations

Human beings greatly benefit from ecosystem services provided by birds that encompasses the four types of services recognised by the United Nations Millennium Ecosystem Assessment of provisioning, regulating, cultural, and supporting services (Millennium Ecosystem Asssessment, 2005; Whelan et al., 2008). Birds provide humans with game meat and guano fertilisers, they control populations of invertebrate and vertebrate pests, they regulate carcasses and waste through scavenging and sustain plants through pollination and seed dispersal, they play an import role in some cultures through art and religion, and through their supporting services, they contribute to nutrient cycling and soil formation (Sekercioglu 2006). Although mammals can be compared with birds with regards to ecosystem services they provide, birds are generally more resilient to extirpation, have twice as many taxa and ten times more flying species (Holbrook et al. 2002; Sekercioglu 2006). However, with global human population expected to reach approximately 9.3 billion by 2050 (UN 2015) and urbanisation is predicted to grow (UN 2014), anthropogenic conversion of natural habitats will rise and the loss of biodiversity will occur at an unprecedented scale and the ecosystem services provided by birds and mammals will significantly diminish. Therefore, through urban ecological research we can have a better understanding of the relationship between nature and city residents and in turn urban ecologists can help in designing cities that augment both infrastructure and ecosystem services (Tanner et al. 2014). The research presented in this thesis attempts to contribute to urban ecological research in South Africa by examining the aspects of the ecology of Trumpeter Hornbill across urban-forest mosaics of KZN. The bird is the largest obligate frugivore in South African and still persist and survives in urban-forest mosaics of KZN especially in urban towns along the coast.

The results from the investigation of home range and habitat use of Trumpeter Hornbills across the urban-forest mosaic of Eshowe highlights the value of protecting and maintaining indigenous forests for wildlife persistence and adaptation to urban environment. In the absence of Dlinza Forest in Eshowe, it is highly likely that Trumpeter Hornbills and many other forest dependent species may withdraw from the area. In order for Ezemvelo KZN wildlife to succeed in conserving indigenous biodiversity in the urban environments of KZN for future generations, conservation efforts should be directed at ensuring that indigenous forests already designated as protected areas are well protected and should encourage the identification and designation of more protected areas in human dominated habitats. Results of home range size and habitat use in Eshowe emphasises that even a small size protected forest (e.g. Dlinza Forest) surrounded by a vegetation rich urban settlement is key in sustaining avian species and other wildlife across the urban-forest mosaic. In urban environments that are highly fragmented, the challenges are many and opportunities are few for many wildlife species and having protected natural habitats in these anthropogenic environments is a safety net for a great diversity of flora and fauna species. In addition, maintaining large frugivorous birds in modified and fragmented landscapes is key for sustaining many tropical plant species as they play an important role in long distance seed dispersal (Holbrook et al. 2002) and they act as mobile links by connecting fragmented habitats through seed dispersal (Sekercioglu 2006; Mueller et al. 2014).

Humans frequently control plant richness, evenness and density in urban areas (Tanner et al. 2014) and most likely the current and future biodiversity loss will be due to anthropogenic land use change as one of the main drivers (Newbold et al. 2013). Consequently, understanding how species are affected by land use change is necessary to guide conservation decisions. Results from point count data revealed that relative human abundance negatively influenced occupancy of Trumpeter Hornbills and housing density negatively affected their detection probability in the urban-forest mosaics of KZN. Results from point count data also indicated that availability of large trees and the presence of fruiting tree across the urban-forest mosaics of KZN are important for Trumpeter Hornbill's occurrence. These results simply support the idea of designing ecologically friendly urban settlements that discourages the complete clearance of natural vegetation when designing and building houses or other anthropogenic structures. Previous urban ecological studies have indicated that urban environments that are vegetation rich attracts or supports more wildlife species than those with poor or intensively modified vegetation (Marzluff et al. 2001; McKinney 2002, 2006; McKinney 2008; Cilliers and Siebert 2012; Magle et al. 2012; Larondelle and Haase 2013; Newbold et al. 2013; Tanner et al. 2014). KZN province being a biodiversity hotspot as it is part of the Maputaland-Pondoland-Albany (Steenkamp et al. 2004), planning for infrastructure development (for example with low housing density settlements and including gardens with trees) that reduces environmental damage is crucial for biodiversity persistence and ultimately conservation in human-dominated environments.

The Trumpeter Hornbill is a mobile animal and moving animals connect our world, spread pollen, seeds, nutrients and parasites as they go about their daily lives (Kays et al. 2015). Results on the movement patterns and cores areas of Trumpeter Hornbills across the urban-forest mosaic of Eshowe indicated that the pattern of movements were similar and they did not cover longer daily distances and individual core areas were small. Trumpeter Hornbill's intensely used spaces were mainly concentrated in Dlinza Forest and the surrounding urban gardens. The small core areas and the short daily distances covered by Trumpeter hornbills simply suggest that the urban-forest mosaic of Eshowe is highly productive and able to support the persistence of this large obligate frugivore as four of the individuals tagged exhibited site fidelity to urban-Dlinza Forest environment and only one individual moved to a nearby Entumeni Forest which is about 8 km away from Dlinza Forest. In the urban environment of Eshowe, we identified seven species of figs *Ficus* that were widely distributed and it appears that they are a key food resource drawing Trumpeter Hornbills to this urban environment as their diet is mainly fruits, especially figs (Kemp and Woodcock 1995; Poonswad et al. 2013). However, the urban environment of Eshowe is not spared from invasive and alien fruiting trees and cultivate fruits that are also eaten by Trumpeter Hornbills. Consequently, these alien plants are spread to indigenous Dlinza Forest and other isolated fragmented natural habitats in the surrounding agricultural landscapes possibly through seed dispersal by Trumpeter Hornbills. To stop this, the onus is with Ezemvelo KZN Wildlife to sensitise and educate the community not to plant alien species in their gardens.

What will determine the success and failure of avian species in human-dominated environments? Well, although the densities of most species are reduced in human-dominate environments (Alkemade et al. 2009; Phalan et al. 2011), different species respond differently to environmental changes and the responses usually depends on the species ecological and morphological traits (Newbold et al. 2013). As such, those species that will succeed in human-dominated environments needs to demonstrate greater environmental tolerance (Bonier et al. 2007).

6.4 Concluding remarks and future works

This thesis explains the persistence and utilisation of the urban-forest mosaics of KZN by Trumpeter Hornbills. The results presented here have shown that urban environments characterised by low housing density with relatively low human abundance and supports healthy natural environments with more large trees and fruiting trees are important for persistence of Trumpeter hornbills in human-dominated environments. The thesis presents necessary information with regards to home range size, core areas and habitat use of Trumpeter Hornbills across urban-forest mosaics of Eshowe. In addition, valuable information with regards to important environmental variables that affect the occupancy and detection probability of Trumpeter

Hornbills across urban-forest mosaics of KZN have been provided. However, the fact that all aspects of the ecology of Trumpeter Hornbills were not addressed, the following future works are proposed:

1. Despite our efforts to capture and fit transmitters, in the current research we only managed to tracked five individuals of Trumpeter Hornbills for less than a year and only one male was tagged. A long-term study with a greater sample size with an equal number of males and females tagged is required for further understanding of how home range sizes vary between the sexes and how home range sizes and cores areas vary annually and across seasons.
2. In this study a snap shot of fig distribution and abundance in urban Eshowe was provided. To comprehensively understand how fruiting trees influence Trumpeter Hornbills movement patterns, a long term fruiting phenology study of important tree species to Trumpeter Hornbill's diet is need across the urban-forest mosaics of KZN and linking such data to the long term movement pattern of Trumpeter Hornbills.
3. Although the global population of Trumpeter Hornbills is considered to be stable, little information on the number of individuals present both locally and globally are known. With the current anthropogenic threats of habitat loss, hunting and illegal international trade that affects many hornbill species, a study that determines a population estimate for local population of Trumpeter Hornbills in KZN is needed for their effective management and conservation and possibly evaluation of their current status as prescribed by IUCN red

list of threatened species. Furthermore, ongoing annual estimates are required to determine how the species is affected by anthropogenic change.

4. A comprehensive study that looks at the breeding biology of Trumpeter Hornbills is needed. It is still not clear as to whether the female is only fed by a single male when it is sealed in the nest during the breeding season. In addition, despite much effort nest cavities were difficult to locate in the present study. As the hornbills require secondary cavities in trees, anthropogenic effects may be affecting the availability of these. Consequently, it appears that nest sites may be limiting and this will potentially have a negative effect on the population.

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