

**The effects of fragmentation and forest structural
components on the diversity of forest bird species in
Durban, South Africa**

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ABSTRACT

The pressure to meet the demands of the growing human population has resulted in the conversion of a large proportion of the Earth's natural habitats into modified landscapes. With increasing urbanisation, generalist species may persist and thrive within the anthropogenically-modified landscapes, whereas specialist species are likely to decline in numbers and possibly become extinct. Consequently, this affects biodiversity and threatens the long-term functioning of the ecosystem as some species' functional traits are lost. Therefore, understanding the ecological requirements of species with various functional traits to persist within human-modified landscapes is crucial for biodiversity conservation. The present research was conducted in the urban mosaic of Durban (eThekweni Municipality), KwaZulu-Natal, South Africa. The landscape of the study region comprised of extensive patches of Indigenous Forest and Thicket/Dense Bush. Previous research has highlighted the dramatic loss of natural forests within Durban as a result of anthropogenic activities, such as tourism-orientated development and human settlements. However, only a few studies have documented the threats that losing these coastal forests pose to wildlife, particularly birds. With the loss of forests and increasing urbanisation in Durban, the present study aimed to investigate the (i) forest fragmentation effects via patch size and isolation distance of habitat patches on the diversity of forest birds (ii) differences in vegetation structures of Indigenous Forest (hereafter IF) and Thicket/Dense Bush (hereafter TDB; a secondary forest habitat representing regenerating IF) with the aim of showing their importance in the provisioning of habitat and a diversity of niches and resources for avian species in Protected Areas (PAs) within an urban mosaic.

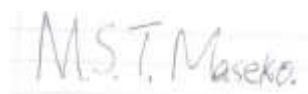
During the southern Africa breeding season between October 2016 and March 2017, we conducted bird point-count surveys in IF and TDB patches within five PAs within eThekweni Municipality. We further recorded microhabitat vegetation structure at each survey point. We recorded a total of 75 bird species. Furthermore, we recorded the species richness per patch as overall species richness, and then calculated the functional diversity of the community recorded as a difference matrix of each species' functional traits. Principally, we found that taxonomic richness was not affected by increasing isolation distance between forest patches, and that habitat patch size positively influenced taxonomic richness and functional diversity. Furthermore, the number of

avian forest specialist species increased with patch size, probably because of the diverse environmental niches and resources present in larger patches. Secondly, the overall vegetation structures and species richness of IF and TDB did not differ significantly. However, the presence of avian specialist species in TDB survey sites was of interest because TDB vegetation structure was not predicted to be ideal habitat for forest specialist species, as TBD was expected to represent secondary forested habitat. The lack of significant differences in vegetation structures between IF and TBD, and the provisioning of forest specialist species in TBD led us to conclude that the TDB in our study region is at an advanced stage of regeneration into IF. Overall, the present study highlighted the diverse avian species that may exist within the urban mosaic forests, provided that availability of specialised niches persist. Therefore, the findings of this study highlight the conservation importance of natural landscapes in human modified landscapes. Furthermore, they emphasize the necessity of legally protecting both forest and thicket dense bush.

PREFACE

The data described in this thesis were collected in Durban, Republic of South Africa, from October 2016 to March 2017. Experimental work was carried out while registered at the School of Life Sciences, University of KwaZulu-Natal, Pietermaritzburg, under the supervision of Professor Colleen T. Downs, and co-supervision of Dr David A. Ehlers Smith and Dr Yvette C. Ehlers Smith.

This thesis, submitted for the degree of Master of Science in the College of Agriculture, Engineering and Science, University of KwaZulu-Natal, School of Life Sciences, 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. The thesis is structured with each data chapter written in manuscript format with the aim to publish in scientific journals. Any repetition was unavoidable.



.....
Mfundo Sibongakonke Terrance Maseko

January 2018

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

January 2018

COLLEGE OF AGRICULTURE, ENGINEERING AND SCIENCE**DECLARATION 1 - PLAGIARISM**

I, Mfundo Sibongakonke Terrance Maseko, declare that

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

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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

MST Maseko, MM Zungu, DA Ehlers Smith, YC Ehlers Smith & CT Downs

Microhabitats and their importance to the diversity of forest birds in five Protected Areas within Durban, South Africa

Author contributions:

MSTM conceived paper with CTD, DAES and YCES. MSTM and MMZ collected data. MSTM analysed data, and wrote the paper. MMZ, DAES, YCES, & CTD contributed valuable comments to the manuscript.

Publication 2

MST Maseko, MM Zungu, DA Ehlers Smith, YC Ehlers Smith & CT Downs

The effects of habitat-patch size and patch isolation on the diversity of forest birds in Durban, South Africa.

Author contributions:

MSTM conceived paper with CTD, DAES and YCES. MSTM and MMZ collected data. MSTM analysed data, and wrote the paper. MMZ, DAE, YCES, & CTD contributed valuable comments to the manuscript.



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January 2018

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To God be the glory

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CHAPTER 1

INTRODUCTION

1.1 Urbanization

Urbanization is the main form of landscape modification by humans globally, leading to the replacement of natural ecosystems (McKinney, 2008; Coleman and Barclay, 2011; Cilliers et al., 2014) and it will probably remain the major cause for as long as humans live. Urban landscapes have been generally perceived as unsuitable environments for wildlife, but with urban expansion and research, it is shown that these landscapes have the ability to play a role in conserving and supporting a variety of wildlife species (Blaustein, 2013; Adams et al., 2014; Concepcion et al., 2015). Globally, cities are occupied by more than three billion people, with most of them located in biologically diverse regions, and although they cover only a small portion (approximately 3%) of the Earth's terrestrial surface (Dearborn and Kark, 2010; Aronson et al., 2014; Cilliers et al., 2014), their impacts to the biosphere are disproportionately large. Human population growth dictates expansion of cities geographically, and this usually forces a continuation in the use of natural environments for anthropogenic structures such as buildings (Aronson et al., 2014). Natural environments are replaced by man-made systems during the process of urbanization, and these changes are rarely reversible and can have long-term detrimental effects on both flora and fauna (Ortega-Alvarez and MacGregor-Forsb, 2009, McKinney, 2008). In recent decades, studies have successfully shown the impacts and implications of urbanization on biota, particularly at a local and regional scale (e.g. Clergeau et al., 1998; Marzluff et al., 2001; Bladley and Altizer, 2007; Widdows and Downs, 2015), but there is a need for global-scale syntheses documenting the impact of urbanization on biodiversity at a global scale, especially since it is key to understanding the impacts at both local and global scales (Aronson et al., 2014).

Different species respond differently to urbanisation processes in a way that some species will thrive and dominate urban landscapes, while others will decline in numbers and/or possibly move away from urban areas (Bonier et al., 2007; Kark et al., 2007). Heterogeneous landscapes within urban areas are a combination of vegetation components, infrastructure and anthropogenic

activities within the urban area (Ortega-Alvarez and MacGregor-Forsb, 2009). There are many ways birds respond to such environmental settings, and the usual trend observed is high abundance and lower species diversity (Blair, 1996; Kark et al., 2007; Ortega-Alvarez and MacGregor-Forsb, 2009). In recent literature, this trend is explained to be as a result of reduced predation risk and high availability of food within urban areas (Shochat, 2004; Singh and Downs, 2016). However, the presence of diverse species along urban gradients is determined by the level of urbanization (Kark et al., 2007). For example, studies have shown that diversity of birds may decrease with level of urbanisation (e.g. Blair 1996; Blair 2004). Birds that are tolerant to extreme heterogeneous landscapes thrive and persist in these areas mainly because they can exploit urban structures and resources, hence they are called urban exploiters (Kark et al., 2007; Singh and Downs, 2016). However, the thriving and persistence of some urban exploiters (e.g. crows – *Corvus spp.*) arise from their behaviour of preying on the native bird's juveniles and their nests contents and ultimately reduce reproduction success, consequently affecting the persistence of native species in urban landscapes (Marzluff et al., 2001). Furthermore, species which adapt to human modified landscapes and use some of these structures within an urban landscape are referred to as urban adapters, and these species are generally in landscapes with intermediate levels of urbanization (Lowry et al., 2011). For example, urban adapters and tolerant/exploiters species utilise building structures in urban landscapes for roosting and nesting (Singh and Downs, 2016; Widdows and Downs, 2016), and they also exploit the food readily available in rubbish bins around urban areas (Kark et al., 2007). The ability to utilize various resources and ecological niches has allowed such species with generalised functional traits to dominate urban landscapes worldwide (Kark et al., 2007). Also, other species that may persist within urban landscapes are adaptable invasive species, comprising generalist species' traits adapting to various environmental niches. Traits are individual species characteristics which are critical for survival and persisting in a given environment (Ricklefs, 1991; Ehlers Smith et al., 2017a). Shochat (2004) discuss that for most urban tolerant bird species (e.g. Common Mynas (*Acridotheres tristis*) and Hageda Ibis (*Bostrychia hagedash*) thriving in urban areas, predators and nest predation effects are not significant to cause much effect on the abundance of these species, hence we observe an increase in the abundance of urban birds. During habitat transformation in urban areas, ecological niches and resources are lost, thus resulting in certain species' habitat requirements not being met

(Marzluff et al., 2001). Therefore, these species relocate from areas with human modified landscapes in pursuit of more natural landscapes with a high diversity of niches, or areas with low disturbances of the natural environments, and these species are referred to as urban avoiders (Kark et al., 2007), and they comprise of traits which are contrary to those generalist species, i.e. specialists.

Currently, it is obvious that transformation of natural environments will continue for indefinitely and affect global biodiversity (Dearborn and Kark, 2010), particularly around already-urbanised areas. Recent literature on the effects of urbanisation shows that urban landscapes may also play a pivotal role in biodiversity conservation (Dearborn and Kark, 2010; Magle et al., 2012), if conservation is prioritised. Blaustein (2013) argues that with proper restoration of abiotic elements such as topography, soil types, water level and light in human-dominated environments, ideal landscapes to provide habitat for biodiversity (i.e. species) are viable. Therefore, it is important that all stakeholders around the world devise efficient conservation management strategies that will reduce the threat urbanisation poses on both flora and fauna. For example, if during the development of cities, creating landscapes which are key for biodiversity conservation can lead to several species being protected within urban areas (see Blaustein, 2013). For instance, development plans in Singapore included “greening Singapore” of which entailed the creation of natural environments, and this had a significantly positive effect with regards to biodiversity conservation (Blaustein, 2013) because a total of 22 natural areas were created, and four of those areas have since legally gained the status of nature reserves. Currently, the biodiversity of Singapore is hugely reliant on these natural areas as they are home to most of the country’s biodiversity (Blaustein, 2013), and such programmes as that of Singapore can be applicable in other urban areas worldwide. In addition to the crucial role these green spaces/areas within urban landscapes, they connect people to nature of which can be instrumental when it comes to establishment of nature awareness and environmental education programmes (Magle et al., 2012).

1.2 Forests

Approximately 0.25% of Southern Africa’s surface is covered by native subtropical forest, but this biome holds a disproportionately large percentage of the regions biodiversity (Low and Rebelo 1996; Eeley et al., 2001), of which include 14% of mammals and terrestrial avian species

(Geldenhuys and MacDevette, 1989). The size of forests and their vulnerability to fragmentation via anthropogenic activities hinders the development of efficient management strategies which can be vital for conservation (Lawes et al., 2000; Eeley et al., 2001). Natural tropical forests are characterised by complex multi-layered vegetation structures that constitute closed canopies of wide areas, and in South Africa the two major forest types are Indian Coastal Belt Forest and Afromontane Forest, both of which are present within the boundary of KwaZulu-Natal Province (Cooper, 1985; Eeley et al., 1999; Mucina and Rutherford, 2011). The Indian Ocean Coastal Belt Forests include the forest subtypes of swamp forest, dune forest, coast lowland forest, riverine forest, coastal scarp forest and riverine forest, whereas Afromontane Forests comprise of mixed *Podocarpus/ Afrocarpus* forest (mist belt) and *Podocarpus/ Afrocarpus* forest (montane) (Eeley et al., 1999). Contrary to the forest subtype numbers, the Afromontane forest is the oldest and thriving forest type in southern Africa (Eeley et al., 1999). The presence of such diverse forests within KwaZulu-Natal calls for urgent conservation measures which will protect both fauna and flora within these forests.

Forest threats

The state and total area covered by forests in South Africa has dramatically declined because more than half of indigenous forests have been altered for human activities, and this is more profound in KwaZulu-Natal (Macdonald, 1989; Eeley et al., 2001; Mucina and Rutherford, 2011) where > 80% of Indigenous Forest has been lost (Olivier et al., 2013). Because habitat loss arises from the disturbance, degradation and unsustainable management and consumption of forests, these are considered to be the main driver of biodiversity loss worldwide (Villard et al., 1999; Fahrig, 2001; Fahrig, 2003; Lawes et al., 2004; Secretariat of the Convention on Biological Diversity, 2010; Ehlers Smith et al., 2017a). Between the two major forest types within KwaZulu-Natal, Coastal Forests are the most threatened because of large scale transformation of natural environments for tourism-orientated development, agricultural expansion (including commercial plantations), buildings and roads (Geldenhuys and MacDevette, 1989; Low and Rebelo, 1996).

Effects of forest disturbance on avian species and traits

Many terrestrial species are threatened because of the intense transformation of their natural habitats, which consequently results in habitat loss (Turner, 1996; Munday, 2004; Fahrig, 2003; Maseko et al., 2017), but forest species are more at risk since they occupy the smallest biome which is highly susceptible to anthropogenic activities (Geldenhuys and MacDevette 1989; Eeley et al., 2001). Literature has well documented the detrimental effects of habitat conversion and transformation on fauna, particularly on birds and mammals (e.g. Marzluff et al., 2001; Benitez-Lopez et al., 2010; Ehlers Smith et al., 2017a, b; Maseko et al., 2017). In the last forty years, fragmentation has become a central issue for conservation biology (Turner, 1996; Fahrig 2003), and the effect it has on wildlife has been well documented. Fragmentation is described as a process of dividing contiguous habitat into more than one isolated segments, and this process is mainly because of intensification of human anthropogenic activities (Andren, 1994; Turner, 1996; Fahrig, 2003; Gaublomme et al., 2008; Ehlers Smith et al., 2017a). For avian species, fragmentation has been reported to have negative impacts on species richness, and overall thriving of species in fragmented landscapes (Andren, 1994; Turner, 1996; Fahrig, 2003; Maseko et al., 2017).

The Island Biogeography Theory (MacArthur and Wilson, 1967) has played a pivotal role in conservation biology/ ecology with regards to studying the effects fragmentation has in a habitat-patch species richness. The idea of the IBT mainly proposes that low or high species richness in a habitat patch is mainly driven-by the size of the patch and how isolated is the patch from other patches, particularly a “core” habitat patch containing the source population of a species (MacArthur and Wilson, 1967; Andren, 1994). Fragmentation negatively affects wildlife because it results in loss of habitat, resources and niches (Fahrig, 2003; Ehlers Smith et al., 2017a). Also, because a continuous habitat is divided, the remaining segments become small and the distance between them is increased (Andren, 1994; Fahrig, 2003). The main effect of increased isolation between patches is that the probability or rate of immigration and colonization is greatly reduced and this significantly reduces genetic diversity, thus increasing the chances of extinction (Cushman, 2006, Wilson et al., 2010; Schooley and Branch, 2011). As fragmentation reduces the size of habitat remnants, it exposes edge-preferring species to parasitism and increased predation risk (McIntyre, 1995; Broadbent et al., 2008; Gaublomme et al, 2008). Also, non-native species may eliminate habitat interior species at the edges where environmental niches may not be ideal

for specialist species, subsequently reducing the remaining habitat available to be occupied by habitat interior and specialist species (Wilson et al., 2010). This consequently increases competition for space and resources, which increases these populations' risk of becoming extinct (McIntyre, 1995; Broadbent et al., 2008).

During fragmentation, habitat heterogeneity is reduced via habitat-structure homogenisation and loss of ecological niches, and as remnants become small, the relative “edge habitat” is increased of which favours generalist species and consequently results in biological homogenisation (Andren, 1994; Fahrig, 2003; Ehlers Smith et al., 2015). Normally, species which are specialised to specific environmental niches/habitats are vulnerable during the transformation of natural landscapes in urban areas, because if the transformation of landscapes is intense, they will lose their natural habitats and subsequently decline in numbers or even become extinct as their specialist traits mean they are unable to adapt to more generalised niches (Bennum et al., 1996; McAlpine et al., 2006; Maseko et al., 2017). A growing body of literature has documented the effects landscape variables pose on biodiversity, and the overall findings report that patch size and connectivity directly influence species diversity (Turrini and Knop, 2015). However, it is mainly habitat patch size that enhances high species diversity; because of the likelihood of larger patches having diverse environmental niches to be utilised by species with different traits, allowing high species diversity. Also, there may be connectivity between larger patches and smaller patches, which facilitates dispersal success, allowing larger patches to be occupied by more migrants and colonisers, hence they will have high species diversity, especially if there are diverse niches and resource in the patch (Bender et al., 2003; Guldemon and van Aarde, 2010; Schooley and Branch, 2011). Biodiversity is crucial for long-term functioning of the ecosystems because the more diverse species in an ecosystem, the more traits available (i.e. high functional diversity), and the more ecosystem services provided (Boyer and Jetz 2014; Brose and Hillebrand, 2016; Ehlers Smith et al., 2017a) such as pest control, pollination and seed dispersal (Kremer et al, 2007; Sekercioglu, 2010). Therefore, conservation of species with diverse traits should ensure that ecosystem services are provided, and ensure the long-term functioning of the ecosystem.

1.3 Study sites

Our forest study sites were within an urban mosaic of eThekweni Municipality (i.e. Durban), KwaZulu-Natal, South Africa (Figure 1.1). The study region is within the Indian Ocean Coastal Belt (IOCB), and comprises of forest fragments which occur naturally and those fragmented as a result of anthropogenic activities (Mucina and Rutherford, 2011; Olivier et al., 2013; Ehlers Smith et al., 2017a). anthropogenic activities have resulted in a massive decrease of coastal forest extent. Also, the region has an abundance of Thicket/ Dense Bush patches, which are crucial in maintaining avian diversity and facilitating dispersal (Ehlers Smith et al., 2017a), which represents secondary/regenerating Indigenous Forest (Ehlers Smith et al., 2017a, b). Urbanisation and habitat transformation threaten the persistence of the Indian Ocean Coastal Forests, yet the IOCB has been reported to hold and support high biodiversity, including mammals and birds (Olivier et al., 2013; Ehlers Smith et al., 2017a, b).

Usually viewed as mostly urban, a significant amount of area of the 2300 km² Durban (eThekweni Municipality) landscape is made up of a mosaic of peri-urban and rural areas, mainly dictated by Apartheid and its segregation laws (eThekweni Municipality, 2010; eThekweni Municipality, 2013b; Boon et al., 2016). The approximate number of people residing in Durban is 3.5 million, with a yearly 1% growth rate (eThekweni Municipality, 2012; Boon et al., 2016). With the largest and busiest port in the continent, one would expect that the proportion of people living in poverty would relatively low, but statistics are showing that people from Durban are the ones mostly living below the poverty line in comparison to the other two largest cities in South Africa, Cape Town and Johannesburg (eThekweni Municipality, 2013; Boon et al., 2016). Additional to the pressure of the growing human population, degradation and fragmentation of natural environments, illegal development and agriculture within these landscapes as means of trying to escape poverty also pose detrimental effects on the natural environments in Durban (Boon et al., 2016). As means of mitigating the effects of the extreme development happening within the city, an open space system called Durban Metropolitan Open Space System was initiated to protect indigenous flora and fauna affected, and also to enhance the long-term quality life of city dwellers (Roberts 1994; Adams, 2005; eThekweni Municipality, 2012; eThekweni Municipality, 2013). The approximate landscape area covered by DMOSS areas is about 75 000 ha whereby 16.2% proclaimed Nature Reserves, 28% Municipal Nature Reserves, 31.3% state managed, 4.4% private

Nature Reserve, 5% Special Rating Area and 15% Non-User Conservation Servitudes (Adams, 2005; eThekweni Municipality, 2012; eThekweni Municipality, 2013). Some of the other open spaces under DMOSS are golf courses, sport fields, private gardens, sport fields and parks. The initial plan for creating DMOSS areas was to have corridors connecting all the nature reserves; which are natural landscapes/environments where conservation is the main priority (Adams, 2005). The planning of these areas was mainly shaped by the Island Biogeography Theory which explains the relationship between species richness and patch size and the importance of connectivity between habitat patches (MacArthur and Wilson, 1967; Adams, 2005). High biological diversity in the aquatic and terrestrial ecosystems within the eThekweni Municipality is constituted by the distinct abiotic factors; soils, geology, physiography, climate and biogeographical position (Boon et al., 2016). The eight major vegetation types in the eThekweni Municipality consist of more than 2000 plant species (eThekweni Municipality, 2010). The above-mentioned vegetation types comprise KwaZulu-Natal Coastal Belt, Scarp Forest, Eastern Valley Bushveld, Ngongoni Veld, KwaZulu-Natal Sandstone Sour-veld, KwaZulu-Natal Hinterland Thornveld, Mangrove Forest, Ngongoni Veld and Northern Coastal Forest (eThekweni Municipality, 2010).



Figure 1.1 Boundary map of eThekweni Municipality, KwaZulu-Natal, South Africa.

1.4 Problem statement and significance of the study

Globally, and in South Africa, conversion of natural environments for human activities has become a norm. In recent decades, globally and in South Africa, natural environments are mainly converted for practices such as agroforestry (e.g. plantations of *Eucalyptus spp.*), agriculture, livestock farming or developed into human settlements (Posa and Sodhi, 2006; Lambin and Meyfroidt, 2011; Carrara et al., 2015; Millan et al., 2015). Forests and thickets are the smallest biomes in South Africa, and with the extensive development and escalating human population growth these biomes are on the verge of being converted for human activities. Conversions of indigenous forests for human-induced modifications have posed a significant threat to forest - dependent wildlife (Naoe et al., 2012). For example, in eThekweni Municipality the human population is escalating at an unprecedented rate, of which increase the risk of natural

environments be converted, and consequently affects wildlife. However, the impacts of habitat transformations for anthropogenic activities on terrestrial wildlife are not well documented, particularly the impacts to avian communities. Furthermore, urbanisation poses a threat to IOCB, but only a few studies have tried to show the diversity of flora and fauna it holds, especially in eThekweni Municipality. Therefore, this study will explore the effects of habitat transformation within an urban mosaic, and highlight the crucial role the Indian Ocean Coastal Forests play with regards to avian biodiversity conservation.

1.6 Aim and objectives

The overall aim of the study was to investigate the effects of fragmentation and importance of forest Protected Areas on the diversity of forest birds within eThekweni Municipality, Durban, South Africa.

- Our main objective was to investigate the effects of fragmentation via measures of patch size and isolation on the taxonomic richness and functional diversity of forest bird community.
- Vegetation structures provide the diversity of niches and resources for the avian community; therefore, we aimed to investigate differences in vegetation structures of Indigenous Forest and Thicket/ Dense Bush, and their importance in providing habitat for species.
- Extinctions of certain species is mainly constituted by loss of habitat, therefore we investigated the importance of habitat patch size on forest dependent species.
- Colonization and dispersal is hindered by an increase in isolation distance between patches, therefore we investigated the effects isolation distance has on the species richness of forest birds.

The thesis is structured with each data chapter written in a manuscript format for submission to an international peer-reviewed journal. Any repetition was unavoidable.

1.7 References

- Adams, A. L., Recio, M. R., Robertson, B. C., Dickinson, K. J. M., Van Heezik, Y., 2014. Understanding home range behaviour and resource selection of invasive common brushtail possums (*Trichosurus vulpecula*) in urban environments. *Biol. Invasions* 16, 1791-1804.
- Adams, L. W., 2005. Urban wildlife ecology and conservation: a brief history of the discipline. *Urban Ecosyst.* 8, 139-156.
- Andren, H., 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. *Oikos*, 355-366.
- Aronson, M. F., La Sorte, F. A., Nilon, C. H., Katti, M., Goddard, M. A., Lepczyk, C. A., Dobbs, C., 2014. A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proc. R. Soc. B.* 281, 1-8.
- Bender, D. J., Tischendorf, L., Fahrig, L., 2003. Using patch isolation metrics to predict animal movement in binary landscapes. *Landsc. Ecol.* 181, 17-39.
- Bennun, L., Dranzoa, C., Pomeroy, D., 1996. The forest birds of Kenya and Uganda. *J. East Afr. Nat. Hist.* 85, 23-48.
- Blair, R. B., 1996. Land use and avian species diversity along an urban gradient. *Ecol. Appl.* 6, 506-519.
- Blair, R., 2004. The effects of urban sprawl on birds at multiple levels of biological organization. *Ecol. Soc.* 9, 1- 21.
- Blaustein, R., 2013. Urban biodiversity gains new converts: cities around the world are conserving species and restoring habitat. *BioScience* 63, 72-77.
- Bonier, F., Martin, P. R., Wingfield, J. C., 2007. Urban birds have broader environmental tolerance. *Biol. Lett.* 3, 670-673.
- Boon, R., Cockburn, J., Douwes, E., Govender, N., Ground, L., Mclean, C., Slotow, R., 2016. Managing a threatened savanna ecosystem KwaZulu-Natal Sandstone Sourveld in an urban biodiversity hotspot: Durban, South Africa. *Afr. Biodivers. Conserv.* 46, 1-12.
- Boyer, A. G., Jetz, W., 2014. Extinctions and the loss of ecological function in island bird communities. *Global Ecol. Biogeogr.* 236, 679-688.
- Bradley, C. A., Altizer, S., 2007. Urbanization and the ecology of wildlife diseases. *Trends Ecol. Evol.* 22, 95-102.
- Broadbent, E. N., Asner, G. P., Keller, M., Knapp, D. E., Oliveira, P. J., Silva, J. N., 2008. Forest fragmentation and edge effects from deforestation and selective logging in the Brazilian Amazon. *Biol. Conserv.* 141, 1745-1757.
- Brose, U., Hillebrand, H., 2016. Biodiversity and ecosystem functioning in dynamic landscapes. *Philos. Trans. Royal Soc. B.* 19, 371.
- Carrara, E., Arroyo-Rodríguez, V., Vega-Rivera, J. H., Schondube, J. E., de Freitas, S.M., Fahrig, L., 2015. Impact of landscape composition and configuration on forest specialist and generalist bird species in the fragmented Lacandona rainforest, Mexico. *Biol. Conserv.* 184, 117-126.
- Chibesa, M., Downs, C. T., 2017. Factors determining the occupancy of Trumpeter Hornbills in urban-forest mosaics of KwaZulu-Natal, South Africa. *Urban Ecosyst.* 1-8.
- Cilliers, S., Du Toit, M., Cilliers, J., Drewes, E., Retief, F., 2014. Sustainable urban landscapes: South African perspectives on transdisciplinary possibilities. *Landsc. Urban Plan.* 125, 260-270.

- Clergeau, P., Savard, J. P. L., Mennechez, G., Falardeau, G., 1998. Bird abundance and diversity along an urban-rural gradient: a comparative study between two cities on different continents. *Condor* 100, 413-425.
- Coleman, J. L., Barclay, R. M. R., 2011. Influence of urbanization on demography of little brown bats (*Myotis lucifugus*) in the prairies of North America. *PLoS One* 6, e20483.
- Concepción, E. D., Moretti, M., Altermatt, F., Nobis, M. P., Obrist, M. K., 2015. Impacts of urbanisation on biodiversity: the role of species mobility, degree of specialisation and spatial scale. *Oikos* 124, 1571-1582.
- Cooper, K. H., 1985. The conservation status of indigenous forests in Transvaal, Natal and O.F.S., South Africa. Wildlife Society of South Africa, Durban Ltd.
- Cushman, S. A., 2006. Effects of habitat loss and fragmentation on amphibians: a review and prospectus. *Biol. Conserv.* 128, 231-240.
- Dearborn, D. C., Kark, S., 2010. Motivations for conserving urban biodiversity. *Conserv. Biol.* 24, 432-440.
- Eeley, H. A. C., Lawes, M. J., Piper, S. E., 1999. The influence of climate change on the distribution of indigenous forest in KwaZulu-Natal, South Africa. *J. Biogeogr.* 26, 595-617.
- Eeley, H. A. C., Lawes, M. J., Reyser, B., 2001. Priority areas for the conservation of subtropical indigenous forest in southern Africa: A case study from KwaZulu-Natal. *Biodivers. Conserv.* 10, 1221-1246.
- Ehlers Smith Y. C., Ehlers Smith D. A., Seymour C. L., Thébault E., van Veen F. J. F., 2015. Response of avian diversity to habitat modification can be predicted from life-history traits and ecological attributes. *Landsc. Ecol.* 30, 1225-1239.
- Ehlers Smith, D. A., Ehlers Smith, Y. C., Downs, C. T., 2017a. Indian Ocean Coastal Thicket is of high conservation value for taxonomic and functional diversity of forest-dependent bird communities in a landscape of restricted forest availability. *For. Ecol. Manage.* 390, 157-165.
- Ehlers Smith., Y. C., Ehlers Smith D. A., Ramesh T., Downs C. T., 2017b. The importance of microhabitat structure in maintaining forest mammal diversity in a mixed land-use mosaic. *Biodivers. Conserv.* DOI: 10.1007/s10531-017-1360-6.
- eThekwin Municipality, 2010. State of biodiversity. Durban.
- eThekwin Municipality, 2012. State of biodiversity. Durban.
- eThekwin Municipality, 2013. State of biodiversity. Durban.
- Fahrig, L., 2001. How much habitat is enough? *Biol. Conserv.* 100, 65-74.
- Fahrig, L., 2003. Effects of habitat fragmentation on biodiversity. *Annu. Rev. Ecol. Evol. Syst.* 34, 487-515.
- Gaublomme, E., Hendrickx, F., Dhuyvetter, H., Desender, K., 2008. The effects of forest patch size and matrix type on changes in carabid beetle assemblages in an urbanized landscape. *Biol. Conserv.* 141, 2585-2596.
- Geldenhuys, C.J., MacDevette, D.R., 1989. Conservation status of coastal and montane evergreen forest. In: Huntley, B.J. ed. *Biotic diversity in southern Africa: Concepts conservation*. Cape Town: Oxford University Press, pp. 224-238.
- GeoterraImage, 2014., KZN Province Land-Cover Mapping (from SPOT5 Satellite imagery circa 2013). Prepared for Ezemvelo KZN Wildlife (Biodiversity Research), South Africa.

- Guldemon, R. A., van Aarde, R. J., 2010. Forest patch size and isolation as drivers of bird species richness in Maputaland, Mozambique. *J. Biogeogr.* 37, 1884-1893.
- Kark, S., Iwaniuk, A., Schalimtzek, A., Banker, E., 2007. Living in the city: can anyone become an 'urban exploiter'? *J. Biogeogr.* 34, 638-651.
- Kremen, C., Williams, N. M., Aizen, M. A., Gemmill-Herren, B., LeBuhn, G., Minckley, R., Winfree, R., 2007. Pollination and other ecosystem services produced by mobile organisms: a conceptual framework for the effects of land-use change. *Ecol. Lett.* 10, 299-314.
- Lambin, E. F., Meyfroidt, P., 2011. Global land use change, economic globalization, and the looming land scarcity. *Proc. Natl. Acad. Sci.* 108, 3465-3472.
- Lawes, M. J., Eeley, H. A. C., Piper, S. E., 2000. The relationship between local and regional diversity of indigenous forest fauna in KwaZulu-Natal Province, South Africa. *Biodivers. Conserv.* 9, 683-705.
- Lawes, M. J., Obiri, J. A. F., Eeley, H. A. C., 2004. The uses and value of indigenous forest resources in South Africa. In: Lawes, M. J., Eeley, H. A. C., Schackleton, C. M., Geach, B. G. S. (Eds.), *Indigenous forests and woodlands in South Africa: Policy, people and practice*. Pietermaritzburg: University of Natal Press, pp. 227-235.
- Low, B., Rebelo, A. G., 1996. *Vegetation of South Africa, Lesotho and Swaziland*. Pretoria, South Africa: Department of Environmental Affairs and Tourism.
- MacArthur, R. H., Wilson, E. O., 1967. *The Theory of Island Biogeography*. Princeton, New Jersey: Princeton University Press.
- Macdonald, I. A. W., 1989. Man's role in changing the face of southern Africa. In: Huntley B. J. (Ed.), *Biotic Diversity in Southern Africa: Concepts and Conservation*. Cape Town: Oxford University Press, pp. 51-77.
- MacGregor-Fors, I., Ortega-Álvarez, R., 2011. Fading from the forest: bird community shifts related to urban park site-specific and landscape traits. *Urban For. Urban Greening* 10, 239-246.
- Magle, S. B., Hunt, V. M., Vernon, M., Crooks, K. R., 2012. Urban wildlife research: past, present, and future. *Biol. Conserv.* 155, 23-32.
- Marzluff, J. M., McGowan, K. J., Donnelly, R., Knight, R. L., 2001. Causes and consequences of expanding American crow populations. In: Marzluff, J. M., Bowman, R., Donnelly, R. (Eds.), *Avian ecology and conservation in an urbanizing world*. Boston, MA: Springer, pp. 331-363.
- Maseko, M. S., Ramesh, T., Kalle, R., Downs, C. T., 2017. Response of Crested Guinea-fowl (*Guttera edouardi*), a forest specialist, to spatial variation in land use in iSimangaliso Wetland Park, South Africa. *J. Ornithol.* 158, 469-477.
- McAlpine, C. A., Rhodes, J. R., Callaghan, J. G., Bowen, M. E., Lunney, D., Mitchell, D. L., Possingham, H. P., 2006. The importance of forest area and configuration relative to local habitat factors for conserving forest mammals: a case study of koalas in Queensland, Australia. *Biol. Conserv.* 132, 153-165.
- McIntyre, N. E., 1995. Effects of forest patch size on avian diversity. *Landsc. Ecol.* 10, 85-99.
- McKinney, M. L., 2002. Urbanization, biodiversity, and conservation: the impacts of urbanization on native species are poorly studied, but educating a highly urbanized human population about these impacts can greatly improve species conservation in all ecosystems. *Bioscience* 5210, 883-890.

- Millan, C. H., Develey, P. F., Verdade, L. M., 2015. Stand-level management practices increase occupancy by birds in exotic Eucalyptus plantations. *Forest Ecol. Manage.* 336, 174–182.
- Mucina, L., Rutherford, M. C., 2011. The vegetation of South Africa, Lesotho and Swaziland. *Strelitzia* 19. Pretoria: South African National Biodiversity Institute.
- Munday, P. L., 2004. Habitat loss, resource specialization, and extinction on coral reefs. *Glob. Chang. Biol.* 10, 1642–1647.
- Naoe, S., Sakai, S., Masaki, T., 2012. Effect of forest shape on habitat selection of birds in a plantation-dominant landscape across seasons: comparison between continuous and strip forests. *J. For. Res-JPN.* 17, 219–223.
- Olivier, P. I., van Aarde, R. J., Lombard, A. T., 2013. The use of habitat suitability models and species-area relationships to predict extinction debts in coastal forests, South Africa. *Divers. Distr.* 19, 1353–1365.
- Ortega-Álvarez, R., MacGregor-Fors, I., 2009. Living in the big city: Effects of urban land-use on bird community structure, diversity, and composition. *Landsc Urban Plan.* 90, 189–195.
- Posa, M. R. C., Sodhi, N. S., 2006. Effects of anthropogenic land use on forest birds and butterflies in Subic Bay, Philippines. *Biol. Conserv.* 129, 256–270.
- Ricklefs, R.E., 1991. Structures and transformations of life histories. *Funct. Ecol.* 5, 174–183.
- Roberts, D. C., 1994. The design of an urban open-space network for the city of Durban (South Africa). *Environ. Conserv.* 21, 11–17.
- Schooley, R. L., Branch, L. C., 2011. Habitat quality of source patches and connectivity in fragmented landscapes. *Biodivers. Conserv.*, 20, 1611–1623.
- Secretariat of the Convention on Biological Diversity, 2010. *Global Biodiversity Outlook 3* (Vol. 104). Montreal, Canada: Secretariat of the Convention on Biological Diversity.
- Sekercioglu, C. H., 2010. Ecosystem functions and services. In: Sodhi, N. S., Ehrlich, P. R. (Eds.), *Conservation biology for all*. Oxford: Oxford University Press, pp. 45–72.
- Shochat, E., 2004. Credit or debit? Resource input changes population dynamics of city-slicker birds. *Oikos* 106, 622–626.
- Singh, P., Downs, C. T., 2016. Hadedda Ibis (*Bostrychia hagedash*) urban nesting and roosting sites. *Urban Ecosyst.* 19, 1295–1305.
- Turner, I. M., 1996. Species loss in fragments of tropical rain forest: a review of the evidence. *J. Appl. Ecol.* 200–209.
- Turrini, T., Knop, E., 2015. A landscape ecology approach identifies important drivers of urban biodiversity. *Glob. Change Biol.* 21, 1652–1667.
- Verburg, P. H., Overmars, K. P., Huigen, M. G., de Groot, W. T., Veldkamp, A., 2006. Analysis of the effects of land use change on protected areas in the Philippines. *Appl. Geogr.* 262, 153–173.
- Villard, M. A., Trzcinski, M. K., Merriam, G., 1999. Fragmentation effects on forest birds: relative influence of woodland cover and configuration on landscape occupancy. *Conserv. Biol.* 13, 774–783.
- Widdows, C. D., Downs, C. T., 2015. A genet drive-through: are large spotted genets using urban areas for “fast food”? a dietary analysis. *Urban Ecosyst.* 18, 907–920.
- Widdows, C. D., Downs, C. T., 2016. Urban roost temperatures of large-spotted-genets: The effect of anthropogenic structures. *J. Therm. Biol.* 57, 66–71.

Wilson, J. W., Stirnemann, R. L., Shaikh, Z. S., Scantlebury, M., 2010. The response of small mammals to natural and human-altered edges associated with Afromontane forests of South Africa. *For. Ecol. Manage.* 259, 926-931.

CHAPTER 2

The effects of habitat-patch size and patch isolation on the diversity of forest birds in

Durban, South Africa

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2.1 Abstract

Loss of habitat structural heterogeneity and area because of anthropogenic fragmentation poses a threat to the survival of wildlife, particularly in urban areas. Understanding the underlying ecological processes that influence species' distributions in fragmented landscapes is vital for conservation. Our main objective was to describe the effects of forest fragmentation via measures of patch size and isolation on the taxonomic richness and functional diversity of forest bird communities in five Protected Areas within an urban-forest mosaic in the Durban Metropolitan Area, South Africa. We conducted point counts in 41 distinct patches and recorded a total of 75 bird species. Patch size had a significant effect on all bird diversity measures, with larger patches generally having higher species diversity than small patches. Isolation distance did not have a significant influence on the diversity of birds. Habitat amount was not significant for measures of bird diversity, but based on the Akaike's weight it was important for specialist species. Shape index was significant for species richness. These results show the importance of large forest fragments/patches for the conservation of forest birds and for maintaining ecosystem functioning of forests in increasingly urbanizing landscapes.

Keywords: Forest fragmentation, taxonomic richness, functional diversity, shape index, habitat amount, habitat heterogeneity

2.2 Introduction

Anthropogenic land-use change has detrimental effects on biodiversity worldwide (Gaston et al., 2003). The main cost to biodiversity of converting natural environments for anthropogenic activities is the loss of habitat for species (Andren, 1994; Ehlers Smith et al., 2015; Maseko et al., 2017). The increase in the human population in urban areas has put direct pressure on natural environments as these areas are converted for human activities (Brown et al., 2014). Development (e.g. building of roads, houses) or land-use change within a natural habitat often poses threats to the survival of species and communities (Zurita et al., 2006); Kale et al. (2012) found that urbanization favours the persistence of a few habitat and resource-generalist species but significantly negatively affects the overall species richness of birds. Understanding the effects that human-induced modifications pose on global biodiversity has become a key aspect in landscape ecology and conservation biology (Fischer et al., 2007; Ehlers Smith et al., 2015). Despite the ongoing awareness and research on anthropogenic land-use change and fragmentation effects (e.g. Laurance 2008; Fahrig, 2013; Haddad et al., 2015; Bohnert et al., 2016; Collins et al., 2017), the conversion of natural environments for human-induced modifications is continuing at an unprecedented rate. Studies predict that at least half of the present biodiversity will be lost because of anthropogenic activities (Cushman, 2006). Most studies document the vital role that biodiversity plays in ecosystem functioning, with some suggesting that its loss may have significant impacts on the long-term functioning of ecosystems (Alvey, 2006; Boyer and Jetz 2014; Brose and Hillebrand, 2016).

Landscape modifications can cause a dramatic decline in species richness, which subsequently affects ecosystem functioning. Functional traits are individual species' attribute features and requirements which are essential for survival and thriving in a given environment (Ricklefs, 1991); ecosystem functioning is not solely dependent on taxonomic richness, but instead is mainly driven by the functional traits of all the species within an ecosystem (Flynn et al., 2009). Diversity of traits indicates the availability of resources and a variety of environmental characteristics such as landscape matrices, and the changes to traits may indicate changing environmental conditions (Ehlers Smith et al., 2017a). Thus, diverse traits are key indicators of a well-functioning ecosystem (Flynn et al., 2009; Mayfield et al., 2010); where there are changes to environmental conditions or landscape configuration, specialist species are at risk of extinction

(causing a decrease in functional diversity), consequently affecting ecosystem functioning (Tscharntke, et al., 2005; Ehlers Smith et al., 2017a).

Many studies have documented the impacts of fragmentation and its consequences on birds, detailing how bird communities are changed or affected (Andren, 1994; McIntyre, 1995). Fragmentation is a process of partitioning any contiguous habitat type into smaller and more isolated segments, mainly because of conversion for human activities (Villard et al., 1999; Fahrig 2003). Most case-studies found that factors such as escalating human population growth and intensification of anthropogenic land-use are the major causes of fragmentation (e.g. Andren, 1994; McKinney, 2002; Scharlemann et al., 2004). Fragmentation effects are habitat loss, resource and niche diversity losses, increasingly smaller habitat patches, and an increase in the distance between segmented patches of the original habitat, with an increase in “edge” habitat at the expense of internal habitat structures (Andren, 1994; Fahrig, 2003). Subsequently, this results in increased dispersal and colonisation challenges (because of habitat-patch isolation) and a homogenisation of resources and niches in smaller patches (through disturbance and “edge effects”; i.e. a decrease in the diverse structures and resources for nesting and foraging, resulting in species in small patches becoming more susceptible to invasion of alien flora and fauna, and an increase in predation and competition (McIntyre, 1995; Murcia, 1995; Broadbent et al., 2008; Gaublomme et al., 2008). These consequences result in a dramatic decline in species diversity and abundance in the fragmented habitat.

Globally, forests play a key role in provisioning and protecting biodiversity. However, it has been observed that forest habitats are transformed mainly for agriculture and human settlements (Scharlemann et al., 2004; Ehlers Smith, 2014). Severe loss and fragmentation of forests is occurring in areas of high biodiversity value, with a detrimental effect within forest fragments utilised by forest-dependent species (Scharlemann et al., 2004; Aratrakorn et al., 2006); changes in forest environments by human-induced modifications result in reduction of specific habitat structures and niches, thus threatening the survival of forest-dependent species (Bennun et al. 1996; McAlpine et al., 2006; Maseko et al., 2017). The mobility and visibility of birds, and their varied response to land-use change has helped researchers document the impacts of human-induced modifications (McIntyre, 1995).

Persistence of suitable habitat patches usually ensures that species dependent on them survive; however, the main drivers for colonisation of a particular patch by avian species are the size of a patch and isolation distances between them (Walker et al, 2003; Martensen et al., 2008). For many decades, the effects of fragmentation have been documented in reference to the Island Biogeography Theory (IBT) (MacArthur & Wilson, 1967; Andren, 1994; Norton et al., 2000; Cook et al., 2002), which states that species richness is directly influenced by size and isolation distance between patches or from “mainland” habitat patches (MacArthur and Wilson, 1967; Cook et al., 2002). However, research conducted in the past two decades shows that factors such as the configuration and amount of habitat surrounding a patch also influences species richness in fragments as it aids dispersal and colonization between patches (Norton et al., 2000; Fahrig, 2001; Fahrig, 2013). For example, if the surroundings of a patch are buildings and roads it will be more difficult for species to disperse rather than when having a matrix of natural habitat with a diversity of structures around the patch.

In many cases, the primary predictor for species richness is patch size (Walker et al., 2003; Martensen et al., 2008). Generally, specialist species avoid small patches as there is a high chance of extinction because of the lack of available resources and diversity of niches (Walker et al, 2003). Therefore, it is expected that the larger a patch, the greater the diversity of species that will occupy it, in comparison to those found in a small patch. In this study, we documented how forest fragmentation and patch configuration impacts the diversity of forest birds. We aimed to describe the effects of landscape matrices on species richness and functional diversity of forest birds within an urban mosaic. Our main objectives were to investigate the effects of (1) patch size and isolation of patches, and (2) patch shape on the taxonomic richness and functional diversity of forest-bird species in five forested Protected Areas. Since patch size and increasing isolation reduce the probability of colonization (Bender et al., 2003), we predicted that an increase in isolation distance would negatively affect bird species richness in isolated patches. Assuming larger patches provision more diverse ecological niches and resources (Soga et al., 2013), we further predicted that these patches would contain the highest avian species richness, functional diversity and the most specialist species. According to the habitat amount hypothesis, the availability of suitable habitat patches around a given/surveyed patch would positively influence species richness in that patch (Fahrig, 2013). Therefore, we predicted that the amount of habitat surrounding a surveyed

patch would positively influence avian species richness and number of forest specialist because of increased availability of niches and resources. We predicted that patch-shape irregularity would negatively affect forest specialist species because it represents an increase of “edge” habitat, in relation to interior habitat. Also, we predicted that patch shape irregularity would increase species richness as there would be an increase in edge-tolerant and generalist species. Isolation distance negatively affect sedentary species when they try to disperse (from natal range, find territory, mates etc) because they are unable to cross isolation distance (Harris and Reed, 2002; Ehlers Smith et al., 2017a). Therefore, we predicted that non-migratory species will be affected by isolation distance.

2.3 Methods

Study sites

The study was conducted in five forested Protected Areas within eThekweni Municipality (Durban), KwaZulu-Natal (KZN), South Africa. The city of Durban (29.8587° S, 31.0218° E) is on the Eastern coast of KZN, within the Indian Ocean Coastal Belt (Fig. 2.1). Like many other Metropolitan areas in the world, the remaining natural environments in Durban are on the verge of being converted for agriculture, buildings, roads and human settlements. The area of eThekweni Municipality is 2,297 km², and most of the landscape is utilised for settlements and industries (eThekweni Biodiversity Report, 2010). However, within the city there are open areas used for biodiversity conservation through the Durban Metropolitan Open Space System (D'MOSS) programme, which is concerned with either natural/Protected Areas or those restored back to natural environments with the main aim to protect fauna and flora. Currently, ~ 4 million people (and rising) reside in Durban (eThekweni Biodiversity Report, 2010; Boon et al., 2016), and with the current escalating human population growth, most of the remaining natural environments are likely to be affected because of urbanization. D'MOSS areas mainly comprise of indigenous coastal scarp and coastal lowland forests, coastal thicket/dense bush (which may be considered secondary/regenerating indigenous forest; Ehlers Smith et al., 2017a, b), and grasslands. The area has a subtropical climate and receives an annual rainfall of ~1000 mm, the majority of which falls during summer (McPherson et al., 2016). We chose five Protected Areas in eThekweni Municipality for our study with differing size as detailed in Table 2.1.

Table 2.1. The five Protected Area forest study sites within the greater Durban area, eThekwin Municipality, KwaZulu-Natal, South Africa used in the present study.

Nature reserve	Size (ha)	Coordinates	Forest type
Krantzkloof Nature Reserve	584	29°45'51"S 30°51'03"E	Coastal lowland
Kenneth Stainbank Nature Reserve	253	29°54'26"S 30°56'03"E	Coastal lowland
Palmiet Nature Reserve	98	29°49'14"S 30°55'53"E	Coastal lowland
Virginia Bush Nature Reserve	38	29°46'11"S 31°02'40"E	Coastal scarp
Pigeon Valley Nature Reserve	10	29°51'52"S 30°59'19"E	Coastal scarp

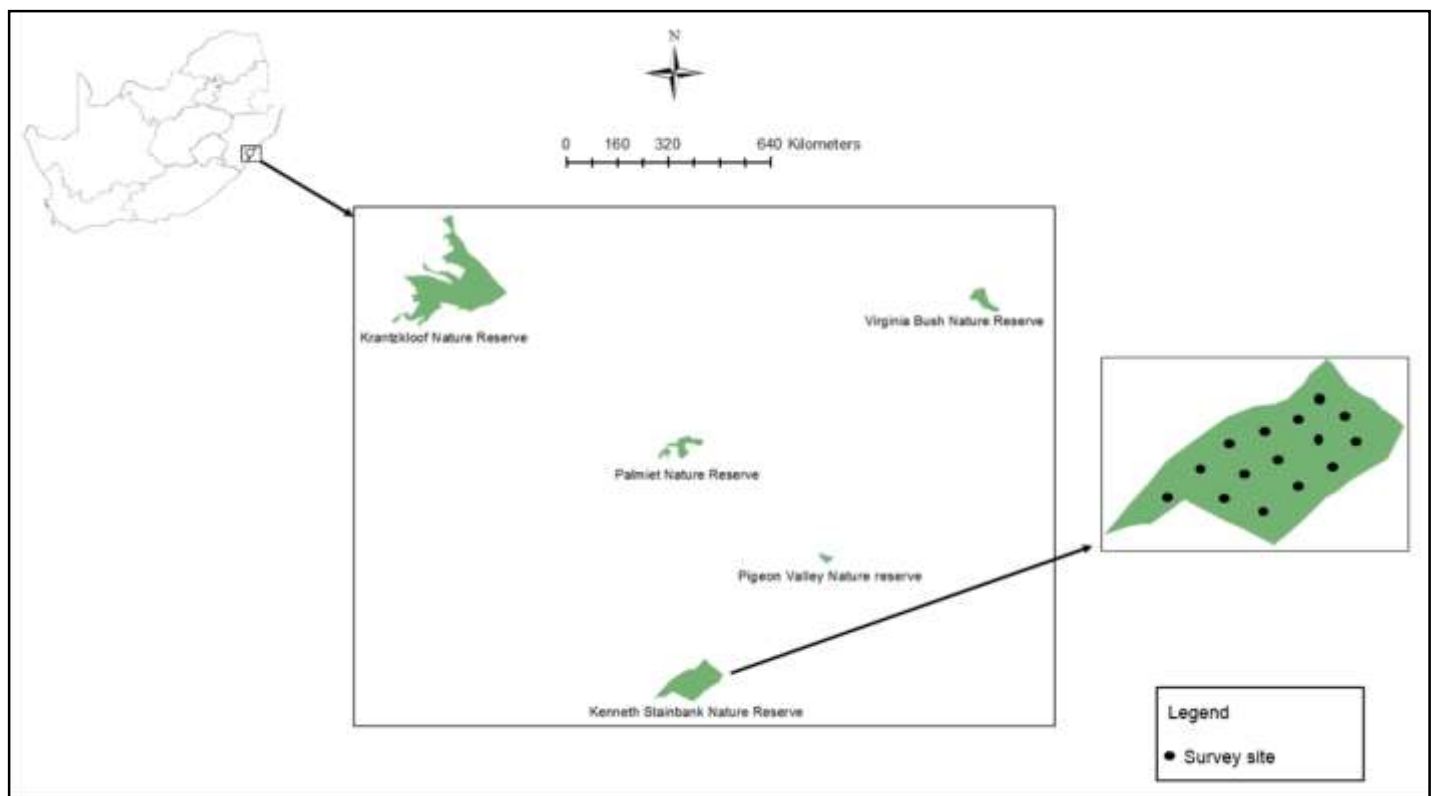


Figure 2.1. Location of the five Protected Areas (PAs) surveyed in Durban, eThekwin Municipality, KwaZulu-Natal, South Africa. An example illustrating point survey sites in one of the of the PAs is shown.

Data collection and initial analyses

Using the most recent land cover maps (GeoTerra Image, 2014), we overlaid the forest and coastal thicket/dense bush layers of our five Protected Area study sites (Table 2.1) using ArcGIS v10.2 (Environmental Systems Research Institute 2011, USA). We then created a systematic point grid-system of 200 m x 200 m, the axes of which were used as locations to conduct surveys (Fig. 2.1). The point locations were projected onto a global position system (Garmin GPSMAP 64 Mapping Handheld GPS) and located in the field. Some of the physical points were not geographically accessible, resulting in some identified points not being surveyed. However, we ensured that the sampled point was not 50 m further from the projected point and the 200 m inter-point distance was maintained. At each point, we conducted a fixed-radius point-count survey wherein we identified all bird species within a 100 m radius using visual and auditory cues. Surveys were done in the first 3 h after sunrise, and at each point we surveyed for 10 min. Each Survey was consistently done by the same three people between October 2016 and March 2017, to cover the breeding season of southern Africa. To ensure detectability of forest birds we conducted the surveys on days when there was no rain and the wind was < 4 km/h (Bibby et al., 2000). In a case where we were unsure if an individual was identified correctly, we did not record it in the list of species identified in that point survey (Ehlers Smith et al., 2015, 2017a).

Using ArcGIS v10.2 (Environmental Systems Research Institute 2011, USA), we calculated the patch size of all the distinct patches surveyed within each of our Protected Area study sites (Table 2.1). We then selected the largest patch in our survey region to be the mainland/core patch. We calculated the distance of all the other patches in relation to the core patch and recorded that distance as the isolation distance. To determine the species richness in each patch, we pooled the total number of species recorded during each survey conducted within a particular patch. We categorized birds according to their feeding guilds and then determined the number of insectivores, carnivores, frugivores, nectarivores, omnivores and granivores within each patch (Hockey et al. 2005). Additionally, we calculated the number of forest-dependent species in every patch (c.f. Oatley 1989; Hockey et al. 2005).

With reference to Hockey et al. (2005), we created a matrix of functional traits comprising body mass, foraging strategy, primary diet, and nesting strategy. We further created a presence/absence binary matrix for each distinct patch (Ehlers Smith et al., 2017a). We then

calculated the functional diversity score of each surveyed patch in package FD using R v3.3.1 (R Core Team, 2015). Functional diversity was calculated using the Functional Richness (FRic) calculation which describes functional richness as a convex hull of traits present in the community, determined using a Gower's distance matrix.

The average inter-patch distances for coastal thicket/dense bush, and indigenous forest patches is 0.66km and 0.42 km, respectively (Olivier et al., 2013). Therefore, using ArcGIS v10.2 (Environmental Systems Research Institute 2011, USA), we created a mean 0.54 km buffer around each survey patch to determine the number of available patches around our surveyed patches. Additionally, we calculated the ideal total habitat amount (i.e. area of dense bush and indigenous forests) which can be utilised by the birds within the 0.54 km buffer. Contrary to the IBT, landscape ecology highlights the importance of fragment shape (Laurence and Yensen, 1991). Therefore, to determine fragment shape and its deviation from circularity we calculated the Shape Index (SI) using a model from Paton (1975). The ordinary perimeter to area/ratio only focuses on the size of the patch, whereas the SI examines both the patch size and irregularities of the patch perimeter (Hill and Curran, 2003). The standard SI value of a circular patch is 1.0, and it increases with irregularity /complexity of a patch (Laurence and Yensen, 1991; Hill and Curran, 2003; Ewers and Didham, 2007). Perimeter and area of each fragment were calculated in ArcGIS v10.2 (Environmental Systems Research Institute 2011, USA, Table 2.2).

$$SI = P / 200(\pi.TA)^{0.5}$$

Table 2.2. Description of variables in the Shape Index model.

Abbreviation	Covariate name
P	Perimeter (m)
TA	Fragment area (ha)

Data analyses

We created General Linear Models (GLMs) in SPSS version 20.0 to determine the importance of explanatory variables (i.e. area, isolation distance, shape index and habitat amount)

on our response variables (i.e. species richness, functional diversity, individual feeding guilds and forest specialist species). We used the AIC values to select the best model (Burnham and Anderson, 2002), with the highest model for each response variable being that containing the most important explanatory variables. Also, we calculated each model's Akaike weight to determine the strength of the explanatory variables on the response variables. Furthermore, for each response variable we tested the significance of the variables in the best model against that particular response. We tested for correlation between the explanatory variables and found that there was strong correlation between (1) patch size and shape index (Spearman rank, $N = 41$, $r = 0.67$, $P < 0.001$). To avoid problems associated with multi-collinearity, we retained patch size as our explanatory measure of landscape effects on diversity. However, patch size and isolation distance had no multi-collinearity (Spearman rank, $N = 41$, $r = 0.139$, $P = 0.391$), hence we kept both as variables exploring the effects of fragmentation. We performed linear regression models of (i) patch size vs functional guilds (ii) isolation distance vs species richness, functional diversity, forest specialists and sedentary species. All statistical analyses were conducted in IBM SPSS statistics version 20.0.

2.4 Results

We conducted a total of 137 fixed-radius point-counts within 42 distinct forest patches in the five Protected Areas. The total avian species richness recorded across the five study sites was 75. Based on model selection, GLMs showed that (i) area and shape index were the most important variables for species richness (ii) area and shape index was the most important variable for functional diversity, and lastly area and habitat amount were the most important variables for forest specialists (Table 2.2). All the explanatory variables in the top models had a significant effect on the measures of bird diversity except habitat amount on forest specialists (Table 2.3). Patch size had a significant effect on functional feeding guilds: insectivores ($F_{1,40} = 82.140$, $P < 0.001$, $r^2 = 0.678$), carnivores ($F_{1,40} = 16.657$, $P < 0.001$, $r^2 = 0.30$), frugivores ($F_{1,40} = 28.571$, $P < 0.001$, $r^2 = 0.42$), nectarivores ($F_{1,40} = 20.291$, $P < 0.001$, $r^2 = 0.34$), omnivores ($F_{1,40} = 17.384$, $P < 0.001$, $r^2 = 0.31$) and granivores ($F_{1,40} = 41.059$, $P < 0.001$, $r^2 = 0.51$). Isolation distance and habitat amount had no significant effect on any measure of avian diversity ($P > 0.05$). Also, isolation distance did not have a significant effect in sedentary species ($P > 0.05$). However, habitat amount was important for forest specialist species (Table 2.3 and Figure 2.2).

Table 2.3: General Linear Models showing a summary of models for measures of bird diversity in surveyed sites within eThekweni Municipality, South Africa.

Response variable	Model	AIC	Delta AIC	AIC wgt	Model likelihood	No. of Parameters	2LL
Species richness	SI + Area	289.98	0.00	0.270	1.000	2	-140.99
	SI + Area + Isol	291.40	1.41	0.134	0.494	3	-140.70
	SI	291.49	1.51	0.127	0.471	1	-142.75
	SI + Area + HA	291.90	1.92	0.104	0.383	3	-140.95
	HA + Isol + SI + Area	292.49	2.51	0.077	0.285	4	-140.23
	Area	292.70	2.72	0.070	0.257	1	-143.35
	SI + HA	293.24	3.26	0.053	0.196	2	-142.62
	SI + Isol	293.47	3.49	0.047	0.175	2	-142.74
	Area + Isol	293.70	3.71	0.042	0.156	2	-142.85
	Area + HA + Isol	294.58	4.59	0.027	0.101	3	-142.29
Functional diversity	Area	219.126	0.00	0.263	1.000	1	-106.56
	Area + Isol	220.458	1.33	0.135	0.514	2	-106.23
	SI	220.985	1.86	0.104	0.395	1	-107.49
	SI + Area	221.104	1.98	0.098	0.372	2	-106.55
	Area + HA	221.117	1.99	0.097	0.370	2	-106.56
	Isol + HA + Area	221.851	2.72	0.067	0.256	3	-105.93
	Area + Isol + SI	222.432	3.31	0.050	0.191	3	-106.22
	SI + Isol	222.505	3.38	0.049	0.185	2	-107.25
	SI + HA	222.936	3.81	0.039	0.149	2	-107.47
	SI + Area + HA	223.095	3.97	0.036	0.137	3	-106.55
Forest specialists	SI + Area + HA	196.059	0	0.208	1.000	3	-93.03
	SI + HA	196.494	0.435	0.168	0.805	2	-94.247
	SI vs Area	196.627	0.568	0.157	0.753	2	-94.316
	SI	197.431	1.372	0.105	0.504	1	-95.716
	SI + HA + Isol + Area	197.782	1.723	0.088	0.423	4	-92.891
	SI + Isol	198.097	2.038	0.075	0.361	2	-95.049
	SI + Area + Isol	198.226	2.167	0.070	0.338	3	-94.113
	SI + HA + Isol	198.485	2.426	0.062	0.297	3	-94.243
	Area	200.472	4.413	0.023	0.110	1	-97.236
	Area + HA	200.538	4.479	0.022	0.107	2	-96.269

Index: Area, SI=Shape index, HA=Habitat amount, Isol= Isolation distance.

Table 2.4. Significance of important variables on the different measures of bird diversity, based on the top models for each response variable.

Response variable	Explanatory variables	B	St. Error	Wald Chi-square	P value
Species richness	Shape index	0.148	0.0625	5.597	0.018
	Area	0.235	0.0536	19.286	0.001
Functional diversity	Area	0.003	0.0009	13.09	0.0001
Forest specialists	Shape index	0.186	0.0652	8.112	0.004
	Habitat amount	0.002	0.003	0.626	0.429
	Area	0.075	0.0164	20.867	0.001

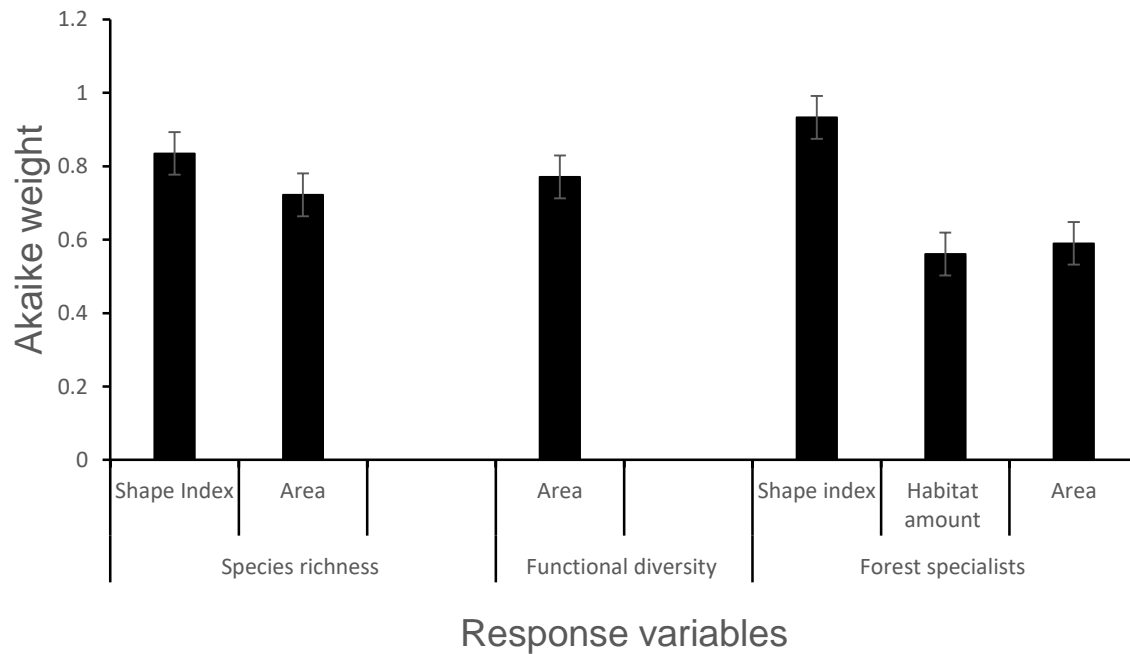


Figure 2.2. Akaike's weight of all the variables in the top models for the different measures of bird diversity in the present study.

2.5 Discussion

Our study region is highly threatened in terms of being fragmented by anthropogenic activities such as human settlements, industry and agriculture. Habitat patch-size is one of the key parameters to the persistence of species in fragmented habitats (Fahrig et al., 2003; Uezu et al., 2005; Soga et al., 2013). Our results showed that avian species richness increased with increasing forest patch size, which supported the prediction that larger forest patches would have more diverse ecological niches and resources, thus supporting diverse avifauna species. Moreover, habitat patch size was important for the occurrence of forest-dependent species. Since we assumed that larger forest patches would have more diverse ecological niches, we assumed that the strongly positive relationship between large forest patches and forest specialists was because of the availability of specific ecological niches (which are key for supporting a diverse avian community) within these patches. This finding highlights the importance of ensuring that these forest fragments are protected from any anthropogenic disturbances to ensure persistence of, and reduce the chances of losing forest-dependent species. Our data showed that patch irregularity increased with forest patch size. This indicated that larger forest patches would have more niches, resources and “edges”. The increase in edges will generally support many generalist and edge-tolerant avian species, hence avian species richness will increase (Germanie et al., 1997; Bender et al., 2003; Ewers and Didham, 2007). This supported our prediction of increasing avian species richness with patch shape irregularity. On the contrary, smaller forest patches with numerous edges are more likely to have more generalists and less specialists/low functional diversity overall (Germanie et al., 1997; Soga et al., 2013). Isolation distance had no significant effect on the overall diversity measure in our study but as this was conducted during the breeding season, we assumed initially that the presence of Palearctic, intra-African and altitudinal avian migrants (Hockey et al., 2005) in our data set was the reason we did not see any impact of isolation distance as these species would be less affected by the isolation effect. However, after testing the significance of isolation distance without Palearctic, intra-African and altitudinal avian migrants in the data set, we still found isolation distance to be insignificant.

Our results showed that functional diversity increased with patch size, thus supporting our prediction that larger forest patches would have high functional diversity, which were presumably provisioned for by the diversity of resources and niches available in the larger patches.

Importantly, taxonomic richness and functional diversity changes were correlated, indicating that as functional traits were pressured and ultimately lost by patch-size and resource and niche reduction, the species that possessed vulnerable traits were also lost. This can be problematic results in a homogenisation of communities by generalist species with similar traits at the expense of a diverse community with many specialised traits. For conservation purposes, this result is key because it shows that larger patches are not only crucial for species richness, but also for functional diversity which is essential for ecosystem functioning. Many studies are documenting the vital role that biodiversity plays in ecosystem functioning, with some suggesting that its loss could have significant impacts on the long term-functioning of ecosystems (Alvey, 2006; Boyer and Jetz 2014; Brose and Hillebrand, 2016). For example, loss of seed-dispersing bird species within an environment can have detrimental effects on the long-term survival of plants and food availability for many species, thus affecting the functioning and persistence of an ecosystem (Cordeiro and Howe, 2003). Via ecosystem services provided by birds, some important needs for the survival of humans are met (Alho, 2012). For instance, through seed dispersal and pollination birds ensure that abundant plant species are produced, and the production of plants is crucial to humans as they are important in producing medicine/drugs (Alho, 2002). Also, agricultural crops are also produced during these processes and these crops are either consumed or sold (Alho, 2002; Whelan et al., 2015), thus humans have food to sustain themselves or benefit financially. Furthermore, through recreational activities such as bird-watching humans benefit financially since people are employed for their skills when there are citizen science projects (Greenwood, 2007). Furthermore, our results showed that the richness of all functional guilds (insectivores, carnivores, frugivores, nectarivores, omnivores and granivores) increased with increasing patch size. Various food sources in larger forest patches provision for all avian feeding guilds to persist and thrive, hence we observed a positive relationship between larger forest patches and all avian feeding guilds (Uezu et al., 2005). Therefore, we concluded that larger forest patches played a significant role in ensuring the survival of diverse avian communities in an urbanised landscape.

Current studies show the importance of the habitat matrix, i.e. habitat amount and land-use surrounding fragmented forest landscapes, with regard to facilitating immigration and colonisation, which allow species to disperse and persist (Laurance, 2008; Fahrig, 2013; Neuschulz et al., 2013). We predicted that the amount of habitat available around a surveyed forest patch

would have a positive effect on avian species richness and forest specialist, but our results showed that there was no significant effect on either avian species richness nor forest specialist species. Fahrig (2013) discussed that a relationship between avian species richness and habitat amount is dependent on matrix quality surrounding the surveyed patch: high quality matrix influences the relationship between habitat amount and avian species richness whereas low quality matrix can sometimes result in the two variables having no relationship. Therefore, considering the study region (relatively high-density housing), one would expect the matrix quality of the surrounding landscape to be low, hence our results are interesting in showing that the low matrix quality around our forests is the reason there is a lack of relationship between habitat amount and overall avian species richness. Although our results did not find the significance of habitat amount in measures of bird diversity, but the Akaike weight showed the importance of habitat amount in the provision of forest specialist species. In conclusion, our study highlighted the crucial role that forests/forest fragments play in protecting and providing habitat for various bird communities. Our results show the importance of larger forest patches in ecosystem functioning and the likelihood of disappearance of forest specialist species in small patches. Therefore, because of the crucial role these forest fragments play in conservation of various bird communities, we urge South African policy makers and different stakeholders involved in decision making to ensure that the protection of natural environments (particularly forests) against anthropogenic activities is prioritised. In protecting natural forests as a country we are ensuring that the future generations also benefit through ecosystem services provided by species occupied within these forests.

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2.7 References

- Alho, C. J., 2012. The importance of biodiversity to human health: an ecological Perspective. *Estud. Av.* 26, 151-166.
- Alvey, A. A., 2006. Promoting and preserving biodiversity in the urban forest. *Urban For. Urban Green.* 5, 195-201.
- Andren, H., 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. *Oikos* 71, 355-366.
- Aratrakorn, S., Thunhikorn, S., Donald, P. F., 2006. Changes in bird communities following conversion of lowland forest to oil palm and rubber plantations in southern Thailand. *Bird Conserv. Int.* 16, 71-82.
- Bender, D. J., Tischendorf, L., Fahrig, L., 2003. Using patch isolation metrics to predict animal movement in binary landscapes. *Landsc. Ecol.* 181, 17-39.
- Bennun, L., Dranzoa, C., Pomeroy, D., 1996. The forest birds of Kenya and Uganda. *J. East Afr. Nat. Hist.* 85, 23-48.
- Bibby, C., Burgess, N., Hill, D., 2000. Bird census techniques. Academic Press, London.
- Böhnert, T., Wenzel, A., Altenhövel, C., Beeretz, L., Tjitrosoedirdjo, S. S., Meijide, A., Kreft, H., 2016. Effects of land-use change on vascular epiphyte diversity in Sumatra Indonesia. *Biol. Conserv.* 202, 20-29.
- Boon, R., Cockburn, J., Douwes, E., Govender, N., Ground, L., Mclean, C., Slotow, R., 2016. Managing a threatened savanna ecosystem KwaZulu-Natal Sandstone Sourveld in an urban biodiversity hotspot: Durban, South Africa. *Afr. Biodivers. Conserv.* 46, 1-12.
- Boyer, A. G., Jetz, W., 2014. Extinctions and the loss of ecological function in island bird communities. *Global Ecol. Biogeogr.* 236, 679-688.
- Broadbent, E. N., Asner, G. P., Keller, M., Knapp, D. E., Oliveira, P. J., Silva, J. N., 2008. Forest fragmentation and edge effects from deforestation and selective logging in the Brazilian Amazon. *Biol. Conserv.* 141, 1745-1757.
- Brose, U., Hillebrand, H., 2016. Biodiversity and ecosystem functioning in dynamic landscapes. *Philos. Trans. Royal Soc. B*, 19, 371.
- Brown, M. L., Donovan, T. M., Schwenk, W. S., Theobald, D. M., 2014. Predicting impacts of future human population growth and development on occupancy rates of forest-dependent birds. *Biol. Conserv.* 170, 311-320.
- Burnham, K.P., Anderson, D.R., 2002. Model selection and multimodel inference: a practical information-theoretic approach. Springer, New York.
- Collins, C. D., Banks-Leite, C., Brudvig, L. A., Foster, B. L., Cook, W. M., Damschen, E. I., Holt, R. D., 2017. Fragmentation affects plant community composition over time. *Ecography* 401, 119-130.
- Cook, W. M., Lane, K. T., Foster, B. L., Holt, R. D., 2002. Island theory, matrix effects and species richness patterns in habitat fragments. *Ecol. Lett.* 5, 619-623.
- Cordeiro, N. J., Howe, H. F., 2003. Forest fragmentation severs mutualism between seed dispersers and an endemic African tree. *Proc. Nat. Acad. Sci.* 100, 14052-14056.
- Cushman, S. A., 2006. Effects of habitat loss and fragmentation on amphibians: a review and prospectus. *Biol. Conserv.* 128, 231-240.

- Ehlers Smith D. A., 2014. The effects of land-use policies on the conservation of Borneo's endemic *Presbytis* monkeys. *Biodivers. Conserv.* 23, 891–908.
- Ehlers Smith Y. C., Ehlers Smith D. A., Seymour C. L., Thébault E., van Veen F. J. F., 2015. Response of avian diversity to habitat modification can be predicted from life-history traits and ecological attributes. *Landsc. Ecol.* 30, 1225–1239.
- Ehlers Smith, D. A., Ehlers Smith, Y. C., Downs, C. T. 2017a. Indian Ocean Coastal Thicket is of high conservation value for taxonomic and functional diversity of forest-dependent bird communities in a landscape of restricted forest availability. *For. Ecol. Manage.* 390, 157–165.
- Ehlers Smith., Y. C., Ehlers Smith D. A., Ramesh T., Downs C.T., 2017b. The importance of microhabitat structure in maintaining forest mammal diversity in a mixed land-use mosaic. *Biodivers. Conserv.* DOI: 10.1007/s10531-017-1360-6.
- ESRI Environmental Systems Research Institute 2011 ArcGIS Desktop 10.
- eThekweni Municipality, 2010. State of biodiversity. Durban.
- Ewers, R. M., Didham, R. K., 2007. The effect of fragment shape and species' sensitivity to habitat edges on animal population size. *Conserv. Biol.* 214, 926-936.
- Fahrig, L., 2001. How much habitat is enough? *Biol. Conserv.* 100, 65-74.
- Fahrig, L., 2003. Effects of habitat fragmentation on biodiversity. *Ann. Rev. Ecol. Evol. Syst.* 34, 487-515.
- Fahrig, L., 2013. Rethinking patch size and isolation effects: the habitat amount hypothesis. *J. Biogeogr.* 409, 1649-1663.
- Fischer, J., Lindenmayer, D. B., 2007. Landscape modification and habitat fragmentation: a synthesis. *Glob. Ecol. Biogeogr.* 163, 265-280.
- Flynn, D. F., Gogol-Prokurat, M., Nogeire, T., Molinari, N., Richers, B. T., Lin, B. B., DeClerck, F., 2009. Loss of functional diversity under land use intensification across multiple taxa. *Ecol. Let.* 12, 22-33.
- Gaston, K. J., Blackburn, T. M., Goldewijk, K. K., 2003. Habitat conversion and global avian biodiversity loss. *Proc. R Soc. Lond. B Biol. Sci.* 270, 1293-1300.
- Gaublomme, E., Hendrickx, F., Dhuyvetter, H., Desender, K., 2008. The effects of forest patch size and matrix type on changes in carabid beetle assemblages in an urbanized landscape. *Biol. Conserv.* 141, 2585-2596.
- GeoterraImage, 2014. KZN Province Land-Cover Mapping (from SPOT5 Satellite imagery circa 2013). Prepared for Ezemvelo KZN Wildlife (Biodiversity Research), South Africa.
- Germaine, S. S., Vessey, S. H., Capen, D. E., 1997. Effects of small forest openings on the breeding bird community in a Vermont hardwood forest. *Condor* 708-718.
- Greenwood, J. J., 2007. Citizens, science and bird conservation. *J. Ornithol.* 148, 77-124.
- Haddad, N. M., Brudvig, L. A., Clobert, J., Davies, K. F., Gonzalez, A., Holt, R. D., Cook, W. M. 2015. Habitat fragmentation and its lasting impact on Earth's ecosystems. *Sci. Adv.* 12, 1-9.
- Harris R.J., Reed, J.M., 2002. Behavioral barriers to non-migratory movements of birds. *Ann. Zool. Fenn.* 39, 275–290.
- Hill, J. L., Curran, P. J., 2003. Area, shape and isolation of tropical forest fragments: effects on tree species diversity and implications for conservation. *J. Biogeogr.* 309, 1391-1403.
- Hockey, P.A.R., Dean, W.R.J. Ryan, P.G., 2005 Roberts birds of southern Africa. 7th edition. John Voelcker Bird Book Fund, Cape Town, South Africa.

- Kale, M., Dudhe, N., Kasambe, R., Chakane, S., Bhattacharya, P., 2012. Impact of urbanization on avian population and its status in Maharashtra State, India. *IJAES*. 71, 59-76.
- Laurance, W. F., Yensen, E., 1991. Predicting the impacts of edge effects in fragmented habitats. *Biol. Conserv.* 551, 77-92.
- Laurance, W.F., 2008 Theory meets reality: how habitat fragmentation research has transcended island biogeographic theory. *Biol. Conserv.* 141, 1731–1744.
- MacArthur, R.H., Wilson, E.O., 1967. *The Theory of Island Biogeography*. Princeton University Press, Princeton, New Jersey.
- Martensen, A. C., Pimentel, R. G., Metzger, J. P., 2008. Relative effects of fragment size and connectivity on bird community in the Atlantic Rain Forest: implications for conservation. *Biol. Conserv.* 141, 2184-2192.
- Maseko, M. S., Ramesh, T., Kalle, R., Downs, C. T., 2017. Response of crested guinea-fowl (*Guttera edouardi*), a forest specialist, to spatial variation in land use in iSimangaliso Wetland Park, South Africa. *J. Ornithol.* 158, 469-477.
- Mayfield, M. M., Bonser, S. P., Morgan, J. W., Aubin, I., McNamara, S., Vesk, P. A., 2010. What does species richness tell us about functional trait diversity? Predictions and evidence for responses of species and functional trait diversity to land-use change. *Glob. Ecol. Biogeogr.* 19, 423-431.
- McAlpine, C. A., Rhodes, J. R., Callaghan, J. G., Bowen, M. E., Lunney, D., Mitchell, D. L., Possingham, H. P., 2006. The importance of forest area and configuration relative to local habitat factors for conserving forest mammals: a case study of koalas in Queensland, Australia. *Biol. Conserv.* 132, 153-165.
- McIntyre, N. E., 1995. Effects of forest patch size on avian diversity. *Landsc. Ecol.* 10, 85-99.
- McKinney, M. L., 2002. Urbanization, biodiversity, and conservation: the impacts of urbanization on native species are poorly studied, but educating a highly urbanized human population about these impacts can greatly improve species conservation in all ecosystems. *Bioscience* 5210, 883-890.
- McPherson, S. C., Brown, M., Downs, C. T., 2016. Crowned eagle nest sites in an urban landscape: Requirements of a large eagle in the Durban Metropolitan Open Space System. *Landsc. Urban Plan.* 146, 43-50.
- Murcia, C., 1995. Edge effects in fragmented forests: implications for conservation. *Trends Ecol. Evol.* 10, 58–62.
- Neuschulz, E.L., Brown, M. Farwig, N., 2013 Frequent bird movements across a highly fragmented landscape: the role of species traits and forest matrix. *Anim. Conserv.* 16, 170–179.
- Norton, M. R., Hannon, S. J., Schmiegelow, F. K., 2000. Fragments are not islands: patch vs landscape perspectives on songbird presence and abundance in a harvested boreal forest. *Ecography* 23, 209-223.
- Oatley, T.B., 1989. Biogeography of the forest avifauna in South Africa. In: Geldenhuys, C.J. ed. *Biogeography of the mixed evergreen forests of southern Africa*. Occasional Report No. 45. Pretoria, Foundation for Research and Development, CSIR, pp. 48-59.
- Olivier, P. I., van Aarde, R. J., Lombard, A. T., 2013 The use of habitat suitability models and species-area relationships to predict extinction debts in coastal forests, South Africa. *Divers. Distr.* 19, 1353–1365.

- Patton, D. R., 1975. A diversity index for quantifying habitat "edge". *Wildl. Soc. Bull.* 34, 171-173.
- R Core Team 2015 R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.
- Ricklefs, R. E., 1991. Structures and transformations of life histories. *Funct. Ecol.* 5, 174–183.
- Scharlemann, J. P., Green, R. E., Balmford, A., 2004. Land-use trends in Endemic Bird Areas: global expansion of agriculture in areas of high conservation value. *Glob. Change Biol.* 10, 2046-2051.
- Soga, M., Kanno, N., Yamaura, Y., Koike, S., 2013. Patch size determines the strength of edge effects on carabid beetle assemblages in urban remnant forests. *J. Insect Conserv.* 172, 421-428.
- Tscharntke, T., Klein, A. M., Kruess, A., Steffan-Dewenter, I., Thies, C., 2005. Landscape perspectives on agricultural intensification and biodiversity and ecosystem service management. *Ecol. Lett.* 8, 857–874.
- Uezu, A., Metzger, J. P., Vielliard, J. M., 2005. Effects of structural and functional connectivity and patch size on the abundance of seven Atlantic Forest bird species. *Biol. Conserv.* 123, 507-519.
- Villard, M. A., Trzcinski, M. K., Merriam, G., 1999. Fragmentation effects on forest birds: relative influence of woodland cover and configuration on landscape occupancy. *Conserv. Biol.* 13, 774-783.
- Walker, R. S., Novaro, A. J., Branch, L. C., 2003. Effects of patch attributes, barriers, and distance between patches on the distribution of a rock-dwelling rodent *Lagidium viscacia*. *Landsc. Ecol.* 182, 185-192.
- Whelan, C. J., Şekercioğlu, Ç. H., Wenny, D. G., 2015. Why birds matter: from economic ornithology to ecosystem services. *J. Ornithol.* 156, 227-238.
- Wiens, J. A., 1995. Habitat fragmentation: island v landscape perspectives on bird conservation. *Ibis* 137, 97-104.
- Zurita, G. A., Rey, N., Varela, D. M., Villagra, M., Bellocq, M. I., 2006. Conversion of the Atlantic Forest into native and exotic tree plantations: Effects on bird communities from the local and regional perspectives. *For. Ecol. Manage.* 2351, 164-173.

CHAPTER 3

Microhabitats and their importance to the diversity of forest birds in five Protected Areas within Durban, South Africa

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3.1 Abstract

Forest disturbance through anthropogenic activities poses a significant threat to the persistence and thriving of wildlife. Generally, disturbance in natural environments alters the vegetation structure, and consequently affects ecologically-dependent avian species. We investigated the presence of various vegetation structures and microhabitats covariates on the diversity of avian forest species in Durban, eThekweni Municipality, South Africa. During the breeding season, we conducted 135 bird point counts between Indigenous Forest (IF) and Thicket Dense Bush (TDB; a secondary wooded-habitat that represents regenerating Indigenous Forest) vegetation-class sites, and further recorded microhabitat covariates in each survey site. We compared taxonomic richness of IF and TDB. An RLQ analysis was conducted to examine the association between avian functional traits and microhabitat structures present at each survey site. There was no significant difference in vegetation structure and species richness between IF and TDB. Species with specialised ecological niches (e.g. large-bodied frugivores; cavity-nesters) were present in both IF and TDB. This may indicate that TDB in Durban represents an advanced stage of forest regeneration. Our study shows the importance of both IF and TDB in terms of providing habitat for both avian generalist and specialist functional traits. More importantly, it shows that forested areas under conservation management in urbanised areas allows species with different traits to persist and thrive.

Keywords: Indigenous forest, thicket dense bush, forest specialists, forest complexity, species richness, traits

3.2 Introduction

Globally, biodiversity loss remains one of the major challenges in conservation biology (Fischer and Lindenmayer, 2007; Sliwinski et al., 2016). One of the core problems associated with biodiversity loss is the intensification of land use which is accelerated by escalating human population growth (Falcucci et al., 2007; Brown et al., 2014). The main consequences of habitat destruction and land-use change are habitat loss and reduced habitat availability for species (Fahrig, 2003). As a result, loss of natural habitat is the main threat to the conservation of many terrestrial species (Villard et al., 1999; Maseko et al., 2017). For several decades, Protected Areas (PAs) have been vital for ensuring that plant and animal species are protected from human activities (Verburg et al., 2006). PAs are often portrayed as natural landscapes where disturbances to natural environments do not occur. However, studies have shown that anthropogenic activities within these areas could pose a significant threat to biodiversity (Verburg et al., 2006; Maseko et al., 2017), and that many PAs are degraded (Nelleman et al., 2007).

Globally, urbanization continues to threaten biodiversity, especially in developing countries such as South Africa (McKinney 2002; Chase and Walsh, 2004). Due to rapid changes in urban landscapes, it is important that research is conducted and that conservation efforts focus on the remaining natural habitat fragments within urban areas, as these landscapes play an important role in biodiversity conservation within the urban mosaic, and may provision natural ecosystem functioning within anthropogenic landscapes (Aronson et al., 2014; Brown et al., 2014). The rapid increase in human population growth directly increases the demand for agriculture expansion and more human settlements (Villasenor et al., 2016). It is projected that by the year 2030, an additional 2 billion people will be residing in cities worldwide (McDonald et al., 2008; UN, 2015). Consequently, the conversion of natural environments will increase to meet the needs of the growing human population. Subsequently, specific ecological niches for some species will be completely removed, thus resulting in a decline in species diversity and high chance of extinction for species with specialised ecological niches (McKinney 2002; Maseko et al., 2017).

Most current landscape-ecology research is documenting the effects that fragmentation has on biodiversity, including avian diversity (e.g. Betts et al., 2014; Bregman et al., 2014; Hermes et

al., 2016). The process of fragmentation occurs when formerly intact habitats are divided into smaller and more isolated segments (Andren 1994; Fahrig 2003). During this process, there is a change in the structural configuration of the habitat fragment (Villard et al., 1999). Land-use change and fragmentation research is usually conducted in areas fragmented for the expansion of agriculture or areas where the fragmented landscape is surrounded by land used for agricultural purposes (e.g. Carrara et al., 2015; Ehlers Smith et al., 2015). Results from research where effects of natural conversions occurred because of agriculture are not ideal proxies when documenting land-use change in urban areas (FitzGibbon et al., 2007). However, this is because unlike in agriculture dominated landscapes, the matrix around fragments in urban areas is made up of human-induced modifications such as extensive buildings and roads (McAlpine et al., 2006; Villasenor et al., 2015). This impermeable environment surrounding the forest fragments in urban landscapes usually affects mobility of species when attempting to move from one fragment to another, consequently affecting colonisation and dispersal for some species (Renjifo 2001; McAlpine et al., 2006; FitzGibbon et al., 2007).

Forest structural complexity plays an important role in maintaining high biodiversity of species in forests (Dewalt et al., 2003). Understanding the significant role of forest characteristics in the distribution of species is important for conservation of forests and biodiversity (Gil-Tena et al., 2007). Globally, forest structures of many fragments (both large and small) have been changed because of anthropogenic activities such as deforestation and fragmentation (Watson et al., 2004; Diaz et al., 2005). Several studies have shown that changes in forest structural components (e.g. logs, snags and canopy trees) significantly affects the abundance of many terrestrial species, particularly bird communities (Villard et al., 1999; Diaz et al., 2005; Fischer and Lindenmayer, 2007; Carrara et al., 2015; Hermes et al., 2016; Ehlers Smith et al., 2017c, d). For example, snags play a crucial role in the success of nesting for many cavity-nesting bird species (Mannan et al., 1980; Tomasevic and Marzluff, 2017), therefore any disturbance affecting logs consequently threatens the persistence of avian communities within a forest fragment. Moreover, changing the landscape matrix by deforestation reduces nesting sites and increase the risk of nest predation because of edge and area effects (Parker et al., 2005).

To determine the importance of microhabitats with regards to conservation, it is important to conduct research focusing on forest structural components on a broader spectrum. In this study,

we investigated the effects microhabitats on the diversity of forest birds in forests within a South African Metropolitan, Durban. The coastal forests we studied mainly comprise of Indigenous forest (IF) and Thicket Dense Bush (TDB) vegetation; (i) IF traits include indigenous trees with a mean canopy height greater than 5 metres (m) with the canopy covering large areas, and diverse understory vegetation structures (ii) while TDB traits include bush dominated areas and indigenous trees with the average height of 2-5m (also with a canopy covering large areas) and reduced structural complexity (Ehlers Smith et al., 2017a, b). However, both classifications comprise > 75% canopy cover, and in IF patches in southern KwaZulu-Natal, tree species overlap by ~77%, and pioneer vs climax tree species in TDB follows the forest regeneration chronosequence, indicating that TDB is secondary/regenerating IF (Ehlers Smith et al., 2017a, b). We examined whether there was a difference in bird species richness and vegetation structures between IF and TDB in the Durban Metropolitan. Based on previous research (c.f. Ehlers Smith et al., 2017a, b), we hypothesized that there would be a significant difference in IF and TDB vegetation structures, and consequently, avian species richness would differ between IF and TDB. Specifically, our study sought to investigate the importance of IF and TDB for the diversity of forest birds in the five Protected Areas within an urban mosaic, and specifically describe the role forested habitats play in conserving biodiversity of birds within an urban mosaic.

3.3 Materials and methods

Study site

We conducted the study in five protected areas (PAs) within Durban, KwaZulu-Natal, South Africa (Table 2.1). The city of Durban is one of the most populated city in South Africa (Boon et al., 2016) and this increases the risk of converting the current available natural landscapes for anthropogenic activities such as human settlements and agriculture. The human population continue to increase in Durban (eThekweni Municipality, 2010), but the eThekweni Municipality continue to strive in protecting the green areas under the Durban Metropolitan Open Space System (DMOSS). These urban green areas significantly contribute in conservation of wildlife species in a way that they are suitable habitats for some species (MacGregor-Fors and Ortega-Alvarez, 2011). Within DMOSS areas some of the vital indigenous vegetation are KwaZulu-Natal Sandstone Sourveld and riverine forests. IF are mainly in PAs whereas TDB are widely distributed in PA, human

modified areas and agricultural landscapes (Olivier et al., 2013; GeoTerra Image, 2014; Ehlers Smith et al., 2017a).

Data collection

We created a 200 m x 200 m systematic grid after overlaying the forest and dense bush layers of the five PAs in ArcGIS ArcMap v10.2 (Environmental Systems Research Institute 2011, USA, Chapter 2). The cross-points of each grid-line were selected as point locations for conducting surveys. Survey points in the field were located using global position system (Garmin GPSMAP 64 Mapping Handheld GPS). It was not geographically possible to reach some points, therefore we ensured that the sampled point was not more than 50 m away from the actual point. Using visual and audio cues, we conducted fixed-radius point-surveys to record all the bird species within a 100 m radius and these were consistently done by three people. To ensure that we cover the breeding season, point surveys were conducted between October 2016 and March 2017. The point surveys were done during the first 3 h after sunrise and the time interval for each point was 10 min. The ability to detect birds is reduced when there is rain or wind, therefore all surveys were done on dry and wind-still days (Bibby et al., 2000; Ehlers Smith et al., 2015). If we were uncertain about any species during the survey, we did not record it.

Habitat data

To determine the ecological habitat of forest-bird species, we recorded the local microhabitat-scale variables in a 20 m radius at each sampling point (Bibby et al, 2000; Ehlers Smith et al, 2015, 2017 a, b, c, d). The microhabitat variables recorded were: average height of herbaceous plants, grass, trees and scrub. The vertical section at each sampling point was divided into vegetation height bands to estimate the percentage cover of foliage in that particular height band. Furthermore, we recorded the number of trees and the height of the tallest tree at each sampling point using a measuring rule (Ehlers Smith et al., 2015, 2017a, b, c, d).

Avian functional traits

We generated a functional trait list incorporating the species primary diet (insectivory; nectarivory; granivory; carnivory; omnivory; frugivory) and feeding strategy (terrestrial probe; arboreal probe; harvest [seed, nectar, fruit]; various [omnivory]; perch and swoop; hawk and glean), species body mass [(very small: <10 g; small: 10.1–50 g; medium: 50.1–100 g; medium-

large: 100.1–500 g; large: >500 g)] and nesting strategy (cavity; ground; platform; bowl/ball/oval/cup) (Hockey et al., 2005; Ehlers Smith et al., 2017a). These traits are indicators of how bird species utilise niches and resources in the landscape, and indicate what niches and resources the habitat can provision for (Flynn et al., 2009).

Data analyses

Foliage profile of vegetation classes were converted into foliage height diversity using the Shannon-Weiner Formula (SWDI) to examine the habitat diversity of each survey site,

$$H = - \sum p_i \ln(p_i)$$

and p is a segment/portion of the total foliage in the i th layer) (Bibby et al., 2000). There was no multi-collinearity between all the explanatory variables, thus we retained all the variables. The averages of all site covariates were not normally distributed (Kolmogorov-Smirnov, $P < 0.05$). We performed Mann Whitney U tests to compare the averages of local site variables of both TDB and IF.

Our data set met all the rarefaction assumptions, hence we conducted rarefaction analyses in EstimateS 9.1 (Gotelli and Colwell, 2011). Rarefaction curves of the two sites (i.e. IF and TDB) were conducted, and to test if there was a significant difference between the two sites species richness, we checked using the 95% confidence intervals of the sample with a larger sample size (i.e. IF). If the rarefaction curve of a smaller sample (i.e. TDB) is within the 95% confidence interval of the larger sample, then there would be a significant difference between the two samples sites species richness, vice versa (Gotelli and Colwell, 2011). In the programme PAST, we conducted non-metric multidimensional scaling (MDS) and similarity percentage analysis (SIMPER) (Harper and Ryan, 2001). Using presence/absence (Jaccard index), we used MDS to visually show the difference in survey sites between TDB and IF. The statistical difference between TDB and IF species richness was tested using rarefaction curves, whereas SIMPER showed avian species accountable for the similarities and differences between TDB and IF. We conducted an RLQ analysis (Doledec et al., 1996) to examine the relationship between habitat structures in each vegetation class, local site variables at each point and species richness. An RLQ was used as we wanted to investigate the effects of vegetation classes and local site variables on functional diversity. RLQ analysis use a three-table ordination that aims to augment the covariance between local site covariates (R) traits (Q) and species recorded at each survey site (L).

3.4 Results

We conducted 84 bird survey point count surveys in IF patches and 51 in TDB patches and recorded 75 avian species in total (Appendix 3.1). Overall, there was no significant difference between the vegetation classes' microhabitat variables of TDB and IF ($P > 0.05$), except for a significant difference in the stem density of 2-5 m trees between the two vegetation classes ($P < 0.05$, Fig. 3.1 and Fig.3.3). In IF, we recorded 74 species (mean \pm SD, 16.58 ± 4.13 /survey site), whereas in TDB, we recorded 67 species (mean \pm SD, 13.65 ± 4.12 /survey site). We observed a 36.37 similarity in avian species between IF and TDB (3 3.1). However, there was no significant difference in avian species richness between TDB and IF (Fig.3.2c). Green-backed Camaroptera (*Camaroptera brachyura*), Fork-tailed Drongo (*Dicrurus adsimilis*) and Collared Sunbird (*Hedydipna collaris*) had the highest contribution to the average similarity between IF and TDB (Table 3.1). Species that were present in IF and absent in TDB were African Green Pigeon (*Treron calvus*), Cape Turtle Dove (*Streptopelia capicola*), Grey Sunbird (*Cyanomitra veroxii*), Klaas's Cuckoo (*Chrysococcyx klaas*), Narina Trogon (*Apaloderma narina*), Sweet Waxbill (*Coccyzygia melanotis*), Village Weaver (*Ploceus cucullatus*) and Grey-headed Bushshrike (*Malaconotus blanchoti*), while only Cape Starling (*Lamprotornis nitens*) was absent in IF and present in TDB. The results of RLQ analyses showed that the relationship between species traits and local site variables was significant (Monte-Carlo, $P = 0.001$, 9999 permutations, Fig. 3.4). Moreover, the analysis showed that the total variance (55 %) between traits and site variables was explained by the combination of the first two axes (Table 3.2). The correlation of RLQ was also explained by the first two axes; for R it was 15.4% and 12.6%, for Q it was 7.2% and 6.0%, for L it was 7.2 % and 6 %. Correspondence analysis in RLQ showed that percentage cover of bare ground and leaf litter correlated with TDB sites while SWDI, stem density of 2-5 m trees, number of horizontal dead trees and stem density of 6-10 m trees corresponded with IF sites. Bare-ground corresponded with large birds. We observed a correspondence between stem density of 6-10 m trees and nectarivores.

Table 3.1: RQL analysis of site covariates (R), individual species traits (Q), and species richness (L).

	Axis 1: Eigen values	Variance explained (%)	Axis 2: Eigen values	Variance explained (%)
R (PCA)	1.54	1.38	1.23	1.13
L (COA)	7.04	5.58	4.67	4.47
Q (Hill-Smith)	2.93	2.45	2.30	2.10
RLQ combined	0.05	39.57	0.20	15.59
Covariance	0.06	0.02	0.02	0.02
Projected variance: R	17.10	15.37	13.71	12.59
Projected variance: Q	0.26	0.20	0.17	0.16
Correlation: L	8.26	7.20	6.77	6.17

Table 3.2. SIMPER analysis results showing ten key avian species responsible for the similarity between Indigenous forest (IF) and Thicket Dense Bush (TDB).

Species		Av. Dissim ¹	Contrib.(%) ²	Cumulative (%) ³
Green-backed Camoroptera	<i>Camoroptera brachyura</i>	1.785	2.804	2.804
Fork-tailed Drongo	<i>Dicrurus adsimilis</i>	1.777	2.793	5.598
Collared Sunbird	<i>Hedydipna collaris</i>	1.756	2.76	8.358
Yellow-rumped Tinkerbird	<i>Pogoniulus bilineatus</i>	1.74	2.735	11.09
Dark-capped Bulbul	<i>Pycnonotus tricolor</i>	1.729	2.718	13.81
Tawny-flanked Prinia	<i>Prinia subflava</i>	1.723	2.708	16.52
Spectacled Weaver	<i>Ploceus ocularis</i>	1.72	2.703	19.22
Crested Barbet	<i>Trachyphonus vaillantii</i>	1.714	2.69	21.92
Olive Sunbird	<i>Cyanomitra olivacea</i>	1.704	2.678	24.59
Cape White-eye	<i>Zosterops virens</i>	1.703	2.676	27.27

1 = Average dissimilarity between vegetation classes. 2 = Percentage contribution to dissimilarity. 3 = Cumulative percentage dissimilarity.

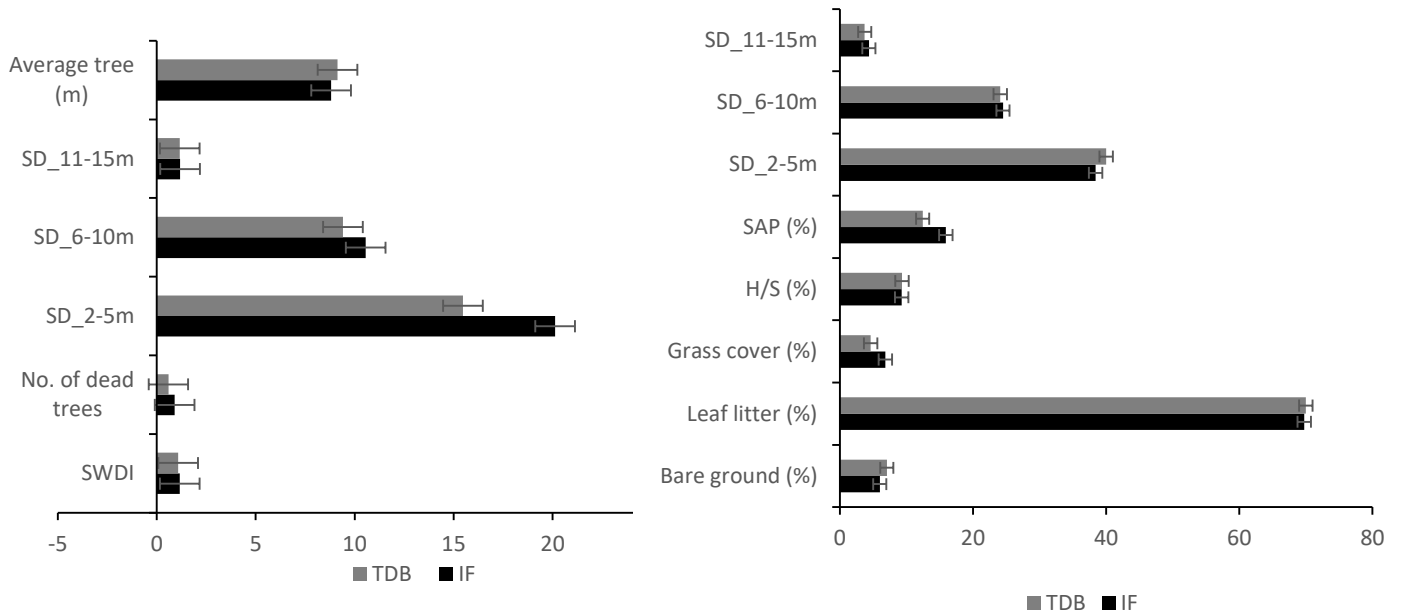


Figure 3.1: Microhabitat variables obtained from survey sites in Thicket/Dense Bush (TDB) and Indigenous Forest in Durban, South Africa where (a) shows mean tree height, number of dead trees (DT), Shannon-Weiner Diversity Index (SWDI) and stem density of trees (SD 2 – 5 m), 6 – 10 m (SD6 – 10 m) and 11–15 m (SD 11 – 15 m) in height whereas (b) shows percentage cover of trees (SD 2 – 5 m), 6 – 10 m (SD 6 – 10 m) and 11–15 m (SD 11 - 15m) in height, saplings and scrub/woody vegetation < 2 m (SAP), percentage cover of herbaceous plants and seedlings (H/S), grass cover, leaf litter and bare ground.

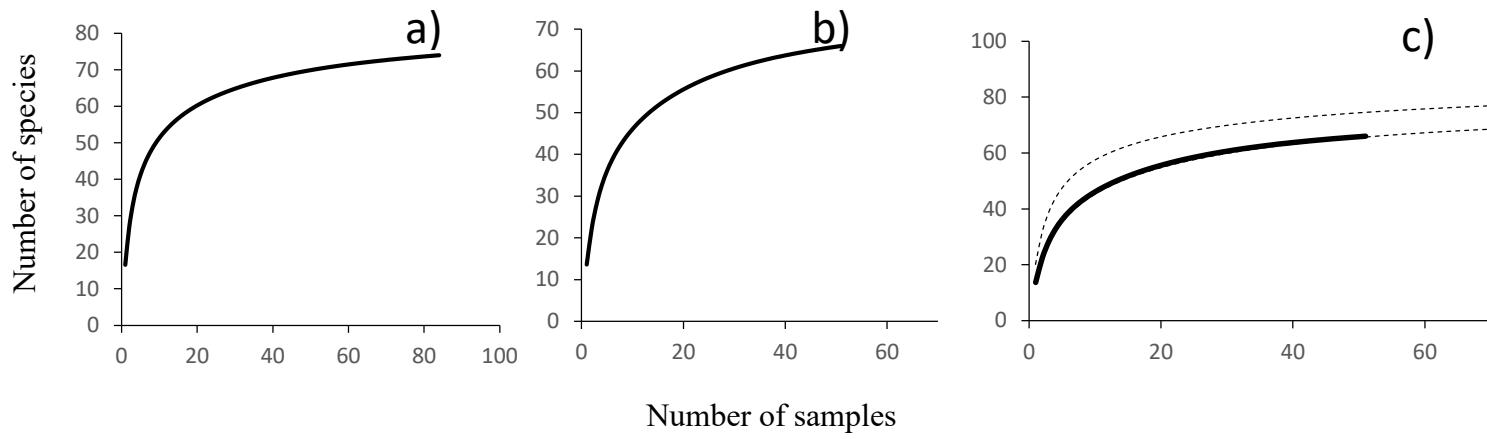


Figure 3.2: Rarefaction curves of Indigenous Forest (a) and Thicket Dense Bush (b) in surveyed sites in Durban South Africa and (c) the rarefaction curve of TDB within the 95% confidence interval of Indigenous Forest.

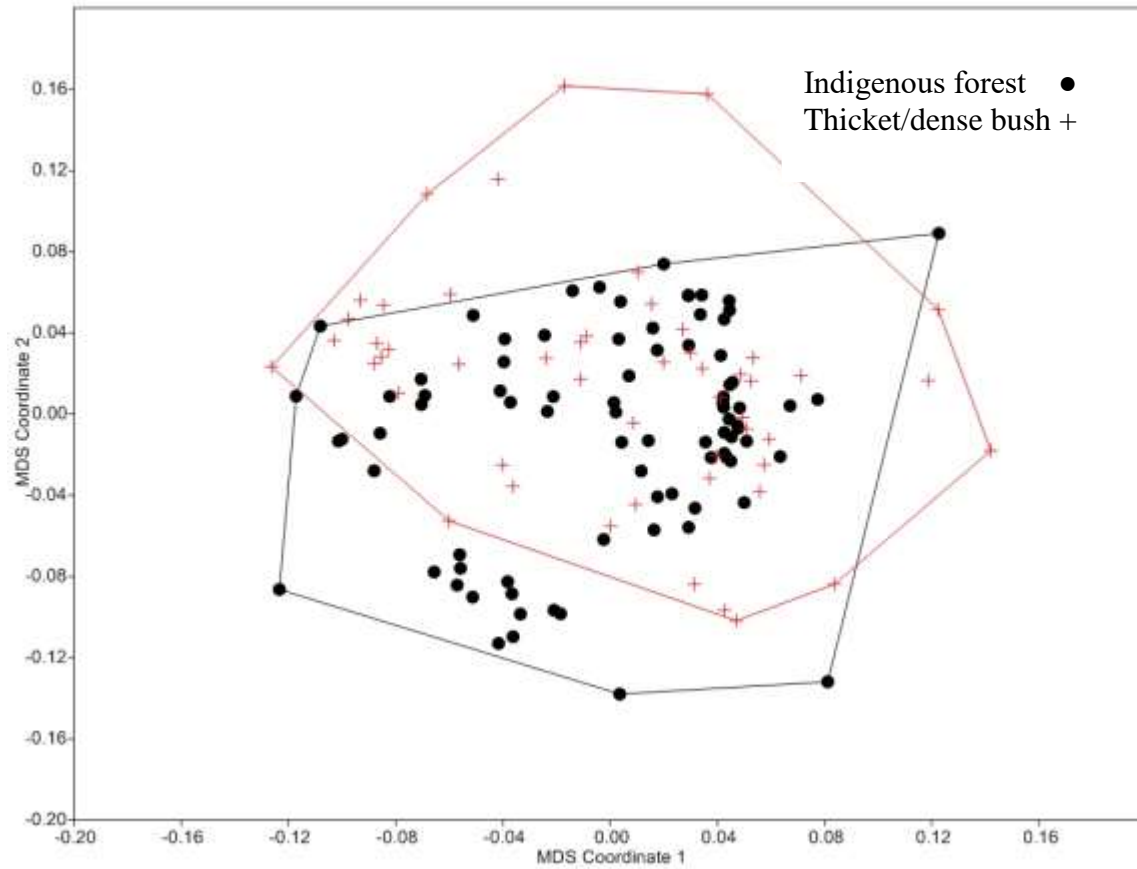
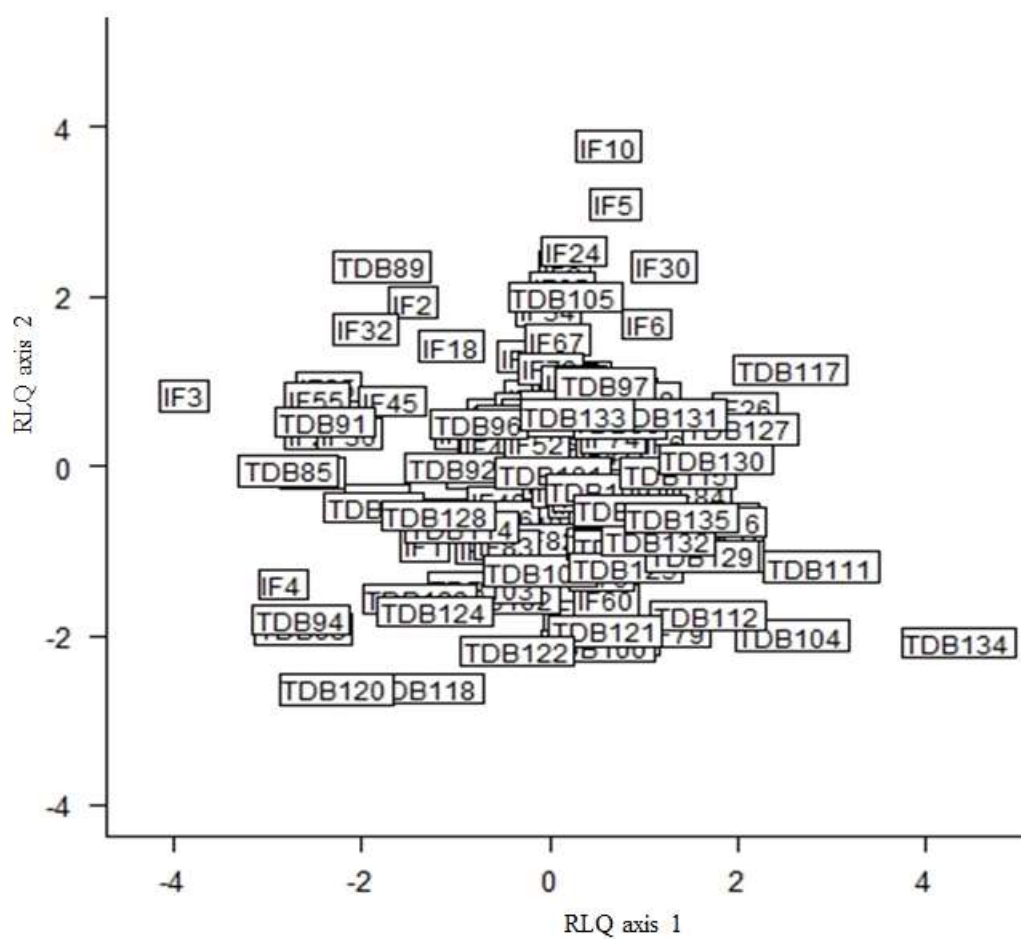
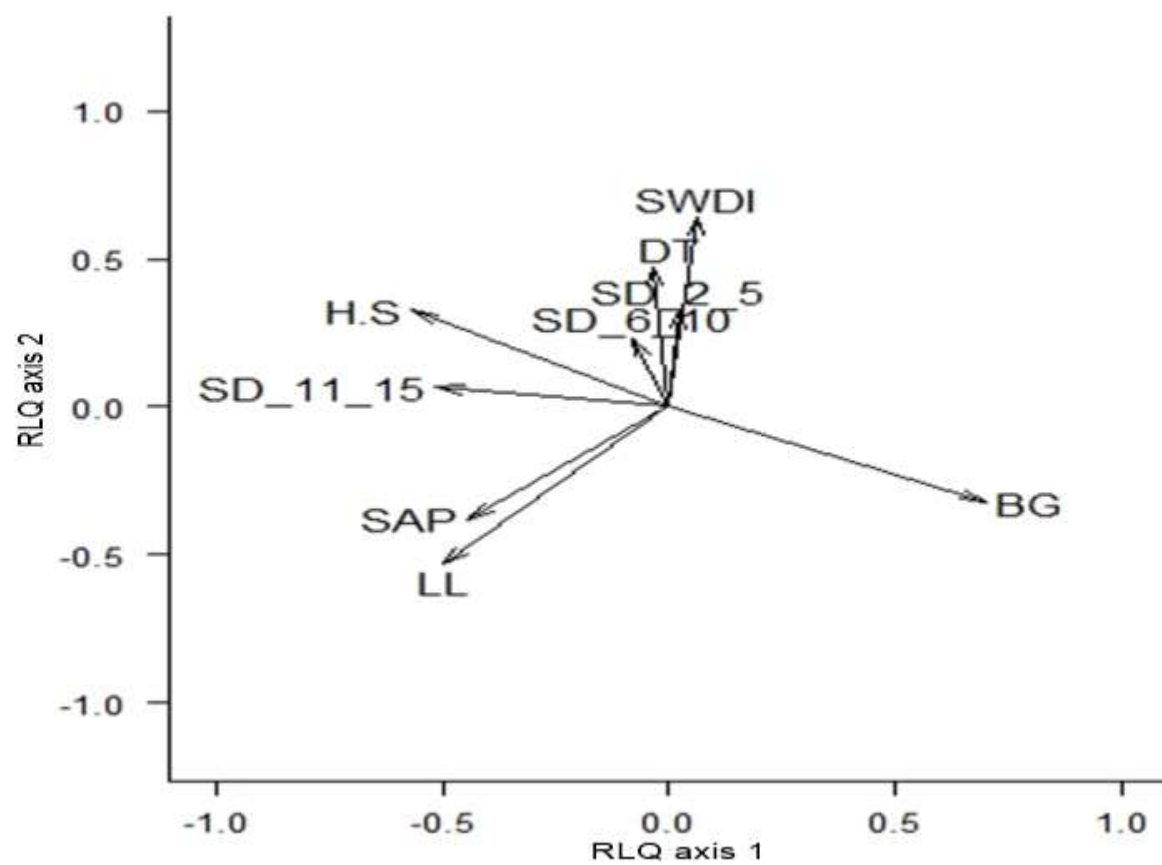


Figure 3.3. Non-metric multidimensional scaling showing distinct clustering Thicket/Dense Bush and Indigenous Forests survey sites based on the avian species present/absence in the present study.



a)



b)

Figure 3.4. RQL analysis showing (a) distribution of survey sites conducted in Indigenous Forests and Thicket/Dense Bush (b) microhabitats associated with (c) species traits in surveyed sites in Durban, South Africa.

3.5 Discussion

Indigenous forests are diminishing globally and in South Africa, and this is generally because of anthropogenic phenomena such as urbanisation and agricultural expansion (Ehlers Smith et al., 2017a, b.). The main implication of losing natural forests, is the decline in biodiversity as a result of habitat loss. Given the high diversity of the avian taxonomic community (75 species) recorded in our study, it is important that the current forests in our area are properly conserved as they provide habitat for a diverse community of avian species. Moreover, our data showed that the forest areas provided habitat for avian species with diverse traits and specific ecological niches, including avian forest specialists such as Square-tailed Drongo (*Dicrurus ludwigii*) and Spotted Ground Thrush (*Zoothera guttata*). Therefore, we conclude that our study highlights the importance of conservation of forests within an urban mosaic area, especially with regards to conserving biodiversity and ecosystem functioning.

The similarity in vegetation structure of TDB and IF in our study area implies that TDB was also a preferred habitat for bird species and requires special conservation attention. There was no significant difference in avian species richness between IF and TDB, which did not support our hypothesis of differences in the avian communities between IF and TDB. However, it is important to highlight that avian species traits overlapped between IF and TDB, and that TDB had all the forest specialist species that were also found in IF, with the exception of the Narina Trogon. We concluded that the similarity in vegetation classes was the one of the main possible explanation of this non-significant result of species richness between IF and TDB. Interestingly, the presence of avian forest specialist species in TDB, such as Olive Bushshrike (*Telophorus olivaceus*), Green-backed Camaroptera, Spotted Ground Thrush (c.f. Oatley, 1989), and the presence of specialised niches, structures and species in TDB in a relatively heavily urbanised mosaic environment, suggests that the TDB has reached an advanced state of forest regeneration. This is because during the degradation of natural forests, avian specialists are most at risk of being local extinction

(Maseko et al., 2017); thus, their presence in TDB suggests forest regeneration and not degradation of IF.

Habitat heterogeneity and vegetation structure are important factors for species diversity in a particular area (Tews et al., 2004). Overall, our results showed that there was no significant difference between the microhabitat structures of TDB and IF, which does not support our first prediction of a significant difference between IF and TDB vegetation structures. The only difference was in the stem density of 2-5 m trees, indicating a higher proportion of saplings and smaller/younger trees in IF. However, these results may also imply that the IF has been degraded to a point that it now structurally represents a more TDB-type habitat, or that the TDB has successfully regenerated to become IF. Our results are contrary to those of Ehlers Smith et al., (2017a), who found a significant difference in microhabitat structures between IF and TDB on the southern coast of the Indian Ocean Coastal Belt Forest. Our study was conducted in a highly urbanised (high density housing) area whereas Ehlers Smith's study was conducted in a more rural/natural area (less housing density and more farmlands). Therefore, it is important to conduct more research to investigate any other factors influencing the similarities and differences between vegetation structures of IF and TDB.

Overall, there was a similarity between the association of microhabitats variables and survey sites of IF and TDB. However, stem density of 2-5 m trees in height and number of dead trees (DT) was more prevalent in IF than TDB. There was an association between herbaceous plants and seedlings and ground nesting species. We assumed this association was mainly because these ground nesting species use herbaceous plants to line and camouflage their nests. There was a strong association between large-harvesting bird species and bare ground, which was more prevalent in TDB. This is encouraging, because large, frugivorous birds are often most negatively affected by the fragmentation effect (Restrepo et al., 1999; Galletti et al., 2003). Nectarivorous species were more commonly associated with a high heterogeneity index, and IF sites, which is interesting in comparison to Ehlers Smith et al., (2017a), who found no difference in the distribution of avian nectarivores across the IF and TDB landscape. Globally, human-induced modifications (e.g. fragmentation) in natural environments significantly threatens ecological processes such as pollination (Neuschulz et al., 2016). Since avian nectarivorous species in this study were associated with high heterogeneity forest areas and IF, this suggests that in such a

highly transformed landscape, protection of these areas is important to ensure the continuity of pollination. Moreover, pollination, as an ecosystem service (Kremen et al., 2007) is aided by avian nectarivorous species, therefore it is important to conserve them via protecting their preferred habitat to ensure that the ecosystem service is protected. Cavity-nesters were associated with taller trees, which were more prevalent in TDB in the present study. This is a positive trend as the specialised niche of cavity nesting is often threatened by wood removal and forest degeneration (e.g. du Plessis, 1995; Downs and Symes, 2004; Wilson et al., 2017). This may further offer support that TDB in the study areas is at an advanced stage of forest regeneration, as specialised niches and associated avian forest specialists were present.

In conclusion, vegetation structures and taxonomic richness in our study region were not significantly different between vegetation classes. Additionally, species traits between the two vegetation classes displayed a high proportion of overlap, which indicated that the TDB in Durban represented an advance stage of IF regeneration, as avian specialist species are generally rapidly lost during the forest-degradation process (Maseko et al., 2017). Our study showed that avian species with specific ecological niches can persist in an anthropogenic/urban environment provided structures are maintained and protected. Our results show the importance of both IF and TDB in protecting the avian species within an urban area. Furthermore, our study showed that TDB needs protection status as it represents regenerating forest, a highly restricted and Critically Endangered biome.

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3.7 References

Andren, H., 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. *Oikos*, 71, 355-366.

- Aronson, M. F., La Sorte, F. A., Nilon, C. H., Katti, M., Goddard, M. A., Lepczyk, C. A., Dobbs, C., 2014. April. A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proc. R. Soc. B.* 281, 1-8.
- Betts, M. G., Fahrig, L., Hadley, A. S., Halstead, K. E., Bowman, J., Robinson, W. D., Lindenmayer, D. B., 2014. A species centered approach for uncovering generalities in organism responses to habitat loss and fragmentation. *Ecography* 376, 517-527.
- Bibby, C., Burgess, N., Hill, D., 2000. *Bird census techniques*. Academic Press, London.
- Boon, R., Cockburn, J., Douwes, E., Govender, N., Ground, L., Mclean, C., Slotow, R., 2016. Managing a threatened savanna ecosystem (KwaZulu-Natal Sandstone Sourveld) in an urban biodiversity hotspot: Durban, South Africa. *Afr. Biodivers. Conserv.* 46, 1-12.
- Bregman, T. P., Sekercioglu, C. H., Tobias, J. A., 2014. Global patterns and predictors of bird species responses to forest fragmentation: implications for ecosystem function and conservation. *Biol. Conserv.* 169, 372-383.
- Brown, M. L., Donovan, T. M., Schwenk, W. S., Theobald, D. M., 2014. Predicting impacts of future human population growth and development on occupancy rates of forest-dependent birds. *Biol. Conserv.* 170, 311-320.
- Carrara, E., Arroyo-Rodríguez, V., Vega-Rivera, J. H., Schondube, J. E., de Freitas, S. M., Fahrig, L., 2015. Impact of landscape composition and configuration on forest specialist and generalist bird species in the fragmented Lacandona rainforest, Mexico. *Biol. Conserv.* 184, 117-126.
- Chace, J. F., Walsh, J. J., 2004. Urban effects on native avifauna: a review. *Landsc. Urban Plan.* 741, 46-69.
- DeWalt, S. J., Maliakal, S. K., Denslow, J. S., 2003. Changes in vegetation structure and composition along a tropical forest chronosequence: implications for wildlife. *Forest Ecol. Manag.* 1821, 139-151.
- Díaz, I. A., Armesto, J. J., Reid, S., Sieving, K. E., Willson, M. F., 2005. Linking forest structure and composition: avian diversity in successional forests of Chiloé Island, Chile. *Biol. Conserv.* 1231, 91-101.
- Doledec, S., Chessel, D., Braak, C.J.F., Champely, S., 1996. Matching species traits to environmental variables: a new three-table ordination method. *Environ. Ecol. Stat.* 3, 143-166.
- Downs, C. T., Symes, C. T., 2004. Snag dynamics and forest structure in Afromontane forests in KwaZulu-Natal, South Africa: implications for the conservation of cavity-nesting avifauna. *S. Afr. J. Bot.* 70, 265-276.
- Du Plessis, M., 1995. The effects of fuelwood removal on the diversity of some cavity-using birds and mammals in South Africa. *Biol. Conserv.* 74, 77-82.
- Ehlers Smith, D. A., Ehlers Smith, Y.C., Downs, C.T., 2017d. Seasonal habitat requirements of Lemon Dove (*Aplopelia larvata*) in Coastal Forest: camera-trap surveys of a reclusive species. *Afr. Zool.* 52: 199-207.
- Ehlers Smith, D.A., Ehlers Smith, Y.C., Downs, C.T., 2017a. Indian Ocean Coastal Thicket is of high conservation value for taxonomic and functional diversity of forest-dependent bird communities in a landscape of restricted forest availability. *Forest Ecol. Manag.* 390, 157-165.

- Ehlers Smith, D.A., Ehlers Smith, Y.C., Ramesh, T., Downs, C.T., 2017c. Camera-trap data elucidate habitat requirements and conservation threats to an endangered forest specialist, the Spotted Ground Thrush (*Zoothera guttata*). *Forest Ecol. Manag.* 400, 523–530.
- Ehlers Smith, Y. C. E., Ehlers Smith, D. A. E., Seymour, C. L., Thébault, E., van Veen, F. F., 2015. Response of avian diversity to habitat modification can be predicted from life-history traits and ecological attributes. *Landsc. Ecol.* 30, 1225-1239.
- Ehlers Smith, Y.C., Ehlers Smith, D.A., Ramesh, T., Downs, C.T., 2017b. The importance of microhabitat structure in maintaining forest mammal diversity in a mixed land-use mosaic. *Biodivers. Conserv.* <http://dx.doi.org/10.1007/s10531-017-1360-6>.
- ESRI (Environmental Systems Research Institute) (2011) ArcGIS Desktop 10.
- eThekweni Municipality, 2010. State of biodiversity. Durban.
- Fahrig, L., 2003. Effects of habitat fragmentation on biodiversity. *Ann. Rev. Ecol. Evol. Syst* 34, 487-515.
- Falcucci, A., Maiorano, L., Boitani, L., 2007. Changes in land-use/land-cover patterns in Italy and their implications for biodiversity conservation. *Landsc. Ecol.* 224, 617-631.
- Fischer, J., Lindenmayer, D. B., 2007. Landscape modification and habitat fragmentation: a synthesis. *Glob. Ecol. and Biog.* 163, 265-280.
- FitzGibbon, S. I., Putland, D. A., Goldizen, A. W., 2007. The importance of functional connectivity in the conservation of a ground-dwelling mammal in an urban Australian landscape. *Landsc. Ecol.* 2210, 1513.
- Flynn, D. F. B., Gogol-Prokurat, M., Nogeire, T., Molinari, N., Richers, B. T., Lin, B. B., Simpson, N., Mayfield, M. M., DeClerck, F. 2009. Loss of functional diversity under land use intensification across multiple taxa. *Ecol. Lett.* 12, 22–33.
- Galetti, M., Alves-Costa, C. P., Cazetta, E., 2003. Effects of forest fragmentation, anthropogenic edges and fruit colour on the consumption of ornithocoric fruits. *Biol. Conserv.* 111, 269-273.
- GeoterraImage, 2014. KZN Province Land-Cover Mapping (from SPOT5 Satellite imagery circa 2013). Prepared for Ezemvelo KZN Wildlife (Biodiversity Research), South Africa.
- Gil-Tena, A., Saura, S., Brotons, L., 2007. Effects of forest composition and structure on bird species richness in a Mediterranean context: implications for forest ecosystem management. *Forest Ecol. Manage.* 2422, 470-476.
- Gotelli, N. J., Colwell, R.K., 2011. Estimating species richness. In: Magurran, A.E., McGill, B.J., (Eds.), *Biological diversity: frontiers in measurement and assessment*. New York: Oxford University Press, pp.39-54.
- Harper D, Ryan P., 2001. PAST: paleontological statistics software package for education and data analysis. *Palaeontol Electron* 4:1–9
- Hermes, C., Döpper, A., Schaefer, H. M., Segelbacher, G., 2016. Effects of forest fragmentation on the morphological and genetic structure of a dispersal-limited, endangered bird species. *Nat. Conserv.* 16, 39-58.
- Hockey, P.A.R., Dean, W.R.J., Ryan, P., 2005. *Birds of Southern Africa*, 7th edn. John Voelcker Bird Book Fund, Cape Town.
- Kremen, C., Williams, N. M., Aizen, M. A., Gemmill-Herren, B., LeBuhn, G., Minckley, R., Winfree, R., 2007. Pollination and other ecosystem services produced by mobile organisms: a conceptual framework for the effects of land-use change. *Ecol. Lett.* 10, 299-314.

- MacGregor-Fors, I., Ortega-Álvarez, R., 2011. Fading from the forest: bird community shifts related to urban park site-specific and landscape traits. *Urban For. Urban Greening* 10, 239-246.
- Mannan, R. W., Meslow, E. C., Wight, H. M., 1980. Use of snags by birds in Douglas-fir forests, western Oregon. *J. Wildl. Manage.* 44, 787-797.
- Maseko, M. S., Ramesh, T., Kalle, R., Downs, C. T., 2017. Response of Crested Guinea-fowl *Guttera edouardi*, a forest specialist, to spatial variation in land use in iSimangaliso Wetland Park, South Africa. *J. Ornithol.* 1582, 469-477.
- McAlpine, C. A., Rhodes, J. R., Callaghan, J. G., Bowen, M. E., Lunney, D., Mitchell, D. L., Possingham, H. P., 2006. The importance of forest area and configuration relative to local habitat factors for conserving forest mammals: a case study of koalas in Queensland, Australia. *Biol. Conserv.* 132, 153-165.
- McDonald, R. I., Kareiva, P., Forman, R. T., 2008. The implications of current and future urbanization for global protected areas and biodiversity conservation. *Biol. Conserv.* 1416, 1695-1703.
- McKinney, M. L., 2002. Urbanization, biodiversity, and conservation: the impacts of urbanization on native species are poorly studied, but educating a highly urbanized human population about these impacts can greatly improve species conservation in all ecosystems. *Bioscience* 5210, 883-890.
- Nellemann, C., et al., 2007. The Last Stand of the Orangutan - State of Emergency: Illegal Logging, Fire and Palm Oil in Indonesia's National Parks. United Nations Environment Programme, GRID-Arendal, Oslo.
- Neuschulz, E. L., Mueller, T., Schleuning, M., Böhning-Gaese, K., 2016. Pollination and seed dispersal are the most threatened processes of plant regeneration. *Sci. Rep.* 6,1-6.
- Oatley, T.B., 1989. Biogeography of the forest avifauna in South Africa. In: Geldenhuys, C.J. ed. *Biogeography of the mixed evergreen forests of southern Africa*. Occasional Report No. 45. Pretoria, Foundation for Research and Development, CSIR, pp. 48-59.
- Olivier, P.I., van Aarde, R.J., Lombard, A.T., 2013. The use of habitat suitability models and species-area relationships to predict extinction debts in coastal forests, South Africa. *Divers. Distr.* 19,1353-1365.
- Parker, T. H., Stansberry, B. M., Becker, C. D., Gipson, P. S., 2005. Edge and area effects on the occurrence of migrant forest songbirds. *Conserv. Biol.* 94, 1157-1167.
- Renjifo, L. M., 2001. Effect of natural and anthropogenic landscape matrices on the abundance of subandean bird species. *Ecol. Appl.* 111, 14-31.
- Restrepo, C., Gomez, N., Heredia, S., 1999. Anthropogenic edges, tree fall gaps, and fruit-frugivore interactions in a Neotropical montane forest. *Ecology*, 80, 668-685.
- Sliwinski, M., Powell, L., Koper, N., Giovanni, M., Schacht, W., 2016. Research design considerations to ensure detection of all species in an avian community. *Methods Ecol. Evol.* 74, 456-462.
- Tews, J., Brose, U., Grimm, V., Tielbörger, K., Wichmann, M. C., Schwager, M., Jeltsch, F., 2004. Animal species diversity driven by habitat heterogeneity/diversity: the importance of keystone structures. *J. Biogeog.* 31, 79-92.
- Tomasevic, J. A., Marzluff, J. M., 2017. Cavity nesting birds along an urban-wildland gradient: is human facilitation structuring the bird community? *Urban Ecosyst.* 202, 435-448.

- UN., 2015. World population prospects: working paper no. ESA/P/WP. 241. United Nation Department of Economic and Social Affairs, Population Division, New York.
- Verburg, P. H., Overmars, K. P., Huigen, M. G., de Groot, W. T., Veldkamp, A., 2006. Analysis of the effects of land use change on protected areas in the Philippines. *Appl. Geogr.* 262, 153-173.
- Villard, M. A., Trzcinski, M. K., Merriam, G., 1999. Fragmentation effects on forest birds: relative influence of woodland cover and configuration on landscape occupancy. *Conserv. Biol.* 13, 774-783.
- Villaseñor, N. R., Blanchard, W., Driscoll, D. A., Gibbons, P., Lindenmayer, D. B., 2015. Strong influence of local habitat structure on mammals reveals mismatch with edge effects models. *Landsc. Ecol.* 302, 229-245.
- Villaseñor, N. R., Blanchard, W., Lindenmayer, D. B., 2016. Decline of forest structural elements across forest–urban interfaces is stronger with high rather than low residential density. *Basic Appl. Ecol.* 175, 418-427.
- Watson, J. E., Whittaker, R. J., Dawson, T. P., 2004. Habitat structure and proximity to forest edge affect the abundance and distribution of forest-dependent birds in tropical coastal forests of south eastern Madagascar. *Biol. Conserv.* 1203, 311-327.
- Wilson, A. L., Bowker, M., Shuttleworth, A., Downs, C. T., 2017. Characteristics of snags and forest structure in southern mistbelt forests of the Amatole region, South Africa. *Afr. J. Ecol.* 55, 518–529.

Appendix 3.1

Appendix 3.1. Avian species recorded during the Southern Africa breeding season (i.e. October 2016 March 2017) in five protected areas (PAs) within Durban, eThekweni Municipality, KwaZulu-Natal, South Africa.

Common name	Scientific Name	Forest specialist or Generalist
African Crowned Eagle	<i>Stephanoaetus coronatus</i>	Specialist
African Dusky Flycatcher	<i>Muscicapa adusta</i>	Generalist
African Goshawk	<i>Accipiter tachiro</i>	Specialist
African Green Pigeon	<i>Treron calvus</i>	Generalist
African Hoopoe	<i>Upupa africana</i>	Generalist
African Paradise Flycatcher	<i>Terpsiphone viridis</i>	Generalist
African Pied Wagtail	<i>Motacilla aguimp</i>	Generalist
Amethyst Sunbird	<i>Chalcomitra amethystina</i>	Generalist
Bar Throated Apalis	<i>Apalis thoracica</i>	Specialist
Black Backed Puffback	<i>Dryoscopus cubla</i>	Generalist
Black Collared Barbet	<i>Lybius torquatus</i>	Generalist
Black Headed Oriole	<i>Oriolus larvatus</i>	Generalist
Black Sparrowhawk	<i>Accipiter melanoleucus</i>	Generalist
Blue Mantled Crested Flycatcher	<i>Trochocercus cyanomelas</i>	Specialist
Bronze Mannikin	<i>Spermestes cucullatus</i>	Generalist
Brown Hooded Kingfisher	<i>Halcyon albiventris</i>	Generalist
Brown Scrub Robin	<i>Cercotrichas signata</i>	Specialist
Cape Batis	<i>Batis capensis</i>	Specialist
Cape Glossy Starling	<i>Lamprotornis nitens</i>	Generalist
Cape Robin Chat	<i>Cossypha caffra</i>	Generalist
Cape Turtle Dove	<i>Streptopelia capicola</i>	Generalist
Cape White Eye	<i>Zosterops virens</i>	Generalist
Cardinal Woodpecker	<i>Dendropicus fuscescens</i>	Generalist
Chorister Robin Chat	<i>Cossypha dichroa</i>	Specialist

Collared Sunbird	<i>Hedydipna collaris</i>	Specialist
Common Fiscal	<i>Lanius collaris</i>	Generalist
Common Waxbill	<i>Estrilda astrild</i>	Generalist
Crested Barbet	<i>Trachyphonus vaillantii</i>	Generalist
Crowned Hornbill	<i>Tockus alboterminatus</i>	Generalist
Dark Capped Bulbul	<i>Pycnonotus tricolor</i>	Generalist
Forest Canary	<i>Crithagra scotops</i>	Specialist
Fork Tailed Drongo	<i>Dicrurus adsimilis</i>	Generalist
Gorgeous Bushshrike	<i>Telophorus viridis</i>	Generalist
Green Backed Camaroptera	<i>Camaroptera brachyura</i>	Specialist
Green Wood Hoopoe	<i>Phoeniculus purpureus</i>	Generalist
Grey Cuckooshrike	<i>Coracina caesia</i>	Specialist
Grey Headed Bushrike	<i>Malaconotus blanchoti</i>	Generalist
Grey Sunbird	<i>Cyanomitra veroxii</i>	Generalist
Grey Waxbill	<i>Estrilda perreini</i>	Specialist
Hadedda Ibis	<i>Bostrychia hagedash</i>	Generalist
Klaas's Cuckoo	<i>Chrysococcyx klaas</i>	Generalist
Knysna Turaco	<i>Tauraco corythaix</i>	Specialist
Kurrichane Thrush	<i>Turdus libonyana</i>	Generalist
Lemon Dove	<i>Aplopelia larvata</i>	Specialist
Little Sparrowhawk	<i>Accipiter minullus</i>	Generalist
Marsh Warbler	<i>Acrocephalus palustris</i>	Generalist
Narina Trogon	<i>Apaloderma narina</i>	Specialist
Natal Spurfowl	<i>Pternistis natalensis</i>	Generalist
Olive Bushshrike	<i>Chlorophoneus olivaceus</i>	Specialist
Olive Sunbird	<i>Cyanomitra olivacea</i>	Specialist
Olive Thrush	<i>Turdus olivaceus</i>	Generalist
Purple Crested Turaco	<i>Tauraco porphyreolopha</i>	Generalist
Red Capped Robin Chat	<i>Cossypha natalensis</i>	Generalist
Red Eyed Dove	<i>Streptopelia semitorquata</i>	Generalist

Red Fronted Tinkerbird	<i>Pogoniulus pusillus</i>	Generalist
Red Winged Starling	<i>Onychognathus morio</i>	Generalist
Sombre Greenbul	<i>Andropadus importunus</i>	Generalist
Southern Black Flycatcher	<i>Melaenornis pammelaina</i>	Generalist
Southern Boubou	<i>Laniarius ferrugineus</i>	Generalist
Speckled Mousebird	<i>Colius striatus</i>	Generalist
Spectacled Weaver	<i>Ploceus ocularis</i>	Generalist
Spotted Ground Thrush	<i>Zoothera guttata</i>	Specialist
Square Tailed Drongo	<i>Dicrurus ludwigii</i>	Specialist
Swee Waxbill	<i>Coccygia melanotis</i>	Generalist
Tambourine Dove	<i>Turtur tympanistra</i>	Specialist
Tawny Flanked Prinia	<i>Prinia subflava</i>	Generalist
Terrestrial Brownbul	<i>Phyllastrephus terrestris</i>	Generalist
Thick Billed Weaver	<i>Amblyospiza albifrons</i>	Generalist
Trumpeter Hornbill	<i>Bycanistes bucanitor</i>	Specialist
Village Weaver	<i>Ploceus cucullatus</i>	Generalist
White Bellied Sunbird	<i>Cinnyris talatala</i>	Generalist
White Eared Barbet	<i>Stactolaema leucotis</i>	Generalist
White Starred Robin	<i>Pogonocichla stellata</i>	Specialist
Yellow Fronted Canary	<i>Crithagra mozambica</i>	Generalist
Yellow Rumped Tinkerbird	<i>Pogoniulus bilineatus</i>	Specialist

CHAPTER 4

CONCLUSIONS

The increase in urbanisation exacerbates the transformation of natural landscapes for anthropogenic activities such as human settlements and agricultural expansion (McKinney 2002; UN, 2015; Chibesa and Down, 2017). From a global perspective, studies have well-documented the response of wildlife to habitat transformations and disturbances (e.g. Fahrig, 2003; Ehlers Smith et al., 2015; Maseko et al., 2017). Although certain species may persist and thrive within transformed landscapes, the overall effects of habitat transformations on species are negative, and threaten global biodiversity. Considering the continuous fragmentation and destruction of natural landscapes, there is an urgent necessity of studies to provide effective management strategies which will aid in the conservation of flora and fauna. For example, the establishment of natural environments within urban areas facilitates persistence of a high proportion of species which would otherwise become locally extinct within urban landscapes (Blaustein, 2013). This is particularly pertinent in some regions where Protected Areas are degraded or at risk, compromising their effectiveness to safeguard species and communities because of illegal activities (e.g. logging and agricultural expansion; (Verburg et al., 2006).

The present study investigated the effects of fragmentation via patch size and isolation on the diversity of forest birds. The study investigated the differences in vegetation structures of Indigenous Forest (IF) and Thicket Dense Bush (TDB) with the aim of showing their importance in the provision of habitat and different niches and resources to avian species in Protected Areas (PAs) within an urban mosaic of eThekweni Municipality (Chapters 2 and 3). The study region comprised of lowland coastal and scarp forest patches which were either naturally patchily distributed (scarp forest) or fragmented via anthropogenic activities (lowland coastal forest; Mucina and Rutherford, 2011; Olivier et al., 2013), with several patches of TDB (GeoTerra Image, 2014). We conducted bird point count surveys during the southern African breeding season (i.e. October 2016 - March 2017) to determine the taxonomic richness within IF and TDB patches, and recorded a total of 75 avian species (Chapters 2 and 3). This study showed that the five PAs are a habitat for a diverse avian community, which is of conservation importance considering that they are located within a region dominated by human-induced modifications (Chapters 2 and 3).

Our results showed that both IF and TDB hold avian generalists and forest specialist species, which are crucial for long-term functioning of the ecosystem (Chapters 2 and 3). Overall, vegetation structures and species richness of IF and TDB did not differ significantly (Chapter 3). Interestingly, although we did not predict the presence of forest specialists in TDB because of the vegetation structures associated with TDB, based on previous research within the habitat subclass which indicated less structural diversity than that found in IF (Ehlers Smith et al., 2017), we found these species (Chapter 3). We concluded that TDB represented an advanced stage of regenerating to IF since there was a presence of specialist species in TDB survey sites (Chapter 3). Furthermore, our results showed that taxonomic richness and functional diversity increased with habitat-patch size and that isolation distance negatively affected taxonomic richness (Chapter 2). Also, the number of specialist species increased with habitat patch size, highlighting the importance of forest fragments for biodiversity conservation (Chapter 2). Finally, the overall conclusion from our study is that, if habitat structures persist and concomitant environmental niches are well protected, species with diverse traits (i.e. specialists and generalists) may persist within urban landscapes.

4.1 Recommendations for future studies

A major finding from this study was the presence of forest specialist species in TDB, which, together with no significant differences between IF and TDB habitat structures, indicating that the TDB within eThekweni Municipality is at an advanced stage of regeneration. Considering that specialist species are particularly at risk of becoming extinct if specific ecological niches and resources are lost, it is important to further assess the TDB state, which will enable us to understand to fully understand why there were forest specialist in TDB. For example, our study did not explore the availability of resources such as food in IF and TDB, which may further explain our findings. Furthermore, vegetation structures of IF and TDB did not differ in our study region (high urbanisation), which was contrary to those of Ehlers Smith et al. (2017), a study conducted in an area with low-medium urbanisation levels. Therefore, we recommend more research to be conducted to fully elucidate the reason behind the differences. The largest patch was selected to be the mainland patch, but we also had other patches which were relatively large. Therefore, it would be interesting to determine the patch-size threshold to be considered a mainland, hence we

recommend that other studies explore this, especially in urban landscapes with high level of urbanisation. Such information may be crucial for decision-making during development in a way that conservationist can argue that such forest patches must be protected from further fragmentation, which would reduce the size, and possibly have detrimental effects on specialist species and ultimately affect biodiversity negatively. Lastly, from a management perspective, it is crucial to protect the TDB along corridors to simultaneously facilitate connectivity, dispersal and colonisation, permit regeneration of the critically endangered and range-restricted biome of the Indian Ocean Coastal Belt Forest, and provide ecosystem services to the mixed land-use mosaic.

4.2 References

- Andren, H., 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. *Oikos*, 71, 355-366.
- Blaustein, R., 2013. Urban biodiversity gains new converts: cities around the world are conserving species and restoring habitat. *BioScience*, 63,2, 72-77.
- Chibesa, M., Downs, C. T., 2017. Factors determining the occupancy of Trumpeter Hornbills in urban-forest mosaics of KwaZulu-Natal, South Africa. *Urban Ecosyst.* 1-8.
- Cushman, S. A., 2006. Effects of habitat loss and fragmentation on amphibians: a review and prospectus. *Biol. Conserv.* 128, 231-240.
- Ehlers Smith, Y. C., Ehlers Smith, D. A., Seymour, C. L., Thébault, E., van Veen, F. F., 2015. Response of avian diversity to habitat modification can be predicted from life-history traits and ecological attributes. *Landscape Ecol.* 30, 1225-1239.
- Ehlers Smith, D. A., Ehlers Smith, Y. C., Downs, C. T., 2017. Indian Ocean Coastal Thicket is of high conservation value for taxonomic and functional diversity of forest-dependent bird communities in a landscape of restricted forest availability. *Forest Ecol. Manag.* 390, 157–165.
- Fahrig, L., 2003. Effects of habitat fragmentation on biodiversity. *Ann. Rev. Ecol. Evol. Syst* 34, 487-515.
- GeoterraImage, 2014. KZN Province Land-Cover Mapping (from SPOT5 Satellite imagery circa 2013). Prepared for Ezemvelo KZN Wildlife (Biodiversity Research), South Africa.
- McKinney, M. L., 2002. Urbanization, biodiversity, and conservation: the impacts of urbanization on native species are poorly studied, but educating a highly urbanized human population about these impacts can greatly improve species conservation in all ecosystems. *Bioscience* 5210, 883-890.
- Mucina, L., Rutherford, M. C., 2011. The vegetation of South Africa, Lesotho and Swaziland. *Strelitzia* 19. South African National Biodiversity Institute, Pretoria.
- Olivier, P. I., van Aarde, R. J., Lombard, A. T., 2013 The use of habitat suitability models and species-area relationships to predict extinction debts in coastal forests, South Africa. *Divers. Distr.* 19, 1353–1365.

- UN., 2015. World population prospects: working paper no. ESA/P/WP. 241. United Nation Department of Economic and Social Affairs, Population Division, New York.
- Verburg, P. H., Overmars, K. P., Huigen, M. G., de Groot, W. T., Veldkamp, A., 2006. Analysis of the effects of land use change on protected areas in the Philippines. *Appl. Geogr.* 262, 153-173.