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THE INFLUENCE OF *Lantana camara* INVASION ON PLANT, MAMMAL AND AVIFAUNA DIVERSITY IN KATONGA WILDLIFE RESERVE, WESTERN UGANDA

BY

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JANUARY, 2026

DECLARATION

I Stephen Ssemwaka declare that this dissertation is my original work it has never been submitted to any institution of learning for any kind of award. I take full responsibility for the correctness of the said information.

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DEDICATION

I dedicate this work to Mr. Raymond Katebaka, whose belief in my potential inspired me to pursue this Master's program. To my parents, Mr. Lumu Charles Kizza and Ms. Nakibuuka Maxencia, for their unwavering love and support, and to Mr. Kateregga Doddovick, whose encouragement kept me focused and determined to complete this journey.

ABSTRACT

Invasive plant species are a growing threat to biodiversity and ecosystem integrity worldwide. In Uganda *Lantana camara*, is one of the most aggressive invaders that have spread across Uganda's protected areas, yet its ecological impacts in savanna ecosystems remain poorly documented. This study assessed the influence of *Lantana camara* invasion on the diversity of native plants, avifauna, and medium-to-large mammals in Katonga Wildlife Reserve, Western Uganda. Six transects (three invaded and three non-invaded) were surveyed using quadrats, point count, and line transect methods, for plants, avifauna and mammals respectively A total of 103 plant species, 92 bird species, and 14 mammal species were recorded. Plant species diversity was significantly higher in non-invaded sites than the invaded ($t_{(26)} = -4.201$, $p = 0.0003$). In contrast, avifauna diversity across ecological categories and feeding guilds, as well as mammal diversity based on feeding guilds, did not differ significantly between invaded and non-invaded sites. Community composition differed significantly between invaded and non-invaded sites for plants (Global RANOSIM = 0.377, $p = 0.001$), avifauna (Global RANOSIM = 0.110, $p = 0.003$), and mammals (Global RANOSIM = 0.407, $p = 0.001$). SIMPER analysis indicated that plant community dissimilarity was primarily driven by *Cymbopogon afronardus* Stapf (17.04%), *Setaria sphacelata* (Schumach.) Stapf & C.E. Hubb. ex Moss (9.53%), *Cynodon dactylon* (L.) Pers. (5.07%), and *Panicum maximum* Jacq. (4.61%). For avifauna, the species contributing most to dissimilarity between invaded and non-invaded transects included *Cuculus solitarius* Stephens (3.59%), *Colius striatus* Gmelin (2.99%), *Pycnonotus barbatus* Desfontaines (2.88%), and *Camaroptera brachyura* Vieillot (2.80%). Mammalian dissimilarity was mainly influenced by *Kobus ellipsiprymnus* Ogilby (3.94%), *Tragelaphus scriptus* Pallas (3.94%), *Xerus erythropus* É. Geoffroy Saint-Hilaire (3.94%), and *Piliocolobus tephrosceles* Elliot (3.46%). The findings highlight that *Lantana camara* significantly alters plant diversity and may indirectly affect faunal assemblages. Based on these findings, it is recommended that the Uganda Wildlife Authority (UWA**)** prioritize targeted control of *Lantana camara* in invaded areas and investigate the impacts of invasion of forage quality in the reserve.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LISTS OF ABBREVIATIONS AND ACRONYMS

ANOSIM – Analysis of Similarities

CBD – Convention on Biological Diversity

ENH – Empty Niche Hypothesis

ERH – Enemy Release Hypothesis

IUCN – International Union for Conservation of Nature

KWR – Katonga Wildlife Reserve

MAAIF – Ministry of Agriculture, Animal Industry and Fisheries

MHU – Makerere University Herbarium

NEMA – National Environment Management Authority

NWH – Novel Weapons Hypothesis

PPH – Propagule Pressure Hypothesis

SIMPER – Similarity Percentage Analysis

UWA – Uganda Wildlife Authority

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CHAPTER ONE

1 INTRODUCTION

1.1 Background

Globally invasive plant species are a serious threat to biodiversity and ecosystems (Stewart et al., 2021). Their spread has been detrimental to both native flora and fauna leading to species displacement, ecosystems alterations, and disruption of ecological functions (Lowe et al., 2004; Sharma et al., 2005; Shuvar et al., 2021).

Invasive alien species are plants that unintentionally or deliberately enter natural habitats where they are not typically present, causing substantial harm to these new environments (Shuvar et al., 2021). These species are known for their rapid spread, often suppressing native species; some possess allelopathic properties that inhibit the growth of other plants (Cheema et al., 2013; Hejda et al., 2009a; Kato-Noguchi & Kurniadie, 2021). Additionally, invasive plants have adversely affected local fauna by reducing forage availability for herbivores and altering habitat structure and dynamics (Barahukwa et al., 2023).

Globally, alien invasive species have been recognized for negatively impacting the ecosystems. This attracted international efforts, such as those under the Convention on Biological Diversity, emphasize the need to manage and mitigate the impacts of invasive species to preserve global biodiversity(Essl et al., 2020; NEMA, 2016). An ecosystem is said to be invaded if the invasive species cover exceed a threshold (O'Loughlin et al., 2021). Although this threshold may vary among different invasive species, a cover of 20% is generally accepted as the benchmark for most invasions (de Francesco et al., 2023; O'Loughlin et al., 2021)

Africa is particularly vulnerable to the invasion of alien species because of its rich and diverse ecosystems (UNEP, 2010). Significant threats to the continent's biodiversity hotspots, like the Albertine Rift and the Cape Floristic Region, come from invasive species like *Ricinus communis, Eichhornia crassipes* and *Lantana camara* (Bitani et al., 2022; J. R. Wilson et al., 2014; Witt et al., 2018).

Despite being a significant socio-ecological challenge, invasive species remain under-researched in Africa (Egoh et al., 2020; Witt, 2010). Egoh et al. (2020) report that, as of 2020, only 36 scholarly articles on invasive species had been published in Africa, with 22 of these originating from South Africa. This highlights a substantial knowledge gap regarding invasive species in other regions of the continent.

Witt et al. (2018) identified 164 invasive species in East Africa with 30 species considered to have the greatest impacts in terms of transforming natural ecosystems. In Uganda, various invasive species pose significant threats to protected ecosystems, with *Lantana camara, Senna spectabilis*, and *Dichrostachys cinerea* being among the most severe. This study aims to assess the impacts of *Lantana camara on* the native biodiversity of Katonga Wildlife Reserve (KWR) in western Uganda.

Lantana camara was introduced in Uganda during the 1960s as an ornamental plant, it swiftly became a common weed, particularly in savannah systems like lake Mburo National Park (Kiwuso & Otara, 2007). Today, it is classified as an alien invasive species and has spread throughout many of the country's protected areas.

Research on the ecological effects of *Lantana camara* has primarily focused on parameters such as native plant diversity and composition, where invasion often leads to significant reductions in species richness, evenness, and cover through mechanisms like allelopathy and resource competition (Barahukwa et al., 2023; Hejda et al., 2009b; Sharma et al., 2005). Studies have also examined impacts on soil properties, including changes in nutrient cycling and microbial communities, as well as broader ecosystem functions like forage availability and habitat structure (Linders et al., 2019; Paudel et al., 2024; Ruwanza, 2020). For fauna, investigations have highlighted declines in invertebrate diversity and alterations in herbivore assemblages, with cascading effects on higher trophic levels (Hamad et al., 2022; C. Liu et al., 2021; Samways et al., 1996; Suraci et al., 2021).

The drivers of *L. camara* invasion in Africa include anthropogenic disturbances such as land clearance, grazing, and fire regimes, which create open niches for establishment (Kisame & Wanyama, 2017; Shackleton et al., 2017). Propagule pressure from repeated introductions (e.g., as ornamental plants or hedges) and climatic suitability further facilitate spread, while enemy release from natural predators in non-native ranges enhances proliferation (Bhagwat et al., 2012; Day et al., 2003). In East Africa, historical pastoral activities and agricultural expansion have been key accelerators, leading to dense thickets in protected areas (Raphela & Duffy, 2022; Witt et al., 2018)

Birds are particularly affected by L. camara through habitat homogenization, where dense invasions reduce structural complexity, limiting nesting sites, foraging opportunities, and insect prey for insectivorous guilds (Aravind et al., 2010; Bitani et al., 2022). Specialist species often decline in abundance and diversity, while generalists may benefit from increased fruit resources, potentially altering community composition and promoting further seed dispersal (Gooden et al., 2009; Stewart et al., 2021).

A multi-taxa assessment is necessary for understanding L. camara impacts because invasive species exert cascading effects across trophic levels, and single-taxon studies overlook interactions, such as reduced plant diversity affecting herbivores and, in turn, predators (Clusella-Trullas & Garcia, 2017; McCary et al., 2016). This holistic approach reveals ecosystem-level resilience or vulnerabilities, informing comprehensive management in biodiverse savannas (Pyšek et al., 2012; Linders et al., 2019; McGeoch et al., 2024).

However, despite its widespread expansion, limited information exists on its specific impact on native flora and fauna. If the ecological impacts of *Lantana camara* in Katonga Wildlife Reserve remain unstudied, the reserve management risk overlooking critical biodiversity losses and may implement control strategies without scientific backing, potentially resulting in inefficient use of limited conservation resources. Assessing the effects of *L. camara* invasion on plant, bird, and mammal diversity is therefore essential for generating evidence-based knowledge to guide the Uganda Wildlife Authority (UWA) in prioritizing management actions and monitoring biodiversity responses.

1.2 Statement of the research problem

Although *Lantana camara* is widely recognized as a major invasive species in Uganda (Barahukwa et al., 2023; Kisame & Wanyama, 2017; Shackleton et al., 2017), its ecological impacts remain insufficiently studied, especially in savanna ecosystems. Existing research has largely focused on forested ecosystems and has primarily emphasized vegetation responses to invasion (e.g., Barahukwa et al., 2023; Chrispine & Dominic, 2008; Totland et al., 2005). In contrast, the effects of *L. camara* invasion on faunal communities, especially birds and mammals, have received limited attention, and evidence from savanna landscapes is scarce.

In Katonga Wildlife Reserve, there is a paucity of data on the native flora and fauna diversity between *Lantana camara* invaded and non-invaded areas. The extent to which avifaunal assemblages respond to invasion, as well as how medium- to large-sized mammal communities are affected, remains poorly understood. *Lantana camara's* rapid spread in Katonga Wildlife Reserve is attributed to the ecological disturbances caused by pastoralist communities that previously occupied the reserve (Kisame & Wanyama, 2017; Zwick et al., 1997). The species has since expanded extensively across the reserve, posing a great threat to the native species (Kisame & Wanyama, 2017).

1.3 Objectives

1.3.1. Major objective

The study sought to assess the influence of *Lantana camara* invasion on the plant and animal diversity in the Katonga Wildlife Reserve, Uganda.

1.3.2. Specific objectives

The specific objectives of the study were to:

i. Assess the native plant species diversity in *Lantana camara*-invaded and non-invaded areas of Katonga Wildlife Reserve,

ii. Assess the diversity of avifauna assemblages in *Lantana camara*-invaded and non-invaded areas of Katonga Wildlife Reserve,

iii. Assess the diversity of Medium to large-sized mammals in *Lantana camara*-invaded and non-invaded areas of Katonga Wildlife Reserve

1.4 Hypotheses

In this study it was hypothesized that;

(i) Plant diversity does not vary between *Lantana camara* invaded and non-invaded areas.

(ii) Avifauna diversity does not vary between *Lantana camara* invaded and non-invaded areas.

(iii) Medium to large sized mammal diversity does not vary between *Lantana camara* invaded and non-invaded areas.

1.5 Significance of the study

The study generates a comprehensive assessment of how *Lantana camara* invasion influences biodiversity in Katonga Wildlife Reserve by simultaneously examining native plant diversity, avifaunal assemblages, and medium- to large-sized mammals. This moves beyond vegetation-based assessments and provides a multi-taxa perspective that better reflects ecosystem-level impacts. The findings address the gap in understanding invasion dynamics in savanna ecosystems, which differ from the forested ecosystems where most Ugandan studies on *Lantana camara* have been conducted. The results offer evidence-based guidance to the Uganda Wildlife Authority and conservation partners in designing targeted control and monitoring strategies for KWR and other Protected areas. Additionally, the results serve as a baseline for ecological researchers investigating invasive species impacts and ecosystem resilience worldwide.

CHAPTER TWO

2 LITERATURE REVIEW

2.1 Invasive Species and their characteristics

Invasive species pose a worldwide threat, endangering native biodiversity, disrupting ecosystem functions, depleting natural resources, and impacting local economies and human health (C. Liu et al., 2021; Ricciardi, 2014; Simba et al., 2013). They are introduced, either intentionally or accidentally, into new geographic areas where they spread and become established. In most cases, nonnative species fail to establish self-sustaining populations in new habitats due to unfavorable climate, and soils, as a result, they don't spread far away from their point of introduction and often cause no significant impacts to their new environments. However, a few species become invasives, they spread very fast, replace native species hence leading to environmental degradation (Ricciardi, 2014).

Invasive species are characterized by their ability to spread very fast, producing large number of seeds, easily adapting and being resistant to environmental changes, having several dispersal pathways, allelopathy, among others some of which are shared by all invasives while others are species specific (Bhagwat et al., 2012; Devin & Beisel, 2007; Van Kleunen et al., 2015). Through using a combination of these characteristics, they dominate new ecosystems, replace native species thereby disrupting the native ecosystems (Van Kleunen et al., 2015). This affects both plants and animals, for example, it is estimated that *Lantana camara* invasion is threatening 1400 native species worldwide (The State of Queensland, 2023). Invasive species displace native plants, reducing available forage for animals, which in turn leads to a decline in wildlife populations. This can trigger both ecological and economic downturns (Bhagwat et al., 2012).

As a result, several actors worldwide have come up with frameworks to control, prevent and manage the spread of invasive species. The International Union for Conservation of Nature came up with the top one hundred invasive species of the world (Lowe et al., 2004). The Convention on Biological Diversity (CBD) has called member countries to work together in the control of invasive species, as a result, the CBD has overtime come up with different guidelines to control the spread of invasive species including the Post- 2020 target on invasive alien species (Essl et al., 2020). Several countries worldwide have come up with different strategies and frameworks to fight biological invasions and restore the affected ecosystems (Abebe, 2018; Barahukwa et al., 2023; Bitani et al., 2022; IUCN, 2013; Keller et al., 2011; Shuvar et al., 2021).

2.2 Major invasive species of the world

The International Union for Conservation of Nature published a list of the a hundred of the world's worst invasive species of the world (Lowe et al., 2004). The listed plant species included *Spathodea campanulata, Acacia mearnsii, Schinus terebinthifolius, Imperata cylindrica, Pinus pinaster, Opuntia stricta, Myrica faya, Arundo donax, Ulex europaeus, Hiptage benghalensis, Fallopia japonica, Hedychium gardnerianum, Clidemia hirta, Pueraria montana var. lobata, Lantana camara, Euphorbia esula, Leucaena leucocephala, Melaleuca quinquenervia, Prosopis glandulosa, Miconia calvescens, Mikania micrantha, Mimosa pigra, Ligustrum robustum, Cecropia peltate, Lythrum salicaria, Cinchona pubescens, Ardisia eliptica, Chromolaena odorata, Psidium cattleianum, Tamarix ramosissima, Sphagneticola trilobata,* and *Rubus ellipticus*.

In Uganda, the most recorded invasive species *include Lantana camara, Dichrostachys cinerea, Brugmansia suaveolens, Senna spectabilis, Capparis tomentosa, Imperata cylindrica Mimosa pigra*, *Senna didymobotrya*, *Caesalpinia decapetala, Cuscuta spp*, among others (Mungatana & Ahimbisibwe, 2012; Okia & Amuno, 2021; Ronald et al., 2016)

Of these, *Dichrostachys cinerea, Senna didymobotrya, Lantana camara, Mimosa pigra and Capparis tomentosa have been recorded* to have detrimental impacts (Barahukwa et al., 2023; MAAIF, 2015; Mungatana & Ahimbisibwe, 2012).

This study focuses on *Lantana camara* and its impacts on the native flora and fauna in Katonga wildlife reserve.

2.3 Theories that explain biological invasions

a) The Invasion Meltdown

This theory was established by Simberloff & Holle (1999). It explains how the establishment of one invasive species in an ecosystem can lead to the establishment and spread of additional invasive species. The presence of invasive species can lead to cascading effects, such as changes in nutrient cycles, trophic interactions, or habitat structure, which make the ecosystem increasingly hospitable to invasives and less supportive of native species. This can be illustrated by an invasive plant which alter soil chemistry in a way that benefits an invasive herbivore, invasive predator or other invasive plants that may reduce native competitors, allowing another invader to thrive. For example, the introduction of the Japanese white-eye, *Zosterops japonicus* facilitated the dispersal of *Myrica faya* seeds in the natural areas of Hawaiian ecosystems, *Myrica faya* invaded nitrogen-poor volcanic soils in the area, enriching the soil and creating conditions that facilitate the invasion of other non-native plant species (Simberloff & Holle, 1999). Another example is the invasive yellow crazy ant (*Anoplolepis gracilipes*) which formed mutualistic relationships with introduced scale insects, leading to high ant densities that devastate native red crab populations (O'Dowd et al., 2003). This disruption has cascading effects on the island's ecosystem, demonstrating a severe invasion meltdown.

b) Biotic Resistance Paradigm

This theory suggests that ecosystems with high native species diversity and strong ecological interactions are less susceptible to invasions by non-native species (Levine & D'Antonio, 2018). This is because the native species effectively utilize available resources, leaving little opportunity for invaders to establish and thrive. For example, a study in California grasslands found that high native plant diversity reduced the establishment and growth of invasive grasses by effectively utilizing the resources thereby creating a competitive environment for invaders (Levine, 2000). Another study found out that in rocky intertidal zones, native species like barnacles and mussels occupy physical space, preventing the establishment of invasive algae (Stachowicz et al., 1999).

c) Enemy Release Hypothesis (ERH)

This theory was proposed by Keane and Crawley (2002). It states that plant species, on introduction to an exotic region, experience a decrease in regulation by herbivores and other natural enemies, resulting in a rapid increase in distribution and abundance. This theory is one of the most commonly recognized in the study of biological invasions and is founded on three key principles: (1) natural predators play a critical role in managing plant populations; (2) these predators tend to affect native species more significantly than they do exotic ones; and (3) when there is a decrease in the regulation by enemies, plants can take advantage of this, leading to higher population growth.

Numerous studies provide empirical support for the ERH. For instance, *Lantana camara*, which originates from Central and South America, has become invasive in multiple locations, including Africa, Asia, and Australia. In East Africa, especially Uganda, it grows rapidly due to the lack of natural herbivores and pathogens that would normally control its populations unlike in its native range. This liberation from natural foes enables *Lantana camara* to outcompete indigenous plant species (Shackleton et al., 2017). Another case is *Melaleuca quinquenervia* (Paperbark tree), introduced from Australia, which has invaded wetland areas in Florida, USA, mainly in the Everglades. The absence of natural herbivores in its new surroundings has allowed it to expand significantly. However, after the introduction of host-specific herbivores from Australia, its invasive spread was somewhat controlled (Center et al., 2014) Furthermore, *Solidago canadensis* (Canada goldenrod), originally from North America, has become invasive in parts of Europe and China (H. Liu & Stiling, 2006). Evidence suggests that it encounters significantly fewer herbivores and pathogens in its introduced regions, reinforcing the ERH (H. Liu & Stiling, 2006).

Although the ERH is strongly supported by evidence, it has several drawbacks, one being its universal applicability. Some invasive species can prosper even when natural enemies are present, indicating that other mechanisms, such as phenotypic plasticity, allelopathy, or rapid evolutionary changes, may significantly contribute (Colautti et al., 2004). The interaction of multiple factors, including propagule pressure, disturbance regimes, and new competitive strategies, rather than merely enemy release, is another critique of the hypothesis.

Despite the criticisms of the ERH, it still holds important practical importance for managing invasive species, such as the implementation of biological control strategies. These strategies may involve reintroducing natural enemies that co-evolved with the invasive species from its native range. For instance, in India and South Africa, various leaf-mining and flower-feeding insects from *Lantana camara*'s native habitat have been released to limit its growth and reproduction (Day et al., 2003). However, caution is necessary when implementing biological controls, as poorly selected control agents can negatively impact non-target native species, resulting in unintended ecological outcomes. Therefore, comprehensive risk assessments and extensive ecological studies are crucial prior to initiating biocontrol projects. While biocontrol based on ERH principles can be successful, it proves most effective when combined with other control methods, such as mechanical removal, habitat restoration, and efforts to raise community awareness.

d) The Novel Weapons Hypothesis (NWH)

The Novel Weapons Hypothesis (NWH) presents an alternative perspective on the successful establishment of invasive plant species in non-native ecosystems. It was initially proposed by Callaway in 2000 but further explained by Callaway and Ridenour in 2004 (Bais et al., 2003;

Callaway & Ridenour, 2004), the NWH suggests that certain invasive species possess distinctive biochemical or physical traits called "novel weapons" that confer a competitive advantage over native species in the areas they invade. These "weapons" are typically secondary metabolites or allelopathic chemicals against which native species have not evolved any resistance. For example, allelopathic substances released by invasive plants into the soil or surrounding environment can inhibit seed germination, suppress growth, or disrupt beneficial interactions (such as those with mycorrhizal fungi) that are vital to native plants. Such chemicals are unfamiliar to native species, which have evolved without exposure to these compounds, leaving them without physiological or ecological defenses and, thus, more vulnerable. This biochemical advantage allows the invader to establish, spread, and dominate the new environment. For example, *Alliaria petiolata* (Garlic mustard) was introduced to North America from Europe. It releases glucosinolates that are toxic to arbuscular mycorrhizal fungi, which many native North American plants depend on. It alters soil microbial communities, thereby reducing native plant species survival (Callaway & Ridenour, 2004). Another example is *Centuarea diffusa* (Diffuse knapweed), an invasive species in North America introduced from Eurasia. It produces a compound called (±)-catechin, which is allelopathic to native grasses. It suppresses root growth in native competitors, leading to grassland degradation (Bais et al., 2003). Lastly*, Lantana camara* exerts allelopathic effects on native plants through leaf litter and root exudates that inhibit germination and seedling growth, it alters plant community structure and reduces native plant diversity in invaded ecosystems (Sharma et al., 2005).

The theory however has some limitations such as a Lack of universal application, since some invasives succeed due to growth traits, disturbance tolerance, or propagule pressure. It is also difficult to prove novelty. It can be challenging to demonstrate that the compounds used by invaders are entirely "novel" to the invaded ecosystem or that native species have had no prior exposure. The presence of Confounding factors or processes (e.g., nutrient cycling changes, disturbance regimes) may also explain native species declines rather than the novel compounds (Callaway & Ridenour, 2004)

Despite these criticisms, the NWH provides valuable insight into the below-ground mechanisms of invasion that are often overlooked. For example, the need for reconditioning the soil after removal to allow native species to re-establish themselves, since some invasive plants alter soil chemistry or microbial communities. There is also a need for early detection, monitoring, and response systems upon identification of species with allelopathic properties in the ecosystem.

e) Propagule Pressure Hypothesis (PPH)

The propagule pressure hypothesis has been supported across various taxa and ecosystems, indicating that the pressure exerted by introduced individuals may be a more reliable predictor of invasion success than the characteristics of the species or the habitats themselves (Lockwood et al., 2005). This is a key concept in invasion ecology as it highlights the role of introduction efforts in influencing the establishment and spread of invasive species. Propagule pressure involves both the quantity of non-native individuals introduced and the frequency of these introductions into new environments. The primary assertion is that species introduced in greater numbers, or more frequently, tend to have a greater likelihood of forming self-sustaining populations. Lockwood et al. (2005) discusses several reasons for PPH one of them being the ability to overcome demographic stochasticity. This is explained by the fact that small introduced populations often fail to establish due to random events, such as all individuals being of the same sex or facing disease outbreaks, conversely, introducing a larger number of individuals increases the chances that enough will survive and successfully reproduce. Furthermore, propagule pressure can contribute to invasion success by mitigating Allee effects. Certain species need a minimum number of individuals; this is referred to as the "critical population size needed to create viable populations." It is essential for social interactions, finding mates, or engaging in cooperative behaviors. In addition, multiple introductions from diverse source populations can enhance genetic diversity, which boosts the species' adaptive capacity and resilience in unfamiliar environments. Lastly, heightened exposure opportunities arise when introductions occur more frequently, increasing the likelihood that these propagules will arrive during favorable environmental conditions, such as appropriate seasonal timing or after disturbances.

A common example of this phenomenon is the zebra mussel (*Dreissena polymorpha*), which has been introduced to North America on multiple occasions through ballast water, leading to its extensive establishment in the Great Lakes (Ricciardi, 2007). In a similar way, *Lantana camara* has undergone repeated introductions to East Africa for both ornamental and utilitarian purposes, facilitating its proliferation across various ecosystems (Day et al., 2003). While the Propagule Pressure Hypothesis (PPH) has faced critiques for its descriptive nature rather than a mechanistic approach, it nonetheless offers a valuable framework for managing pathways of introduction and prioritizing strategies for prevention in the realm of invasion ecology (Lockwood et al., 2005).

f) Empty Niche Hypothesis (ENH)

This theory offers an important lens through which we can understand why some ecosystems are particularly vulnerable to biological invasions. At its core, the hypothesis suggests that invaders are successful not just because they are aggressive or fast-growing, but because they find and fill ecological roles that are unoccupied or underutilized by native species (Elton, 2020), that is to say, the invader steps into a gap that the local ecosystem isn't fully using.

For example, nitrogen-fixing trees such as *Leucaena leucocephala* and some Acacia species have become invasive in grasslands across parts of Africa and Australia. These ecosystems often lack native plants that perform nitrogen fixation, leaving a "functional gap" that these introduced species exploit (Richardson et al., 2000). Their ability to improve soil fertility gives them an advantage and can lead to dramatic shifts in plant community composition.

Another case is *Carpobrotus edulis* which has invaded coastal Mediterranean-type environments in California. It thrives in sandy, nutrient-poor soils where native plant cover is naturally sparse, it occupies space and resources that would otherwise remain largely unused thereby outcompeting native species that are not adapted to such environments (D'Antonio & Mahall, 1991).

The theory is limited by the fact that it is not always easy to prove that a niche was truly "empty" before an invader arrived. However, it encourages ecologists and managers to think beyond just species counts and consider the full range of ecological roles within a system and can be helpful in designing restoration strategies, where reintroducing or reinforcing native species with underrepresented traits may help "close" those empty niches and make the habitat more resistant to future invasions (Richardson & Pyšek, 2007).

2.4 Impacts of biological invasions on Ecosystems

a) Alternation of native species dynamics.

Biological invasions may alter ecosystem dynamics such as competition, predation and herbivory (Catford et al., 2012; Hamad et al., 2022; McGeoch et al., 2024). Invasive species may outcompete the native species for resources such as light and nutrients thereby disrupting local ecosystems (Catford et al., 2012). Trophic cascades may be upset by the introduction of invasive species. An invasive predator might, for instance, lower herbivore populations, which could alter the nitrogen cycle and vegetation structure. On the other hand, the ecosystem may be further altered if the invading predator outcompetes or preys on native apex predators, leading to a proliferation of lower-level consumers (Dunham et al., 2022; Ehrenfeld, 2010). The efficiency of predators may be impacted by habitat changes caused by some invasive species. Invasive plants, for instance, may alter the structure of the vegetation, making it harder for native predators to hunt or giving prey species a place to hide, which would alter the dynamics of predator-prey relationships (Boynton et al., 2021; McCary et al., 2016). In herbivory, herbivores may have less access to native vegetation when invasive plants outcompete them for light, nutrients, and water. For native herbivores, this may result in fewer food sources, necessitating dietary changes or population decreases. By displacing native herbivores, bringing in invading ones, and changing the dynamics between plants and herbivores, biological invasions disturb herbivory. Native plants may deteriorate and be overused as a result of invasive herbivores outcompeting them. Additionally, invasive plants may give native herbivores a different source of food, which could change how they eat and encourage the spread of invasives. When overgrazing lowers plant biomass, it can lead to trophic cascades that affect higher trophic levels and other herbivores (McGeoch et al., 2024).

b) Changes in Species Richness and Diversity

The introduction of invasive species tends to have the opposite effect on ecosystem services. They either increase habitat complexity or heterogeneity, which tends to cause abundances and/or species richness (Shiferaw et al., 2018). For example, a study on the impacts *Lantana camara* on Plant Community across Habitat Types in Central Nepal found out that invaded sites faced a significant decline in species diversity (Paudel et al., 2024) while other studies, such as Kumar Rai & Singh (2020) and Lean (2021), have found that some invasive species can increase biodiversity, thereby contributing to more valuable ecosystems by enhancing local species richness and, as a result, improving ecosystem services

c) Disruption of Trophic Interactions

Alterations in plant types can directly impact herbivores by decreasing the quantity or quality of their plant hosts. Meanwhile, changes in habitat structure, microenvironment, and soil or litter characteristics can indirectly influence the availability of prey or the number of predators, thereby modifying trophic interactions (Clusella-Trullas & Garcia, 2017). A study examining the impact of invasive species on the trophic structure of green and brown food webs in terrestrial ecosystems found that invasive plants had no detectable effect on higher trophic levels in grasslands. In wetlands, however, they had strong, opposing effects on upper trophic levels of green food webs, while in woodlands, invasive species altered the abundance of at least one trophic level in both green and brown webs (McCary et al., 2016). Invasive species can also modify trophic interactions via trophic cascades, where invasive predators decrease the populations of consumers responsible for nutrient transfer between habitats. This reduction leads to decreased nutrient inputs and changes in nutrient cycling within the ecosystem. (Ehrenfeld, 2010).

d) Evolutionary response

Ecosystems and species often exhibit evolutionary responses to biological invasions as they adapt to the pressures introduced by invasive species (Strauss et al., 2006). Native species may evolve new traits, such as improved defences against invasive predators or herbivores, or enhanced competitive abilities to contend with invasive plants (Lau, 2006). These adaptations can include changes in morphology, behaviour, or reproductive strategies (Melotto et al., 2020; Oduor, 2022; Strauss et al., 2006). For example, the population of red cedars evolved greater resistance to deer browsing within 100 years post-invasion, similar to native mollusks in marine ecosystems that developed thicker shells to reduce predation by introduced crabs (Strauss et al., 2006). The same study explains how *Rana aurora* frogs in ponds invaded by bullfrogs evolved responses to chemical cues whereby frogs from invaded ponds reduce their foraging activity and increase refuge use when presented with chemical cues from bullfrogs, a trait which was not exhibited by frogs in non-invaded ponds hence the genotypes from invaded regions experience less predation than genotypes from uninvaded ponds.

2.5 Invasive *Lantana camara*

Lantana camara is a compact perennial shrub reaching heights of up to 2 meters, it establishes dense thickets across various environments (Witt et al., 2018). *Lantana camara* has its origin from South and Central America, it ranks among the globe's ten most detrimental weeds, infiltrating millions of hectares of natural and cultivated landscapes, inflicting considerable ecological and socio-economic harm (Qin et al., 2016). Today, it is widespread throughout Asia, Europe and Africa (Devin & Beisel, 2007; IUCN, 2013; C. Liu et al., 2021; Raphela & Duffy, 2023). Similarly it is found in Uganda were it was introduced during the 1960s, and it has swiftly became a common weed, particularly in savannah ecosystems systems like lake Mburo National Park (Kiwuso & Otara, 2007). It is classified as an alien invasive species according to the Invasive Species Specialist Group (ISSG) by the

International Union for the Conservation of Nature (IUCN) and has spread throughout many of the country's protected areas (Lowe et al., 2004).

Its adaptability to various climatic conditions allows it to thrive in a wide range of habitats, including riverbanks, mountain slopes, valleys, pastures, and commercial forests, hindering access, land use, reduced animal quality when consumed, decreased crop yield, and compromised functionality of ecosystems (Sharma et al., 2005).

2.6 Impacts of *Lantana camara on* native flora

Studies by (Kato-Noguchi & Kurniadie, 2021; Paudel et al., 2024; Ruwanza, 2020; Sharma et al., 2005) revealed that *Lantana camara* roots, leaves and stems produce allelopathic compounds which hinder other native species from germinating and establishing themselves, this as a result reduces interspecies competition making *Lantana camara* to swiftly establish its self in with limited competition (Kato-Noguchi & Kurniadie, 2021; Paudel et al., 2024).

Comparative studies have shown that *Lantana camara* invasion is associated with a reduction in plant species diversity in invaded ecosystems (Ruwanza, 2020). According to Ruwanza, (2020), *Lantana camara* invasion affects the trees and shrubs more than herbs and grasses, however, this contradicts with Paudel et al. (2024) who reported a sharp decline in herbs and grasses abundance in *Lantana camara* invaded sites.

However much *Lantana camara* is known for replacing the native vegetation, there are some species which adapt and can be recorded in both invaded and non-invaded sites (Barahukwa et al., 2023; Paudel et al., 2024; Ruwanza, 2020).

2.7 Impacts of *Lantana camara on* native fauna

Lantana camara has a significant negative impact on native plant species and can gradually diminish endemic plants, which are a crucial source of forage and browse for herbivores (Simba et al., 2013). This can be escorted with a decline in species diversity and abundance which affects ecosystem functionality such as a disruption in the food chain (McCary et al., 2016).

In addition, studies have found out that *Lantana camara* can be poisonous to both domestic and wild animals (Kumar et al., 2016). It is known to cause hepatotoxicity, photosensitization, and intrahepatic cholestasis in nearly all animals (Kumar et al., 2016). Lantana toxicity also disrupts both haematological and biochemical parameters in the body, its toxic compound lantadene leads to fatty degeneration, bile duct hyperplasia, gallbladder edema, degeneration of liver cells, and portal fibrosis. According to The State of Queensland (2023), *Lantana camara* poisoning primarily occurs when animals are introduced to areas infested with the plant. Animals raised in lantana-affected regions tend to avoid it unless forced by a lack of other food. Young animals are especially vulnerable. The severity of poisoning symptoms depends on the amount and type of *Lantana camara* consumed, as well as the intensity of light exposure in certain cases.

For avifauna species, a study in Mahadeshwara reserve forest, south India revealed that *Lantana camara* is known to alter bird community structure by reducing diversity, with certain bird guilds being more affected than others, this study also concluded that *Lantana camara* invasion affects the bird species diversity, species richness, abundance and evenness (Aravind et al., 2010).

CHAPTER THREE

3 STUDY AREA AND METHODS

3.1 Description of Study area.

Katonga Wildlife Reserve covers an area of 207 km^2 and lies between 30°34' and 30°55'E, and 0°12' and 0°18N within Kiruhura, Kazo and Kamwenge Districts in Western Uganda. According to Game (Preservation and Control) Act Chapter 198, (2000), its boundaries Commence at a point where the Kaisunga river joins the Katonga river; continues in a north-easterly direction following the eastern bank of Kaisunga river to its confluence with the Nyandagura river; it then continues in a north-easterly direction to the highest point of Kabuya hill; moves in an easterly direction to Kitemba hill; then to the peak of Zina hill; follows the western side of Mparo-Kabogole road (track) to the Katonga river; extends following the northern bank of Katonga river to the point of commencement.

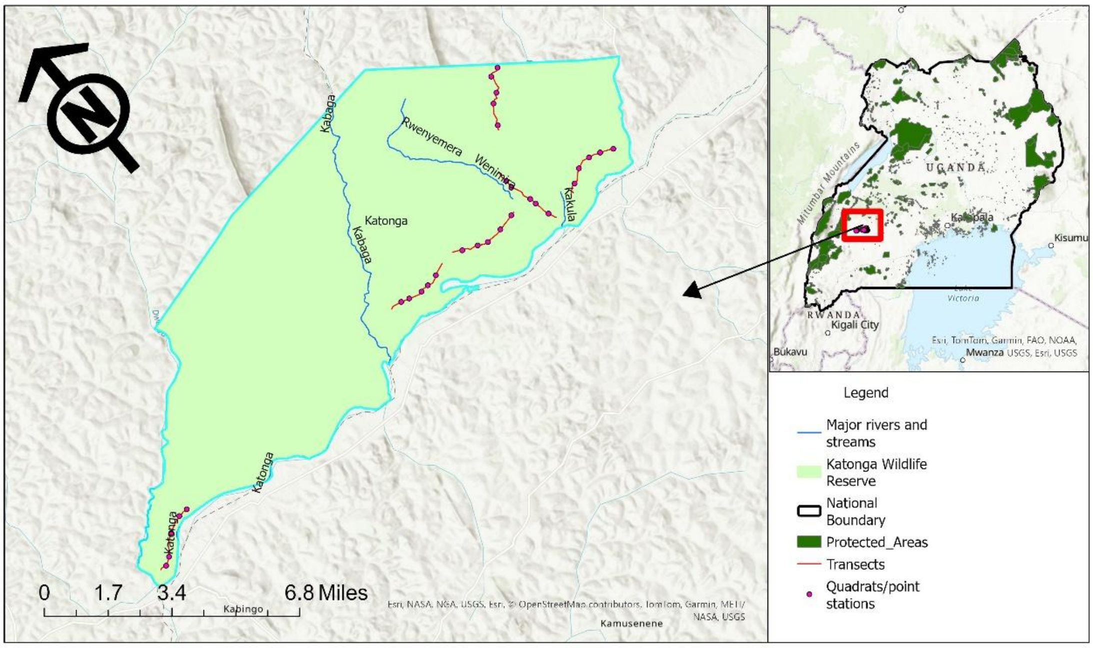

Figure 1: A location map of Katonga Wildlife Reserve showing the study transects and sampling points

The vegetation Katonga Wildlife Reserve is a combination of grasslands, woodlands, wetlands and riverine forests (Zwick et al., 1997). The grasslands are dominated by Cymbopogon species and Themeda species, while the woodlands are dominated by a mixture of Acacia and Combretum trees, while the wetlands are dominated by Cyperus species.

Zwick et al. (1997) recorded A total of 65 mammal species including 30 medium-to-large sized species and 35 small mammals have been recorded. And a total of 154 bird species, 17 reptile species, 15 amphibian species, 99 butterfly species, and eight dragonfly and damselfly species.

3.2 Methods

3.2.1 Study design.

The study employed a cross section mixed methods approach where quantitative data on plant, bird, and mammal communities were collected concurrently from the same sites (invaded and non-invaded) and integrated during interpretation to provide a multi-taxa assessment of *Lantana camara* invasion effects. Six transects were laid in the study area each measuring three kilometers (Bennun et al., 2002). The location for each transects was selected purposively with 3 transects established in *Lantana camara* - invaded areas, and 3 in the non-invaded areas. Each transect was laid at least 1km away from each other. This was to minimize the chances of double counting and to ensuring that variability within each habitat is considered. Along each transect served as a sampling unit for the different taxa.Quadrats measuring 20m X 20m, 10m X 10m and 1m X 1m were laid to study vegetation diversity. *Lantana camara*-invaded areas were defined as sites where *L. camara* cover exceeded a threshold of 20%, following established criteria for invasive plant dominance (de Francesco et al., 2023; O'Loughlin et al., 2021). Percentage cover of *L. camara* was estimated visually in the field.

3.2.2 Data collection.

3.2.2.1 Native plant diversity in the *Lantana camara* infested areas of the Katonga Wildlife Reserve

Along each transect, five nested quadrats (Hill, 2005) were established at random distances using a random number generator. Each nested quadrant had a 20m X 20m plot to study trees, a 10m X 10m plot to study shrubs and a 1m X 1m for herbaceous plants.

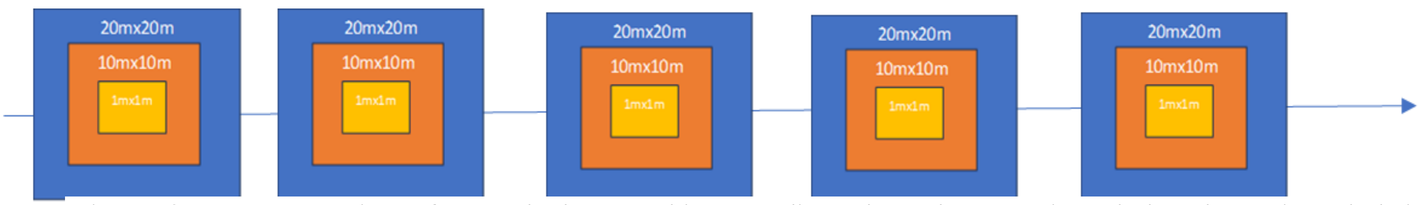

Figure 2: Representation of Nested Plots used in sampling plants between invaded and non-invaded sites in Katonga Wildlife Reserve

Plant species were identified in each quadrant, and voucher specimens were collected for those that could not be identified in the field. These specimens were pressed and sent to the Makerere University Herbarium (MHU) for further identification.

3.2.2.2 Avifauna assemblages in the *Lantana camara* infested areas of the Katonga Wildlife Reserve

The point count method due to its efficiency in detecting species richness and abundance (Bennun et al., 2002; Hill, 2005; Larsen, 2016) was adopted for surveying bird diversity in the study area. Each point count lasted 15 minutes, during which all birds seen or heard within an approximate radius of about 100 metres were recorded. Bird surveys in each transect were conducted between 7:00 a.m. and 11:00 a.m. when birds were most active. Birds were identified using a Field guide to the birds of East Africa: Kenya, Tanzania, Uganda, Rwanda, Burundi (Stevenson & Fanshawe, 2020).

3.2.2.3 Medium to large sized mammal species distribution in the *Lantana camara* infested areas of the Katonga Wildlife Reserve

Mammal counts were conducted along each transect during early morning hours (approximately 7:00 a.m. to 11:00 a.m.), when mammals are generally more active (Larsen, 2016). All mammals seen or heard within a distance of 100-meter on either side of the transect line were recorded. Each transect measured 3 km in length and was walked slowly at an average speed of approximately 1 km per hour to maximize detectability.

3.2.3 Data analysis

Data analysis was done for species richness, alpha diversity and beta diversity. Species richness was assessed by counting the total number of species recorded in each transect. Alpha diversity was analyzed using the Shannon–Wiener Index, expressed as

$$\mathrm{H}' = -\sum_{i=1}^{S} (p_i) ln(p_i)$$

where is the Shannon–Wiener diversity index, S is the total number of species in the sample or transect, and $p_i$is the proportional abundance of species i, calculated as $p_i = n_i/N$. Here, $n_i$represents the number of individuals of species i, and N is the total number of individuals of all species in the sample. The natural logarithm $\ln(p_i)$ is used. The negative sign ensures that H'yields a positive value, because $\ln(p_i)$ is negative when $0 < p_i < 1$. The index increases with both the number of species (richness) and how evenly individuals are distributed among them (evenness). In other words, a higher Shannon value indicates a community with many species that are more evenly represented, while a lower value indicates fewer species or dominance by a few species. The index was analyzed in species diversity and richness - IV (SDR-IV) software. For plants and avifauna, diversity indices were computed for each quadrat and point count station, respectively, in both invaded and non-invaded sites. Birds were further categorized by ecological traits, including ecological categories and feeding guild, and diversity indices were computed separately for these groups to assess invasion impacts. Avifauna ecological categories followed the classification from the *Bird Atlas of Uganda* (Carswell et al., 2007), including forest generalists (F-species), forest visitors (f-species), waterbird specialists (W), waterbird non-specialists (w), grassland species (G), and cross-habitat users such as fw, Fw, and WG species. Feeding guilds were assigned based on Bennun et al., (2002) and included insectivores, frugivores, granivores, omnivores, and carnivores. For medium- to large-sized mammals, diversity indices were calculated along each transect and species were grouped by feeding guild as grazers, browsers, omnivores, carnivores, and browser-grazers.

Statistical comparisons and graphical visualization for alpha diversity were performed in R version 4.3.1, using tidyverse, vegan, and ggplot2 packages for data wrangling, diversity computation, visualization, and hypothesis testing (Oksanen et al., 2024; Team R Core, 2016). Data were first tested for normality using the Shapiro–Wilk test. Where assumptions of normality and homogeneity of variances were met, differences between invaded and non-invaded sites were evaluated using independent-samples *t*-tests. When these assumptions were violated, the non-parametric Wilcoxon rank-sum test was applied as an alternative.

Variations in species composition between invaded and non-invaded transects were tested using Analysis of Similarities (ANOSIM) and Similarity Percentage (SIMPER) in CAP 3.1 (Seaby et al., 2006). The ANOSIM test is based on a rank similarity matrix, producing the statistic:

$$R = \frac{r_B - r_W}{\frac{1}{2}N(N-1)}$$

Where R = ANOSIM test statistic, measures separation between groups (–1 to 1). r_B = average rank of dissimilarities between groups (e.g., invaded vs. non-invaded). r_W = average rank of dissimilarities within groups. N = total number of samples. 1/2N(N−1) = total number of unique sample pairs in the dataset. A higher R means stronger differences between groups; R≈0 means no difference; negative R means more similarity within groups than between them.

SIMPER analysis, on the other hand, decomposes the average Bray–Curtis dissimilarity between groups into contributions from each species:

$$D_{jk} = \frac{a_i + b_i}{a + b + c}$$

Where; D_{jk} is the contribution of species iii to dissimilarity between sample j and sample k. a_i is the species i present only in sample j. b_i species i present only in sample k. c are the number of species shared between both samples while a+b+c are the total species observed between the two samples. This formula calculates pairwise dissimilarity, which SIMPER partitions to show the percentage contribution of each species to the overall difference between groups

CHAPTER FOUR

4 RESULTS

4.1 Plant diversity

A total of 103 plant species were recorded from thirty quadrats distributed along six transects across the invaded and non-invaded sites. Of these, 59 species were recorded along the invaded transects while the non-invaded transects had 87 species. The recorded species belonged to 46 plant families and 76 genera. The species accumulation curve (Figure 3) showed a steep initial rise, indicating rapid discovery of new species in the first few quadrats sampled, followed by a gradual increase tending towards an asymptote. This suggests that the sampling effort was sufficient to capture the majority of the species present in both habitats, with diminishing returns for additional sampling.

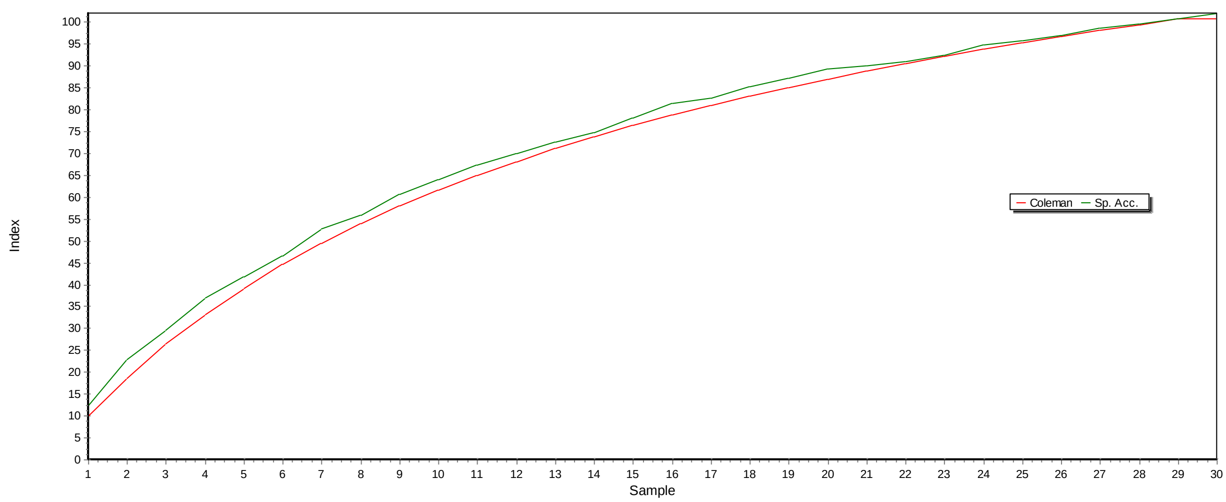

Figure 3: Coleman and species accumulation (Sp. Acc.) curves of the recorded plant species along six transects in Katonga Wildlife Reserve

Plant species diversity ranged between 2.08 to 3.45 in non-invaded sites while for the invaded transects, they ranged between 1.61 to 2.64, with non-invaded sites having higher diversity values (Figure 4)

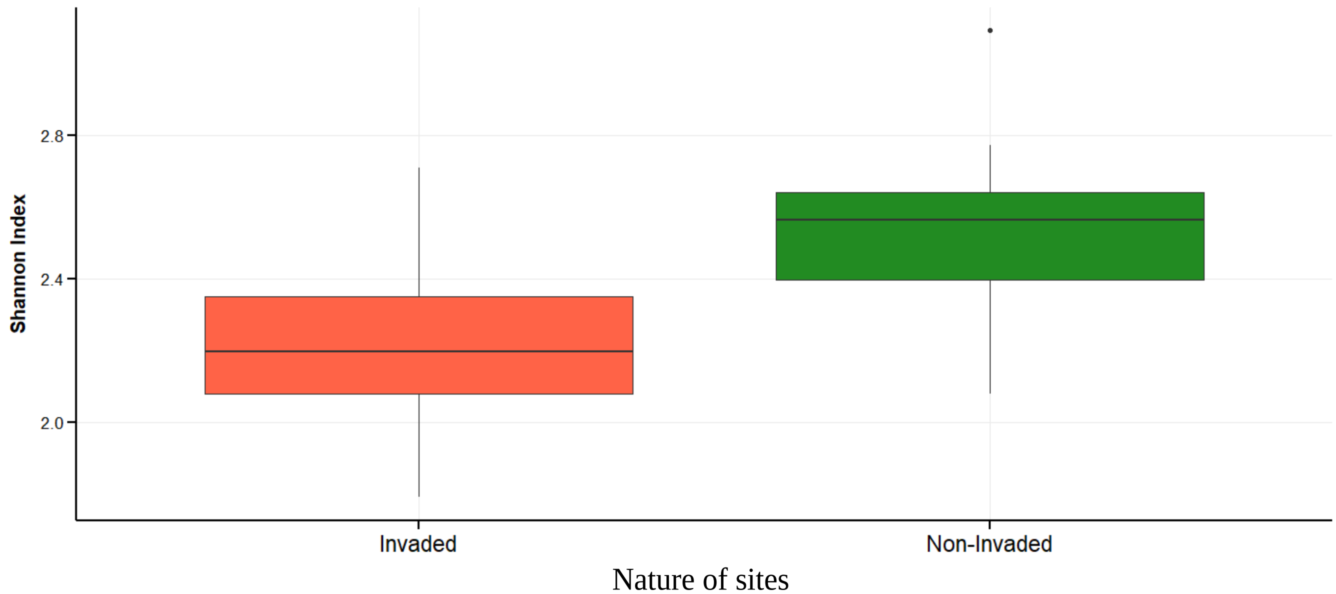

Figure 4 Plant species diversity between invaded and non-invaded sites in Katonga Wildlife Reserve

When the diversity values were analyzed statistically, there was a significant difference between the invaded and non-invaded transects ($t_{(26)} = -4.201$, $p = 0.0003$).

Analysis of Similarities also showed a significant separation in community composition between invaded and non-invaded sites (Global $R_{ANOSIM} = 0.3769$, $p = 0.001$). This suggests that the groups were more similar within than it would be by chance. Based on Similarity Percentage (Table 1 and Table 2), average similarity within the invaded transects was 10.88 and 13.39 in the non-invaded transects. Average dissimilarity between the invaded and non-invaded transects was 95.43

Table 1: Contribution of individual plant species to the overall similarity within the invaded and non-invaded transects in Katonga Wildlife Reserve. Species are arranged according to percentage contribution to the similarity within the ecosystem types and only those contributing >2% are shown. Average similarity and percentage of cumulative similarity are also given.

Species	Average abundance	Average Similarity	% Contribution	Cumulative %
Invaded transects				
Setaria verticillata (L.) P.Beauv.	3.3	2.93	26.86	26.86
Vepris nobilis (Delile) Mziray	1.4	2.11	19.36	46.22
Cissus quadrangularis L.	0.53	0.70	6.46	52.68
Rhus longipes Engl.	0.6	0.62	5.74	58.42
Panicum maximum Jacq.	0.93	0.52	4.78	63.21
Phyllanthus ovalifolius Forssk.	0.47	0.47	4.36	67.57
Desmodium adscendens (Sw.) DC.	0.6	0.43	3.96	71.53
Commelina benghalensis L.	0.67	0.38	3.53	75.06
Setaria chevalieri Stapf	6.13	0.37	3.43	78.49
Grewia similis K.Schum	0.4	0.37	3.38	81.87
Vernonia amygdalina Delile	0.53	0.35	3. 2	85.07
Setaria sphacelata (Schumach.) Stapf & C.E.Hubb. ex Moss	3.13	0.24	2.20	87.27
Non invaded transects				
Cymbopogon afronardus Stapf	30	6.17	46.03	46.03
Setaria sphacelata (Schumach.) Stapf & C.E.Hubb. ex Moss	14.33	1.78	13.30	59.33
Setaria verticillata (L.) P.Beauv.	5.67	0.85	6.3	65.70
Cynodon dactylon (L.) Pers.	8.67	0.68	5.08	70.78
Tagetes minuta L.	6.33	0.5	3.73	74.51
Panicum maximum Jacq.	6.60	0.41	3.08	77.59
Sporobolus pyramidalis P.Beauv.	7.73	0.34	2.55	80.14

Table 2: Results of SIMPER analysis highlighting the species contributing most to the dissimilarity between invaded and non-invaded transects in Katonga Wildlife Reserve. Species are ranked according to their percentage contribution to the dissimilarity between sites and only those contributing >1% are shown. The values of average dissimilarity and the percentage of cumulative dissimilarity are also given

	Average Abundance				
Species	**invaded only**	**non-invaded only**	**Average Dissimilarity**	**% Contribution**	**Cumulative %**
Cymbopogon afronardus Stapf	0	30	16.2642	17.0434	17.0434
Setaria sphacelata (Schumach.) Stapf & C.E.Hubb. ex Moss	3.13	14.33	9.1	9.53	26.56
Cynodon dactylon (L.) Pers.	0	8.67	4.84	5.07	31.64
Panicum maximum Jacq.	0.93	6.6	4.40	4.61	36.26
Sporobolus pyramidalis P.Beauv.	0	7.73	3.90	4.09	40.35
Setaria verticillata (L.) P.Beauv.	3.33	5.67	3.76	3.94	44.3
Brachiaria decumbens Stapf	0	8	3.7	3.87	48.17
Setaria homonyma (Steud.) Chiov.	0	6	3.35	3.51	51.68
Tagetes minuta L.	0.67	6.33	3.10	3.25	54.94
Setaria chevalieri Stapf	6.13	0	2.62	2.74	57.67
Diplocyclos palmatus (L.) C.Jeffrey	0	5.33	2.33	2.44	60.12
Cyperus erectus (Schumach.) Mattf. & Kük.	0	5	2.32	2.43	62.55
Kyllinga alba Nees	0	2.67	2.16	2.26	64.82

4.2 Avifauna diversity

Ninety-two bird species belonging to forty-two and seventy-four genera bird families were recorded in the study area. of these, sixty species were recorded in the invaded transects and seventy-seven species in the non-invaded transects.

The species accumulation curve was approaching an asymptote, indicating that the rate of new species detection had diminished, and that most bird species in the study area had been sufficiently sampled (Figure 5). This suggests that the survey effort, which consisted of 30-point counts, was sufficient, and that enhancing the sampling intensity would probably produce only minimal increases in species richness.

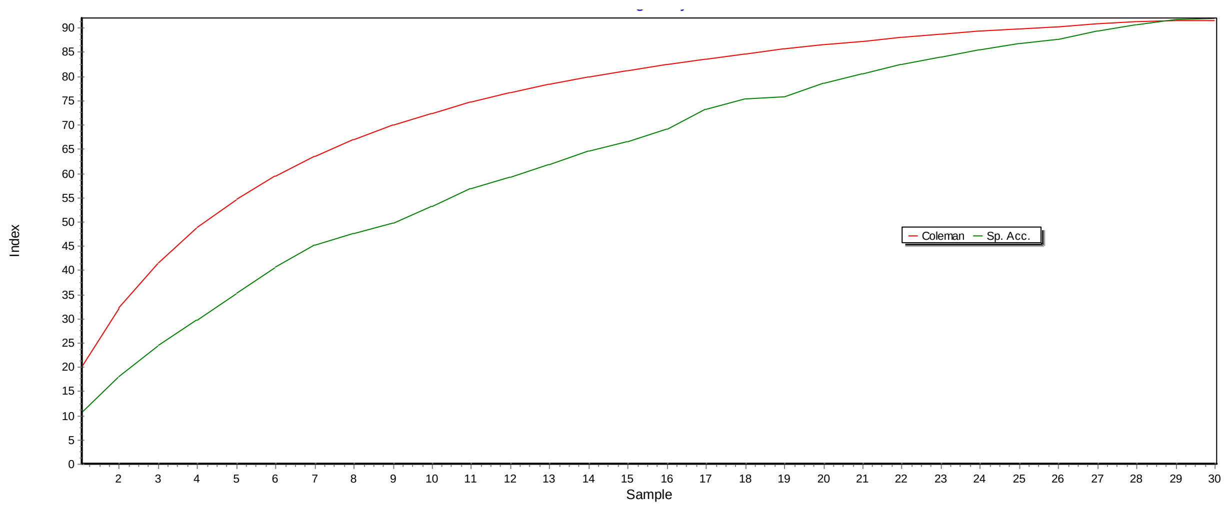

Figure 5: Coleman and species accumulation (Sp. Acc.) curves of the recorded avifauna species along six transects in Katonga Wildlife Reserve.

When analyzed by ecological categories, avifauna species diversity ranged from 0.00 to 2.84 in the invaded transacts and from 0.64 to 2.95 in the non-invaded transects. Statistically, avifauna species diversity by ecological categories did not exhibit any statistical significance between the invaded and non-invaded transects (Table. 3).

Table 3: Statistical comparisons for Avifauna species diversity between invaded and non-invaded sites in Katonga Wildlife Reserve.

Ecological categories	Mean Diversity Invaded	Diversity Mean Non-Invaded	Statistical test	Statistic	DF	P Value
Forest visitors (f)	2.599	2.53	t-test	0.26	3	0.81
Waterbird non-specialists (w)	0.561	0.617	Wilcoxon	6.00	-	0.65
Forest generalists (F)	1.566	1.794	t-test	-0.69	4	0.53
Grassland species (G)	0.785	1.416	t-test	-1.26	3	0.29
Waterbird specialists (W)	0.631	0.297	Wilcoxon	6.00	-	0.64

This pattern is visible in the Shannon diversity box plot (Figure 6), where the median for non-invaded transects lies marginally above that for invaded sites, although the interquartile ranges overlap considerably.

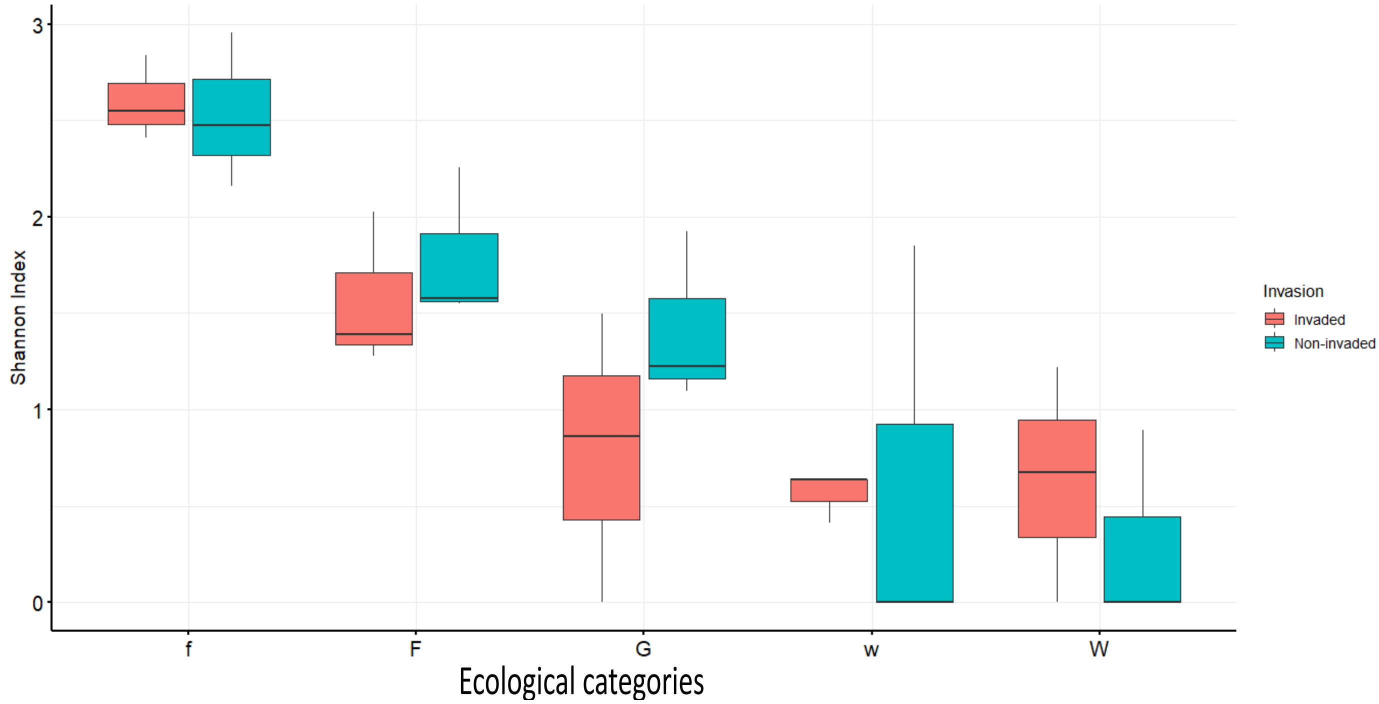

Figure 6: Avifauna Species Diversity based on ecological categories between invaded and non-invaded sites in Katonga Wildlife Reserve. F= forest generalist), f-species= forest visitors, W= waterbird specialists, w = waterbird non-specialists, G= grassland species

For the feeding guild group, Shannon diversity values ranged from 0.00 to 2.57 in the invaded transects and from 0.00 to 2.99 in the non-invaded sites. There was no statistical difference in the observed differences as seen in table 4.

Table 4: Statistical comparisons for avifauna species diversity between invaded and non-invaded sites based on feeding guilds in Katonga wildlife Reserve

Feeding guild	Mean Diversity Invaded	Diversity Mean Non-Invaded	Statistical test	Statistic	DF	P Value
Frugivore	0.879	1.394	Wilcoxon	0	NA	0.1
Insectivore	2.497	2.528	t-test	-0.11	2	0.9221
Granivore	0.88	1.231	t-test	-1.155	2	0.3467

Carnivore	0.853	0.332	Wilcoxon	7	NA	0.3758
Omnivore	0.69	1.161	t-test	-0.936	3	0.4285
Frugivore_Insectivore	0.784	0.911	Wilcoxon	3	NA	0.7
Nectarivore_Insectivore	0.212	0.368	Wilcoxon	3	NA	0.6428

The box plot (Figure 7) shows a slightly higher central tendency in non-invaded sites, with substantial overlap between groups.

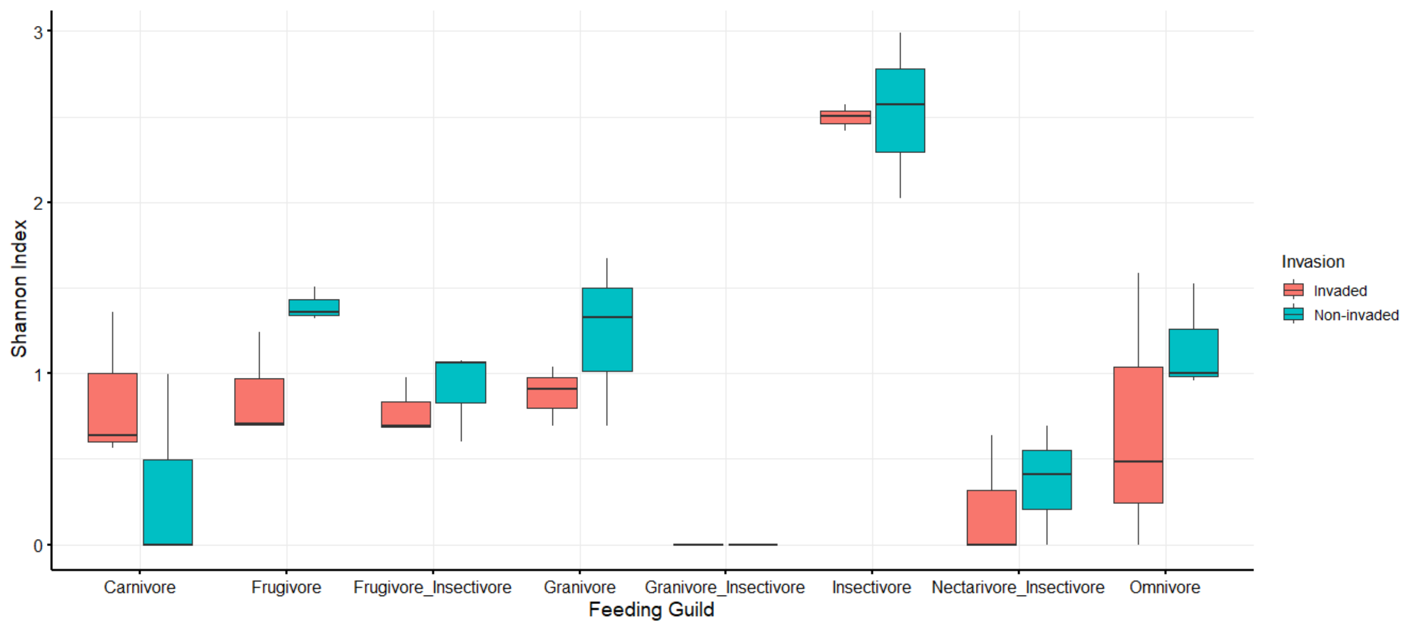

Figure 7: Avifauna Shannon Diversity based on Feeding Guild between invaded and non-invaded sites in Katonga Wildlife Reserve

Avifauna species composition between invaded and non-invaded sites differed significantly (Global R_{ANOSIM} = 0.110; p = 0.003) suggesting that species were more similar within the groups that would be expected by chance. Pairwise comparison did not show a significant difference (Global R_{ANOSIM} = 0.110; p = 0.019). Based on the analysis of SIMPER (Table 5 and Table 6), the species contributing up to 90% of the similarity within each ecosystem were 15 within the invaded sites and 24 in the non-invaded sites. Average similarity within the invaded transects was 22.17 and 17.31 for the non-invaded transects. Between the invaded and non-invaded transects, average dissimilarity was 82.83.

Table 5: Contribution of individual avifauna species to the overall similarity within the invaded and non-invaded transects in Katonga Wildlife Reserve. Species are arranged according to percentage contribution to the similarity within the ecosystem types and only those contributing >2% are shown. Average similarity and percentage of cumulative similarity are also given.

Species	Average Abundance	Average. Similarity	% Contribution	Cumulative %
Invaded transects				
Cuculus solitarius Stephens, JF	0.67	4.6	20.86	20.86
Camaroptera brachyura Vieillot, LJP	0.6	3.68	16.59	37.45
Pycnonotus barbatus Desfontaines	0.4	1.49	6.71	44.16
Cisticola woosnami Ogilvie-Grant, WR	0.4	1.47	6.60	50.76
Laniarius major Hartlaub, KJG	0.4	1.46	6.57	57.34
Colius striatus Gmelin, JF	0.4	1.41	6.37	63.71
Halcyon senegalensis Linnaeus, C	0.33	0.96	4.34	68.05
Chrysococcyx klaas Stephens, JF	0.33	0.94	4.24	72.29
Streptopelia capicola Sundevall, CJ	0.33	0.94	4.24	76.53
Psalidoprocne albiceps Sclater, PL	0.27	0.69	3.13	79.66
Hedydipna collaris Vieillot, LJP	0.27	0.58	2.62	82.27
Nicator chloris Valenciennes, A	0.27	0.55	2.49	84.77
Lamprotornis purpuroptera Rüppell, WPES	0.27	0.54	2.46	87.22
Non-invaded transects				
Camaroptera brachyura Vieillot, LJP	0.67	3.94	22.77	22.77
Colius striatus Gmelin, JF	0.53	2.16	12.48	35.26
Pycnonotus barbatus Desfontaines	0.4	1.29	7.45	42.70

Euplectes axillaris Smith, A	0.4	1.11	6.39	49.09
Corythaeola cristata Vieillot, LJP	0.27	0.68	3.94	53.03
Cisticola woosnami Ogilvie-Grant, WR	0.33	0.65	3.78	56.81
Nicator chloris Valenciennes, A	0.27	0.55	3.18	59.99
Turdus pelios Bonaparte, CLJL	0.27	0.54	3.13	63.12
Crinifer zonurus Rüppell, WPES	0.27	0.51	2.97	66.09
Pogoniulus bilineatus Sundevall, CJ	0.27	0.51	2.94	69.03
Spermestes cucullata Swainson, WJ	0.27	0.42	2.44	71.48
Cuculus solitarius Stephens, JF	0.27	0.36	2.05	73.53
Halcyon senegalensis Linnaeus, C	0.27	0.35	2.02	75.55

Table 6: Results of SIMPER analysis highlighting the avifauna species contributing most to the dissimilarity between invaded and non-invaded transects in Katonga Wildlife Reserve. Species are ranked according to their percentage contribution to the dissimilarity between sites and only those contributing >1% are shown. The values of average dissimilarity and the percentage of cumulative dissimilarity are also given

	Average Abundance				
Species	**Invaded**	**Non-invaded**	**Average Dissimilarity**	**% Contribution**	**Cumulative %**
Cuculus solitarius Stephens, JF	0.67	0.27	2.98	3.59	3.59
Colius striatus Gmelin, JF	0.4	0.53	2.48	2.99	6.59
Pycnonotus barbatus Desfontaines	0.4	0.4	2.39	2.89	9.47
Camaroptera brachyura Vieillot, LJP	0.6	0.67	2.32	2.80	12.27
Cisticola woosnami Ogilvie-Grant, WR	0.4	0.33	2.25	2.72	14.99
Laniarius major Hartlaub, KJG	0.4	0.2	2.10	2.54	17.53
Halcyon senegalensis Linnaeus, C	0.337	0.27	1.97	2.38	19.91
Nicator chloris Valenciennes, A	0.27	0.27	1.94	2.35	22.25
Euplectes axillaris Smith, A	0.07	0.4	1.89	2.29	24.54
Streptopelia capicola Sundevall, CJ	0.33	0.13	1.76	2.12	26.66
Chrysococcyx klaas Stephens, JF	0.33	0.07	1.72	2.08	28.74
Lamprotornis purpuroptera Rüppell, WPES	0.27	0.2	1.72	2.08	30.83
Corythaeola cristata Vieillot, LJP	0.07	0.27	1.66	2.00	32.82

4.3 Medium to large sized mammal diversity

Fourteen mammal species from seven mammal families and fourteen genera were recorded in the study area. twelve mammal species were recorded in the invaded transects and thirteen mammal species were recorded in the non-invaded transects. The species cumulative curve (Figure 8) flattened indicating that majority of the species in the area had been recorded and that the sampling efforts were sufficient.

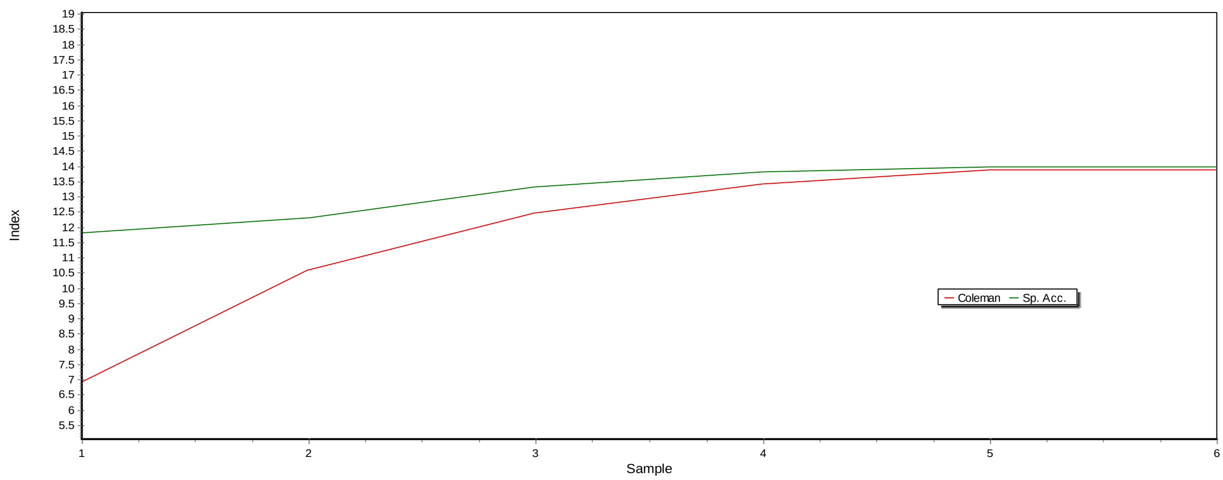

Figure 8: Coleman and species accumulation (Sp. Acc.) curves of the recorded mammal species along six transects in Katonga Wildlife Reserve

Shannon diversity ranged from 0.00 to 0.11 in the invaded transects and from 0.00 to 1.23 in the non-invaded transects. (Figure 9).

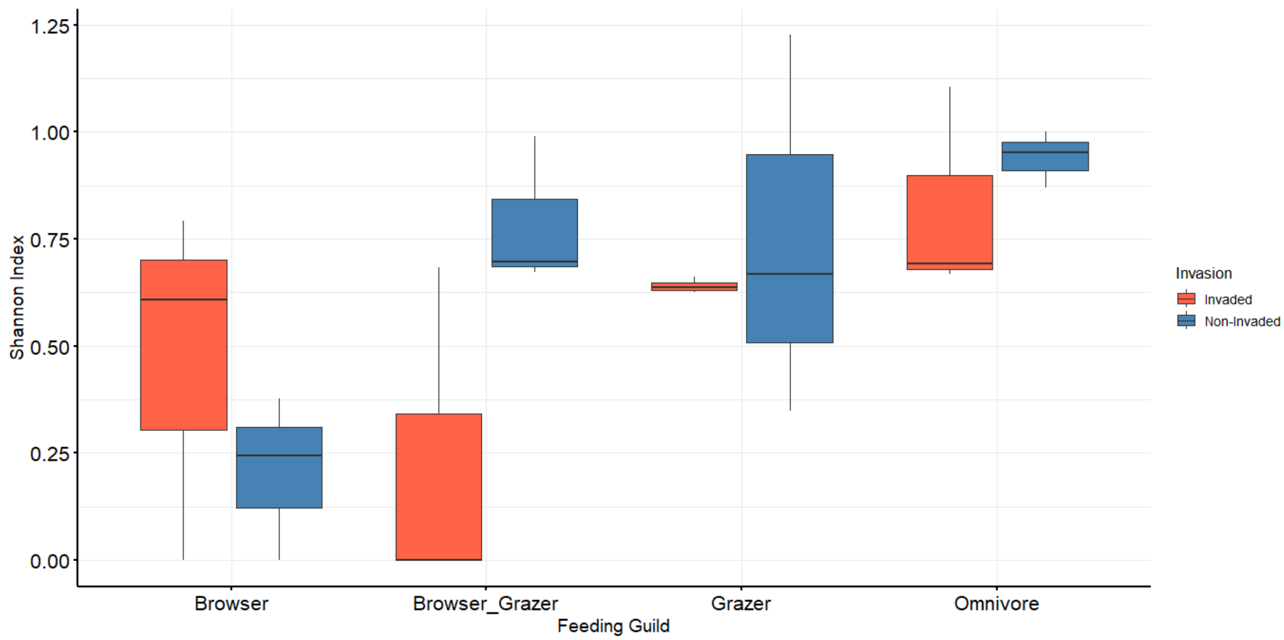

Figure 9: Mammal Shannon Diversity based on Feeding guilds between invaded and non-invaded sites in Katonga Wildlife Reserve

Shannon diversity for the mammals was not statistically significance in all feeding guilds (Table 7)

Table 7: Statistical comparisons for mammal diversity based on feeding guilds in Katonga Wildlife Reserve

Feeding guild	Mean Diversity Invaded	Mean Diversity Non_Invaded	Method	Statistic	DF	P value
Browser	0.467	0.207	t-test	0.99	3	0.4017
Browser_Grazer	0.228	0.786	Wilcoxon	1.00	NA	0.184
Grazer	0.641	0.748	t-test	-0.42	2	0.7183
Omnivore	0.821	0.941	t-test	-0.81	2	0.4927

Analysis of similarity showed a significant difference (Global R_{ANOSIM} = 0.407; p = 0.001) in mammal species composition suggesting that species were more similar within the groups that would be expected by chance. Based on the analysis of SIMPER (Table 8 and Table 9), the species contributing up to 90% of the similarity within each ecosystem were 6 within the invaded sites and 8 in the non-invaded sites. Average similarity within the invaded transects was 72.60 and 83.89 for the non-invaded transects. Between the invaded and non-invaded transects, average dissimilarity was 40.50.

Table 8: Contribution of individual mammal species to the overall similarity within the invaded and non-invaded transects in Katonga Wildlife Reserve. Species are arranged according to percentage contribution to the similarity within the ecosystem types and only those contributing >2% are shown. Average similarity and percentage of cumulative similarity are also given.

Species	Average Abundance	Average. Similarity	% Contribution	Cumulative %
Invaded transects				
Phacochoerus africanus Gmelin	1	12.75	17.57	17.57
Hippopotamus amphibius Linnaeus	1	12.75	17.57	35.13
Chlorocebus pygerythrus F. Cuvier	1	12.75	17.57	52.70
Colobus guereza Rüppell	1	12.75	17.57	70.27
Loxodonta africana Blumenbach	1	12.75	17.57	87.83
Damaliscus lunatus Burchell	0.67	5.13	7.06	94.90
Non-invaded transects				
Kobus ellipsiprymnus Ogilby	1	9.44	11.26	11.26
Hippopotamus amphibius Linnaeus	1	9.44	11.26	22.52
Tragelaphus scriptus Pallas	1	9.44	11.26	33.77
Colobus guereza Rüppell	1	9.44	11.26	45.03
Phacochoerus africanus Gmelin	1	9.44	11.26	56.29
Loxodonta africana Blumenbach	1	9.44	11.26	67.55
Chlorocebus pygerythrus F. Cuvier	1	9.44	11.26	78.80
Xerus erythropus É.Geoffroy Saint-Hilaire	1	9.44	11.26	90.07

Table 9: Results of SIMPER analysis highlighting the mammal species contributing most to the dissimilarity between invaded and non-invaded transects in Katonga Wildlife Reserve. Species are ranked according to their percentage contribution to the dissimilarity between sites and only those contributing >1% are shown. The values of average dissimilarity and the percentage of cumulative dissimilarity are also given

	Average Abundance				
Species	**Invaded**	**Non-invaded**	**Average Dissimilarity**	**% Contribution**	**Cumulative %**
Kobus ellipsiprymnus Ogilby	0.33	1	3.94	12.91	12.91
Tragelaphus scriptus Pallas	0.33	1	3.94	12.92	25.83
Xerus erythropus É.Geoffroy Saint-Hilaire	0.33	1	3.94	12.91	38.74
Piliocolobus tephrosceles Elliot	0.67	0	3.46	11.35	50.09
Aepyceros melampus Lichtenstein, 1812	0	0.67	3.37	11.05	61.15
Equus quagga Boddaert	0	0.67	3.37	11.05	72.20
Damaliscus lunatus Burchell	0.67	0.33	3.22	10.56	82.76
Philantomba monticola Thunberg	0.33	0.67	2.99	9.80	92.57

CHAPTER FIVE

5 DISCUSSION

5.1 Plant species diversity

The 103 plant species recorded in this study represent 60.45% of the 155 species previously documented in Katonga Wildlife Reserve (Zwick et al., 1997). This difference is likely due to the limited spatial coverage of the current survey, which examined only six transects instead of the entire reserve. However, the species accumulation curve (Figure 3) was nearly leveling off indicating that addition transects would yield a few new species (Gotelli & Colwell, 2001). The detection of species not reported in previous studies represents a change in species compsition.

The study found that non-invaded sites exhibited higher diversity than invaded sites, this aligns with findings from temperate and tropical systems where invasive plants have been shown to reduce native plant diversity through competitive exclusion and habitat alteration (Hejda et al., 2009a; Pyšek et al., 2012). The highly significant differences observed (p = 0.0002) indicate that invasion impacts plant community diversity.

The significant difference in community composition between the invaded and non-invaded sites (Global R_{ANOSIM}, R = 0.3769, p = 0.001) suggests that invasive species are not just changing the species present but also creating a more uniform community within each type of habitat (Invaded and non-invaded habitats). This aligns with the idea of biotic homogenization driven by invasions, where the invaded areas end up dominated by species that can thrive in disturbed conditions, often at the expense of native diversity (Olden & Rooney, 2006).

The high dissimilarity (95.43%) between the invaded and non-invaded transects indicates that invasion has substantially altered plant community structure in Katonga Wildlife Reserve. Non-invaded areas were dominated by native grassland species such as *Cymbopogon afronardus*, *Setaria sphacelata*, and *Cynodon dactylon*, which are typical of open savanna habitats and contribute to ecosystem stability and forage productivity (Edroma, 1981; Wronski, 2003). In contrast, invaded sites were characterized by shade-tolerant species such as *Setaria verticillata*, *Vepris nobilis*, and *Cissus quadrangularis*, suggesting that invasion has facilitated a shift toward communities dominated by opportunistic species adapted to altered microhabitats (Hejda et al., 2009b; Pyšek et al., 2012). This implies that *Lantana camara* invasion leads to the replacement of characteristic native grasses with tolerant and secondary species, thereby increasing vegetation uniformity and potentially altering habitat quality for associated fauna. Similar findings have been reported elsewhere, where *Lantana camara* invasion caused a decline in native grass cover and promoted the establishment of generalist or secondary species (Bhagwat et al., 2012; Gooden et al., 2009; Hejda et al., 2009a). In return, this may lead to potential cascading effects on grazing resources and ecosystem processes (Hughes & Denslow, 2005).

5.2 Avifauna diversity

Avifauna species diversity, by ecological categories, showed marginal differences between non-invaded and invaded sites. While non-invaded areas exhibited slightly higher mean Shannon diversity (1.33 vs. 1.23), there was a considerable overlap in interquartile ranges which suggests that these differences are minimal. A similar pattern was observed in the feeding guilds where by non-invaded sites again show slightly higher mean Shannon diversity (0.99 vs 0.85) which indicates that the invasion of *Lantana camara* has not significantly altered local diversity patterns at the site level. These findings resonate with findings from South India's Male Mahadeshwara Hills, where bird species diversity, richness, and abundance declined at higher Lantana densities; however, evenness increased (Aravind et al., 2010).

SIMPER analysis revealed that dissimilarity in bird communities between invaded and non-invaded sites was not driven by a single dominant species but rather by a suite of moderately abundant taxa including *Cuculus solitarius*, *Colius striatus*, *Pycnonotus barbatus*, and *Camaroptera brachyura*. Within invaded sites, *Cuculus solitarius* and *Camaroptera brachyura* together contributed over 37% of the similarity, whereas non-invaded sites exhibited more evenness, with *Camaroptera brachyura* (23%) and *Colius striatus* (12%) being the major contributors. ANOSIM supported these patterns, showing significant but modest differences between invaded and non-invaded assemblages ($R = 0.11$, $p = 0.003$), suggesting that although bird communities overlap considerably, invasion alters their relative composition. This agrees with previous studies where ecological invasions alter community structure and affects specialist species (Bitani et al., 2022; Linders et al., 2019; Paudel et al., 2024).

5.3 Medium to large size mammal species diversity

The absence of statistically significant differences in mammal species diversity between invaded and non-invaded transects suggests that invasion has not (yet) produced a large, consistent effect on local mammal diversity or dominance patterns at the spatial scale of this study. This could be explained by (i) Resilience or behavioral plasticity of mammals. Many medium–large mammals can persist in altered vegetation if sufficient forage or cover remains or if they shift space-use patterns such as change use of microhabitats or time of use (Gaynor et al., 2018; Terborgh et al., 2001). This can maintain similar local diversity even where plant communities change. A study in the continental United States found out that 58% of the studied mammal had responded positively to human disturbances (Suraci et al., 2021). This suggests that there is a possibility for persistence even in cases of invasion, as seen in this study. Another study in the Amazon found out that after a disturbance, some mammal species became more in small patches, even as others disappeared (Michalski & Peres, 2007), this was observed as grazers were highly abundant in non-invaded sites as compared to invaded areas with some species like zebras being recorded only in non-invaded sites. (ii) Subtle compositional shifts rather than richness loss. The SIMPER results (several species each contributing small amounts to overall differences) and the flat species accumulation curve indicate that invasion is correlated with small abundance shifts among species rather than a loss of species from the system altogether.

ANOSIM revealed significant differences in mammal communities between invaded and non-invaded areas (Global $R_{ANOSIM} = 0.407$, $p = 0.001$). SIMPER analysis indicated that these differences were largely driven by changes in herbivore assemblages. Non-invaded sites were characterized by higher abundances of waterbuck, bushbuck, zebra, and impala, while invaded sites supported generalists such as warthogs, elephants, and vervet monkeys. Within invaded sites, similarity was dominated by a few large-bodied species, suggesting homogenization, whereas non-invaded sites exhibited more even contributions across multiple taxa. This indicates that invasion reduces mammal community heterogeneity and favors disturbance-tolerant generalists over specialized grazers and browsers (Aravind et al., 2010; G. Wilson et al., 2013).

5.4 Study Limitations

This study could have been limited by several factors which should be considered when interpreting the findings. For example, the relatively small number of transects, as reflected in species accumulation curves that did not fully level off, may have reduced statistical power to detect subtle effects, particularly for rare or low-abundance species (Christie et al., 2019). Fauna surveys were conducted solely during daytime using line transects, limiting detectability of nocturnal mammals (Bowler et al., 2017) and potentially biasing diversity estimates between invaded and non-invaded areas. Furthermore, the single sampling period did not account for seasonal variations in biodiversity patterns (Ojha et al., 2025), such as rainfall-driven changes in plant diversity, bird movements, or mammal distributions.

CHAPTER SIX

6 CONCLUSION AND RECOMMENDATIONS

6.1 CONCLUSION

- The results showed that *Lantana camara* significantly reduces native plant diversity as evidenced by a significantly higher diversity in the non-invaded sites than in the invaded sites. Community analyses indicated distinct differences in plant community composition between invaded and non-invaded areas, highlighting the invasive species' impact on ecological processes.
- Although overall bird diversity and richness showed only slight declines in invaded areas, results indicated that some groups were more affected than the others, suggesting a potential decline in the habitat quality due to *Lantana camara*.
- For medium-to-large mammals, moderate but nonsignificant diversity differences were noted between invaded and non-invaded areas. Some species, like *Kobus ellipsiprymnus* gilby, *Tragelaphus scriptus* Pallas, *Xerus erythropus* É.Geoffroy Saint-Hilaire *Aepyceros melampus* Lichtenstein, and *Equus quagga* Boddaert were more prevalent in non-invaded habitats, while others, like *Piliocolobus tephrosceles* Elliot and *Philantomba monticola* Thunberg were more common in invaded areas. This indicates that *Lantana camara* may indirectly affect mammal communities by altering vegetation structure and forage quality.

6.2 RECOMMENDATIONS

The study recommends the following:

(a) The Uganda Wildlife Authority (UWA) should prioritize the control of *Lantana camara* in invaded areas within Katonga Wildlife Reserve (KWR), alongside the establishment of long-term monitoring plots to track vegetation recovery and evaluate the effectiveness of control interventions.

(b) Protected area managers should integrate systematic bird community monitoring into invasive species management programs and assess habitat features most affected by invasion in order to support vulnerable avian guilds

(c) A mixed monitoring approach incorporating vegetation and mammal surveys should be maintained to detect long-term shifts in mammal community composition and to investigate changes in forage quality and availability in *L. camara*-invaded areas.

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8 APPENDICES

Appendix 1: A list of plant species and families recorded in *Lantana camara* invaded and non-invaded sites of Katonga Wildlife Reserve

Plant species name	**Plant Family name**
Albizia coriaria Welw. ex Oliv.	Fabaceae
Albizia zygia (DC.) J.F.Macbr.	Fabaceae
Acacia sieberiana Tausch	Fabaceae
Acacia seyal Delile	Fabaceae
Allophylus africanus P.Beauv.	Sapindaceae
Euphorbia candelabrum Welw.	Euphorbiaceae
Turraea robusta Gürke	Meliaceae
Vepris nobilis (Delile) Mziray	Rutaceae
Vernonia amygdalina Delile	Asteraceae
Combretum collinum Fresen.	Combretaceae
Combretum molle R.Br. ex G.Don	Combretaceae
Rhus longipes Engl.	Anacardiaceae
Rhus natalensis Bernh.	Anacardiaceae
Grewia similis K.Schum	Malvaceae
Grewia mollis Juss.	Malvaceae
Croton macrostachyus Hochst. ex Delile	Euphorbiaceae
Erythrina abyssinica Lam	Fabaceae
Sapium ellipticum (Hochst.) Pax	Euphorbiaceae
Bersama abyssinica Fresen	Francoaceae
Senna spectabilis (DC.) H.S.Irwin & Barneby	Fabaceae
Acacia hockii De Wild.	Fabaceae
Solanum incanum L.	Solanaceae
Solanum indicum L.	Solanaceae
Solanum aculeastrum Dunal	Solanaceae
Solanum nigrum L.	Solanaceae
Abutilon mauritianum (Jacq.) Medik.	Malvaceae
Tephrosia nana Kotschy ex Schweinf.	Fabaceae
Tephrosia linearis (Willd.) Pers.	Fabaceae
Hoslundia opposita Vahl	Lamiaceae
Harrisonia abyssinica Oliv.	Rutaceae
Tagetes minuta L.	Asteraceae
Triumfetta macrophylla K.Schum.	Malvaceae
Triumfetta rhomboidea Jacq.	Malvaceae
Physalis peruviana L.	Solanaceae

Phyllanthus ovalifolius Forssk.	Phyllanthaceae
Lantana camara L.	Verbenaceae
Vernonia brachycalyx O.Hoffm.	Asteraceae
Leonotis nepetifolia (L.) R.Br.	Lamiaceae
Indigofera congesta Welw. ex Baker	Fabaceae
Desmodium velutinum (Willd.) DC.	Fabaceae
Carissa spinarum L.	Apocynaceae
Sida cordifolia Forssk.	Malvaceae
Ocimum gratissimum L.	Lamiaceae
Urena lobata L.	Malvaceae
Cissus quadrangularis L.	Vitaceae
Asparagus africanus Lam.	Asparagaceae
Acalypha villicaulis Hochst. ex A.Rich.	Euphorbiaceae
Capparis tomentosa Lam.	Capparaceae
Asystasia gangetica (L.) T.Anderson	Acanthaceae
Clerodendrum rotundifolium Oliv.	Lamiaceae
Chromolaena odorata (L.) R.M.King & H.Rob.	Asteraceae
Cissus rotundifolia Blume	Vitaceae
Pseudarthria hookeri Wight & Arn.	Fabaceae
Bidens pilosa L.	Asteraceae
Cynodon dactylon (L.) Pers.	Poaceae
Setaria chevalieri Stapf	Poaceae
Setaria sphacelata (Schumach.) Stapf & C.E.Hubb. ex Moss	Poaceae
Setaria verticillata (L.) P.Beauv.	Poaceae
Setaria homonyma (Steud.) Chiov.	Poaceae
Panicum maximum Jacq.	Poaceae
Panicum repens L.	Poaceae
Sporobolus africanus (Poir.) Robyns & Tournay	Poaceae
Sporobolus pyramidalis P.Beauv.	Poaceae
Ageratum conyzoides L.	Asteraceae
Achyranthes aspera L.	Amaranthaceae
Digitaria velutina (Forssk.) P.Beauv.	Poaceae
Aspilia kotschyi (Sch.Bip.ex Hochst.) Olove.	Asteraceae
Brachiaria decumbens Stapf	Poaceae
Secamone africana (Oliv.) Bullock	Apocynaceae
Vernonia cinerea (L.) Less.	Asteraceae
Microglossa densiflora Hook.f.	Asteraceae
Abrus canescens Welw. ex Baker	Fabaceae
Un known genus	*Un known genus*
Diplocyclos palmatus (L.) C.Jeffrey	Cucurbitaceae
Eragrostis exasperata Peter	Poaceae

Dyschoriste clinopodioides Mildbr.	Acanthaceae
Abildgaardia hispidula (Vahl) Lye.	Cyperaceae
Vernoniastrum aemulans (Vatke) H.Rob.	Asteraceae
Kyllinga alba Nees	Cyperaceae
Jasminum dichotomum Vahl	Oleaceae
Indigofera erecta Eckl. & Zeyh.	Fabaceae
Thonningia sanguinea Vahl	Balanophoraceae
Maytenus buchananii (Loes.) R.Wilczek	Celastraceae
Rubia cordifolia L.	Rubiaceae
Hewittia sublobata (L.f.) Kuntze	Convolvulaceae
Vernonia campanea S.Moore	Asteraceae
Rhynchosia goetzei Harms	Fabaceae
Momordica foetida Schumach.	Cucurbitaceae
Commelina benghalensis L.	Commelinaceae
Cissampelos mucronata A.Rich.	Menispermaceae
Desmodium adscendens (Sw.) DC.	Fabaceae
Brachiaria brizantha (A.Rich.) Stapf	Poaceae
Brachiaria platynota (K.Schum.) Robyns	Poaceae
Cyperus erectus (Schumach.) Mattf. & Kük.	Cyperaceae
Berkheya spekeana Oliv.	Asteraceae
Cymbopogon afronardus Stapf	Poaceae
Crassocephalum vitellinum (Benth.) S.Moore	Asteraceae
Hyparrhenia filipendula (Hochst.) Stapf	Poaceae
Digitaria abyssinica (Hochst. ex A.Rich.) Stapf	Poaceae
Ipomoea cairica (L.) Sweet	Convolvulaceae
Urtica dioica L.	Urticaceae
Erigeron sumatrensis Retz.	Asteraceae
Aspilia africana (Pers.) C.D.Adams	Asteraceae

Appendix 2: Plant species diversity across quadrats in *Lantana camara* invaded and non-invaded sites of Katonga Wildlife Reserve

Transect/ Quadrat	Shannon	Site
T1Q1	2.079442	Invaded
T1Q2	2.484907	Invaded
T1Q3	1.609438	Invaded
T1Q4	1.609438	Invaded
T1Q5	1.94591	Invaded
T2Q1	2.302585	Invaded
T2Q2	1.94591	Invaded
T2Q3	2.079442	Invaded
T2Q4	1.94591	Invaded
T2Q5	2.197225	Invaded
T3Q1	2.564949	Invaded
T3Q2	2.639057	Invaded
T3Q3	2.079442	Invaded
T3Q4	2.079442	Invaded
T3Q5	2.079442	Invaded
T4Q1	2.397895	Non-Invaded
T4Q2	2.302585	Non-Invaded
T4Q3	2.639057	Non-Invaded
T4Q4	2.564949	Non-Invaded
T4Q5	2.079442	Non-Invaded
T5Q1	2.772589	Non-Invaded
T5Q2	3.044522	Non-Invaded
T5Q3	2.302585	Non-Invaded
T5Q4	2.564949	Non-Invaded
T5Q5	2.639057	Non-Invaded
T6Q1	2.458311	Non-Invaded
T6Q2	2.639057	Non-Invaded
T6Q3	2.564949	Non-Invaded
T6Q4	2.397895	Non-Invaded
T6Q5	2.397895	Non-Invaded

Appendix 3: A list of Avifauna species and their families recorded in *Lantana camara* invaded and non-invaded sites of Katonga Wildlife Reserve

Number in the bird atlas	Species name	Common name	Avifauna family	Ecological categories	Feeding guild
562	*Pycnonotus barbatus* Desfontaines	Common bulbul	Pycnonotidae	f	Frugivore
	Laniarius erythrogaster Cretzschmar	Black-headed gonolek	Malaconotidae	f	Insectivore
841	*Laniarius major* Hartlaub, KJG	Tropical boubou	Malaconotidae	f	Insectivore
677	*Camaroptera brachyura* Vieillot, LJP	Green-backed camaroptera	Cisticolidae	f	Insectivore
283	*Streptopelia semitorquata* Rüppell, WPES	Red-eyed dove	Columbidae	f	Granivore
	Streptopelia capicola Sundevall, CJ	Ring-necked dove	Columbidae	f	Granivore
39	*Bostrychia hagedash* Latham, J	Hadada Ibis	Threskiornithidae	w	Carnivore
307	*Clamator levaillantii* Swainson, WJ	Levaillant's cuckoo	Cuculidae	f	Insectivore
	Cuculus solitarius Stephens, JF	Red-chested cuckoo	Cuculidae	f	Insectivore
	Turtur tympanistria Temminck, CJ	Tambourine dove	Columbidae	f	Granivore
302	*Tauraco rossae* Gould, J	Ross's turaco	Musophagidae	F	Omnivore
872	*Lamprotornis purpuroptera* Rüppell, WPES	Ruppell's starling	Sturnidae	f	Frugivore_Insectivore
387	*Merops oreobates* Sharpe, RB	Cinnamon-chested bee-eater	Meropidae	G	Insectivore
513	*Hirundo rustica* Linnaeus, C	Barn swallow	Hirundinidae	w	Insectivore
420	*Lophoceros nasutus* Linnaeus, C	African grey hornbill	Bucerotidae	F	Insectivore
369	*Colius striatus* Gmelin, JF	Speckled mouse bird	Coliidae	f	Frugivore
422	*Bycanistes subcylindricus* Sclater, PL	Black-and-white-casqued hornbill	Bucerotidae	F	Frugivore
325	*Centropus senegalensis* Linnaeus, C	Senegal coucal	Cuculidae	f	Insectivore

980	*Spermestes cucullata* Swainson, WJ	Bronze mannikin	Estrildidae	f	Granivore
801	*Cinnyris pulchellus* Linnaeus, C	Beautiful sunbird	Nectariniidae	f	Nectarivore_ Insectivore
292	*Poicephalus meyeri* Cretzschmar, PJ	Meyer's parrot	Psittacidae	f	Omnivore
601	*Myrmecocichla nigra* Vieillot, LJP	Sooty chat	Muscicapidae	G	Insectivore
876	*Cinnyricinclus leucogaster* Boddaert, P	Violet-backed starling	Sturnidae	f	Omnivore
871	*Lamprotornis splendidus* Vieillot, LJP	Splendid starling	Sturnidae	F	Frugivore_In sectivore
73	*Elanus caeruleus* Desfontaines, RL	Black winged kite	Accipitridae	G	Carnivore
881	*Passer griseus* Vieillot, LJP	Northern grey-headed sparrow	Passeridae	f	Granivore
296	*Corythaeola cristata* Vieillot, LJP	Great blue turaco	Musophagidae	F	Frugivore
794	*Hedydipna collaris* Vieillot, LJP	Collared sunbird	Nectariniidae	F	Nectarivore_ Insectivore
850	*Oriolus larvatus* Lichtenstein, MHC	African Black-headed Oriole	Oriolidae	f	Omnivore
28	*Scopus umbretta* Gmelin, JF	Hamerkop	Scopidae	w	Omnivore
401	*Eurystomus glaucurus* Müller, PLS	Broad-billed roller	Coraciidae	f	Insectivore
142	*Numida meleagris* Linnaeus, C	Helmeted guineafowl	Numididae	G	Omnivore
907	*Ploceus nigerrimus* Vieillot, LJP	Vieillot's black weaver	Ploceidae	f	Granivore_I nsectivore
908	*Ploceus cucullatus* Müller, PLS	Village weaver	Ploceidae	f	Omnivore
	Platysteira cyanea Müller, PLS	Brown-throated wattle-eye	Muscicapidae	f	Granivore_I nsectivore
319	*Chrysococcyx klaas* Stephens, JF	Klaas's cuckoo	Cuculidae	f	Insectivore
375	*Halcyon senegalensis* Linnaeus, C	Woodland kingfisher	Alcedinidae	f	Insectivore
376	*Halcyon chelicuti* Stanley, ES	Striped kingfisher	Alcedinidae	f	Insectivore
26	*Ardea melanocephala* Children, JG; Vigors, NA	Black-headed heron	Ardeidae	w	Omnivore

323	*Centropus superciliosus* Hemprich, WF; Ehrenberg, CG	White-browed Coucal	Cuculidae	f	Insectivore
419	*Lophoceros alboterminatus* Büttikofer, J	Crowned hornbill	Bucerotidae	f	Omnivore
529	*Macronyx croceus* Vieillot, LJP	Yellow-throated longclaw	Motacillidae	G	Insectivore
638	*Cisticola erythrops* Hartlaub, KJG	Red-faced cisticola	Cisticolidae	w	Insectivore
803	*Cinnyris erythrocercus* Hartlaub, KJG	Red-chested sunbird	Nectariniidae	w	Nectarivore_ Insectivore
221	*Vanellus senegallus* Linnaeus, C	African wattled lapwing	Charadriidae	W	Carnivore
399	*Coracias caudatus* Linnaeus, C	Lilac-breasted roller	Coraciidae	f	Insectivore
30	*Anastomus lamelligerus* Temminck, CJ	African openbill	Ciconiidae	w	Carnivore
70	*Aviceda cuculoides* Swainson, WJ	African-cuckoo hawk	Cuculidae	F	Insectivore
268	*Treron calvus* Temminck, CJ	African green pigeon	Columbidae	F	Frugivore
271	*Turtur afer* Linnaeus, C	Blue-spotted wood dove	Columbidae	F	Granivore
109	*Kaupifalco monogrammicus* Temminck, CJ	Lizard buzzard	Accipitridae	F	Insectivore
185	*Balearica regulorum* Bennett, ET	Grey-crowned crane	Gruidae	W	Omnivore
305	*Crinifer zonurus* Rüppell, WPES	Eastern plantain-eater	Musophagidae	f	Frugivore
443	*Pogonornis bidentatus* Shaw, G	Double toothed barbet	Lybiidae	f	Frugivore
520	*Motacilla aguimp* Temminck, CJ	African pied wagtail	Motacillidae	w	Insectivore
563	*Nicator chloris* Valenciennes, A	Western nicator	Nicatoridae	F	Insectivore
426	*Pogoniulus scolopaceus* Bonaparte, CLJL	Speckled tinkerbird	Lybiidae	F	Frugivore_In sectivore
431	*Pogoniulus bilineatus* Sundevall, CJ	Yellow-rumped tinkerbird	Lybiidae	F	Frugivore_In sectivore
612	*Turdus pelios* Bonaparte, CLJL	African thrush	Turdidae	f	Omnivore
	Chrysococcyx caprius Boddaert, P	Diederik cuckoo	Cuculidae	G	Insectivore
641	*Cisticola woosnami* Ogilvie-Grant, WR	Trilling cisticola	Cisticolidae	G	Insectivore
645	*Cisticola chiniana* Smith, A	Rattling cisticola	Cisticolidae	F	Insectivore
90	*Polyboroides typus* Smith, A	African harrier-hawk	Accipitridae	f	Carnivore
787	*Chalcomitra senegalensis* Linnaeus, C	Scarlet-chested sunbird	Nectariniidae	f	Nectarivore_

					Insectivore
739	*Terpsiphone viridis* Müller, PLS	African paradise flycatcher	Monarchidae	f	Insectivore
578	*Cossypha niveicapilla* de Lafresnaye, NFAA	Snowy-crowned robin-chat	Muscicapidae	F	Insectivore
522	*Anthus cinnamomeus* Rüppell, WPES	African pipit	Motacillidae	G	Insectivore
	Ceuthmochares aereus Vieillot, LJP	Blue malkoha	Cuculidae	F	Omnivore
404	*Phoeniculus purpureus* Miller, JF	Green Woodhoopoe	Phoeniculidae	F	Insectivore
713	*Melaenornis edolioides* Swainson, WJ	Northern Black Flycatcher	Muscicapidae	f	Insectivore
	Ortygornis sephaena Smith, A	Crested francolin	Phasianidae	G	Granivore
815	*Lanius excubitoroides* Prévost, F; des Murs, MAPO	Grey-backed fiscal	Laniidae	f	Insectivore
932	*Euplectes axillaris* Smith, A	Fan-tailed widowbird	Ploceidae	w	Granivore
985	*Vidua macroura* Pallas, PS	Pin-tailed whydah	Viduidae	G	Granivore
191	*Lissotis melanogaster* Rüppell, WPES	Black-bellied bustard	Otididae	G	Omnivore
408	*Upupa epops* Linnaeus, C	Eurasian hoopoe	Upupidae	G	Omnivore
498	*Psalidoprocne albiceps* Sclater, PL	White-headed saw-wing	Hirundinidae	f	Insectivore
966	*Estrilda paludicola* Heuglin, MT	Fawn-breasted waxbill	Estrildidae	G	Granivore
639	*Cisticola cantans* Heuglin, MT	Singing cisticola	Cisticolidae	f	Insectivore
445	*Trachylaemus purpuratus* Verreaux, JP; Verreaux, JBÉ	Eastern yellow billed barbet	Lybiidae	F	Frugivore_Insectivore
897	*Ploceus ocularis* Smith, A	Spectacled weaver	Ploceidae	f	Insectivore
650	*Cisticola natalensis* Smith, A	Croaking cisticola	Cisticolidae	G	Insectivore
380	*Corythornis cristatus* Pallas, PS	Malachite Kingfisher	Alcedinidae	W	Carnivore
435	*Tricholaema hirsut*a Swainson, WJ	Hairy-breasted Barbet	Lybiidae	F	Frugivore_Insectivore
589	*Cercotrichas leucophrys* Vieillot, LJP	White-browed Scrub-robin	Muscicapidae	f	Insectivore
35	*Ephippiorhynchus senegalensis* Shaw, G	Saddle-billed Stork	Ciconiiformes	W	Carnivore
22	*Ardea intermedia* Wagler, JG	Yellow -billed egret	Ardeidae	W	Carnivore

937	*Amblyospiza albifrons* Vigors, NA	Grosbeak Weaver	Ploceidae	w	Frugivore_Insectivore
573	*Cossypha heuglini* Hartlaub, KJG	White-browed robin-Chat	Muscicapidae	f	Omnivore
995	*Crithagra mozambica* Müller, PLS	Yellow-fronted Canary	Fringillidae	G	Granivore_Insectivore
661	*Schistolais leucopogon* Cabanis, JL	White-chinned prinia	Cisticolidae	F	Insectivore
21	*Egretta garzetta* Linnaeus, C	Little egret	Ardeidae	W	Carnivore

All the recorded avifauna species were of least concern according to IUCN

Appendix 4: Avifauna diversity across their habitat affinities and feeding guilds in *Lantana camara* invaded and non-invaded sites of Katonga Wildlife Reserve

Transect	Invasion	Group	Group type	Shannon
T1	Invaded	f	Ecological categories	2.41
T2	Invaded	f	Ecological categories	2.55
T3	Invaded	f	Ecological categories	2.84
T4	Non-invaded	f	Ecological categories	2.16
T5	Non-invaded	f	Ecological categories	2.95
T6	Non-invaded	f	Ecological categories	2.48
T1	Invaded	w	Ecological categories	0.41
T2	Invaded	w	Ecological categories	0.64
T3	Invaded	w	Ecological categories	0.64
T4	Non-invaded	w	Ecological categories	0.00
T5	Non-invaded	w	Ecological categories	1.85
T6	Non-invaded	w	Ecological categories	0.00
T1	Invaded	F	Ecological categories	1.28
T2	Invaded	F	Ecological categories	1.39
T3	Invaded	F	Ecological categories	2.03
T4	Non-invaded	F	Ecological categories	2.26
T5	Non-invaded	F	Ecological categories	1.57
T6	Non-invaded	F	Ecological categories	1.55
T1	Invaded	G	Ecological categories	1.49
T2	Invaded	G	Ecological categories	0.86
T3	Invaded	G	Ecological categories	0.00
T4	Non-invaded	G	Ecological categories	1.23
T5	Non-invaded	G	Ecological categories	1.92
T6	Non-invaded	G	Ecological categories	1.09
T1	Invaded	W	Ecological categories	0.00
T2	Invaded	W	Ecological categories	1.22
T3	Invaded	W	Ecological categories	0.67
T4	Non-invaded	W	Ecological categories	0.00
T5	Non-invaded	W	Ecological categories	0.89
T6	Non-invaded	W	Ecological categories	0.00
T1	Invaded	Frugivore	Feeding guild	0.70
T2	Invaded	Frugivore	Feeding guild	0.69
T3	Invaded	Frugivore	Feeding guild	1.24
T4	Non-invaded	Frugivore	Feeding guild	1.36
T5	Non-invaded	Frugivore	Feeding guild	1.32
T6	Non-invaded	Frugivore	Feeding guild	1.50
T1	Invaded	Insectivore	Feeding guild	2.57
T2	Invaded	Insectivore	Feeding guild	2.50

T3	Invaded	Insectivore	Feeding guild	2.42
T4	Non-invaded	Insectivore	Feeding guild	2.57
T5	Non-invaded	Insectivore	Feeding guild	2.99
T6	Non-invaded	Insectivore	Feeding guild	2.02
T1	Invaded	Granivore	Feeding guild	0.69
T2	Invaded	Granivore	Feeding guild	0.91
T3	Invaded	Granivore	Feeding guild	1.04
T4	Non-invaded	Granivore	Feeding guild	1.33
T5	Non-invaded	Granivore	Feeding guild	1.66
T6	Non-invaded	Granivore	Feeding guild	0.69
T1	Invaded	Carnivore	Feeding guild	0.64
T2	Invaded	Carnivore	Feeding guild	1.36
T3	Invaded	Carnivore	Feeding guild	0.56
T4	Non-invaded	Carnivore	Feeding guild	0.00
T5	Non-invaded	Carnivore	Feeding guild	0.99
T6	Non-invaded	Carnivore	Feeding guild	0.00
T1	Invaded	Omnivore	Feeding guild	0.48
T2	Invaded	Omnivore	Feeding guild	0.00
T3	Invaded	Omnivore	Feeding guild	1.57
T4	Non-invaded	Omnivore	Feeding guild	1.00
T5	Non-invaded	Omnivore	Feeding guild	1.52
T6	Non-invaded	Omnivore	Feeding guild	0.96
T1	Invaded	Frugivore_Insectivore	Feeding guild	0.69
T2	Invaded	Frugivore_Insectivore	Feeding guild	0.68
T3	Invaded	Frugivore_Insectivore	Feeding guild	0.97
T4	Non-invaded	Frugivore_Insectivore	Feeding guild	1.07
T5	Non-invaded	Frugivore_Insectivore	Feeding guild	0.59
T6	Non-invaded	Frugivore_Insectivore	Feeding guild	1.06
T1	Invaded	Nectarivore_Insectivore	Feeding guild	0.00
T2	Invaded	Nectarivore_Insectivore	Feeding guild	0.00
T3	Invaded	Nectarivore_Insectivore	Feeding guild	0.64
T4	Non-invaded	Nectarivore_Insectivore	Feeding guild	0.41
T5	Non-invaded	Nectarivore_Insectivore	Feeding guild	0.69
T6	Non-invaded	Nectarivore_Insectivore	Feeding guild	0.00
T1	Invaded	Granivore_Insectivore	Feeding guild	0.00
T2	Invaded	Granivore_Insectivore	Feeding guild	0.00
T3	Invaded	Granivore_Insectivore	Feeding guild	0.00
T4	Non-invaded	Granivore_Insectivore	Feeding guild	0.00
T5	Non-invaded	Granivore_Insectivore	Feeding guild	0.00
T6	Non-invaded	Granivore_Insectivore	Feeding guild	0.00

Appendix 5: A list of Mammal species Recorded in *Lantana camara* invaded and non-invaded sites of Katonga Wildlife Reserve

English name	Species name	Mammal Family	Feeding guild
Waterbuck	*Kobus ellipsiprymnus* Ogilby	Bovidae	Grazer
Impala	*Aepyceros melampus Lichtenstein, 1812*	Bovidae	Browser_Grazer
Bushbuck	*Tragelaphus scriptus* Pallas	Bovidae	Browser_Grazer
Topi	*Damaliscus lunatus* Burchell	Bovidae	Grazer
Zebra	*Equus quagga* Boddaert	Equidae	Grazer
Elephant	*Loxodonta africana* Blumenbach	Elephantidae	Browser_Grazer
Blue duiker	*Philantomba monticola* Thunberg	Bovidae	Browser
Warthog	*Phacochoerus africanus* Gmelin	Bovidae	Omnivore
Hippopotamus	*Hippopotamus amphibius* Linnaeus	Hippopotamidae	Grazer
Vervet monkeys	*Chlorocebus pygerythrus F.* Cuvier	Cercopithecidae	Omnivore
Black and White Colobus Monkey	*Colobus guereza* Rüppell	Cercopithecidae	Browser
Ashy red colobus monkey	*Piliocolobus tephrosceles* Elliot	Cercopithecidae	Browser
African striped ground squirre	*Xerus erythropus* É.Geoffroy Saint-Hilaire	Sciuridae	Omnivore
Marsh mongoose	*Atilax paludinosus* G. [Baron] Cuvier	Herpestidae	Omnivore

Appendix 6: Mammal diversity indices by feeding guilds recorded in *Lantana camara* invaded and non-invaded sites of Katonga Wildlife Reserve

	Feeding.guild	Transect	Invasion	Shannon
1	Omnivore	T1	Invaded	0.67
2	Omnivore	T2	Invaded	0.69
3	Omnivore	T3	Invaded	1.10
4	Omnivore	T4	Non-Invaded	0.87
5	Omnivore	T5	Non-Invaded	1.00
6	Omnivore	T6	Non-Invaded	0.95
7	Grazer	T1	Invaded	0.63
8	Grazer	T2	Invaded	0.64
9	Grazer	T3	Invaded	0.66
10	Grazer	T4	Non-Invaded	1.23
11	Grazer	T5	Non-Invaded	0.67
12	Grazer	T6	Non-Invaded	0.35
13	Browser_Grazer	T1	Invaded	0.00
14	Browser_Grazer	T2	Invaded	0.00
15	Browser_Grazer	T3	Invaded	0.68
16	Browser_Grazer	T4	Non-Invaded	0.99
17	Browser_Grazer	T5	Non-Invaded	0.69
18	Browser_Grazer	T6	Non-Invaded	0.67
19	Browser	T1	Invaded	0.00
20	Browser	T2	Invaded	0.61
21	Browser	T3	Invaded	0.79
22	Browser	T4	Non-Invaded	0.24
23	Browser	T5	Non-Invaded	0.38
24	Browser	T6	Non-Invaded	0.00